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THE BENTHIC ECOLOGY OF MARINE AGGREGATE DEPOSITS

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THE BENTHIC ECOLOGY OF MARINE AGGREGATE DEPOSITS

By

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DOCTOR OF PHILOSOPHY

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Faculty of Science

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ABSTRACT

Macrobenthic assemblages and sedimentary conditions of five areas were described and compared; the role of abiotic factors in shaping benthic distributions was also examined. Two of the study sites were actively dredged at the time of sample collection and the effects of this aggregate extraction were investigated. It was concluded that trailer dredging (at the level intensity employed at these sites) was not associated with any significant disruption to benthic community structure, despite the differences in abiotic and biotic characteristics of the two dredged areas. A meta-analysis of findings from published dredging impact studies revealed a differential response to dredging that was related to habitat type. Full recovery of the benthos following the cessation of dredging was not observed in any of the habitat types suggesting that dredging is associated with a long-term alteration of macrobenthic resources.

EXECUTIVE SUMMARY

Macrobenthic assemblages and sedimentary conditions of five areas were described and compared, the role of abiotic factors in shaping benthic distributions was also examined. Two of the study sites were actively dredged at the time of sample collection and the effects of this aggregate extraction were investigated.

Samples were collected, using a 0.2 m² Hamon Grab, from North Nab located off the Isle of Wight (131 sample stations), West Bassurelle, offshore in the eastern Channel (100 samples), Coal Pit offshore in the southern North Sea (194 stations), 40 samples were collected from each of two areas off the Suffolk/Norfolk coast, Lowestoft and Shipwash Gabbard.

The faunal assemblages of the five study sites show some species in common but are largely dissimilar in terms of community type. Dynamic, frequently disturbed areas with sediments fine enough to be mobilised regularly, i.e. Lowestoft, were shown to be inhabited by faunal communities that differed to those described for more stable habitats. Tubiculous, sessile and epilithic forms were not evident in significant numbers under such conditions. However, robust, mobile taxa with a high fecundity and rapid regeneration periods such as *Ophelia borealis* were found in large numbers. In contrast the relative stability of West Bassurelle, characterised by deeper water and coarser sediments, supported a diverse range of sessile, tubiculous, epilithic, and slow growing species including *Distomus variolosus* and the gastropod *Crepidula fornicata*.

The role of sediment particle size in shaping the distribution of the benthic macrofauna was investigated by multivariate techniques. It was shown that particle

size was less important at the more stable habitats than in those locations subjected to a more vigorous hydrophysical regime.

Spatial patterns are evident in the communities at North Nab and Coal Pit; inter-sample differences occur at smaller distances at the English Channel site, this may reflect a greater habitat complexity provided by a coarser substrate i.e. spaces between the gravel might be filled with finer particles that provide ideal conditions for small invertebrates favouring fine sediments. The presence of epibenthic taxa such as hydroids and bryozoans also provide refugia for other invertebrates. Such complexity is absent at Coal Pit thus leading to increased sample uniformity. The amphipoda were found to be present in similar patch sizes at both locations, these patches being separated by similar distances. The contrasting environmental conditions at these sites make it probable that patch size and separation in the amphipoda is mediated by behavioural/life history traits of this group.

Impacts of aggregate extraction at the North Nab site were found only in stations subjected to anchor dredging. Anchor dredging was associated with a benthic community shift from one characterised by *Distomus variolosus*, tubicolous polychaetes and the gastropod *Crepidula fornicata* to a community dominated by juvenile *Balanus balanus*. Trailer dredging was not shown to have any significant impacts on macrobenthic community structure. Impacts of trailer dredging on the benthic invertebrate community at the Coal pit site were not demonstrated. It was therefore concluded that trailer dredging (at the level intensity employed at these sites) was not associated with any significant disruption to benthic community structure.

Meta-analysis of a number of dredging studies has shown that gravel deposits exhibit the largest initial negative response to dredging, with sand showing the least and muds being intermediately affected. Complete recovery was not demonstrated in any of the habitat types within the time period examined (approximately 7 years). In gravels, recovery is slower than in sands or muds. It seems likely that a permanent alteration to the composition of the macrobenthos occurs in many gravel deposits. Recovery in sands is highly variable. This may be a result of the unstable nature of such deposits. Impacts derived from dredging possibly fall within the range of impacts associated with the natural disturbances to which these deposits are subjected, i.e. storm episodes. Muddy deposits display the most advanced point of recovery with pre-dredging conditions almost returning within 7 years.

CONTENTS

LIST OF TABLES	1
LIST OF FIGURES.....	3
ACKNOWLEDGEMENTS	12
AUTHOR'S DECLARATION.....	13

CHAPTER 1. INTRODUCTION

1.1. Seabed Disturbances	15
1.1.1. Natural Disturbances	15
1.1.1.1. <i>Waves, Currents and Storms</i>	15
1.1.1.2. <i>Bioturbation</i>	16
1.1.1.3. <i>Geographical Events</i>	17
1.1.2. Anthropogenic Disturbances	18
1.1.2.1. <i>Fishing</i>	18
1.1.2.2. <i>Ocean Disposal and Oil Pollution</i>	20
1.2. Impacts of Aggregate Extraction	22
1.2.1. Summary	30
1.3. Aims of Thesis	31

CHAPTER 2. REGION OF STUDY

2.1. Selected Study Sites	33
2.1.1. Shipwash Gabbard	33
2.1.2. Lowestoft	36
2.1.3 West Bassurelle.....	36
2.1.4 North Nab	39
2.1.5 Coal Pit	42

CHAPTER 3. MATERIALS & METHODS

3.1. Collection and Extraction Procedures	46
3.2. Biomass Determination	48
3.3. Particle Size Analysis	48
3.4. Multivariate Analysis.....	48

CHAPTER 4. DESCRIPTION OF THE BENTHIC COMMUNITY ATTRIBUTES OF FIVE GEOGRAPHICALLY SEPARATED REGIONS WITH CONTRASTING ENVIRONMENTAL CHARACTERISTICS

4.1. Introduction51

4.2. Analytical Methods.....54

 4.2.1. Univariate Analysis 54

 4.2.2. Multivariate Analysis 54

4.3. Site Specific Analysis58

 4.3.1. Shipwash Gabbard58

 4.3.1.1. *Substratum Composition*.....58

 4.3.1.2. *General Features of Benthic Community Structure*62

 4.3.2. Lowestoft70

 4.3.2.1. *Substratum Composition*.....70

 4.3.2.2. *General Features of Benthic Community Structure*75

 4.3.3. West Bassurelle.....81

 4.3.3.1. *Substratum Composition*.....81

 4.3.3.2. *General Features of Benthic Community Structure*81

 4.3.4. North Nab93

 4.3.4.1. *Substratum Composition*.....93

 4.3.4.2. *General Features of Benthic Community Structure*94

 4.3.5. Coal Pit.....106

 4.3.5.1. *Substratum Composition*.....106

 4.3.5.2. *General Features of Benthic Community Structure*111

 4.3.6. Summary118

4.4. Results of Regional Comparisons119

 4.4.1. Univariate Measures.....119

 4.4.2. Multivariate Analysis.....123

 4.4.3. Feeding Groups.....136

 4.4.4. Discussion139

 4.4.5. Summary142

CHAPTER 5. ANIMAL: SEDIMENT RELATIONSHIPS. THE ROLE OF ABIOTIC FACTORS IN SHAPING BENTHIC DISTRIBUTIONS

5.1. Introduction144

5.2. Analytical Methods.....145

5.3. Results146

5.4. Discussion147

5.5. Summary155

CHAPTER 6. SPATIAL RELATIONSHIPS IN BENTHIC MACROFAUNA

6.1. Introduction	157
6.2. Analytical Methods.....	158
6.3. Results	160
6.3.1. Relationship with distance	160
6.3.2. Relationship with distance classes (small scale).....	160
6.3.3. Relationship with distance classes (large scale)	163
6.4. Discussion	169
6.5. Summary	174

CHAPTER 7. IMPACTS OF AGGREGATE EXTRACTION ON THE BENTHOS

7.1. Introduction	176
7.2. Materials and Methods.....	181
7.2.1. Sample Positions.....	181
7.2.1.1. <i>North Nab</i>	181
7.2.1.2. <i>Coal Pit</i>	181
7.2.2. Analytical Methods	184
7.2.2.1. <i>Univariate Methods</i>	184
7.2.2.2. <i>Multivariate Methods</i>	184
7.3. Results	188
7.3.1. <i>North Nab</i>	188
7.3.2. <i>Coal Pit</i>	195
7.3.3. Production Meta-analysis (<i>North Nab</i> and <i>Coal Pit</i> data combined)	205
7.4. Discussion	209
7.4.1. <i>North Nab</i>	209
7.4.2. <i>Coal Pit</i>	212
7.4.3. Comparison of dredging impacts at <i>North Nab</i> and <i>Coal Pit</i>	215
7.5. Summary	221

CHAPTER 8. QUANTITATIVE META ANALYSIS OF DREDGING IMPACTS ON BENTHIC MACROFAUNA

8.1. Introduction	223
8.2. Materials and Methods.....	223
8.3. Results	225
8.4. Discussion	229
8.5. Summary	239
8.6. Recommendations for Dredging Best Practices	240

CHAPTER 9. STUDY CRITQUE	242
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REFERENCES	250
-------------------------	------------

APPENDICES (see attached CD Rom)

LIST OF TABLES

Table 1. Total tonnage of sewage sludge and dredged material disposed of at sea in the UK. during 1989. Figures are also given for the quantities, in tonnes, of a range of trace metal contaminants contained within the disposed material. (MAFF, 1991).

Table 2. Summary of the species that contribute to the similarity of macrofauna recorded within the three sediment groups, identified by multivariate analysis of the samples taken in the Shipwash Gabbard study area. The average abundance, the average similarity, the ratio of average similarity to standard deviation and the % contribution of each species to the similarity of each group is shown. For full results refer to Appendix XXIV.

Table 3. Summary of the species that contribute to the similarity of macrofauna recorded within the two sediment groups, identified by multivariate analysis of the samples taken in the Lowestoft study area. The average abundance, the average similarity, the ratio of average similarity to standard deviation and the % contribution of each species to the similarity of each group is shown. For full results refer to Appendix XXV.

Table 4. Summary of the species that contribute to the similarity of macrofauna recorded within the three sediment groups, identified by multivariate analysis of the samples taken in the West Bassurelle study area. The average abundance, the average similarity, the ratio of average similarity to standard deviation and the % contribution of each species to the similarity of each group is shown. For full results refer to Appendix XXVI.

Table 5. Summary of the species that contribute to the similarity of macrofauna recorded within the four sediment groups, identified by multivariate analysis of the samples taken in the North Nab study area. The average abundance, the average similarity, the ratio of average similarity to standard deviation and the % contribution of each species to the similarity of each group is shown. For full data refer to Appendix XXVII.

Table 6. Summary of the species that contribute to the similarity of macrofauna recorded within the three sediment groups, identified by multivariate analysis of the samples taken in the Coal Pit study area. The average abundance, the average similarity, the ratio of average similarity to standard deviation and the % contribution of each species to the similarity of each group is shown. For full data refer to Appendix XXVIII.

Table 7. One-way ANOVA on a range of ecological indices, also shown are the results of a posteriori power analysis.

Table 8. Average abundance and percentage contribution to the internal similarity of the top five ranked genera recorded from each of the study areas.

Table 9. Pairwise comparisons of each of the five study locations and the average dissimilarity between each pair of sites. Average abundance and percentage contribution to the dissimilarity between each of the study areas made by key taxa is also shown.

Table 10. The optimal combinations and single sedimentary variables responsible for the best match, as measured by weighted Spearman Rank correlation, with the benthos recorded at five locations.

Table 11. Characteristics of opportunistic (*r*-selected) and equilibrium (*k*-selected) taxa (adapted from Zajac *et al.* 1998).

Table 12. Summary of the Spearman Rank correlations between corresponding elements of similarity matrices calculated from abundance of major groups using untransformed and fourth root transformed data and inter-sample distances within the five study areas (* $p < 0.05$, ** $p < 0.01$).

Table 13. Summarising the results of a one-way ANOVA of number of species, number of individuals, species richness (*d*), species evenness (Pielou's *J*), Shannon-Weiner diversity ($H(\log_e)$), Simpson's dominance, total biomass and biomass of major groups. Comparisons are made between control, anchor and trailer dredged stations for the North Nab study area. Results of a *posteriori* power analysis are also shown.

Table 14. Summarising the results of a one-way ANOVA of number of species, number of individuals, species richness (*d*), species evenness (Pielou's *J*), Shannon-Weiner diversity ($H(\log_e)$), Simpson's dominance, total biomass and biomass of major groups. Comparisons are made between control, anchor and trailer dredged stations for the Coal Pit study area. Results of a *posteriori* power analysis are also shown.

Table 15. Summary of ANOVA of time categories (since dredging) in three contrasting habitat types. * $P < 0.05$, ** $P < 0.01$, *** $P < 0.001$.

Table 16. Mean macrobenthic response over time in northern European gravel and sand habitats.

LIST OF FIGURES

Figure 1. A. Location of study areas. B. Sample locations (shown by dots) in the Coal Pit, Lowestoft and Shipwash Gabbard study areas. C. Sample locations (shown by dots) in the West Bassurelle and North Nab study areas.

Figure 2. Location of samples collected from the Shipwash Gabbard study area. ARCS chart 1610 used under licence from the UK. Hydrographic Office.

Figure 3. Location of samples collected from the Lowestoft study area. ARCS chart 1543 used under licence from the UK. Hydrographic Office.

Figure 4. Location of samples collected from the West Bassurelle study area. ARCS chart 2451 used under licence from the UK. Hydrographic Office.

Figure 5. Location of samples collected from the North Nab study area. ARCS chart 2045 used under licence from the UK. Hydrographic Office.

Figure 6. Location of samples collected from the Coal Pit study area. ARCS chart 1187 and 1503 used under licence from the UK. Hydrographic Office.

Figure 7a. Deployment of a 0.2 m² Hamon grab.

Figure 7b. 0.1 m² Day grab prior to deployment.

Figure 8. The relative percentage of silt (<0.063 mm), sand (0.063-1 mm) and gravel (2-32 mm) at benthic stations sampled at the Shipwash Gabbard area in August 1996.

Figure 9. Group average sorting dendrogram showing the percentage similarity of the sediments at stations sampled in the Shipwash Gabbard study area. Based on the Bray-Curtis similarity of untransformed percentage retained particle size data.

Figure 10. Two-dimensional plot of a multi-dimensional scaling (MDS) ordination of the particle size fractions at stations sampled in the Shipwash Gabbard study area. Based on the Bray-Curtis similarity of untransformed percentage retained particle size data.

Figure 11. The distribution of sediment groups identified by non-parametric multivariate analysis of particle size composition of benthic stations sampled in the Shipwash Gabbard study area.

Figure 12. Thematic map showing the number of species of macrofauna (>1 mm) per 0.2 m² based on Hamon grab samples at benthic stations sampled in the Shipwash Gabbard study area.

Figure 13. Thematic map showing the number of individuals of macrofauna (>1 mm) per 0.2 m² based on Hamon grab samples at benthic stations sampled in the Shipwash Gabbard study area.

Figure 14. Thematic map showing the biomass (ash free dry weight) in grams per 0.2 m² based on Hamon grab samples at benthic stations sampled in the Shipwash Gabbard study area.

Figure 15. Two-dimensional plot of a multi-dimensional scaling (MDS) ordination of the benthic macrofauna (>1 mm) sampled at benthic stations in the Shipwash Gabbard study area. Based on the Bray-Curtis similarity of square-root transformed species abundance data. The upper plot shows station numbers, the lower plot depicts superimposed sediment groups.

Figure 16. The relative percentage of silt (<0.063 mm), sand (0.063-1 mm) and gravel (2-32 mm) at benthic stations sampled in the Lowestoft area.

Figure 17. Group average sorting dendrogram showing the percentage similarity of the sediments at stations sampled in the Lowestoft study area. Based on the Bray-Curtis similarity of untransformed percentage retained particle size data.

Figure 18. Two-dimensional plot of a multi-dimensional scaling (MDS) ordination of the particle size fractions at stations sampled in the Lowestoft study area. Based on the Bray-Curtis similarity of untransformed percentage retained particle size data.

Figure 19. The distribution of sediment groups identified by non-parametric multivariate analysis of particle size composition of benthic stations sampled in the Lowestoft study area.

Figure 20. Thematic map showing the number of species of macrofauna (>1 mm) per 0.2 m² based on Hamon grab samples at benthic stations sampled in the Lowestoft study area.

Figure 21. Thematic map showing the number of individuals of macrofauna (>1 mm) per 0.2 m² based on Hamon grab samples at benthic stations sampled in the Lowestoft study area.

Figure 22. Thematic map showing the biomass (ash free dry weight) in grams per 0.2 m² based on Hamon grab samples at benthic stations sampled in the Lowestoft study area.

Figure 23. Two-dimensional plot of a multi-dimensional scaling (MDS) ordination of the benthic macrofauna (>1 mm) sampled at benthic stations in the Lowestoft study area. Based on the Bray-Curtis similarity of square-root transformed species abundance data. The upper plot shows station numbers, the lower plot depicts superimposed sediment groups.

Figure 24. The relative percentage of silt (<0.063 mm), sand (0.063-1 mm) and gravel (2-32 mm) at benthic stations sampled at the West Bassurelle study area.

Figure 25. Group average sorting dendrogram showing the percentage similarity of the sediments at stations sampled in the West Bassurelle study area. Based on the Bray-Curtis similarity of untransformed percentage retained particle size data.

Figure 26. Two-dimensional plot of a multi-dimensional scaling (MDS) ordination of the particle size fractions at stations sampled in the West Bassurelle study area. Based on the Bray-Curtis similarity of untransformed percentage retained particle size data.

Figure 27. The distribution of sediment groups identified by non-parametric multivariate analysis of particle size composition of benthic stations sampled in the West Bassurelle study area.

Figure 28. The relative contributions of the main faunal components to species diversity, population density, and biomass of the benthos recorded in the West Bassurelle study area. The abundance of 'Miscellania' is not comparable with that for other groups due to difficulties in the quantification of colonial organisms (such taxa have been recorded as 'present' and assigned a nominal value of 1).

Figure 29. Thematic map showing the number of species of macrofauna (>1 mm) per 0.2 m² based on Hamon grab samples at benthic stations sampled in the West Bassurelle study area.

Figure 30. Thematic map showing the number of individuals of macrofauna (>1 mm) per 0.2 m² based on Hamon grab samples at benthic stations sampled in the West Bassurelle study area.

Figure 31. Thematic map showing the biomass (ash free dry weight) in grams per 0.2 m² based on Hamon grab samples at benthic stations sampled in the West Bassurelle study area.

Figure 32. Two-dimensional plot of a multi-dimensional scaling (MDS) ordination of the benthic macrofauna (>1 mm) sampled at benthic stations in the West Bassurelle study area. Based on the Bray-Curtis similarity of square-root transformed species abundance data. The upper plot shows station numbers, the lower plot depicts superimposed sediment groups.

Figure 33. The relative percentage of silt (<0.063 mm), sand (0.063-1 mm) and gravel (2-32 mm) at benthic stations sampled at the North Nab study area.

Figure 34. Group average sorting dendrogram showing the percentage similarity of the sediments at stations sampled in the North Nab study area. Based on the Bray-Curtis similarity of untransformed percentage retained particle size data.

Figure 35. Two-dimensional plot of a multi-dimensional scaling (MDS) ordination of the particle size fractions at stations sampled in the North Nab study area. Based on the Bray-Curtis similarity of untransformed percentage retained particle size data.

Figure 36. The distribution of sediment groups identified by non-parametric multivariate analysis of particle size composition of benthic stations sampled in the North Nab study area.

Figure 37. The relative contributions of the main faunal components to species diversity, population density, and biomass of the benthos recorded in the North Nab study area. The abundance of *Miscellania* is not comparable with that for other groups due to difficulties in the quantification of colonial organisms (such taxa have been recorded as 'present' and assigned a nominal value of 1).

Figure 38. Thematic map showing the number of species of macrofauna (>1 mm) per 0.2 m² based on Hamon grab samples at benthic stations sampled in the North Nab study area.

Figure 39. Thematic map showing the number of individuals of macrofauna (>1 mm) per 0.2 m² based on Hamon grab samples at benthic stations sampled in the North Nab study area.

Figure 40. Thematic map showing the biomass (ash free dry weight) in grams per 0.2 m² based on Hamon grab samples at benthic stations sampled in the North Nab study area.

Figure 41. Two-dimensional plot of a multi-dimensional scaling (MDS) ordination of the benthic macrofauna (>1 mm) sampled at benthic stations in the North Nab study area. Based on the Bray-Curtis similarity of square-root transformed species abundance data. The upper plot shows station numbers, the lower plot depicts superimposed sediment groups.

Figure 42. The relative percentage of silt (<0.063 mm), sand (0.063-1 mm) and gravel (2-32 mm) at benthic stations sampled at the Coal Pit study area.

Figure 43. Group average sorting dendrogram showing the percentage similarity of the sediments at stations sampled in the Coal Pit study area. Based on the Bray-Curtis similarity of untransformed percentage retained particle size data.

Figure 44. Two-dimensional plot of a multi-dimensional scaling (MDS) ordination of the particle size fractions at stations sampled in the Coal Pit study area. Based on the Bray-Curtis similarity of untransformed percentage retained particle size data.

Figure 45. The distribution of sediment groups identified by non-parametric multivariate analysis of particle size composition of benthic stations sampled in the Coal Pit study area.

Figure 46. The relative contributions of the main faunal components to species diversity, population density, and biomass of the benthos recorded in the Coal Pit study area. The abundance of *Miscellania* is not comparable with that for other groups due to difficulties in the quantification of colonial organisms (such taxa have been recorded as 'present' and assigned a nominal value of 1).

Figure 47. Thematic map showing the number of species of macrofauna (>1 mm) per 0.2 m² based on Hamon grab samples at benthic stations sampled in the Coal Pit study area.

Figure 48. Thematic map showing the number of individuals of macrofauna (>1 mm) per 0.2 m² based on Hamon grab samples at benthic stations sampled in the Coal Pit study area.

Figure 49. Thematic map showing the biomass (ash free dry weight) in grams per 0.2 m² based on Hamon grab samples at benthic stations sampled in the Coal Pit study area.

Figure 50. Two-dimensional plot of a multi-dimensional scaling (MDS) ordination of the benthic macrofauna (>1 mm) sampled at benthic stations in the Coal Pit study area. Based on the Bray-Curtis similarity of square-root transformed species abundance data. The upper plot shows station numbers, the lower plot depicts superimposed sediment groups.

Figure 51. Plots showing the average number of species, individuals and species richness (Margalef's d) for the five study areas. Error bars show standard deviation.

Figure 52. Plots showing the average species evenness (Pielou's J), species dominance (Shannon-Weiner's H) and Simpson's dominance for the five study areas. Error bars show standard deviation.

Figure 53. Two-dimensional plots of a multi-dimensional scaling (MDS) ordination of genera recorded at five study areas from different locations in the North Sea and English Channel, subjected to various transformations based on Bray-Curtis similarity of genera abundance data.

Figure 54. Two-dimensional plots of a multi-dimensional scaling (MDS) ordination of genera recorded in each of five study areas from different locations in the North Sea and English Channel. Relative abundance of frequently recorded taxa are highlighted.

Figure 55. Two-dimensional plots of a multi-dimensional scaling (MDS) ordination of genera recorded in each of five study areas from different locations in the North Sea and English Channel. Relative abundance of frequently recorded taxa are highlighted.

Figure 56. Two-dimensional plots of a multi-dimensional scaling (MDS) ordination of genera recorded in each of five study areas from different locations in the North Sea and English Channel. Relative abundance of frequently recorded taxa are highlighted.

Figure 57. Two-dimensional plots of a multi-dimensional scaling (MDS) ordination of genera recorded in each of five study areas from different locations in the North Sea and English Channel. Relative abundance of frequently recorded taxa are highlighted.

Figure 58. Cumulative dominance curves plotted for genera recorded in each of five study areas from different locations in the North Sea and English Channel.

Figure 59. The relative abundance (%) of different macrofaunal feeding groups recorded in each of five study areas from different locations in the North Sea and English Channel.

Figure 60. Two-dimensional plots of a multi-dimensional scaling (MDS) ordination of macrofaunal feeding groups recorded in each of five study areas from different locations in the North Sea and English Channel. Distributions of frequently recorded taxa are also shown. Based on the Bray-Curtis similarity of untransformed feeding group abundance data.

Figure 61. *Sabellaria spinulosa* (arrowed) Note the non-reef forming growth form present at North Nab and West Bassurelle.

Figure 62. Rank correlograms of Bray-Curtis Similarities between samples from the North Nab study area assigned to various distance classes. Calculations have been made using untransformed and Root-Root transformed data of macrofauna and different groups (P values <0.025 or >0.975 estimated by Monte Carlo permutation are unshaded).

Figure 63. Rank correlograms of Bray-Curtis Similarities between samples from the Coal Pit study area assigned to various distance classes. Calculations have been made using untransformed and Root-Root transformed data of macrofauna and different groups (P values <0.025 or >0.975 estimated by Monte Carlo permutation are unshaded).

Figure 64. Rank correlograms of Bray-Curtis Similarities between samples from the West Bassurelle study area assigned to various distance classes. Calculations have been made using untransformed and Root-Root transformed data of macrofauna and different groups (P values <0.025 or >0.975 estimated by Monte Carlo permutation are unshaded).

Figure 65. Rank correlograms of Bray-Curtis Similarities between samples from the Shipwash Gabbard study area assigned to various distance classes. Calculations have been made using untransformed and Root-Root transformed data of macrofauna and different groups (P values <0.025 or >0.975 estimated by Monte Carlo permutation are unshaded).

Figure 66. Rank correlograms of Bray-Curtis Similarities between samples from the Lowestoft study area assigned to various distance classes. Calculations have been made using untransformed and Root-Root transformed data of macrofauna and different groups (P values <0.025 or >0.975 estimated by Monte Carlo permutation are unshaded).

Figure 67. Predicted relative levels of importance of factors controlling successional dynamics over different scales of disturbance (redrawn from Zajac *et al.* 1998).

Figure 68. Uses of marine aggregates mined in UK waters (BMAPA 2000).

Figure 69. Trailer dredger (Arco Humber) operating in the North Sea, showing material rejected by overboard screening.

Figure 70. Diagrams showing Trailer and Anchor dredgers (redrawn from BMAPA 2000).

Figure 71. Detailed view of Control, Trailer dredged and Anchor dredged stations sampled in the North Nab study area during March/September 1999.

Figure 72. Detailed view of Control, Dredged and Abandoned stations sampled in the Coal Pit study area.

Figure 73. Percentage contribution to the total biomass by various taxonomic groups in stations subjected to different dredging pressures in the North Nab study area. Control stations are those assumed to be unaffected by dredging activity. Non-dredged stations are those which are not directly dredged but may be influenced by sedimentation of the 'outwash' material.

Figure 74. The mean size of individuals (AFDW/g) recorded at stations subjected to different dredging pressures in the North Nab study area. Control stations are those assumed to be unaffected by dredging activity.

Figure 75. Two-dimensional plot of a multi-dimensional scaling (MDS) ordination of the benthic macrofauna sampled in the North Nab study area, based on the Bray-Curtis similarity of square-root transformed species abundance data. Symbols relating to dredging intensity at each station are also shown. Control stations are those assumed to be unaffected by dredging activity.

Figure 76. Cumulative dominance curves plotted from the benthic macrofauna in stations subjected to different dredging pressures in the North Nab study area. Control stations are those assumed to be unaffected by dredging activity.

Figure 77. Abundance Biomass Comparisons (ABC) curves for Control, Anchor and Trailer dredged samples collected in the North Nab study area. Based on abundance and biomass of Annelida, Crustacea, Mollusca and miscellaneous taxa. Abundance shown in green, biomass shown in blue.

Figure 78. Plot showing the number of individuals belonging to various phyletic groups (per 0.2 m²) in samples taken along a transect running from north to south through an actively dredged site in the North Nab study area. Chordata comprise mainly tunicata in this instance.

Figure 79. Percentage contribution to the total biomass by various taxonomic groups in stations subjected to different dredging pressures in the Coal Pit study area. Control stations are those assumed to be unaffected by dredging activity. Non-dredged stations are those which are not directly dredged but may be influenced by sedimentation of the 'outwash' material.

Figure 80. The mean size of individuals (AFDW/g) recorded in stations subjected to different dredging pressures in the Coal Pit study area. Control stations are those assumed to be unaffected by dredging activity.

Figure 81. Two-dimensional plot of a multi-dimensional scaling (MDS) ordination of the benthic macrofauna sampled in the Coal Pit study area, based on the Bray-Curtis Similarity of square-root transformed species abundance data. Symbols relating to dredging intensity at each station are also shown. Control stations are those assumed to be unaffected by dredging activity.

Figure 82. Cumulative dominance curves plotted for the benthic macrofauna at stations subjected to different dredging pressures in the Coal Pit study area. Control stations are those assumed to be unaffected by dredging activity.

Figure 83. Abundance Biomass Comparisons (ABC) curves for Control, Abandoned and Dredged samples collected in the Coal Pit study area. Based on abundance and biomass of Annelida, Crustacea, Mollusca and miscellaneous taxa. Abundance shown in green, biomass shown in blue.

Figure 84. Plot showing the number of individuals belonging to various phyletic groups (per 0.2 m²) in samples taken along a transect running from south to north through an actively dredged site in the Coal Pit study area.

Figure 85. Two-dimensional plots of a multi-dimensional scaling (MDS) ordination based on the Bray-Curtis similarity of the untransformed approximate productivity values for phyla recorded in samples subjected to various levels of aggregate extraction from the North Nab and Coal Pit study sites.

Figure 86. Schematic diagram of the relationship between environmental stress and taxonomic variability (adapted from Boesch & Rosenberg 1981).

Figure 87a. Predicted means calculated from analysis of variance of initial responses to dredging in contrasting habitat types. 95% confidence limits are shown.

Figure 87b. Dredging response data for categorical variables for which there was a significant interaction term, Logtim = 1+time in days log (base e). Data taken from all habitat types and all regions.

Figure 88a. Scatter plot of recovery data for categorical variables for which there was a significant interaction term Data taken from all habitat types and all regions (Van der Veer *et al.* 1985 study removed).

Figure 88b. Scatter plot of recovery data for categorical variables for which there was a significant interaction term. Data taken from northern European gravel habitats only.

Figure 89a. Scatter plot of recovery data for categorical variables for which there was a significant interaction term. Data taken from northern European sand habitats only (Van der Veer *et al.* 1985 study removed).

Figure 89b. Scatter plot of recovery data for categorical variables for which there was a significant interaction term. Data taken from all mud and muddy sand data only.

Figure 90. Flow diagram showing possible macrobenthic response scenarios in relation to aggregate extraction. References are provided as examples of documented responses in contrasting environmental situations.

Figure 91. Hypothetical dredging impact survey grid.

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AUTHOR'S DECLARATION

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Signed J.E. Robinson
Date 22/02/05

CHAPTER 1. INTRODUCTION



CHAPTER 1. INTRODUCTION

1.1. Seabed Disturbances

The following section identifies the role of varying sources of disturbance in controlling the distributions of benthic communities. To assess the impact of a given disturbance, it is necessary to consider both the frequency and magnitude of the event. For example, sediment resuspensions, induced by anthropogenic and natural activities, differ with respect to their spatial and temporal scales and rates of recurrence. Anthropogenically-induced resuspensions are likely to represent a localised perturbation, whereas storm-induced resuspensions can be considered to be far-field, regional phenomena (Hitchcock 1997). In this thesis 'disturbance' is interpreted as an alteration of the normal function of a system by an episodic event, whether that event is natural (i.e. storm-induced sediment transport) or anthropogenic (i.e. sediment transport induced by deployment of a beam trawl). Outlined below are disturbance processes that are of potential importance in UK waters.

1.1.1. Natural Disturbances

1.1.1.1. Waves, Currents and Storms

Impacts attributed to waves can be considered a dominant controlling force on continental shelves, particularly the extreme wave conditions derived from storm events (Hall 1994). Although disturbance from storm events is greatest in shallow situations, Drake & Cacchione (1985) demonstrated that winter storms might lead to significant levels of sediment resuspension at depths of up to 100m. Furthermore Niedoroda *et al.* (1989) stated that a major storm can deposit a bed several centimetres thick at a depth of 20m and several millimetres thick at 40m. Olafsson *et al.* (1994) noted that currents swept away newly settled juvenile bivalves, precluding their development in an area subjected to currents of

sufficient strength. Whilst it may be argued that sediment transport associated with tidal currents cannot legitimately be categorised as a disturbance due to their consistency, wind-driven currents allied with storms are likely to be an influential source of stress (Hall 1994). Olafsson (1988) demonstrated that juveniles of the clam *Macoma balthica* settled in high densities during periods of reduced wind velocities only to be swept away by autumnal storms. As a consequence, the population density of *M. balthica* was significantly negatively correlated with both mean wind speed and the annual number of days of strong winds.

1.1.1.2. Bioturbation

Bioturbation is probably best defined as the direct disruption of sediments by its inhabitants, whether as a consequence of locomotion through the substrata or feeding upon the organic material contained within it (Snelgrove & Butman 1994). The working of sediments by individuals is likely to represent the smallest scale at which physical disturbance processes operate (Hall 1994). It is probable that impacts of bioturbation are significant only in regions of sand or silt, as regions of coarser (gravel) particles are unlikely to support infaunal taxa that disrupt the sediment to a significant degree. However, in areas where appropriate sedimentary conditions prevail, substratum reworking rates can be extremely large. For example, Thayer (1983) cited values of up to 630 cm³ per day for an individual *Macoma balthica*. Bioturbation could, therefore, be of significance to fauna inhabiting areas characterised by fine sediments.

The physical disruption caused by bioturbation is not necessarily associated with a negative impact on benthic assemblages. Widdicombe *et al.* (2000) demonstrated that bioturbation by the sub-surface deposit feeders *Nuculoma tenuis* (Mollusca)

and *Brissopsis lyrifera* (Echinodermata) promoted increased levels of alpha and beta diversity.

1.1.1.3. Geographical Events

The single most important factor responsible for the present-day nature of the benthic environment around the British Isles is the Pleistocene glaciation of northwestern Europe. This glacial event altered benthic conditions in the coastal waters of the UK in three major ways:

1. **Sea level rise.** At the end of the last ice age, river valleys bearing sand and gravel were submerged by the rising sea levels caused by the melting of the glacial ice. In the intervening period up to the present day further sea level rise has occurred (BMAPA 2000).
2. **Change in water temperature.** It is believed that at the end of the last ice age (around 11200 years ago) there was a modification or possible shutdown of the 'thermohaline circulation' (THC), the circulatory pattern responsible for the Gulf Stream which brings warm water to the north west European coasts. Water temperature may have been lowered by 5 degrees Celcius in the space of a few decades. This was caused by a discharge of fresh water from the melting of the North American ice sheet (Jones & Keen 1993).
3. **Increased sediment transport and deposition.** As relative sea level rose, and the ice sheets retreated landwards during glaciation, sediment deposition to the outer and middle shelf decreased and was restricted largely to fluvial sediment discharge into the inner shelf. During the subsequent temperate climatic period, approximately 10000 years ago to the present day, hydrological processes have driven shelf sediment erosion and transport. Such profound changes to the sedimentary environment have influenced the distributions of the benthic fauna present today (Jones & Keen 1993).

1.1.2. Anthropogenic Disturbances

1.1.2.1. Fishing

Commercial fishing activities must rank amongst the most significant anthropogenic causes of physical disturbance in the marine environment. In recent years, there has been increasing concern regarding the impact on the fauna of the use of heavy trawls and vastly more efficient equipment than in times past. Any fishing gear that is towed over the seabed will disturb the sediment and, therefore, its inhabitants. To some degree, the level of this disturbance is dependent upon the specification of the gear and the sediment type (Hall 1994). Amplitude of disturbance, attributable to trawling in different sediments, has been studied by Churchill (1989) who demonstrated that penetration of fishing gear into soft muds was considerably greater than in areas of hard packed sands, thus leading to variability in sediment suspension rates. It was estimated that penetration of otter boards in coarse sands was approximately 10 mm; this was calculated to result in the resuspension of 39 kg of sediment per second. Penetration figures for fine sands and muds were 20 mm (78 kg sec^{-1}) and 40 mm (112 kg sec^{-1}) respectively. Therefore, physical disturbance of the seabed (and presumably the benthos which it supports) was shown to be related to the habitat over which trawling had occurred as a result of the penetration depth achieved by the fishing gear (Churchill 1989).

The impacts of trawling on the benthic fauna have been studied widely (Van Dolah *et al.* 1991; Bergman & Hup 1992; Kaiser *et al.* 1998). Collie *et al.* (2000) stated that the fauna of stable gravel, mud, and biogenic habitats were more adversely affected by trawling than those in less consolidated coarse sediments. This investigation also indicated that recovery of the benthos following the cessation of

fishing appeared to be most rapid in dynamic habitats characterised by opportunistic taxa (Collie *et al.* 2000).

The potential impacts of trawls and fishing dredges on the benthos and the wider marine environment were summarised by Messieh *et al.* (1991) and included:

- incidental mortality of non-target and target species by direct contact with mobile fishing gears;
- exposure of benthic infauna within dredge tracks leading to increased predation pressure (see also Kaiser & Spencer 1994; Groenewold & Fonds 2000);
- alteration of sediment characteristics, both chemical and physical, leading to changes in community structure;
- resuspension of sediments adversely affecting filter-feeding mechanisms, smothering spawning areas and damaging the respiratory system of many organisms;
- resuspension of contaminants such as heavy metals increasing their bioavailability; and
- increasing phytoplankton production as a result of enhanced benthic/pelagic nutrient fluxes.

The total world-wide areas impacted by trawling is considerable. Messieh *et al.* (1991) estimated that fish trawlers and scallop dredges created tracks criss-crossing the Canadian continental shelf approximately 4.3 million kilometres in length. Watling & Norse (1998) likened the disruption of benthic biodiversity attributable to fishing with mobile gear to that of forest clear-cutting. They estimated that approximately 14.8 million km² of the oceans' continental shelves (an area as large as Brazil, the Congo and India combined) were trawled each year. They concluded that few areas of the world's continental shelves have not been trawled (Watling & Norse 1998). Indeed, Frid & Clark (1999) stated that it is unlikely that, at least in the North Sea, un-fished benthic communities have ever

been studied. Clearly, commercial fishing, particularly with mobile gear, plays an important role (perhaps the single most important) in structuring the benthic communities around our coasts.

1.1.2.2. Ocean Disposal and Oil Pollution

The discarding of many waste products from both domestic and industrial sources is often associated with estuarine or inshore outfalls. However, offshore dumping contributes significantly to 'solving' the dilemma of disposing of many by-products derived from modern day society. The impacts on marine benthos of the dumping of sewage sludge, spoil from mining and capital dredging works, and many industrial wastes, have been studied by many authors (Howell & Shelton 1970; Shelton 1971; Jenkinson 1972; Ellis & Hoover 1990; Johnson & Frid 1995; Smith & Rule 2001). The disposal of such waste materials affects the benthos in several ways and is dependent on the form of the waste being disposed. Large volumes of fine particulates, in sewage sludge or spoil from maintenance dredging, when released to a dump-site are likely to alter significantly the characteristics of the existing sediments in terms of grain size, substratum stability and contaminant levels. Indeed, the types of waste to be disposed of, and the hydrodynamics of the dumpsite, are important in predicting impacts. The deposition of china clay waste was shown to have a marked effect on the benthos of the western English Channel. Howell & Shelton (1970) discovered that filter feeders were prevalent in control stations but largely absent from those affected by clay deposits. Impacted areas contained mainly deposit feeders and were characterised by higher diversities and biomasses than areas unaffected by clay deposition. Johnson & Frid (1995) found that dumping large quantities of coal tailings and minestone resulted in increased diversity along the north east coast of England. In contrast, a study in the eastern Mediterranean carried out by Kress *et al.* (1993) revealed a

severe impoverishment of the benthic fauna at a deep-water coal fly-ash disposal site. Bamber (1984) also noted this severe impoverishment of the benthos at a fly ash disposal site off the north east coast of England. Disposal rates at this site resulted in a layer of waste up to 3 metres thick being deposited in some areas, perhaps unsurprisingly this led to a clear impact. Negative correlations were demonstrated between fly ash content of the substrata and all measured diversity/faunistic parameters.

A further source of environmental stress in offshore situations, and one that has particular relevance to the North Sea, is the dumping of drilling wastes associated with oil and gas platforms. Messieh *et al.* (1991) stated that the extent of deleterious effects of drilling wastes on the benthos in the proximity of such operations is dependent on hydrographic conditions, drilling history (exploratory or production) and the drilling fluid formulation. Impacts in the North Sea, where water-based muds (fluids) were used in drilling operations, are confined to several hundred meters from the well site. In contrast, impacts associated with oil-based muds (fluids) extend to approximately 2-3 km from the well site. This increased 'footprint' is attributable possibly to the increased toxicity of the diesel-based compound. In the immediate vicinity of the drilling platform, benthic macrofauna are likely to be smothered under the drill cuttings or affected by the anoxic conditions resulting from organic enrichment of the seabed (Davies *et al.* 1984).

Macrofaunal community adjustment following the disposal of sewage sludge, capital dredging spoil and drill cuttings can often be dictated by the chemicals contained within the dumped material, rather than the material itself. Trace metals, hydrocarbons, pesticide residues and many other industrially-derived compounds can be present in significant concentrations. A detailed review of the effects of

such compounds, both individually and in combination, is obviously beyond the scope of this study, however, an example of trace metal quantities found in sewage sludge and dredging spoil is given in Table 1.

Table 1. Total tonnage of sewage sludge and dredged material disposed of at sea in the UK. during 1989. Figures are also given for the quantities, in tonnes, of a range of trace metal contaminants contained within the disposed material (MAFF 1991).

Spoil Type	Wet tonnage deposited (t)	Quantities of metal contaminants in sludge (t)						
		Cd	Cr	Cu	Hg	Ni	Pb	Zn
Sewage sludge	9642847	2.68	141	164	1.14	22	184	367
Dredged material	44303995	19.8	1343	1145	10.6	679	2021	5258

Since the end of 1998, sewage sludge is no longer disposed of at sea but may still be an influential factor in shaping the communities in areas once used as dumping grounds by the alteration of sediment make up, increased organic load, raised levels of heavy metals etc. Likewise, the dumping of colliery waste ceased in 1992. Dredged material and refuse now comprises virtually all of the material deposited at sea (Smith & Mackie 1996; DEFRA 2004).

1.2. Impacts of Aggregate Extraction

Early studies into the potential impacts of dredging focussed mainly on the ecological effects of dredging and dumping of solid wastes (e.g. harbour dredging), and resulted in the extensive Dredged Material Research Programme (DMRP) of the corps of Engineers (Becker et al. 1975). As awareness and concern of possible environmental damage caused by dredging works increased, methods were developed to monitor the impacts of aggregate extraction on the seabed. The latter included the use of side scan (side looking) sonar techniques to map the seabed and to give a source of comparison of pre- and post-dredging

features. The value of this method was illustrated by Dickson & Lee (1973a, b) who studied the impact of gravel extraction from a shingle bank south of Hastings. A test pit around 3.5 m in depth was dredged into the bank; ten months later, the pit was mapped and found to be 4.7 m deep (due to the compaction of the layer of fine particles at the base of the pit). One year later, the depth of the pit was unchanged (Dickson & Lee 1973a, b). In accordance with these findings, Eden (1975) stated that, with the exception of areas of mobile sands, backfilling of dredged seabed features is likely to be extremely slow.

A study into the physical and biological effects of trailer dredging off Suffolk concluded that the main impact of dredging on this environment was an alteration of seabed topography and that deposition/suspension of fine sediment did not appear to pose serious environmental consequences as the background turbidity levels were already high (Millner *et al.* 1977). This latter study also stated that dredged tracks or pits were likely to remain persistent features of the seabed topography in areas of gravel substrata (Millner *et al.* 1977). Calculations incorporating measurements of bottom current velocity indicated that particles greater than about 6mm would not be transported even in the strongest of tides (Hall 1994). Work of direct relevance to the present thesis was reported by de Groot (1979b), where the potential impacts of gravel extraction on the seabed in an area in the southern North Sea were investigated. Side scan sonar techniques were employed to map the sea floor of an area subjected to trailer dredging and de Groot (1979b) concluded that the dredging activities may have serious implications for the herring and sand eel populations of the region as the gravels used by these fish for spawning would be removed and the levels of turbidity increased.

In a further study, de Groot (1979a) assessed the potential impacts of large-scale sand dredging in the North Sea. Predictions of the likely impacts on plankton, bottom fauna and fish were made. It was speculated that the plankton would be affected by changes in light regime as a consequence of increased turbidity. Impacts on the benthos were discussed, and it was concluded that physical disturbance by dredging activities (i.e. movement or burial) would account for the destruction of fauna in the dredged areas with survival dependent on the depth of burial and mobility of the organism in question. Possible implications for the fish populations in the region were said to be species specific. Visual feeders, such as mackerel and turbot, were predicted to avoid the area to escape the turbid waters. However, species that employ olfaction for prey location were likely to be relatively unaffected, even being actively drawn to the area, attracted by the odour stream of animals crushed in the dredging process. It was also suggested that the elevated levels of suspended sediments and the subsequent release of organic matter and harmful substances such as PCBs could lead to an increase in mortality of fish and invertebrate larvae (de Groot 1979b).

Jones & Candy (1981) compared the benthic fauna of dredged and non-dredged areas in Botany Bay (Australia), and found that the benthos of dredged areas generally differed from that of nearby non-dredged regions with regard to species composition and species richness (both appeared to be strongly associated with sediment type). Dredging had led to substantial changes in sediment composition, thus indirectly modifying the benthic community structure from one associated with clean sand to a mud-dwelling assemblage. Poiner & Kennedy (1984) studied the macrobenthos of a large sandbank following dredging in Moreton Bay (Australia) and reported decreased numbers of individuals and species in the dredged portions of two transects, and noted an enhancement of the fauna in areas

adjacent to the dredging activities. This enhancement of benthic invertebrates was suggested to have resulted from an increase in nutrients derived from the dispersing plume of suspended particles observed during dredging operations. Gajewski & Uscinowicz (1993) carried out further studies into the potential impacts of sediment plumes associated with dredging. Their study illustrated that the width of the plume, determined by light extinction values in the water column, did not exceed 50 m. Sediment traps positioned at varying distance intervals from the dredging activity revealed that settlement occurred mainly in the immediate vicinity of the dredger, beyond 50 m the amount of material settling on the seabed decreased rapidly. These findings led to the supposition that dredging outwash plumes behave according to Gaussian diffusion principles being disputed. In fact, the plumes appear to act as density currents where particles are held together by cohesion during the initial sedimentation process, indicating that the affected area is likely to be significantly smaller than the worst-case scenarios predicted by early simulation models (Newell *et al.* 1998). Whiteside *et al.* (1995) confirmed these findings when, working in Hong Kong waters, they demonstrated that the greatest proportion of the suspended material settled within 100 m (corresponding to approximately 3 minutes from the overflow) astern of the active dredger. After 30 minutes, the surface plume sediment concentrations had almost reached background levels, suggesting that the settlement rate of the plume was far more rapid than models based on simple particle settlement would suggest.

Van der Veer *et al.* (1985) investigated recovery rates of the benthic macrofauna following dredging in the Dutch Wadden Sea. In areas of high current velocity, both infilling of dredge pits and subsequent recolonisation of macrofauna took approximately 1-3 years. Recovery of areas with low tidal stream velocities was more protracted (5-10 years). A dredge pit located on a tidal flat above mean low

water had no macrobenthic infauna at all, despite being dredged some 15 years previously. This study also showed that the infilling of dredged pits was heavily dependent on a number of factors including current patterns, size and depth of pit, sediment composition and the sediment load of the water mass, therefore, recovery would appear to be largely site specific. These conclusions were echoed by de Groot (1986) who stated that recovery from the impacts of aggregate extraction was somewhat dependant on the extent of substratum alteration. For example, the removal of a sand layer overlying bedrock would result in a vastly different habitat available to initial recolonisation and, therefore, an altered benthic community. In contrast, a pit dredged into a sandy substratum was likely to be filled with similar material, thus attracting a similar assemblage of settling invertebrates.

Van Moorsel (1994) investigated the effects of aggregate extraction in an area of the Klaver Bank (North Sea). Two months after aggregate removal, there was a significant reduction (around 75%) in the densities of the macrofauna, compared to pre-dredging levels. The invasion of the polychaete *Magelona mirabilis* was noted in the fine sands of the extraction furrows two months after the cessation of dredging; this species was absent from control sites. One year on, the densities of common species had returned to pre-extraction levels. It was noted, however, that larger, slow-growing species such as the bivalve *Dosinia exoleta* showed no sign of recovery two years after dredging (although this could have reflected the difficulties in obtaining an accurate cross-section of the population of such patchily distributed organisms by means of conventional sampling techniques).

An area of the seabed on the north Norfolk coast was experimentally dredged and the ensuing recovery of the benthos was monitored both immediately after

dredging and at intervals up to 2 years later (Kenny & Rees 1994,1996). Side-scan sonar techniques, in conjunction with underwater cameras, were employed to observe the physical impacts of the aggregate removal. The dredge head had created well-defined furrows 1-2 m in width and approximately 0.3-0.5 m deep. It was estimated that, in areas where the dredged tracks had crossed several times, the furrows reached 2 m in depth. The average number of species recorded fell from around 38 prior to dredging to 13 immediately after extraction. After seven months, the mean number of species rose to 26, suggesting some recolonisation had taken place. Densities of all taxa and mean biomass showed a marked decrease after dredging. Greater variability was observed in the dredged samples compared to the pre-dredging assemblage, a likely consequence of the lower numbers of individuals. In the intervening seven-month period, the dredged samples became more similar to one another as they returned to a community more closely resembling that observed prior to dredging. Two years after the cessation of dredging, the community had still not recovered fully, with biomass remaining substantially reduced. It was observed that, in terms of community abundance and biomass, two species, *Dendrodoa grossularia* (Chordata) and *Balanus crenatus* (Crustacea), contributed 90% and 70% respectively. As these species have life-history traits characteristic of *r*-selected taxa and were both dominant prior to dredging, it is likely that the community was already exposed to a certain level of disturbance. The authors speculated that if a community is dominated by taxa possessing life history traits of opportunistic or *r*-selected taxa, as described by Pianka (1970) then the ability of that community ability to recover from dredging disturbance is likely to be far higher than that of a more stable community characterised by slow growing, large bodied taxa (Kenny & Rees 1994, 1996).

In an extensive study into the effects of aggregate extraction along the French coast of the English Channel, Desprez (2000) found that the physical properties of the substrata in the region had significantly altered. Over a ten-year monitoring period, the original sediment composition of the dredged zone had been transformed from sandy gravels to fine sands and gravels with a notable proportion of very fine sands. The aggregate removal had eliminated the sandy gravel and exposed the underlying gravels. Fine sand, naturally transported on the strong tidal currents and unnaturally by settling from overflow during the dredging process, had filled the dredge tracks. The seabed topography had also been altered considerably to large furrows (up to 5 m in depth) separated by crests of shingles. The impact of dredging on the benthic communities of the area was shown to exhibit a gradient of response. In the western region of the extraction site, impacts were negligible due to the intense hydrodynamics of the area, thus supporting the hypothesis of Kenny & Rees (1994 & 1996) that, in areas subject to high levels of natural disturbance, communities are well adapted to cope with periodic disruption as caused by dredging. In contrast, the eastern areas of the dredged zone displayed large changes in species richness (80% decrease) and population densities (90% reduction). The species composition of the benthos had shifted from one characterised by *Branchiostoma lanceolatum* (Chordata) to a polychaete-orientated community dominated by *Ophelia borealis*, *Nephtys cirrosa* and *Spiophanes bombyx*. The effect of deposition of fine material from overflow in the vicinity of the dredged site was also investigated. It was found that the biological community of the deposition area was more disturbed than that of the dredged area itself. Sixteen months after the cessation of aggregate extraction, the numbers of species recorded in the deposition area were 60% lower than in controls, whereas in the dredged zone numbers had fully recovered, underlining the importance of this indirect form of impact (Kenny & Rees 1994 & 1996).

Twenty-eight months after dredging activities had ceased, total recovery had almost been achieved in terms of numbers of taxa and biomass, although abundances were still at a reduced level. However, the composition of the community had been changed, probably due to the alteration in the sediment makeup.

Van Dalfsen *et al.* (2000) provided evidence for the site specificity of benthic recovery by comparison of the differential response of the benthos to sand extraction in the North Sea and western Mediterranean. The authors observed significant changes in sediment structure (after dredging) at the Mediterranean site only, presumably due to the relatively weak hydrodynamic and sediment transport regimes of the region. In accordance with the findings of van Moorsel (1994) and Desprez (2000), large bodied, slow growing species (mainly bivalves) were almost completely eliminated after dredging at the Mediterranean sites and were replaced by polychaetes such as *Malococeros* sp. and *Capitella capitata*. In the North Sea, the predominant contributor to biomass, *Echinocardium cordatum* (Echinodermata), was observed to show an initial crash in the adult population but there was rapid recolonisation of juveniles. Van Dalfsen *et al.* (2000) illustrated that the recovery rate of the benthos was largely dependent on the degree and duration of change in sediment composition (van Dalfsen *et al.* 2000). The large alteration in sediment composition at the Mediterranean site led to a recovery time estimated to be in excess of four years, with the recolonisation of commercially-exploited bivalve species taking significantly longer, perhaps as long as a decade. It was demonstrated that at the North Sea sites, because changes to the sediment structure were only minor, the abundance of long-lived macrofaunal species returned to pre-dredging levels within two years, although complete recovery in

terms of population structure (i.e. adult to juvenile ratios) may require up to four years.

1.2.1. Summary

In summary, it appears that a fairly consistent pattern of impact on benthic communities is observed in response to aggregate extraction. The patterns of community response appear to be mediated by the method of dredging employed, the intensity/duration of extraction, the hydrodynamic regime, depth, and the sediment composition of the area in question (Kenny & Rees 1994 & 1996; de Groot 1996; Newell *et al.* 1998; Seiderer & Newell 1999; Desprez 2000; van Dalssen *et al.* 2000). Biological and physical impacts of dredging were reviewed by Newell *et al.* (1998) who showed that initial impacts were dependent on habitat type. It is also evident from the literature that any subsequent recovery is site specific. In areas of fine sediments, shallow depths and high current speeds, the recovery period is relatively short as the high levels of sediment transport quickly fill in any dredge tracks/pits. The existing benthic community in such regions is well adapted to elevated levels of natural stresses and, therefore, supports a high proportion of rapidly colonising, fast-growing, opportunistic species. Conversely, at locations with coarse substrata and deep water, the recovery process is much slower. A depression left on the seabed is likely be refilled with finer sediment as the influence of the currents at increased depth is less significant and the larger particles are not readily mobilised except, perhaps, in storm events. The finer sediments may in turn exclude recolonisation by large encrusting organisms due to the lack of suitable attachment surfaces. The community in such areas often comprise relatively high proportions of large bodied, slow growing taxa. Therefore, even with minimal change to the substrata the structure of the benthos will take

longer to reach pre-dredging levels (van der Veer 1985; van Dalssen *et al.* 2000; Seiderer & Newell 1999; Desprez 2000).

1.3. Aims of Thesis

Chapter 2 provides an overview of the study sites and shows the positioning of benthic samples collected at each site. Methods employed and equipment used in sample collection and processing are described in chapter 3. Benthic ecology, relationships between the benthos and the environment in which they reside, and distribution trends in benthic assemblages of the areas to be investigated, as well as other areas to act as comparisons, are examined. In this thesis, chapter 4 will identify differences and similarities in benthic community structure of five geographically separated regions representing contrasting habitats. This section will also identify possible biological factors responsible for the observed patterns i.e. feeding behaviour. Abiotic factors, that contribute to observed patterns in species diversity, population density, biomass and feeding behaviour are also identified. Leading on from this chapter 5 then focuses on the role of sediment particle size in determining benthic distributions. In chapter 6 spatial relationships are investigated, including patch size, between samples taken from the five areas and possible biotic/abiotic factors controlling these patterns will be highlighted. The impacts of marine aggregate extraction on the benthos of two environmentally contrasting dredging areas are considered in chapter 7. Leading on from the findings of the previous section, chapter 8 contains a meta-analysis of published dredging impact studies, and the results from the two dredged areas used in this thesis, these findings are then used to make general predictions of benthic recovery following aggregate mining in different habitat types. A critique of the current study is provided in chapter 9 along with recommendations for future study.



CHAPTER 2. REGION OF STUDY

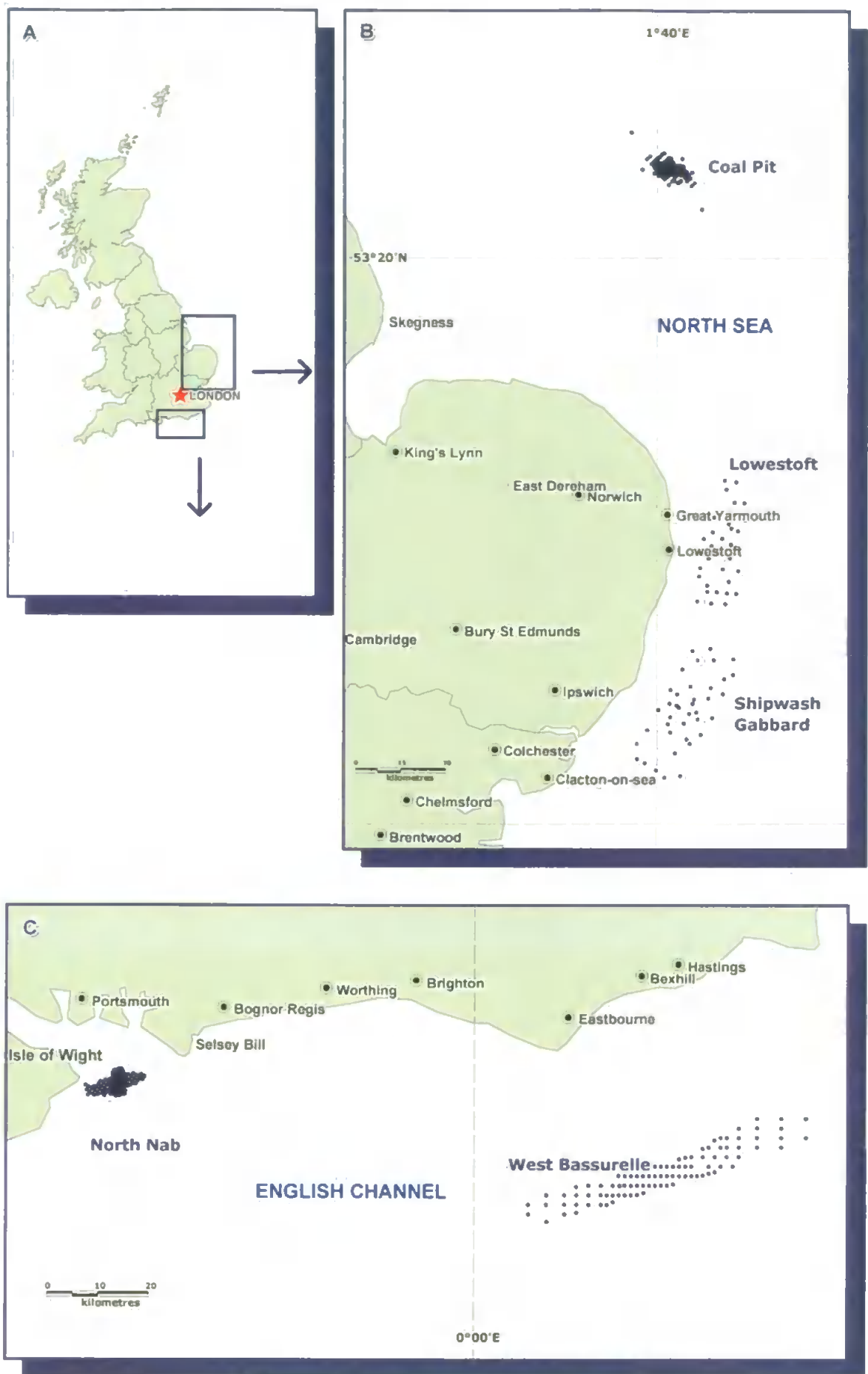
CHAPTER 2. REGION OF STUDY

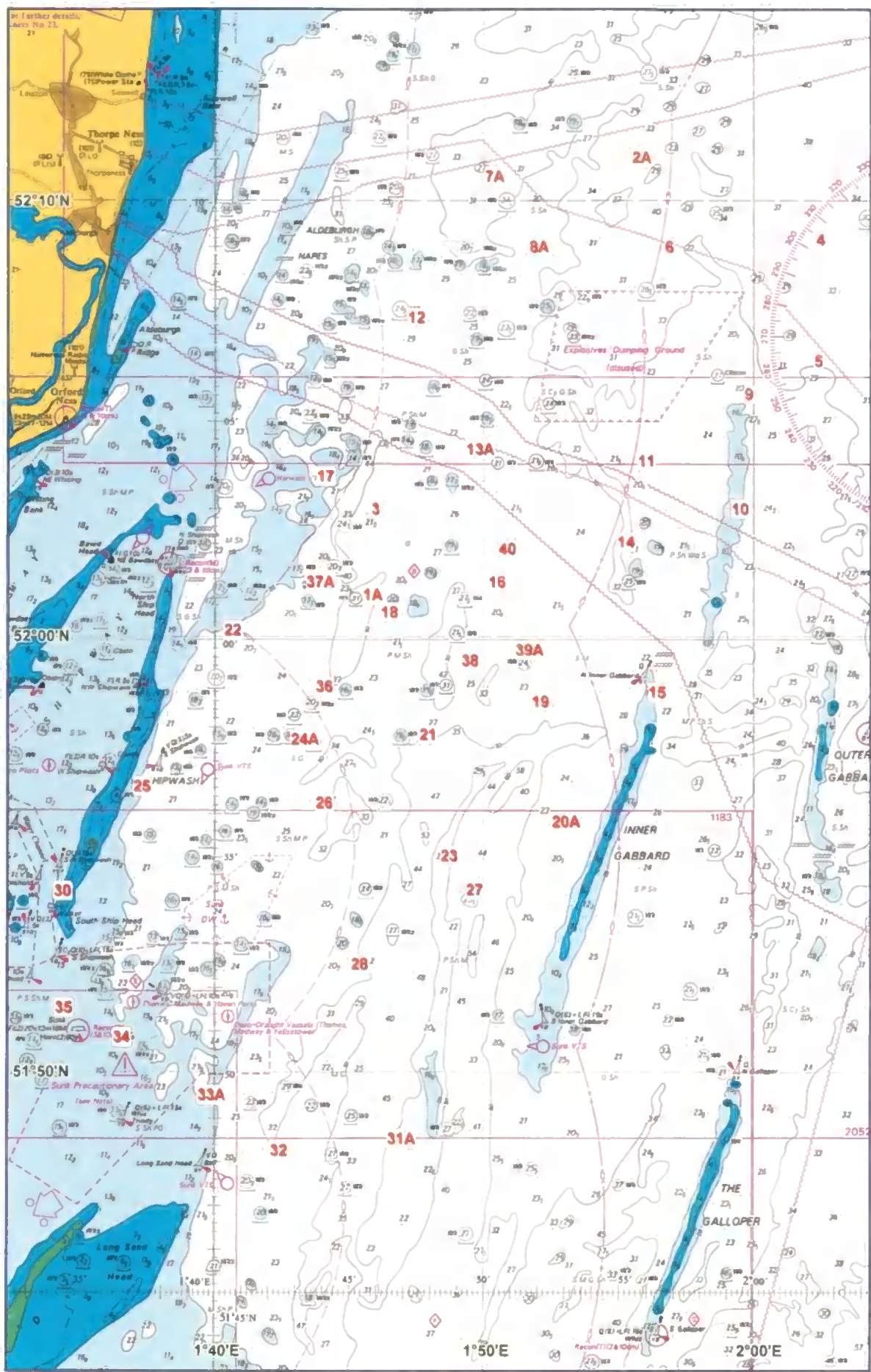
2.1. Selected Study Sites

Figure 1 shows the approximate location of the five study sites (Shipwash Gabbard, Lowestoft, West Bassurelle, North Nab, Coal Pit) featured in the present investigation. Widely separated study areas possessing a range of contrasting hydrophysical characteristics were chosen to highlight the differences and similarities between gravel habitats found in UK waters. Two study sites were actively mined for aggregates at the time of survey (North Nab and Coal Pit). The dredged sites at North Nab and Coal Pit were chosen to allow comparisons of the impacts associated with aggregate extraction at two environmentally contrasting locations. One of the mined areas (North Nab) was subjected to both anchor and trailer dredging, enabling impacts associated with contrasting dredging techniques to be investigated. Positions of all samples used in this investigation, in OSGB 36, are provided in Appendices I - IV.

2.1.1. Shipwash Gabbard (Fig. 1B)

The study area was located 11-22 km east of the Suffolk coastline. Water depth ranged from 15-50 m but was generally between 23-32 m. Seabed deposits comprised mainly sands and gravels, with sporadically-distributed patches of rock. Tidal currents reached a maximum of 2.3 knots at 207° on the flood and 2.3 knots at 0.30° on the ebb. A survey, carried out in August 1996, sampled 40 stations (Figure 2) with a 0.2 m² Hamon Grab, positions were fixed with a Sercel NR103 Differential Global Positioning System (DGPS) using a Trimble 'Hydro' Version 4.02p receiving in WGS 84 and converting Hydro to OSGB 36. The position of the grab on the vessel was offset and the accuracy of the positions was calibrated to within 2 m in port.



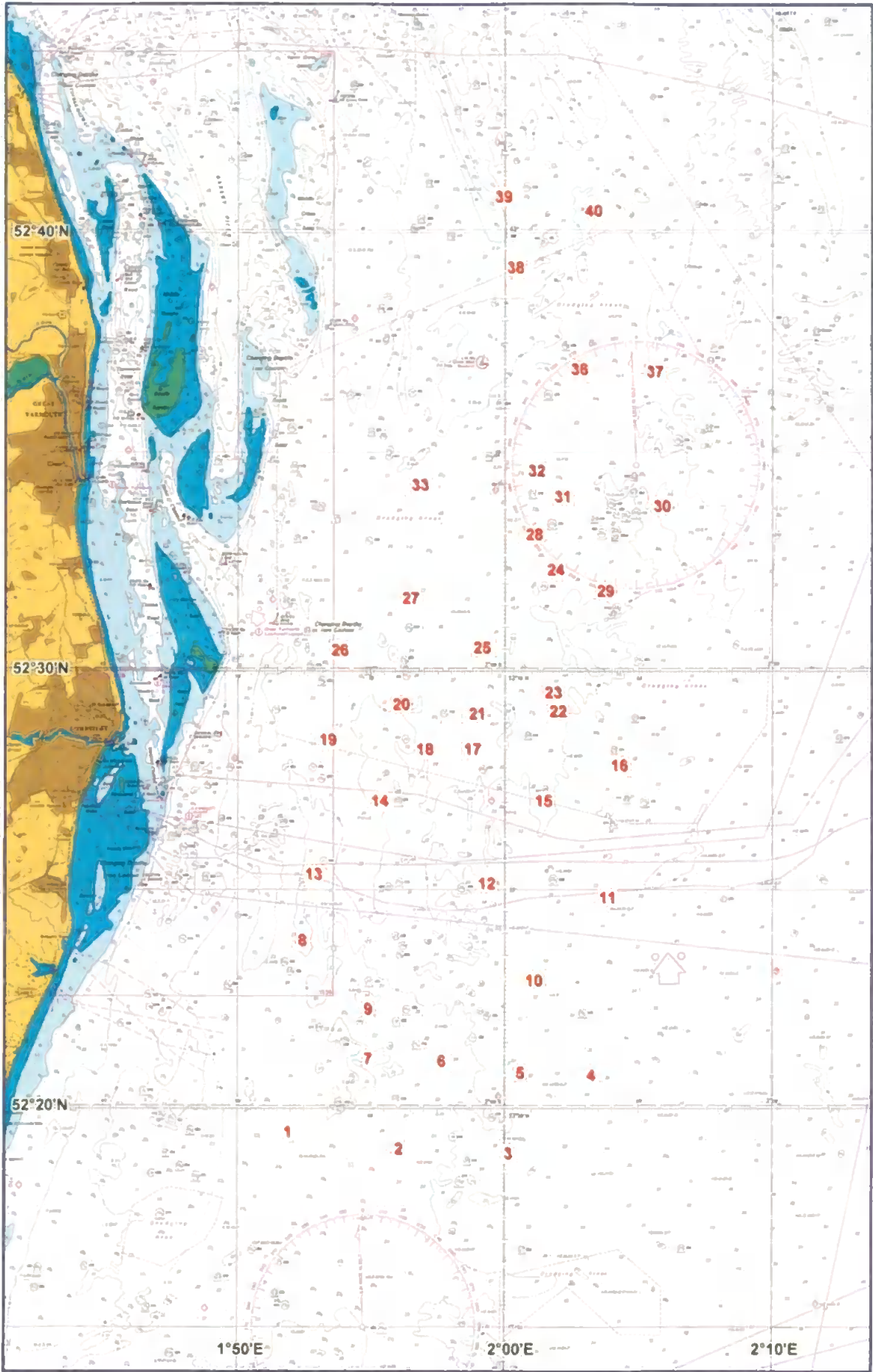


2.1.2. Lowestoft (Fig. 1B)

At the Lowestoft study area, water depth was 23-43 m and the seabed was overlain predominantly with sand, with some gravelly patches. Tidal currents run in a north-south direction with a peak velocity of 2.2 knots at 185° on the flood and up to 2.2 knots at 005° on the ebb. A survey of environmental conditions, approximately 11 km east of Lowestoft, was carried out between 28th and 31st August 1996; the survey grid comprised 40 stations (Figure 3) that were sampled by means of a 0.2 m² Hamon Grab. Positions were fixed with a Sercel NR103 Differential Global Positioning System (DGPS) using a Trimble 'Hydro' Version 4.02P receiving in WGS 84 and converting in Hydro to OSGB 36. The position of the grab on the vessel was offset and the accuracy of the positions was calibrated to within 2 m in port.

2.1.3. West Bassurelle (Fig. 1C)

A survey of the benthic assemblages in the West Bassurelle region of the central mid-Channel was carried out between 27th August and 4th September 1999. The main deposits in the region constituted gravels and sands, with a good deal of coarse shell. The depth ranged from 35 m to 60 m but was, for the most part, 40-50 m. The tidal dispersion curve for tidal diamond (P) at latitude 50°30'N; 00°26'E on British Admiralty Chart No 2451 (Figure 4) shows that tidal streams close to the northern boundaries of the proposed dredging areas comprised a main tidal stream of up to 2.1-knots at 082° on the flood and 1.5-knots at 254° on the ebb. Positions were fixed with a Trimble 4000 SSI with a 300 DCI Differential Global Positioning System (DGPS) using a Trimble 'Hydro' Version 6.05 receiving in WGS 84 and converting to OSGB 36. A total of 100, 0.2 m² Hamon Grab samples were taken (Figure 4). The position of the grab on the vessel was offset and the accuracy of the positions was calibrated to within 2 m in port.



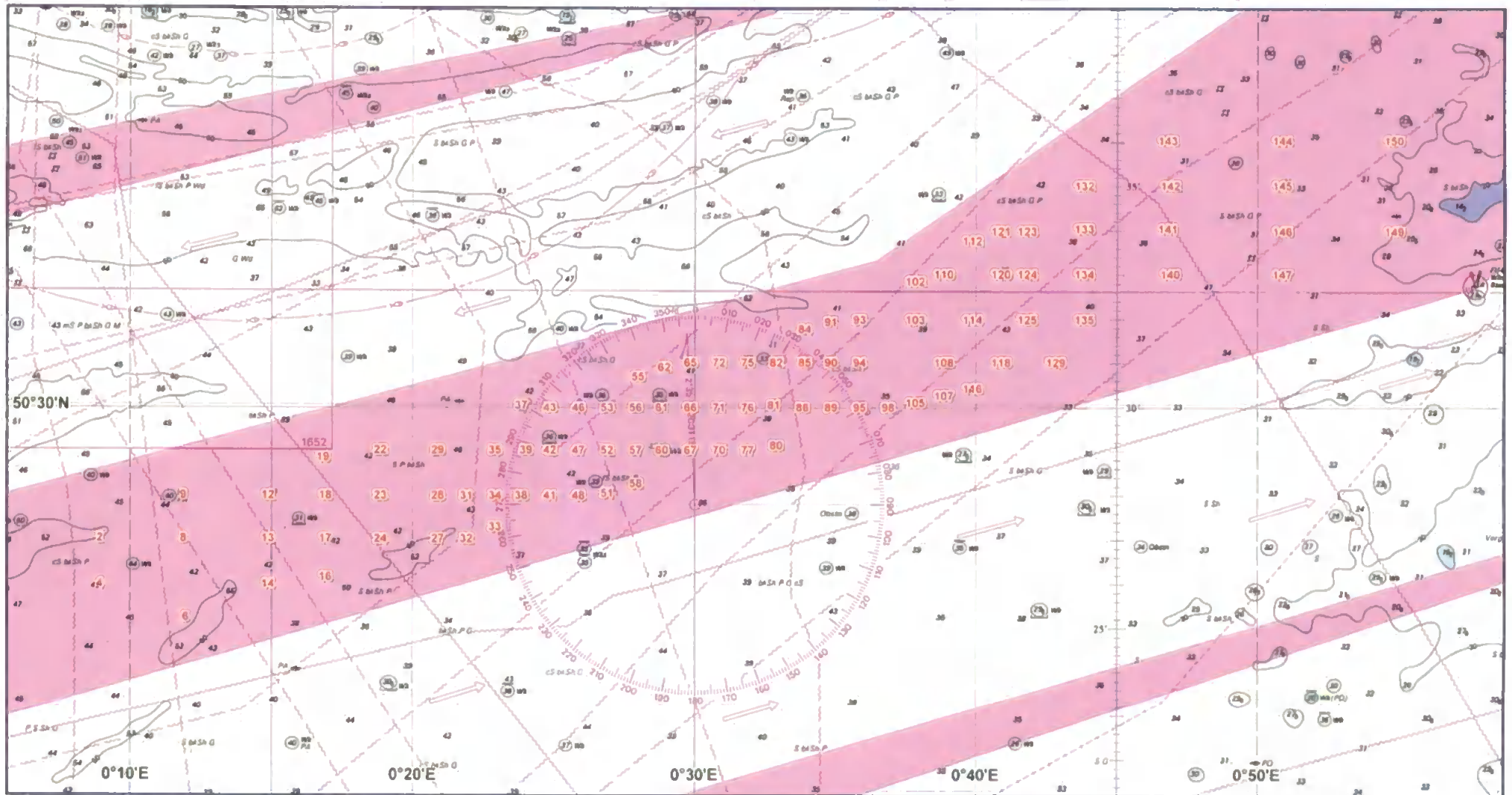


Figure 4. Location of samples collected from the West Bassurelle study area. ARCS chart 2451 used under licence from the U.K. Hydrographic Office.

2.1.4. North Nab (Fig. 1C)

A small, heavily exploited aggregate area to the east of the Isle of Wight, off the south coast of the UK known as the North Nab Production Licence Area, was sampled. The quantity of marine aggregates dredged from this area is relatively low at 150,000 tonnes per year. However, as the area is very small, the amount removed per unit area is probably amongst the highest on the south coast of the United Kingdom. The area had been exploited for almost 10 years prior to the present survey, so any impact of intensive dredging at this site should be apparent in a survey of the benthos. Gravel content of the deposits in this area was sufficiently high to allow minimal screening of fine particles (of approximately eleven Licence Areas on the south coast of the UK, nine involve non-screened cargoes). The North Nab study area, therefore is representative of the majority of the Licence Areas on the south coast of the United Kingdom, in contrast with those of the southern North Sea that are generally heavily screened, as are some on the south coast such as the Owers Bank and English Channel (Hitchcock 1997). The North Nab site has the further advantage that both anchor dredging from a stationary dredger and trailer dredging have been used to exploit the seabed deposits in adjacent locations within the boundaries of the Licence Area. Our data can, therefore, be used to assess the impact of the two principal methods of aggregate exploitation on benthic biological communities.

A survey of the benthic communities (a total of 131 sampling stations) in the vicinity of the North Nab dredging area, approximately 4 nautical miles to the east of the Isle of Wight off the south coast of UK., was carried out between 13th and 16th March 1999. Additional samples were collected on 7th September 1999. Positions were fixed with a Trimble 4000 SSI with a 300 DCI Differential Global Positioning System (DGPS) using a Trimble 'Hydro' Version 6.05 receiving in

WGS 84 and converting to OSGB 36. The position of the grab on the vessel was offset and the accuracy of the positions was calibrated in port to within 2 m. The positions of the sampling stations were chosen to assess the biota both within the dredging areas and in zones potentially affected by deposition, as well as in control zones well outside any area of potential impact of dredging activity. At the time of our initial survey (13th of March 1999), the dredger *City of Cardiff* was loading a cargo at Station 126, allowing the collection of samples within an actively dredged site. The dredge history of other sites within the survey area was obtained from records held by United Marine Aggregates who operate North Nab Production Licence Area.

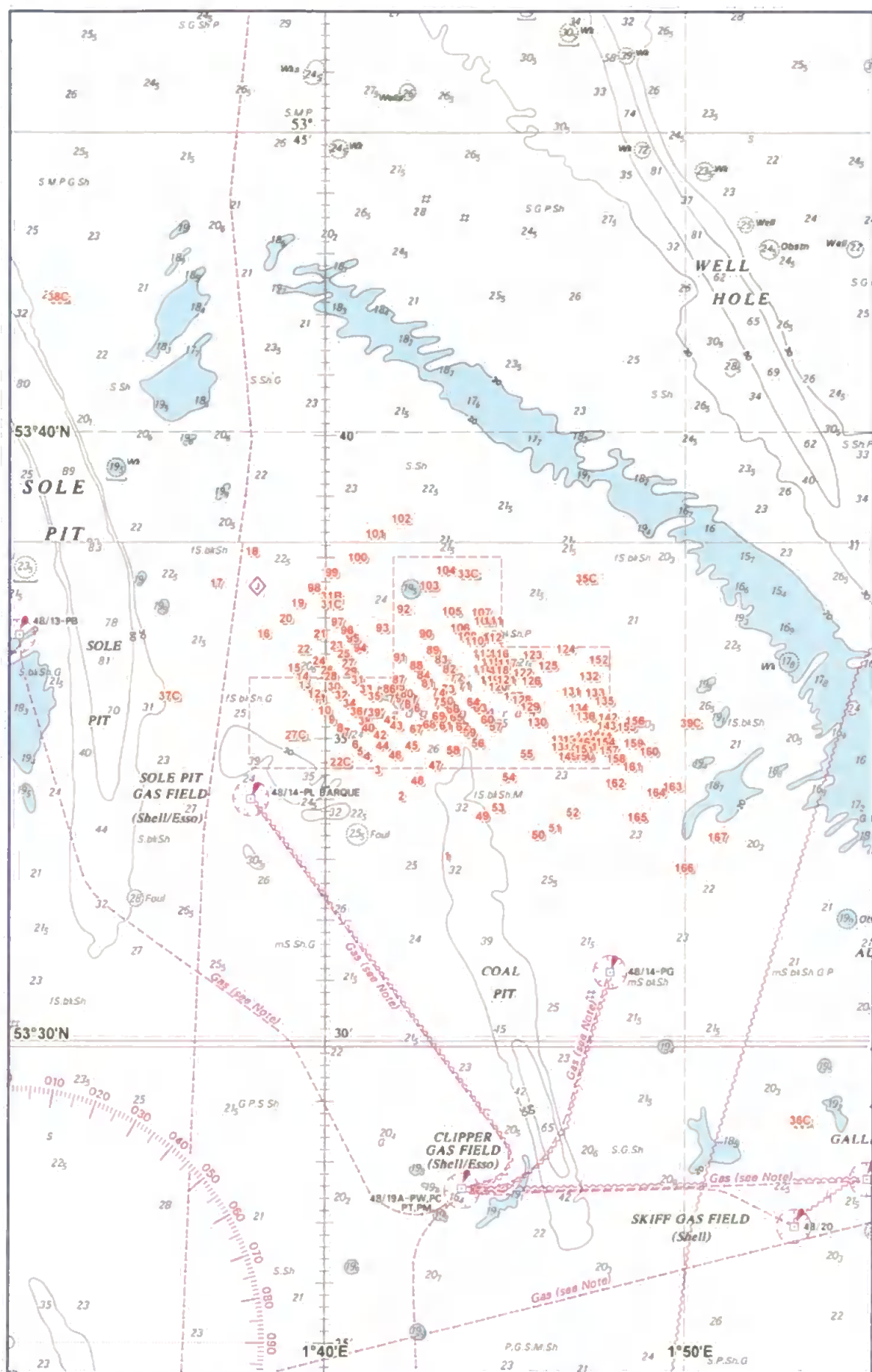
The tidal dispersion curve for tidal diamond N at latitude 50°40.1'N; 00°56.3'W on British Admiralty Chart No 2045 (Figure 5) shows that tidal streams close to the eastern boundaries of the dredging area comprise a main tidal stream of up to 1.8-knots at 078° on the flood and 1.6-knots at 252° on the ebb. The sum of movement during one tidal cycle is a maximum of 6-7 nautical miles during the spring tides. All information available from other survey areas suggests that any impact of deposition of material rejected during the dredging process is confined to within one tidal dispersion, and is likely to be confined to a distance of less than 3 km (see Whiteside *et al.* 1995; Hitchcock & Drucker 1996, reviewed in Newell *et al.* 1998). The positions of the sampling stations were chosen within a zone which extended approximately 3 nautical miles to the east of the eastern boundary of the dredged zone and a similar distance to the west of the western boundary of the Licence Area. These sampling stations covered a potential zone of impact associated with deposition of material from dredging activities in the extraction area. They allow comparison with 'control' sites outside any likely impact of dredging activity as well as with areas affected directly by extraction of marine

aggregates from within the boundaries of the dredging area. The positions of the sampling stations are shown in Figure 5.

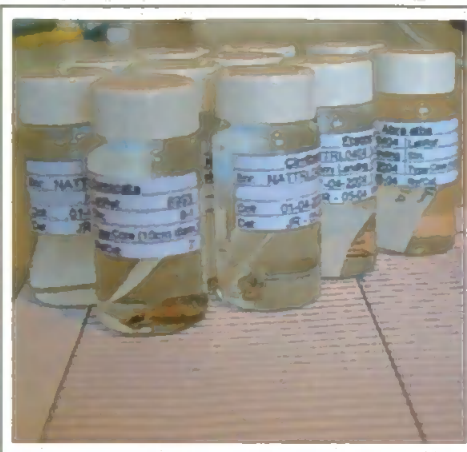
Additional samples were taken on 7th September 1999 to define the impact of dredging within the boundaries of the intensively dredged part of the Licence Area, and to give some information on the relation between past dredging within the site and biological recovery. The positions of stations used to assess the impact of dredging, within the intensively dredged part of the Licence Area, are shown in the survey site map (Figure 5). Samples of the deposits were also taken for subsequent particle size analysis.

2.1.5. Coal Pit (Fig.1B)

A survey of the biota in and adjacent to the 'Coal Pit' dredging area was carried out between the 28th July and 1st August 2000. The site lays approximately 120km off the Humber coastline; trailer dredging for sand and gravel has taken place at the site since 1995. The scale and intensity of dredging is typical for modern dredging licence areas; relatively large vessels operate in the area and employ on-board screening techniques to increase the gravel content of the cargoes. A total of 194 stations was sampled with a standard 0.2m² Hamon Grab (Figure 6). Positions were fixed with an Mk1 Primary System and Veripos DGPS with corrections from Denmark to OSGB 36. The position of the grab on the vessel was offset from the antenna to the grab position on the vessel. The accuracy of the positions was stated to be within 5 m. The tidal streams indicated at Tidal Diamond J on Chart 1503 (53° 17.0'N 01°35.0'E) reach a maximum of 1.4 knots at 320° and 140° on spring tides. The total excursion for one spring tidal cycle is 6.3nM on the northwesterly stream & 5.5nM on the southeasterly stream. Stations were positioned to investigate the effects of the aggregate extraction in the



immediate vicinity of the dredged area and of any likely impacts attributed to material rejected by overboard screening at various distance intervals (100 – 4000m) along a transect running parallel to the axis of tidal movement. Control stations (taken in triplicate) on either side of the direction of tidal flow were taken to eliminate the possibility of any impact attributable to dredging activity. The site consisted of four separate but adjacent dredging blocks, each of which was subjected to various dredging intensities at different times. Within each of the dredged blocks, 15 samples were taken in order to give a reasonable chance of detecting any impact on the benthos (Figure 6).



CHAPTER 3. MATERIALS & METHODS

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3.1. Collection and Extraction Procedures

Samples were taken from a survey vessel using a 0.2 m² Hamon Grab (Figure 7a). This grab has the advantage that any loss of material by 'washout' from the jaws is minimal due to the orientation of the bucket once the sampler has tripped. Sampling in coarse deposits with many conventional grab designs such as the Day Grab (Figure 7b) leads to the jaws being held open by large particles resulting in the subsequent loss of the sample (see Holme & McIntyre 1984; Sips & Wardenburg 1989; Kenny & Rees 1994; van Moorsel 1994). The efficiency of the Hamon grab in coarse substrata has led to this device being recommended in the procedural guidelines for surveys of this type (CEFAS 2002).

Upon retrieval of the grab, the samples were released from the grab onto a 1 mm-mesh sorting tray, the contents of which were transferred to 10l buckets and preserved in 5% formaldehyde solution for subsequent separation and identification. Separation of the biological material was carried out in the laboratory by elution with approximately 30l of tap water through a 1 mm mesh sieve; larger, denser organisms (i.e. bivalves) were separated manually by eye. The residual fauna remaining from the elution process was separated from the remaining sediment using a Zeiss Citoval 2 stereomicroscope at x 20 magnification. The biological material was preserved in 70% methanol for subsequent identification and enumeration. Identified taxa and corresponding abundance values for all samples are shown in Appendices XI - XV.



Figure 7a. Deployment of a 0.2 m² Hamon grab.



Figure 7b. 0.1 m² Day grab prior to deployment.

3.2. Biomass Determination

The blotted wet weight of the main faunal groups was measured as follows: the material was first blotted using absorbent paper and then weighed using a Bel Engineering Mark 205a balance. These data were used to estimate total biomass as ash-free dry weight in grams using conventional conversion factors for each of the faunal groups including Annelida (= wet weight x 0.155), Crustacea (= wet weight x 0.225), Mollusca (= wet weight x 0.085), Echinodermata (= wet weight x 0.08), and Miscellaneous Groups including Porifera and Bryozoa (= wet weight x 0.155) (Eleftheriou & Basford 1989). Values for the biomass of invertebrate material recorded in all samples are given in Appendices XVI - XVIII. Total biomass of each sample was recorded for samples collected at Shipwash Gabbard and Lowestoft.

3.3. Particle Size Analysis

Sub-samples of up to 5 l of sediment were taken from each sample for particle size analysis. This material was sealed in strong plastic bags with a label on both the inside and on the outside of the bag. The material was sieved over the range 50 mm down to 0.063 mm and the results were expressed using conventional Wentworth Classifications to give percentage composition of each particle size. Results of particle size analysis of all samples are shown in Appendices VI - X.

3.4. Multivariate Analysis

Multivariate analysis of biological community structure and the sediment composition followed Field *et al.* (1982) and Clarke & Warwick (1994). Computer software used for these analyses was contained in the PRIMER version V5.2.1 package (Clarke & Gorley 2001). Untransformed data were used for plotting dendrograms and multi-dimensional scaling ordinations (MDS) of the sediment

particle size fractions using the Bray-Curtis index of similarity. Species and generic data (where used) were subjected to a square root transformation to reduce the influence of the numerically dominant taxa (Downing 1979).



CHAPTER 4. DESCRIPTION OF THE BENTHIC COMMUNITY ATTRIBUTES OF FIVE GEOGRAPHICALLY SEPARATED REGIONS WITH CONTRASTING ENVIRONMENTAL CHARACTERISTICS

This section aims to highlight differences/similarities in benthic community assemblages from five study sites distinguished by contrasting hydrophysical regimes. Possible explanations of the role various abiotic parameters play in determining these differences/similarities are also discussed.

CHAPTER 4. DESCRIPTION OF THE BENTHIC COMMUNITY ATTRIBUTES OF FIVE GEOGRAPHICALLY SEPARATED REGIONS WITH CONTRASTING ENVIRONMENTAL CHARACTERISTICS.

4.1. Introduction

Early investigations of the benthic invertebrate assemblages of the English Channel include those of Crawshay (1912) and Ford (1923) who reported on benthic invertebrate assemblages in the western English Channel. Detailed surveys, focusing on molluscs and echinoderms, were undertaken later by Holme (1966). These three investigations highlighted considerable variations in hydrodynamic and sedimentary regimes as well as the associated fauna. Holme (1966) identified four main faunal distribution patterns:

- Species widely distributed over the whole channel region. These included *Nucula* spp. (Mollusca), *Abra alba* (Mollusca), *Gari fervensis* (Mollusca) and *Ophiura albida* (Echinodermata).
- Species confined to western areas including *Gouldia minima* (Mollusca), *Myrtea spinifera* (Mollusca) and *Gari tellinella* (Mollusca).
- Species distributed around the central regions of the channel were *Spisula subtruncata* (Mollusca) and *Crepidula fornicata* (Mollusca).
- 'Northern' species found mainly in eastern areas of the channel were *Spisula elliptica* (Mollusca) and *Buccinum undatum* (Mollusca).

Further to this 1966 investigation, Holme & Wilson (1985) describe three faunal assemblages that were apparently associated with varying levels of sand scour. The first community inhabited hard, non-mobile, coarse substrata and were deemed unaffected by the scouring effects of transitory sand although current speeds were high. Species from this assemblage included many sponges and the delicate bryozoan *Pentapora foliacea*. A second faunal group was observed in areas characterised by coarse substrata that were subjected frequently to sand scour and submerged periodically by finer particles. Taxa commonly associated

with these conditions included *Dysidea fragilis* (Porifera), *Urticina felina* (Cnidaria), *Distomus variolosus* (Chordata), *Flustra foliacea* (Bryozoa), *Balanus crenatus* (Crustacea) and *Sabellaria spinulosa* (Annelida). A third assemblage consisted of many motile species found in the previously described group but differed in that it was characteristically found in areas covered by a few centimetres of sand or a sand/gravel mixture. Due to the unstable surface of this latter habitat, sessile epifaunal taxa were absent. More recently, Migne & Davoult (1995) described the fauna of the Dover Strait, northeast English Channel. The region featured a coarse substratum and was exposed to strong tidal currents. Such conditions appeared to favour dominance of the fauna by three suspension feeding species: *Ophiothrix fragilis* (Echinodermata), *Urticina felina* (Cnidaria) and *Alcyonium digitatum* (Cnidaria).

The North Sea is one of the most environmentally heterogeneous regions in the northeast Atlantic. In the south, it is characterised by relatively shallow water depth, widely changing temperatures, strong tidal currents and mobile sediments. Conversely, the northern North Sea has much greater water depths, small seasonal temperature changes, low tidal currents and stable deposits (Jennings *et al.* 1999). Benthic communities of the North Sea were described by Steven (1933) and McIntyre (1958). More recently, Eleftheriou & Basford (1989) carried out an extensive survey of the northern North Sea and found that macrofaunal diversity was highest in the 120-140 m depth zone. Two separate assemblages were identified. The first, associated with areas of lowered silt content, was dominated by *Aonides paucibranchiata* (Annelida), *Aricidea wassi* (Annelida), *Exogone verugera* (Annelida), *Bathyporeia elegans* (Crustacea) and *Urothoe elegans* (Crustacea). A second assemblage, characteristic of siltier, deeper conditions, was found to have *Heteromastus* sp., *Lumbrineris gracilis* (Annelida), *Phylo norvegica*

(Annelida), *Eriopisa elongata* (Crustacea) and *Thyasira* spp (Mollusca). as important contributors.

A ten-year study executed by Dewarumez *et al.* (1992) in the southern North Sea, highlighted the existence of a transitional community (ecotone) between an assemblage associated with tide-swept region of coarse deposits (pebbles community) and another collection of fauna recorded in muddy sand deposits (*Abra alba* community). The 'pebbles community' had *Ophiothrix fragilis*, *Pisidia longicornis* and *Psammechinus miliaris* as dominant species, whilst the '*Abra alba* community' was characterised by *Spiophanes bombyx*, *Lanice conchilega*, and *Abra alba*. It was noted that a third community associated with heterogeneous muddy sediment, situated between the other two assemblages, was a combination of the two both in bionomic and trophic terms and was, therefore, likely to represent an ecotone. In this instance an ecotone was defined as a transition zone between two ecological communities exhibiting competition between organisms common to both.

Jennings *et al.* (1999) examined and identified the distributions of epifaunal species over a wide area of the North Sea, as having distinct differentiation between the following three geographical regions: northern, central and southern. Important faunal contributors to the southern North Sea assemblage were sand-dwelling species including *Ophiura ophiura* (Echinodermata), *Pagurus bernhardus* (Crustacea), *Asterias rubens* (Echinodermata), and *Liocarcinus holsatus*. (Crustacea). These taxa are often found scavenging on damaged and discarded invertebrates associated with various trawling activities (Kaiser & Spencer 1994), which is in accord with the heavy fishing intensity that the area is subject to. The central North Sea assemblage comprised many of the taxa found in the southern

sector, with *Crangon allmanni* (Crustacea) and *Anapagurus laevis* (Crustacea) increasing in dominance. The northern grouping was characterised by deep-water species such as the sea urchin *Echinus acutus* (Echinodermata) and subarctic species such as the prawn *Pandalus borealis* (Jennings *et al.* 1999).

4.2. Analytical Methods

4.2.1. Univariate Analysis

Histograms were plotted for a range of ecological indices, calculated for each location, including number of species (S), number of individuals (N), species richness (d), species evenness (J), species diversity (H') and dominance (1- λ). One-way ANOVA (analysis of variance) tests were performed on each of the above indices to test for significant differences between the study sites. A posteriori power analysis (Clarke & Green 1988) was carried out for each test. This was done to calculate the statistical power of the ANOVA test or, conversely, to detect the likelihood of a type II error i.e. the probability of failing to detect a difference (in this case a 10% difference) when one actually exists.

4.2.2. Multivariate Analysis

To reduce the possibility of taxonomic discrepancies between the data sets of West Bassurelle, North Nab and Coal Pit, and the data sets compiled previously (Lowestoft and Shipwash Gabbard), the species abundance data were aggregated to the level of genera for the purposes of multivariate analysis. This approach has been shown by Warwick (1988a, 1988b, 1988c), Warwick *et al.* (1990) and Somerfield & Clarke (1995) to yield comparable results with full species abundance data sets. Such data aggregation has the added advantage of lowering the stress measure of the MDS ordinations, making the two dimensional representation more accurate, a feature that is particularly relevant when large

data sets are analysed as in this case. Data were subjected to a range of transformations including untransformed, square root, 4th root, log, and presence/absence.

Trophic groups were categorised according to the groupings assigned in De Grave & Whittaker (1999). The data were not transformed for this analysis. At the actively dredged locations (North Nab and Coal Pit) only control stations were used for inter-regional analyses to ensure patterns reflected the 'naturally shaped' communities as closely as possible. Colonial genera were excluded from the data due to the difficulty in quantifying these taxa; an exception to this was the colonial tunicate *Distomus variolosus*, a frequently recorded and therefore important species at the North Nab site. Quantification of this species was aided by the tendency for this species to form readily identifiable zooids; therefore, each zooid was treated as a single individual. Sample 33A from the Shipwash Gabbard site was omitted from the feeding group analysis because of the presence of an unusually high number of *Lagis koreni* (Annelida), which greatly skewed the results in favour of deposit feeders, the presence of such high numbers of this species being a possible reflection of exceptionally high silt levels noted at this station. This sample was, therefore, deemed unrepresentative of the fauna of the region.

Multivariate analysis was carried out using the PRIMER software package (Clarke & Warwick 1994, Clarke & Gorley 2001) and the following routines were employed.

(a) Multi-dimensional Scaling Ordination (MDS): this technique allows the construction of a 'map' or configuration of the samples in multidimensional space.

This configuration attempts to position the samples as accurately as possible to reflect the similarity between the samples. For example, if sample 1 has a greater similarity to sample 2 than it does to sample 3 then sample 1 will be positioned more closely to sample 2 than it is to sample 3. This 'map' of the relative similarities between samples is plotted in two dimensions. It is important to remember that this two-dimensional plot is a representation of a multidimensional picture. When numbers of samples are analysed, or datasets that include samples that are very different from other samples, the accuracy of the two-dimensional plot may be reduced, a measure of how accurate the two-dimensional representation is (stress) is given on the MDS plot. Stress values <0.1 correspond to a good ordination, values <0.2 gives a useful two-dimensional picture but too much reliance on the fine details of the plot should not be placed, stress >0.3 indicates that the samples are close to be positioned in an arbitrary manner and should be treated with caution, particularly in the upper half of this range.

(b) ANOSIM (Analysis of Similarities): this procedure was used to test the null hypothesis that there were no differences in community composition between the five study sites featured in the present investigation. This routine goes through three main stages in the examination of H_0 , these are:

(i). The calculation of the ANOSIM statistic from the dataset: this reflects the observed differences *between* sites, contrasted with differences *within* sites. The test is based upon the corresponding (rank) similarities between samples. The test statistic R is calculated using the equation:

$$R = (\bar{r}_B - \bar{r}_W)/(M/2)$$

Where \bar{r}_W is the average of all rank similarities among replicates *within* sites, \bar{r}_B is defined as the average of rank similarities from all pairs of samples from *between* sites, and $M = n(n-1)/2$, where n is the total number

of samples in the dataset. R usually lies between 1 and 0; $R = 1$ if all replicates within a particular site are more similar to one another than any sample from a different site. If R is approximately 0, there are no differences between the sites, therefore, between and within site similarities are the same on average. Technically, R can be as low as -1 although this is unlikely as it would indicate that within site similarities were less than between site similarities. The R statistic can be used also to give an indication of the degree of separation between the sites. Well-separated sites exhibit R values greater than 0.75, sites that are clearly separable but show some overlap display R values in the region of 0.5, and sites that are barely separable give R values of 0.25 and below. It should be noted that R can be very small and still be significantly different from zero.

(ii). Recalculation of the R statistic under permutations of the sample labels: Using the null hypothesis of no differences between sites, there is likely to be little effect on average to the R value if the sample labels belonging to each site are arbitrarily reshuffled, because if H_0 is true then all samples are merely replicates of a single site. Therefore, the permutation tests basically consist of rearranging the sample labels, recalculating the R statistic and repeating a large number of times.

(iii). The final stage is the calculation of the significance level: This is carried out by referring the observed value of R , calculated from the 'original' data, to the distribution of R as simulated by permutation. If the observed value of R is unlikely to have come from this distribution, this is evidence to suggest that H_0 should be rejected.

(c) SIMPER (Species Contributions to Similarity): this routine allows comparison between groups of samples from one site to another. Species responsible for the dissimilarity between the two sites are listed in decreasing order of importance in the discrimination of the two regions. Values relating to the contribution to the dissimilarity between the sample groups, and the average abundance of a given taxon at each site, are shown. This routine also provides

information on which species are responsible for the within-site similarities and their contribution to the internal similarity of the group.

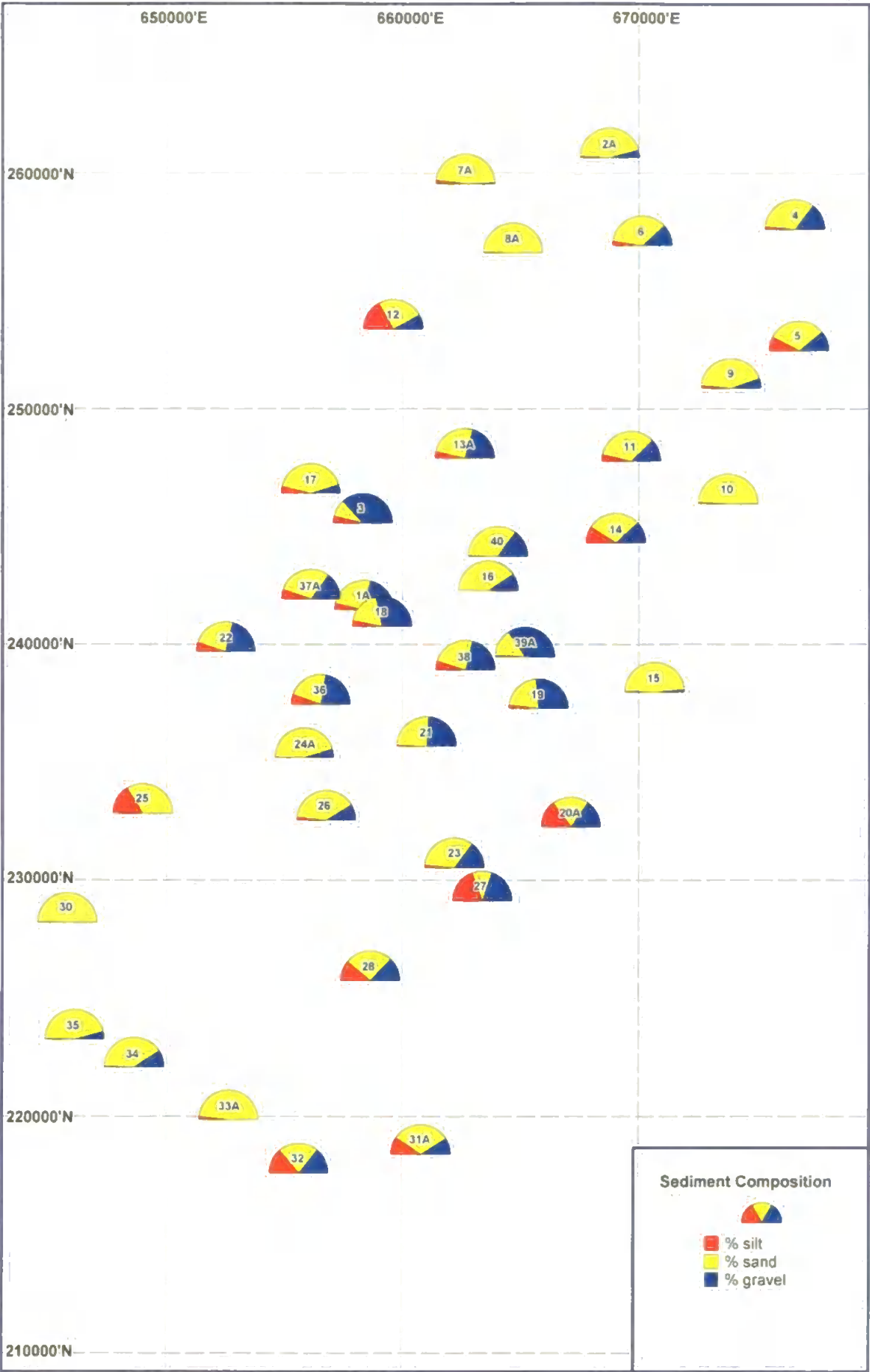
(d) *k*-dominance curves: these plot cumulative ranked abundance against species rank, or log species rank. This procedure has the effect of smoothing the curves. The more elevated the starting point of a curve, the more dominated the group of samples it represents and, therefore, the greater the level of biological 'stress' associated with the sample group.

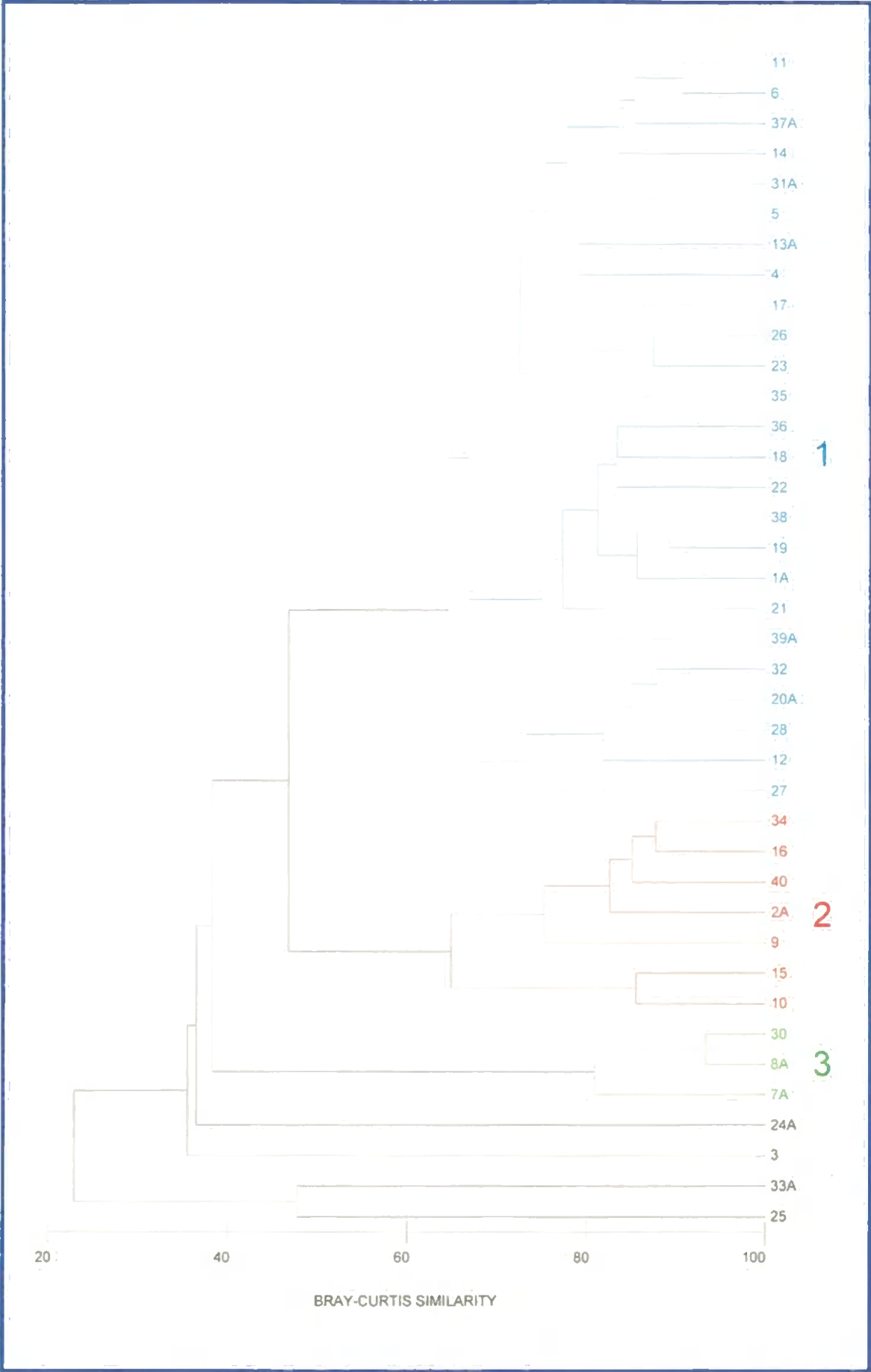
4.3. Site Specific Results

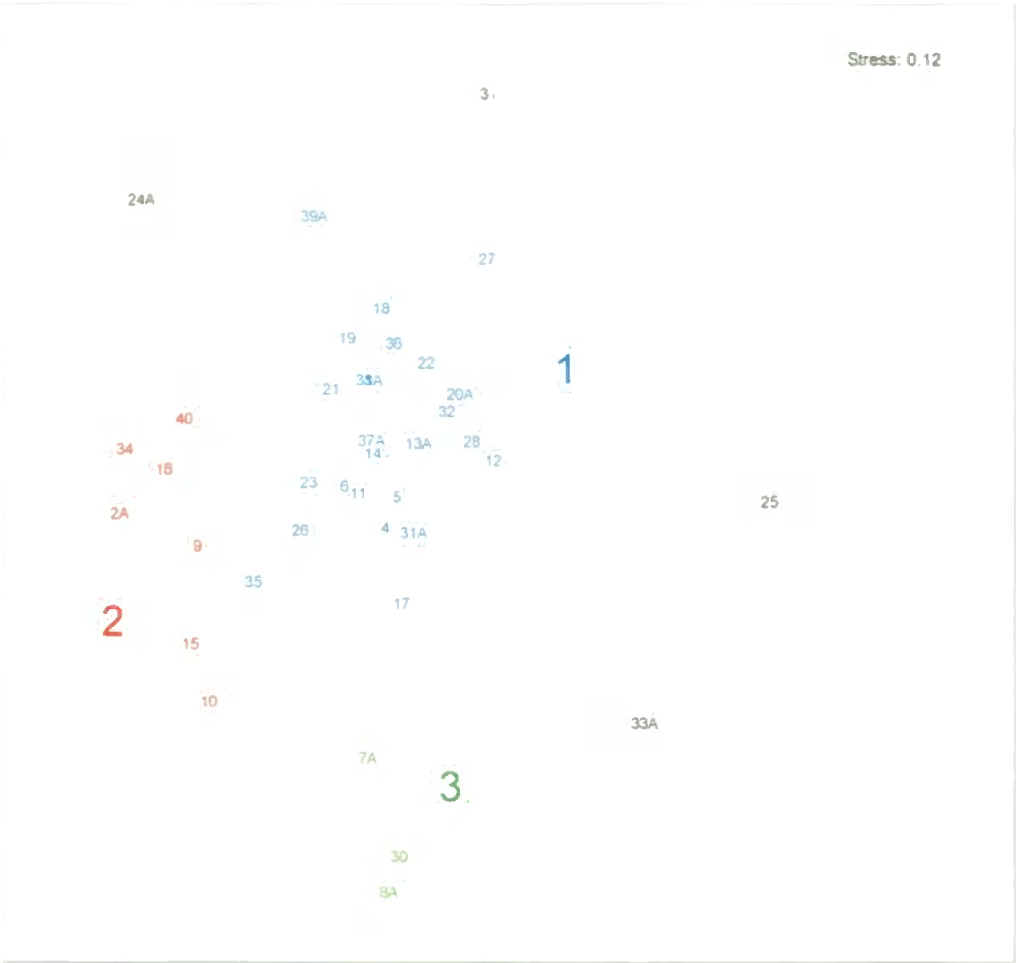
4.3.1. Shipwash Gabbard

4.3.1.1. *Substratum Composition*

The substratum of the Shipwash Gabbard region was, for the most part, sandy with varying proportions of gravel and silt. Coarse deposits were found in stations situated in the central region and those were characterised by sandy gravels. Finer deposits were present in a narrow band running approximately northeast to southwest, whereas in the northwest and south west of the survey area, the proportion of sand in the sediments appeared to increase. Generally, gravels appeared to be present in high proportions with many sediments comprising well over 50% gravel (see Figure 8). A group average sorting dendrogram and corresponding MDS plot based on Bray-Curtis similarity of untransformed percentage retained particle size data are given in Figures 9 & 10. Two main groups of deposit are clearly visible with a third, smaller category also present. The first main group (coded blue) was made up largely of sandy gravels, the second main group (coded red) consisted largely of sand with some gravel, and the third group (coded green) was represented by three samples where sands







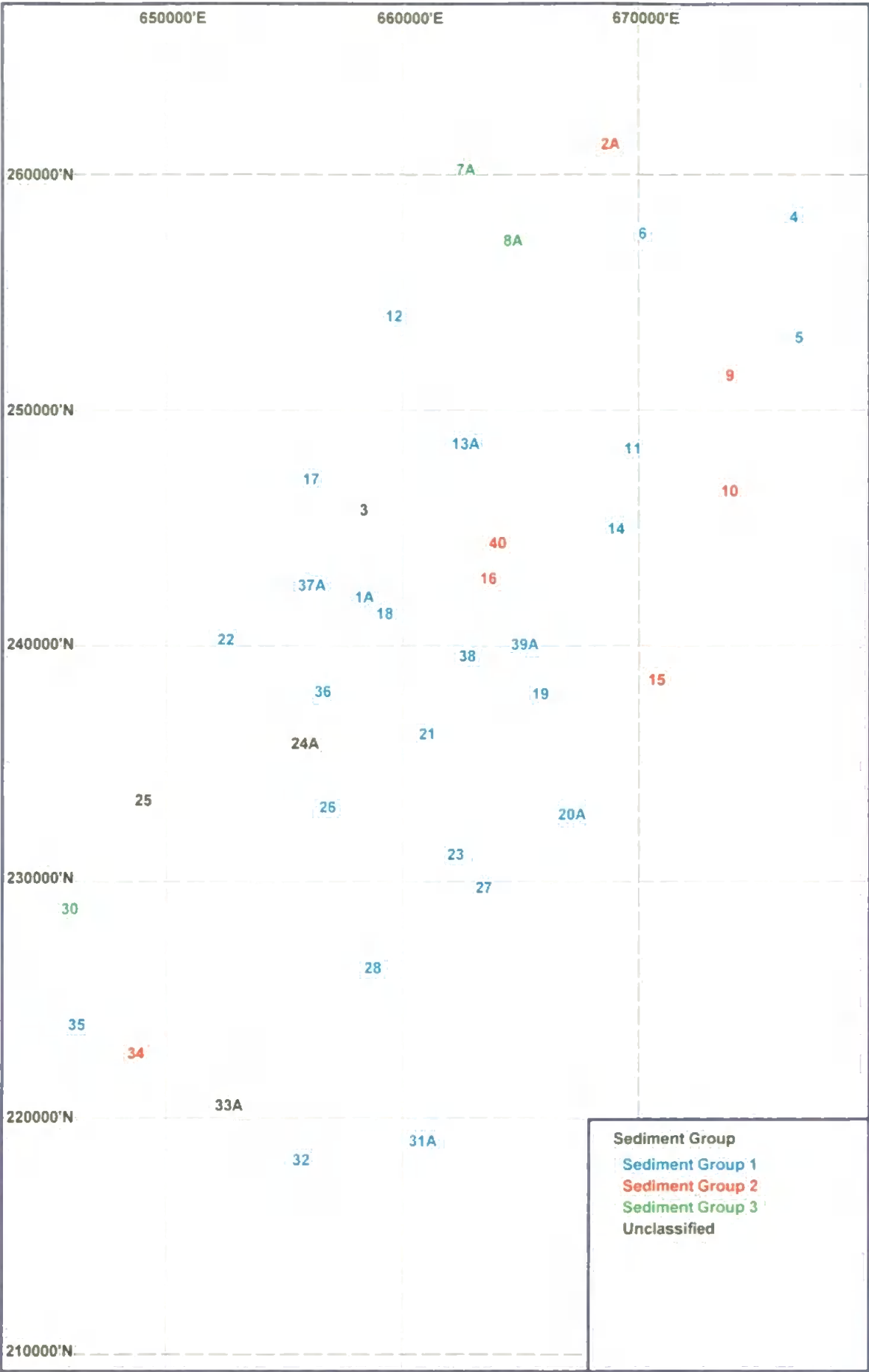
predominate. A further four stations did not fall readily into any closely related group (shown in black).

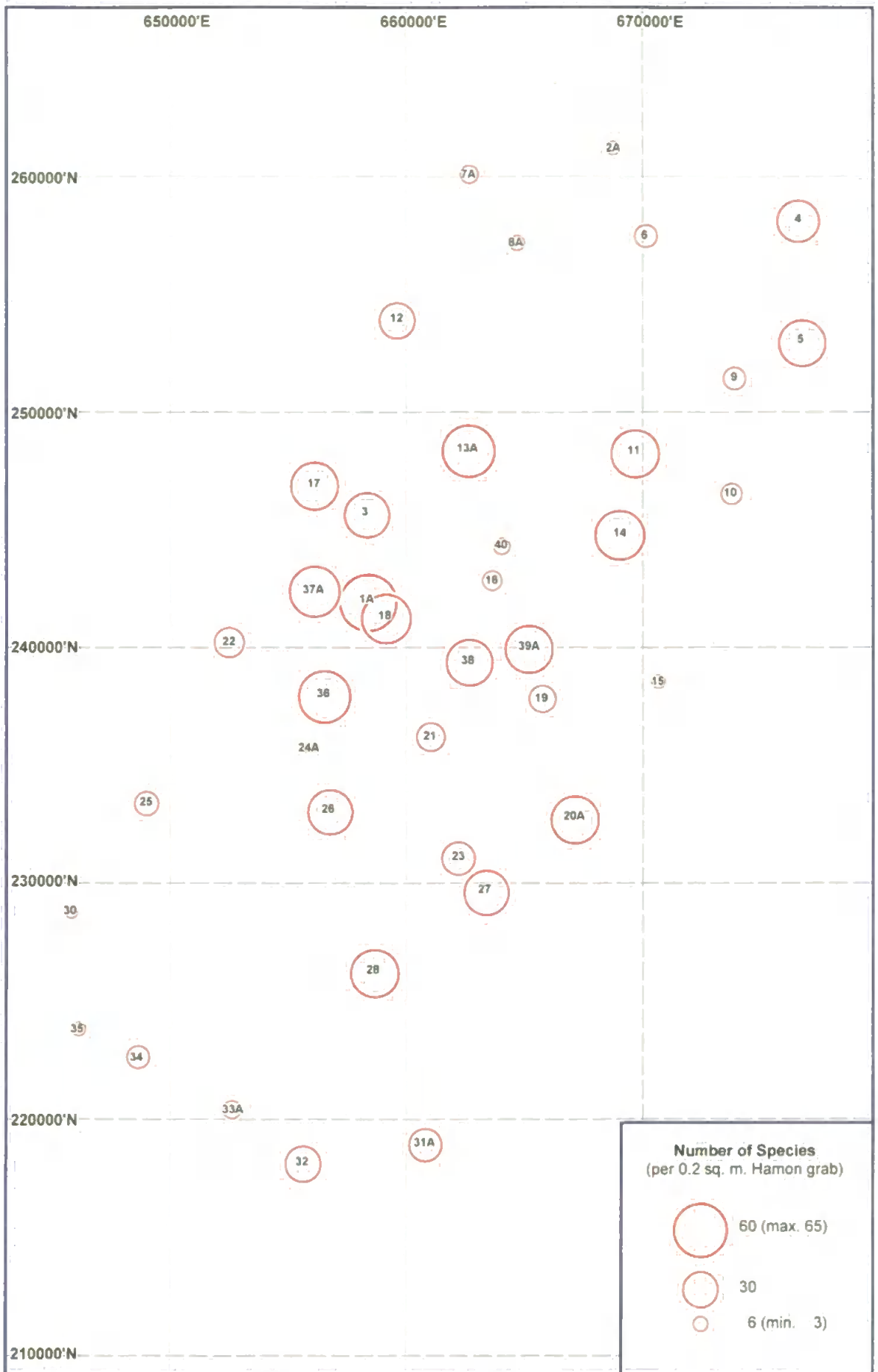
Figure 11 shows the sediment groups identified by multivariate analysis superimposed on the sample grid and confirms the general sediment description given by Figure 8. Group 1 sediments were widely distributed throughout the survey area but gave way to a narrow band of sandy, group 2 deposits in northeastern stations. Group 3 deposits were found in two adjacent stations in the extreme northwest and again in a solitary station in the southwest.

4.3.1.2. General Features of Benthic Community Structure

A total of 198 species was recorded, polychaetes being the dominant group with crustaceans also accounting for a large proportion of the fauna. The sandy substratum, with its lack of suitable attachment surfaces, may be a contributing factor to the apparent absence of many epifaunal taxa. Species variety reached a peak at station 1A with a total of 65 species recorded; the lowest number of species was noted at station 24A where 3 taxa were noted (Figure 12). Diversity was highest in central and northwestern stations, with patches of low diversity associated with the sandy substrata of the north, southwest and central areas.

Population density was highest at station 33A, where over 28000 individuals were recorded, this was attributable mostly to *Lagis koreni* (Annelida), in contrast only 5 individuals were found at station 24A (Figure 13). Generally, high population densities were associated with regions of high species diversity, with central northwestern stations exhibiting high numbers of individuals. Similarly, low numbers of invertebrates were associated with sandy substrata.





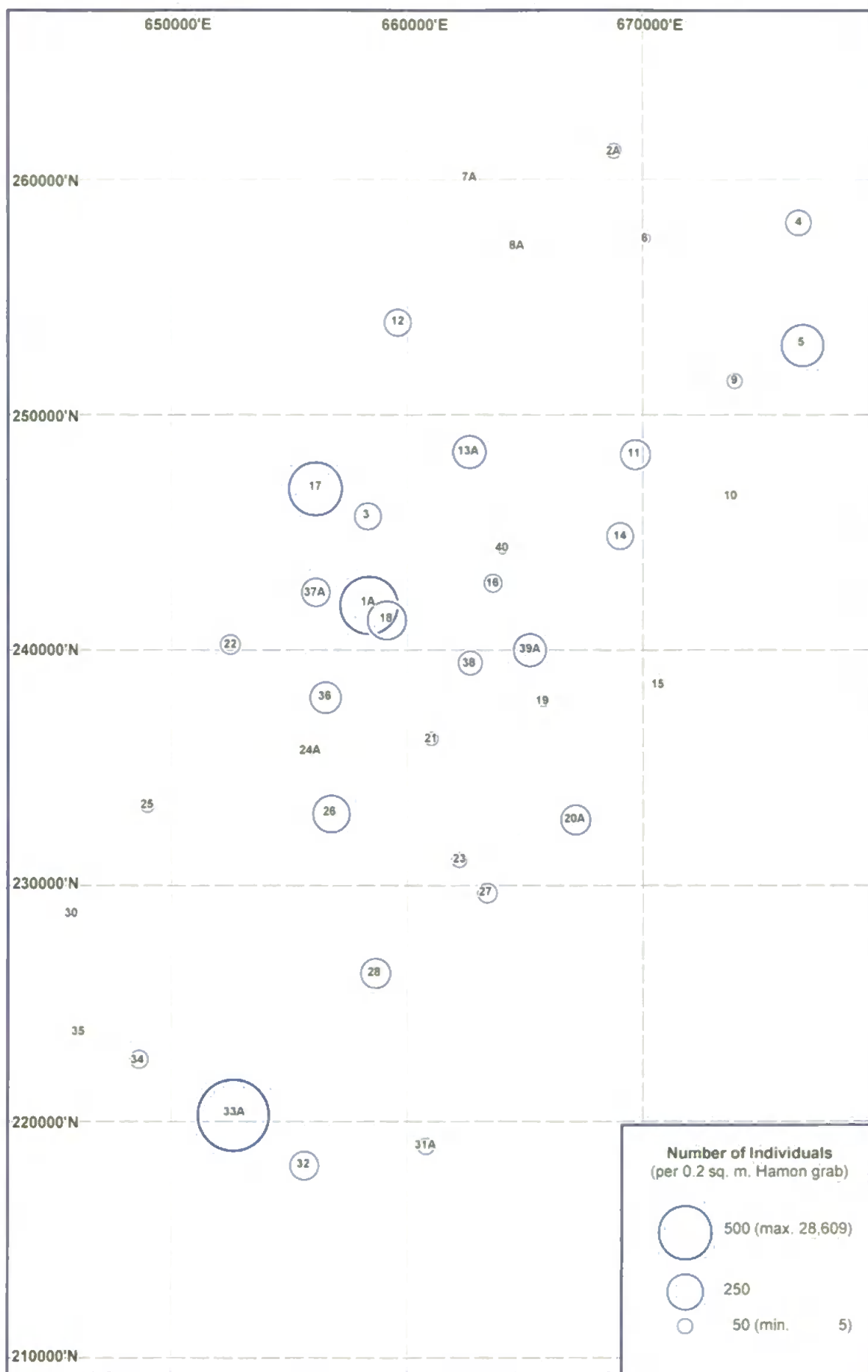
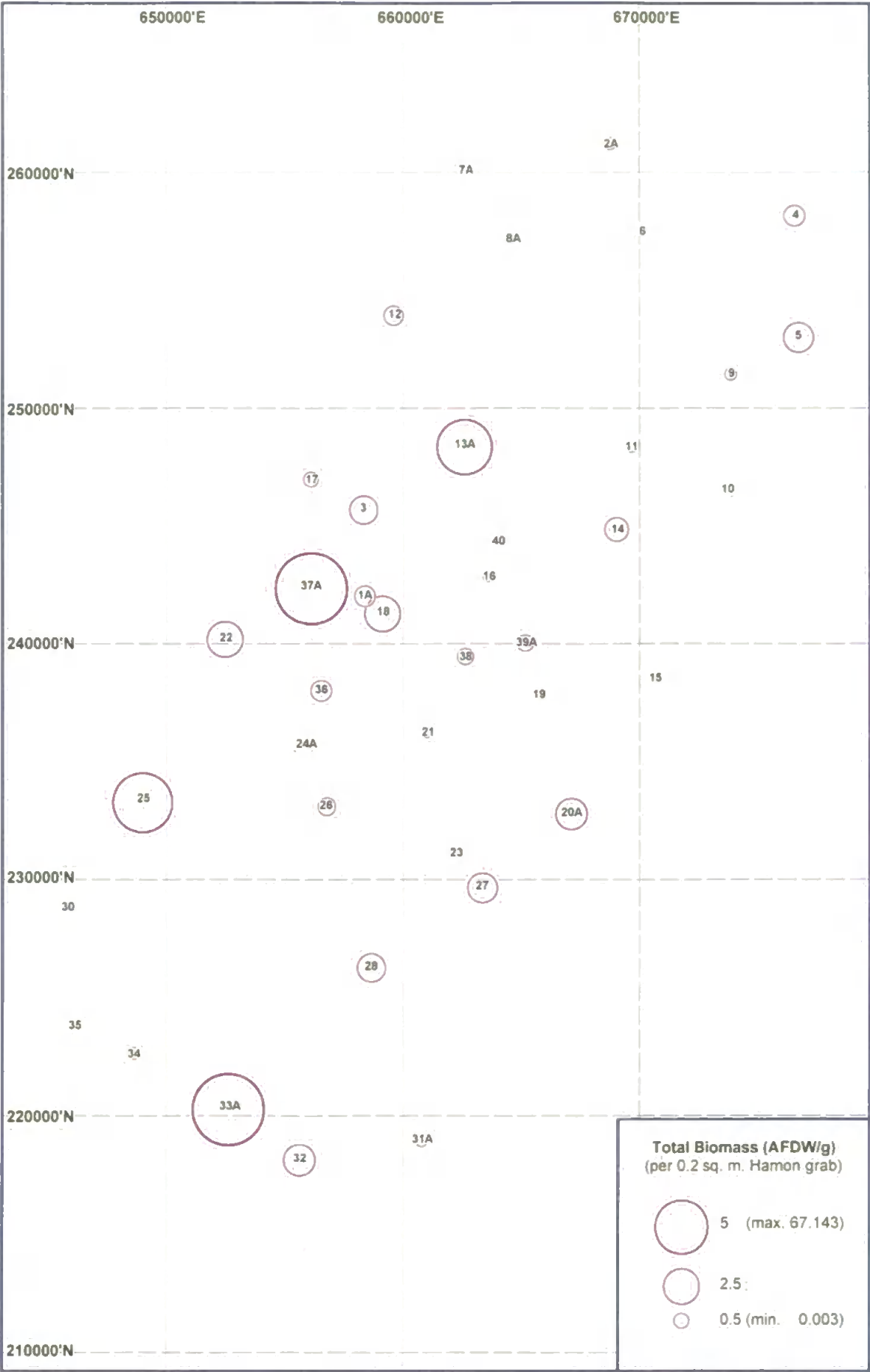


Figure 14 shows the total biomass, given as ash free dry weight (gm), of all faunal components extracted from various samples taken in the Shipwash Gabbard survey area. A very high biomass (67.143 gm) was measured at station 33 and this was due to the unusually high abundance of the polychaete *Lagis koreni*. Low biomass values were documented at stations 24C and 8, where the fauna was present in greatly reduced densities. Recorded biomass for the remaining stations was relatively uniform throughout.

Figure 15, an MDS plot, showing the previously identified sediment groups superimposed on the Bray-Curtis similarity of square root transformed abundance data at each station. From Figure 15, there is a trend between sediment type and benthic community structure. There is evidence of clustering of faunal samples belonging to sediment group 1 (coded blue) and, to a lesser extent, the communities falling into sediment category 2. The samples of the sand-dominated sediments (group 3) do not form a tight cluster and this may result from the low numbers of species recorded at these sites leading to fewer 'species in common' amongst these samples.



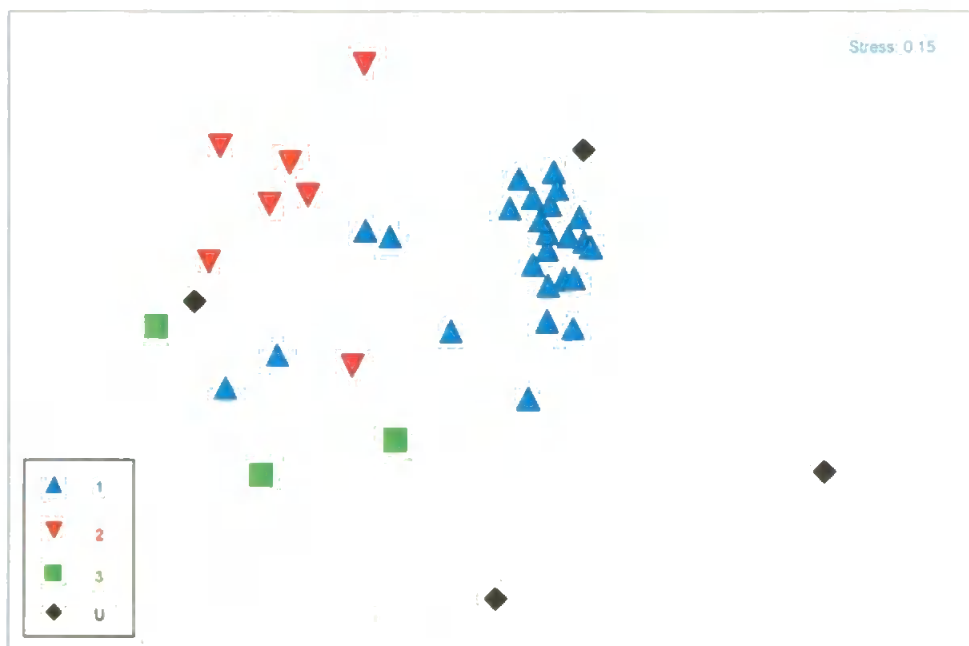
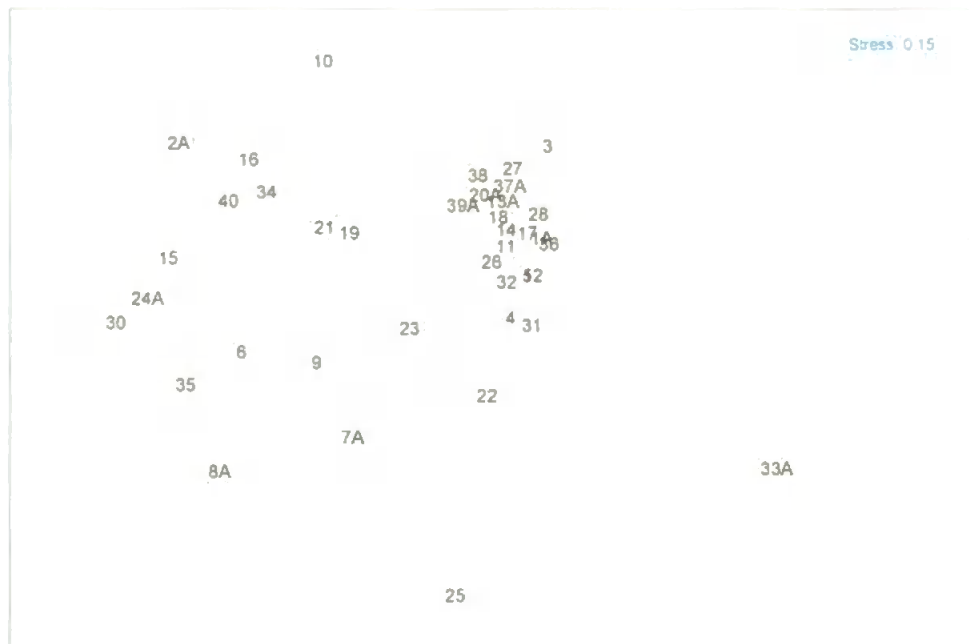


Table 2. Summary of the species that contribute to the similarity within the three groups of benthic macrofauna, identified by multi-variate analysis of the samples taken in the Shipwash Gabbard study area. The average abundance, the average similarity, the ratio of average similarity to standard deviation and the % contribution of each species to the similarity of each group is shown. For full results refer to Appendix XXIV.

Group 1						
Average similarity: 30.76						
Species		Av. Abund	Av. Sim	Sim/SD	Contrib%	Cum. %
<i>Ampelisca</i>	<i>spinipes</i>	17.8	2.67	1.29	8.68	8.68
<i>Lumbrineris</i>	<i>gracilis</i>	8.4	2.21	1.31	7.19	15.87
<i>Glycera</i>	<i>lapidum</i>	3.44	2.06	1.48	6.7	22.57
<i>Lanice</i>	<i>conchilega</i>	8.6	1.77	1.16	5.74	28.31
<i>Pomatoceros</i>	<i>lamarcki</i>	5	1.29	0.67	4.19	32.51
Group 2						
Average similarity: 29.35						
Species		Av. Abund	Av. Sim	Sim/SD	Contrib%	Cum. %
<i>Pisone</i>	<i>remota</i>	19.71	9.86	0.77	33.59	33.59
<i>Glycera</i>	<i>lapidum</i>	3.14	6.77	1.28	23.06	56.65
<i>Glycera</i>	<i>sp.</i>	3.14	4.75	0.87	16.19	72.84
<i>Notomastus</i>	<i>sp.</i>	4.29	2.68	0.56	9.12	81.96
Group 3						
Average similarity: 21.13						
Species		Av. Abund	Av. Sim	Sim/SD	Contrib%	Cum. %
<i>Ophelia</i>	<i>borealis</i>	1.33	12.53	4.14	59.31	59.31
<i>Nephtys</i>	<i>cirrosa</i>	1.33	5.04	0.58	23.83	83.15

The ranked contributions of various species 'belonging' to the sediment groups shown in Figure 15 are given in table 2. Sediment group 1 was characterised by a wide range of taxa including the amphipod *Ampelisca spinipes*, and the polychaetes *Lumbrineris gracilis*, *Glycera lapidum* and *Lanice conchilega*. Sediment group 2 has the greatest contribution to the within group similarity made by polychaetes such as *Pisone remota*, *Glycera lapidum* and juvenile glycerids. Samples belonging to sandy sediments (group 3) were characterised by polychaetes including *Ophelia borealis* and *Nephtys cirrosa*. Indeed, many of the species characterising this grouping have been reported previously in the past to inhabit well-oxygenated sands (Bachelet *et al.* 1996; Desroy *et al.* 2002). Mysids of the genus *Gasterosaccus* burrow in medium sands more frequently than in gravels (Brown & Talbot 1972; Mauchline 1980) and have been described in similar habitats by Bachelet *et al.* (1996) and Vallet *et al.* (2001). A significant quantity of silt at many of the sites comprising sandy gravels (group 1 sediments),

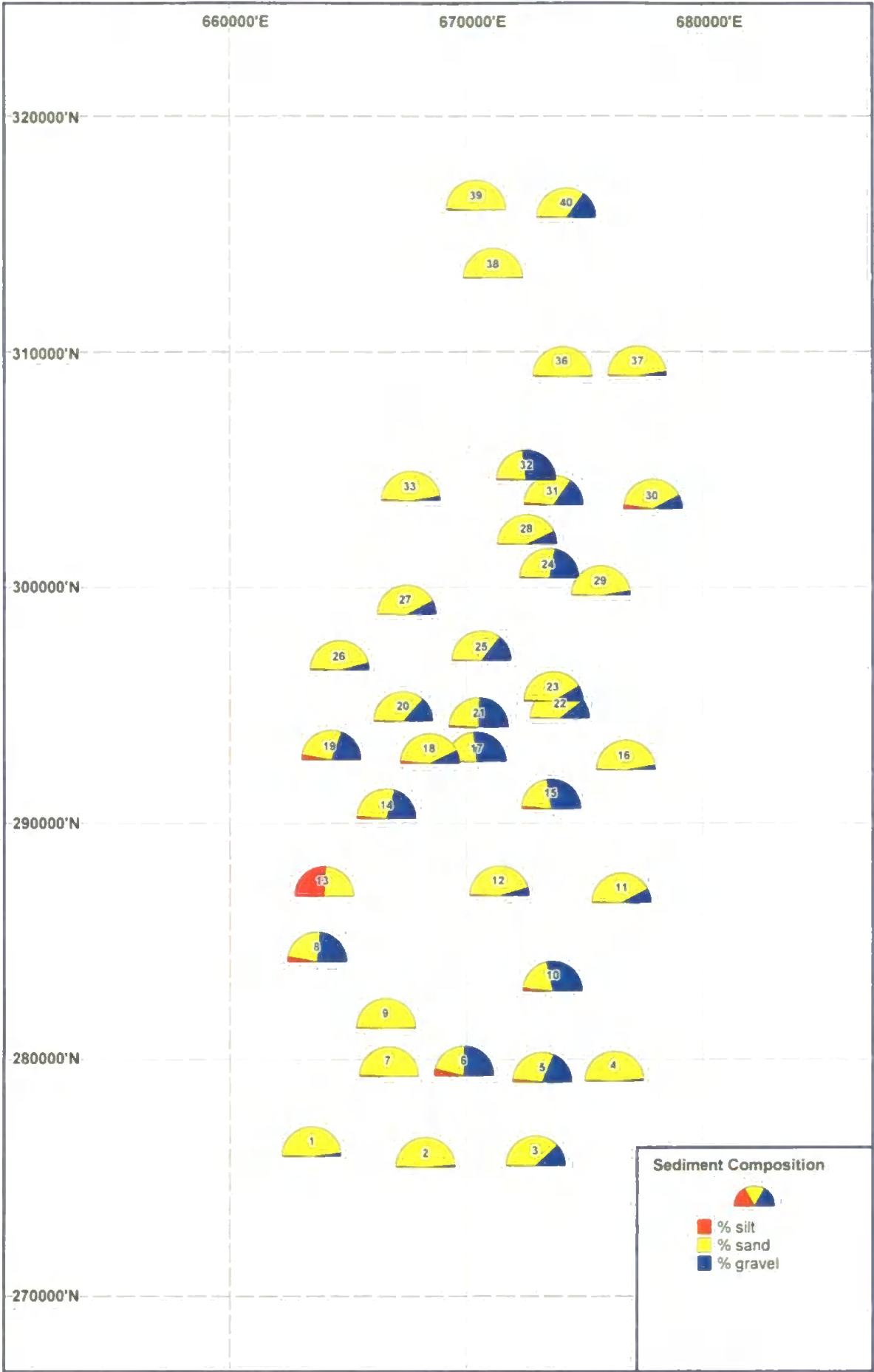
was a probable factor in the presence of *Lagis koreni* (Polychaeta) and the terrebellid *Lanice conchilega* (Dewarumez *et al.* 1992; Fromentin *et al.* 1997; Desroy 2002).

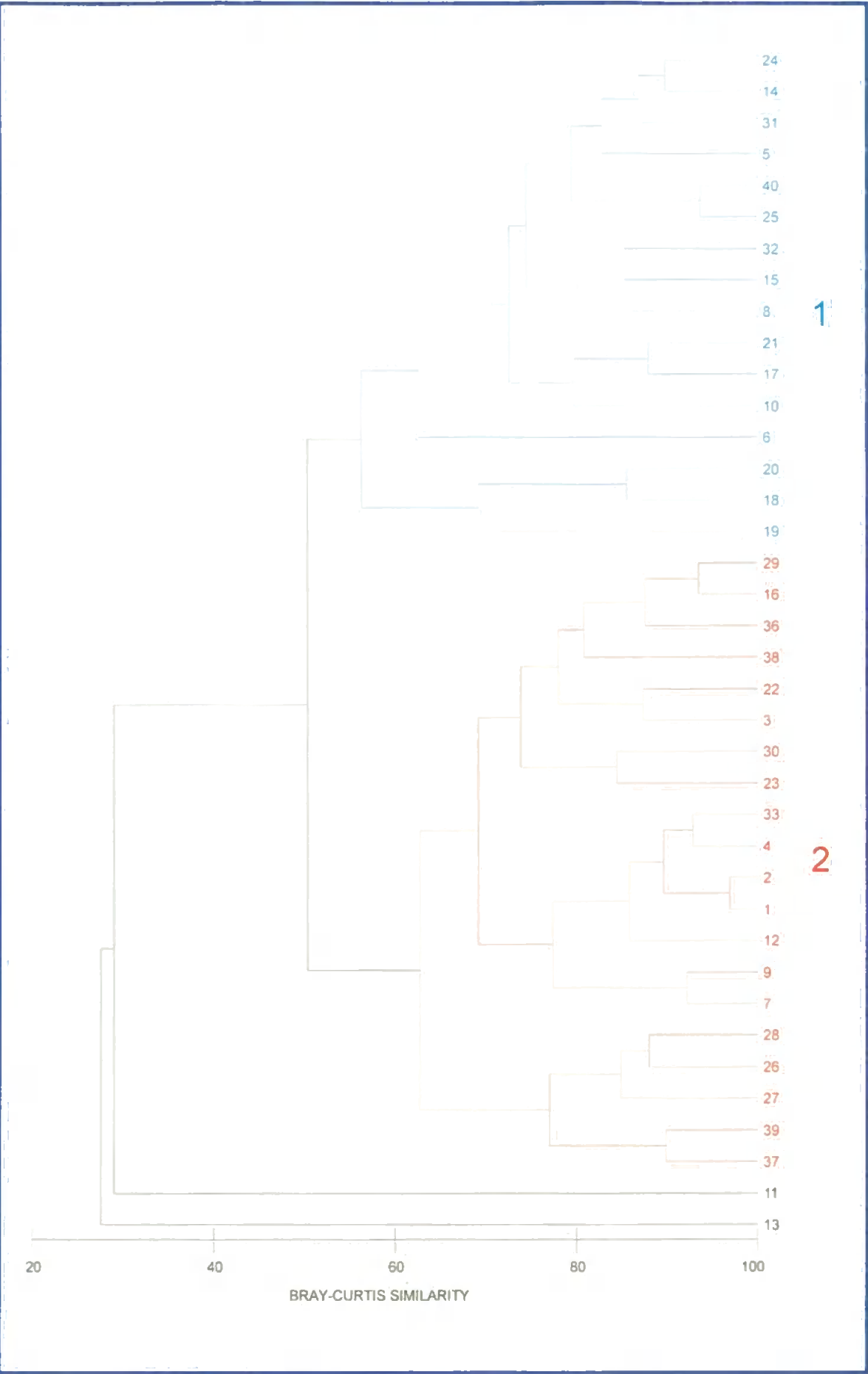
4.3.2. Lowestoft

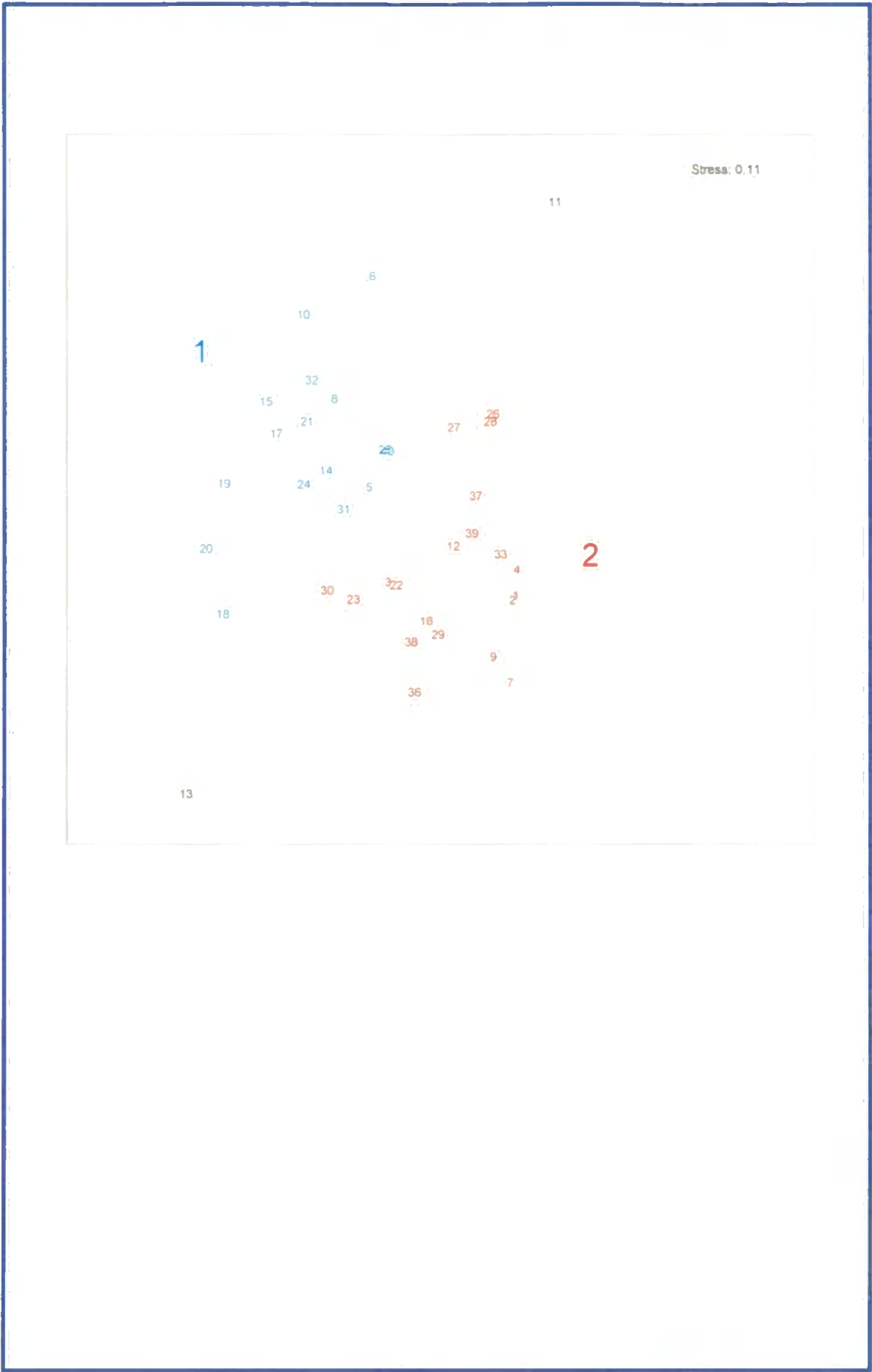
4.3.2.1. Substratum Composition

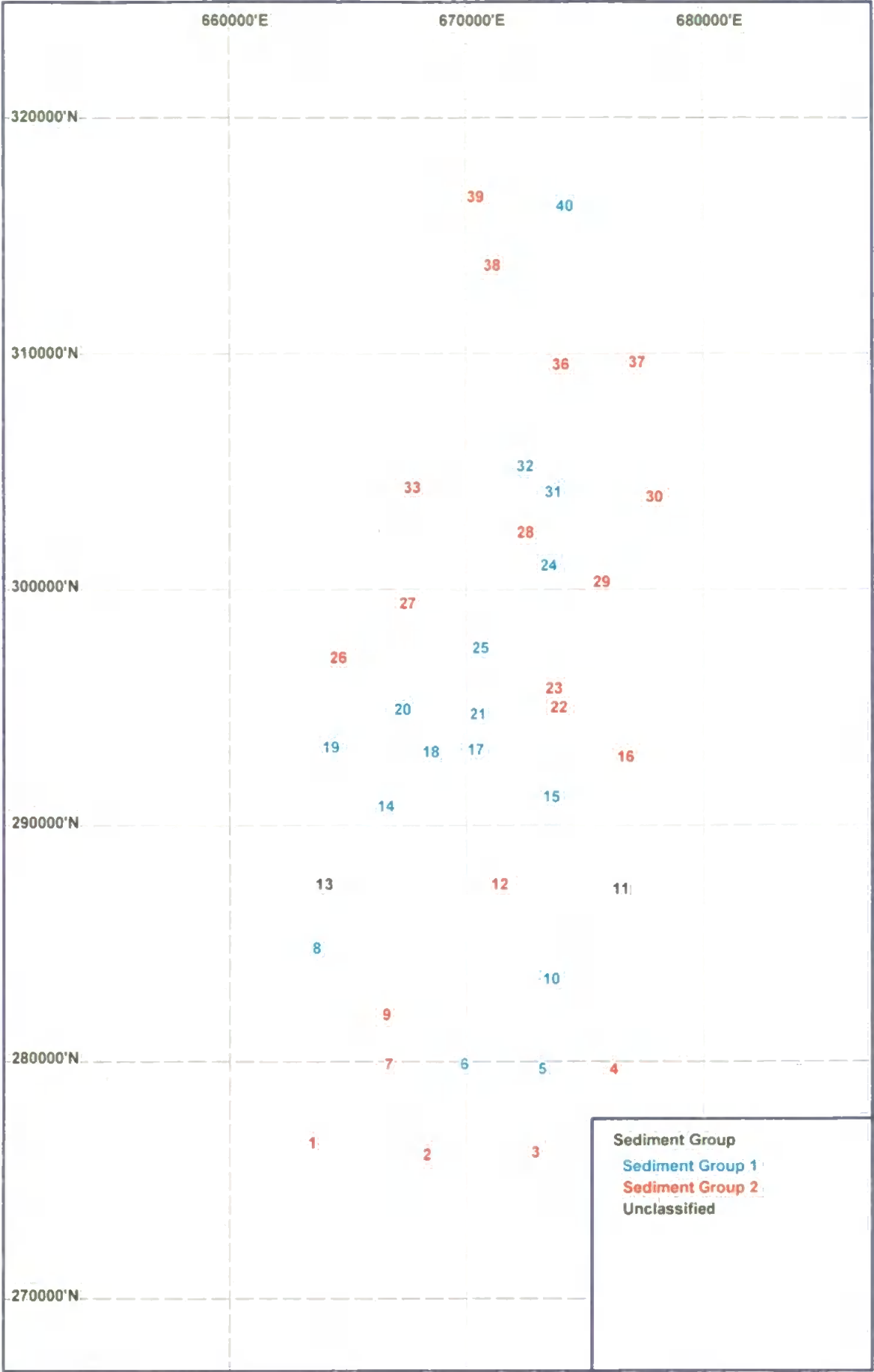
The central area of the Lowestoft sampling grid was characterised by coarse deposits that occurred in a north-south direction. Western and eastern samples comprised finer sediments and, in some cases (e.g. station 13) contained significant proportions of silt and clay. Generally the deposits possessed a high sand component, whether mixed with coarse particles or not, although some samples had a gravel content of 50% or more (Figure 16). Two distinct categories of sediment exist in the Lowestoft survey area (Figures 17 & 18). Group 1 stations (coded blue) consisted of gravely substrata with varying fractions of fine particles, Group 2 (coded red) were sandy deposits with lesser proportions of coarser particles. Station 13, previously identified as possessing a high percentage of silt, did not fall into either group; another outlying sample (site 11) comprised coarse sands with some fine gravel (Figures 17 & 18).

Figure 19 confirms that coarse deposits were found in central stations and for some distance to the north and south before giving way to finer grained substrata (sediment group 2).







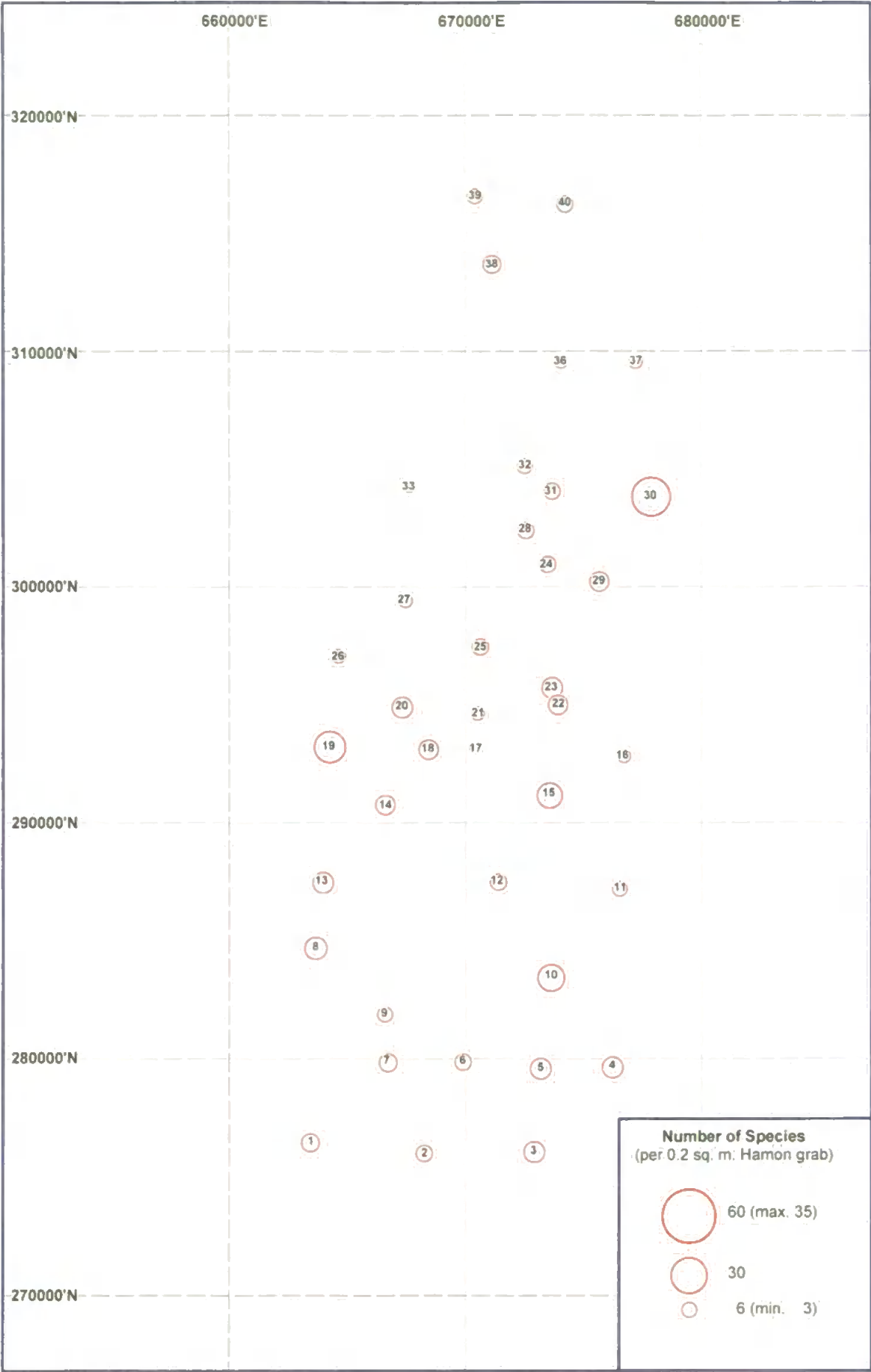


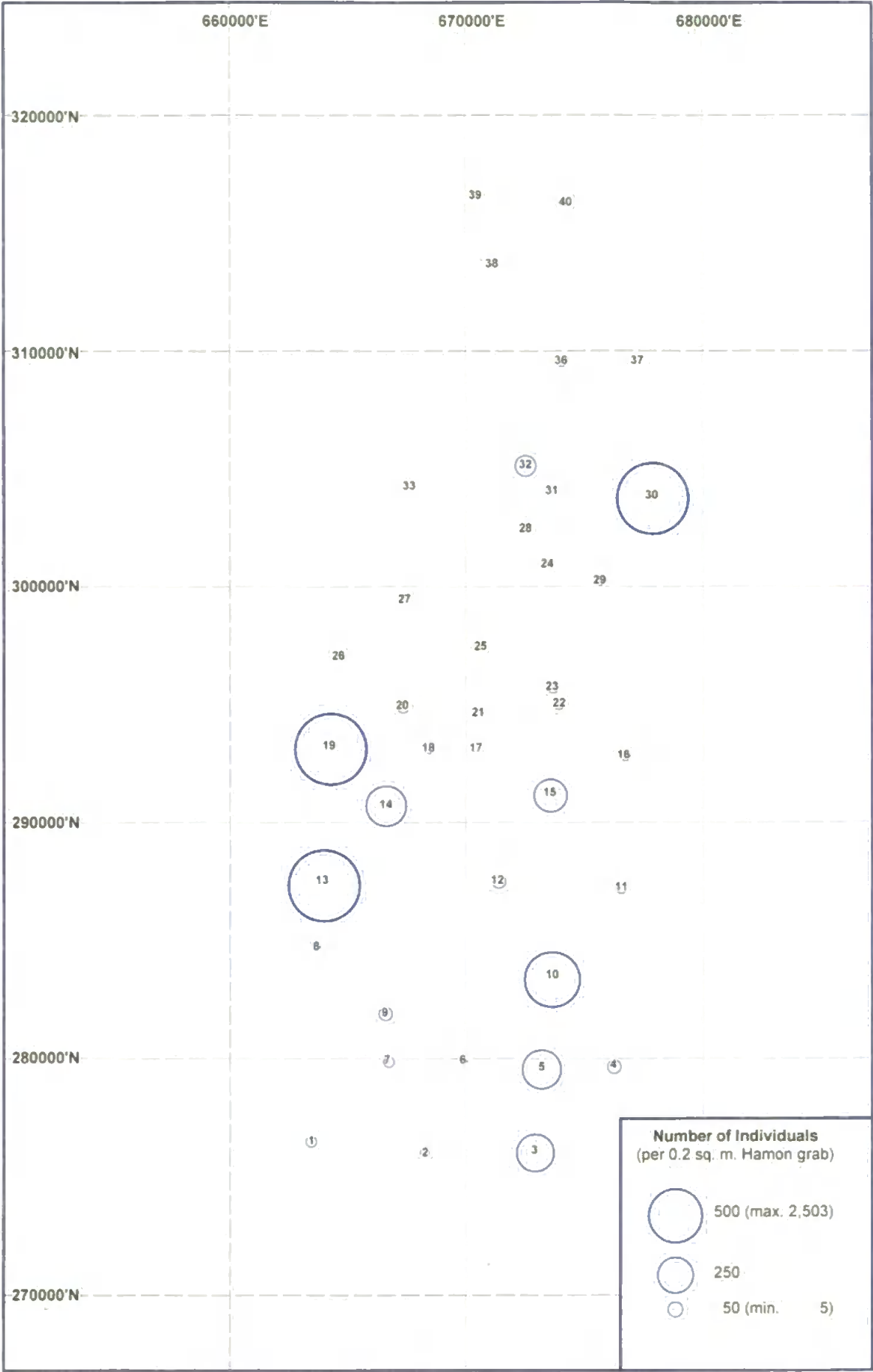
4.3.2.2. General Features of Benthic Community Structure

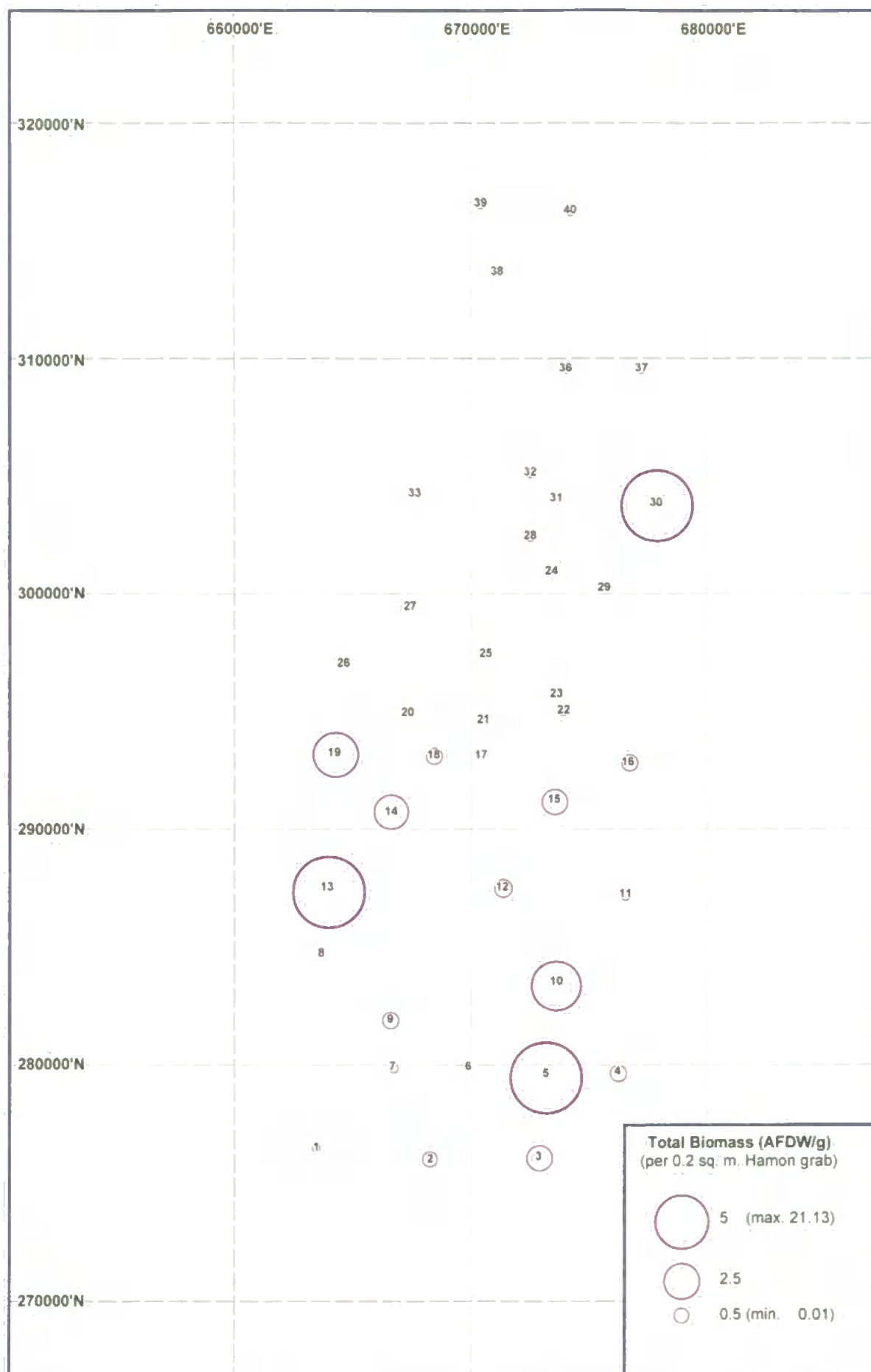
In total, 97 different taxa were noted and polychaetes were the dominant members of the macrofauna (47% of all recorded species). Molluscs, echinoderms and miscellaneous epifaunal components were represented by comparatively few species. Species numbers were highest at station 30 where 35 taxa were discovered; stations 17 and 33 had the lowest species diversity with three species recorded at each site (Figure 20). Species diversity was generally uniform throughout the region with patches of higher diversity seeming to coincide with increased silt content of the substratum. Low numbers of taxa were recorded at stations comprising clean sands.

Macrofaunal population density reached a peak of 2503 individuals at station 13 in the west of the survey area; the high numbers were attributable to the polychaete *Lagis koreni* (Figure 21). In contrast, five individuals were recorded at stations 27 and 37, and northern stations appeared to support fewer individuals than those in the central and southern regions of the survey area (Figure 21). Population density appeared to be uniformly low at most stations, with isolated patches of relatively high macrofaunal abundances; these patches appeared to correspond to a moderate-high silt component of the deposits (Figure 21).

Figure 22 shows biomass values (ash free dry weight in grams), were greatest at station 13, corresponding to the maximum population density. The minimum biomass recorded was at station 21 where a low population density was also noted. Biomass followed a similar pattern to that of population density, with a consistently low biomass at most stations. In the east, stations 5, 10 and 30 exhibited high biomass. In western sites, high biomass values appear to coincided with elevated silt levels.





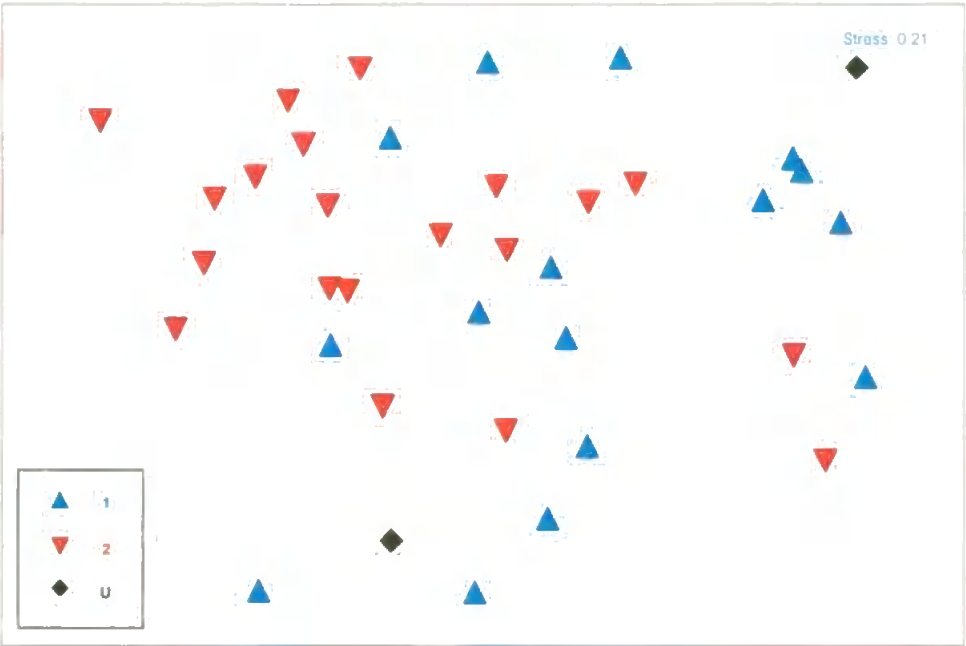


The wide scatter of the stations on the MDS plot (Figure 23) indicates low internal similarity between the samples. Low diversity and, therefore, increased likelihood of samples having fewer taxa 'in common', in many samples from the Lowestoft site may be a reason for such a lack of similarity between stations.

Table 3. Summary of the species that contribute to the similarity of macrofauna recorded within the two sediment groups, identified by multivariate analysis of the samples taken in the Lowestoft study area. The average abundance, the average similarity, the ratio of average similarity to standard deviation and the % contribution of each species to the similarity of each group is shown. For full results refer to Appendix XXV.

Group 2						
Average similarity: 23.67						
Species		Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
<i>Gastrosaccus</i>	<i>spinifer</i>	5.15	7.37	0.73	31.15	31.15
<i>Ophelia</i>	<i>rathkei</i>	3.05	5.32	0.9	22.47	53.61
<i>Nephtys</i>	<i>caeca</i>	1.55	3.64	0.74	15.37	68.99
<i>Ammodytes</i>	<i>tobianus</i>	0.55	1.45	0.4	6.11	75.1
Group 1						
Average similarity: 17.26						
Species		Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
<i>Lagis</i>	<i>koreni</i>	62.5	4.53	0.6	26.23	26.23
<i>Ophelia</i>	<i>rathkei</i>	1.94	2.85	0.51	16.53	42.76
<i>Nephtys</i>	<i>caeca</i>	0.88	1.9	0.55	11.02	53.78
<i>Glycera</i>	<i>rouxii</i>	0.88	1.35	0.37	7.83	61.61
<i>Sabellaria</i>	<i>spinulosa</i>	53	1.16	0.33	6.74	68.36

The SIMPER analysis shows the low internal similarity of the faunal samples belonging to the two sediment groupings (See Table 3). It is also apparent that the two groups have some common species such as *Ophelia rathkei* and *Nephtys caeca*, thus indicating that the two sediment groups do not have contrasting fauna associated with them. The sediments of both groups contained species that occur commonly in sandy deposits (with smaller quantities of coarse particles) and share similarities to the assemblages described by Bachelet *et al.* (1996); JNCC (1997); Olivier & Retiere (1998); and Vallet *et al.* (2001).



4.3.3. West Bassurelle

4.3.3.1. *Substratum Composition*

The substratum comprised sandy gravel with coarser deposits in western and central zones; eastern-most stations were characterised by a greater proportion of sand (Figure 24). The dendrogram and corresponding MDS plot, identified three groups of deposit (Figures 25 & 26). The separation of the samples in the MDS plot represents a gradient from the finest sediments (shaded green) to coarser sediments (coloured blue) with intermediate (red) deposits lying between them. The distribution of sediments identified by multivariate analysis comprised mainly sandy gravels with varying proportions of shell debris (Figure 27). Generally, coarser substrata were found in the southwest and centre of the sampled area, finer, sandy deposits existed in stations in the northeast of the region, with 'transitional' sandier gravels separating the coarse and fine deposits. Gravels were generally present in concentrations of approximately 40%.

4.3.3.2. *General Features of Benthic Community Structure*

A total of 285 different taxa were recorded and relatively high species diversity was found, with an average of 41 species per 0.2 m² Hamon Grab sample. Polychaetes accounted for the most recorded species (33%), followed by crustaceans (29%), with miscellaneous groups and molluscs accountable for 17% and 16% respectively (Figure 28).

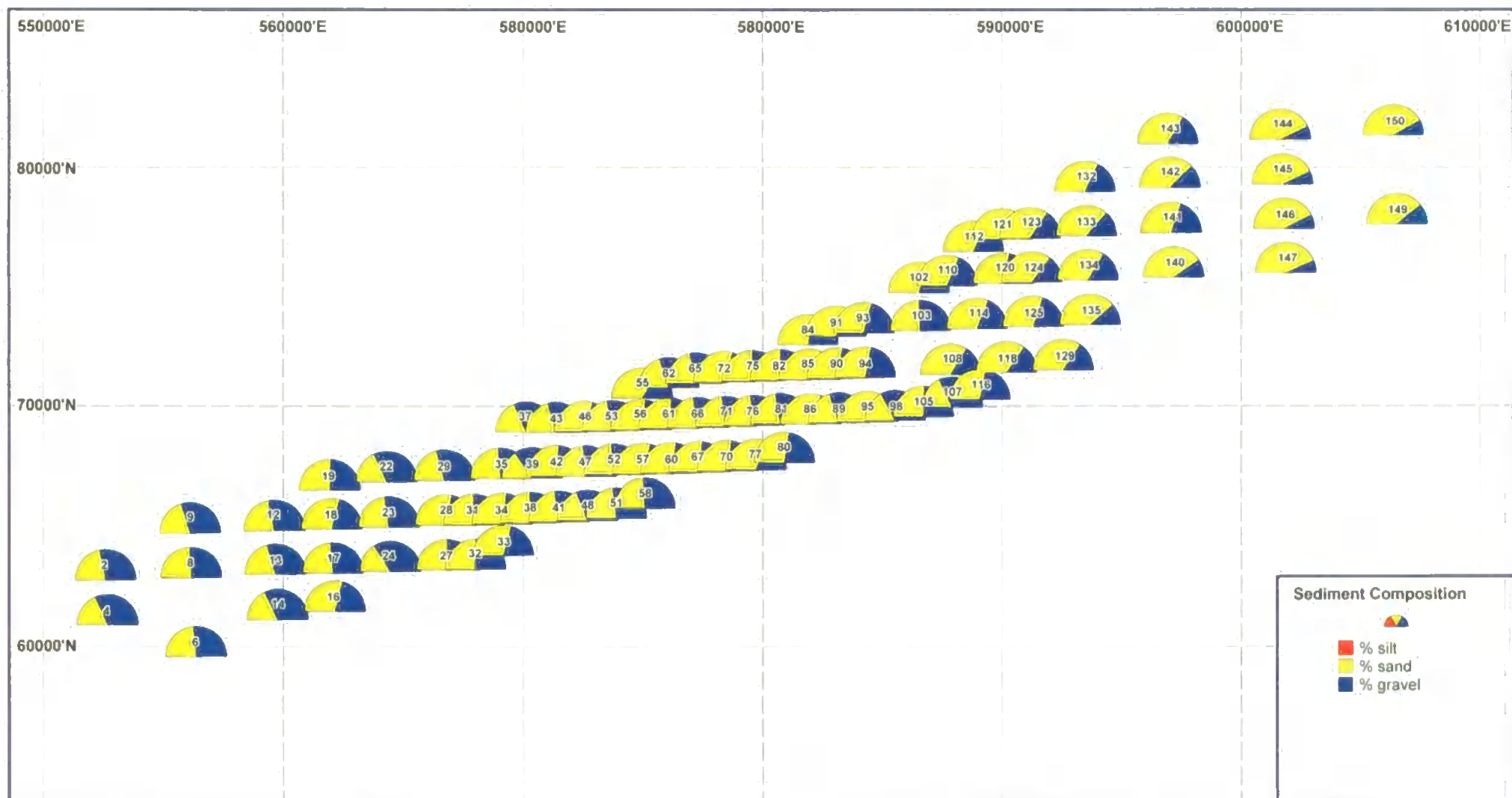
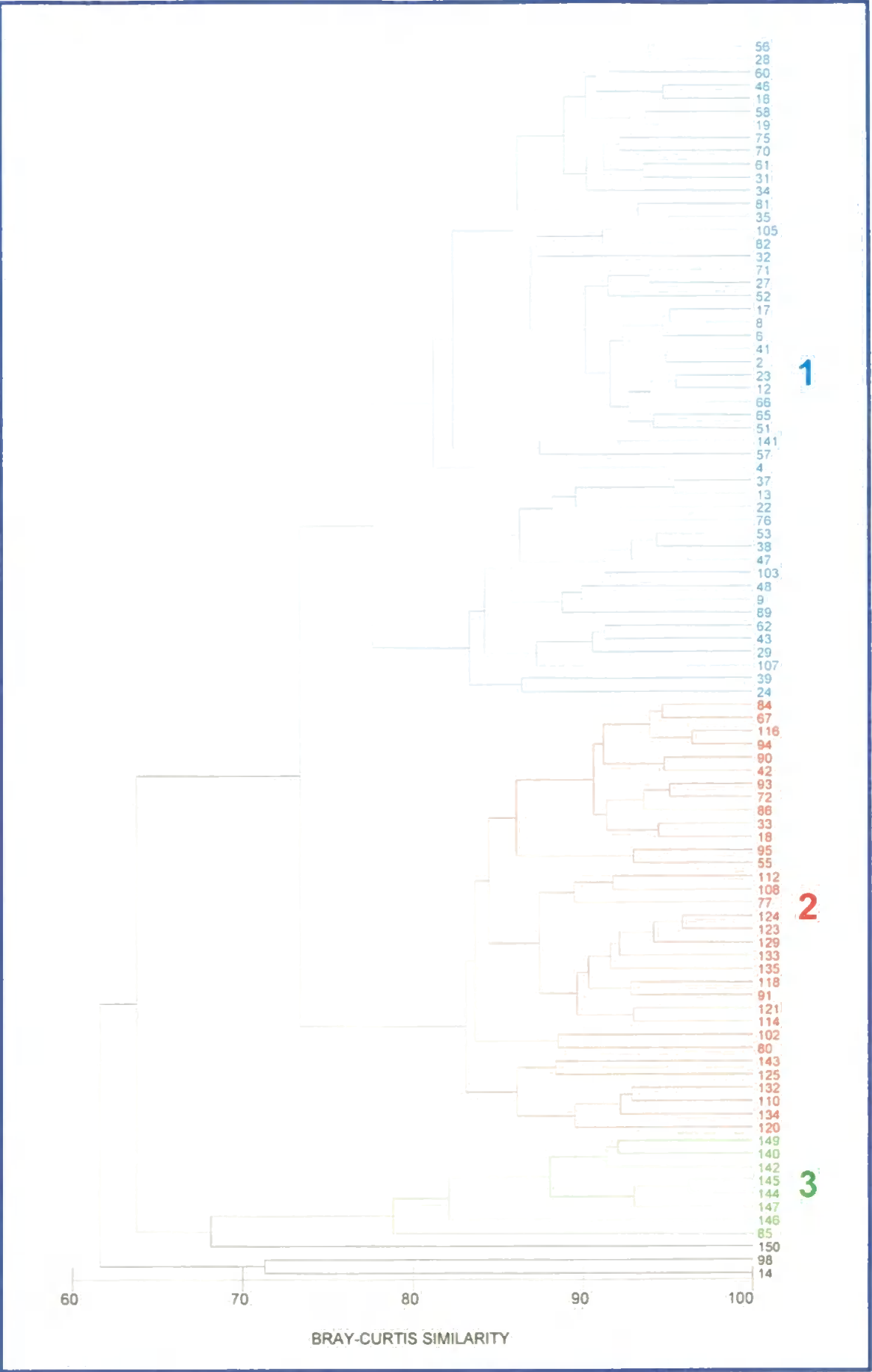


Figure 24. The relative percentage silt (0.063mm), sand (0.063-1mm) and gravel (2-32mm) at benthic stations sampled at the West Bassurelle study area.





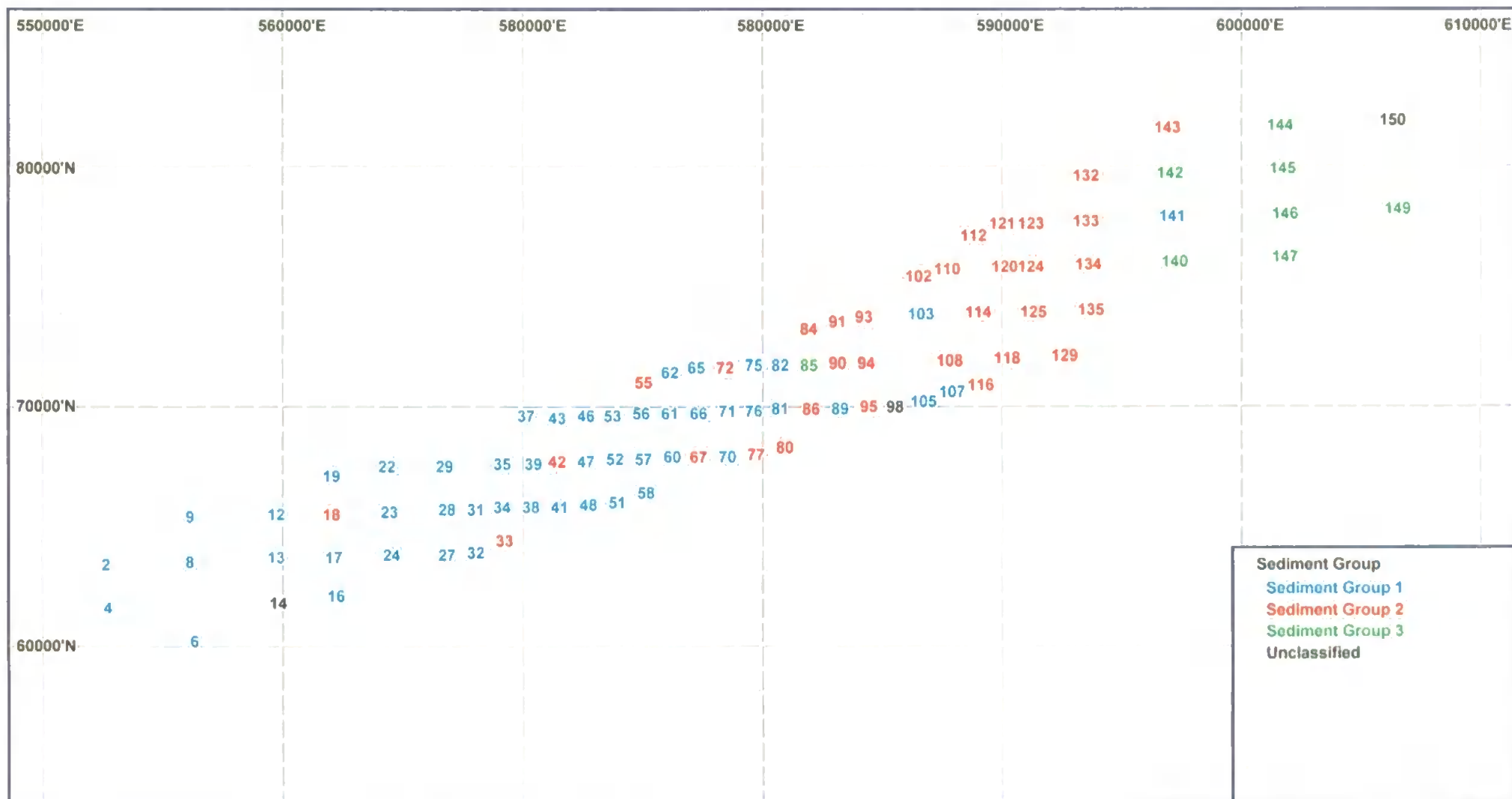


Figure 27. The distribution of sediment groups identified by non-parametric multivariate analysis of particle size composition of benthic stations sampled in the West Bassurelle study area.

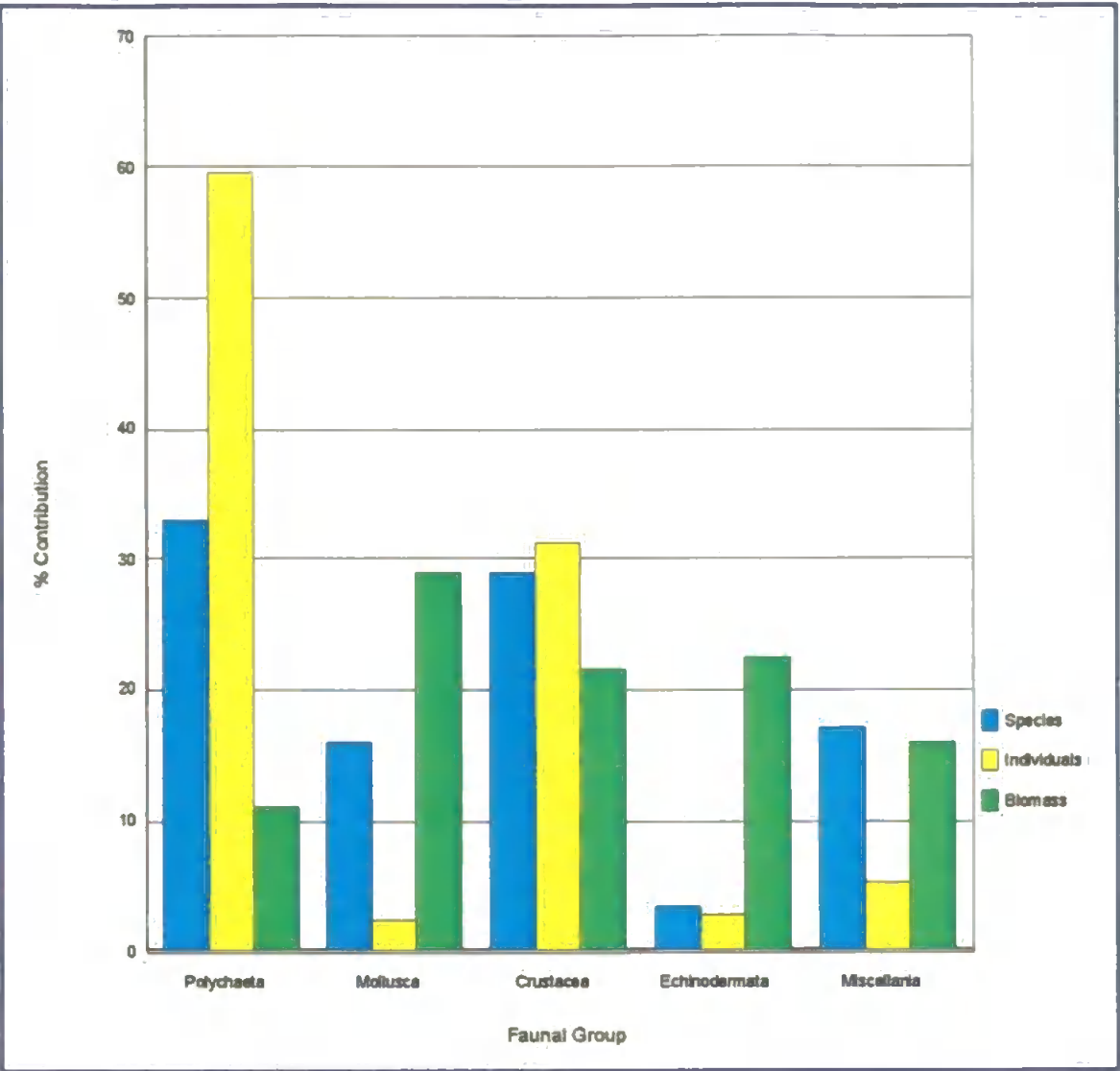


Figure 28. The relative contributions of the main faunal components to species diversity, population density, and biomass of the benthos recorded in the West Bassurelle study area. The abundance of 'Miscellania' is not comparable with that for other groups due to difficulties in the quantification of colonial organisms (such taxa have been recorded as 'present' and assigned a nominal value of 1).

The maximum number of species (79) was recorded at station 37; in contrast, just 7 species were found in the fine shell-sands at station 142. Figure 29 shows diversity to be uniformly high throughout much of the region, with low diversity corresponding to the finer deposits of eastern stations. The coarse deposits, in the southwest and central sectors of the survey area, supported a greater range of species than the finer deposits in the northeast.

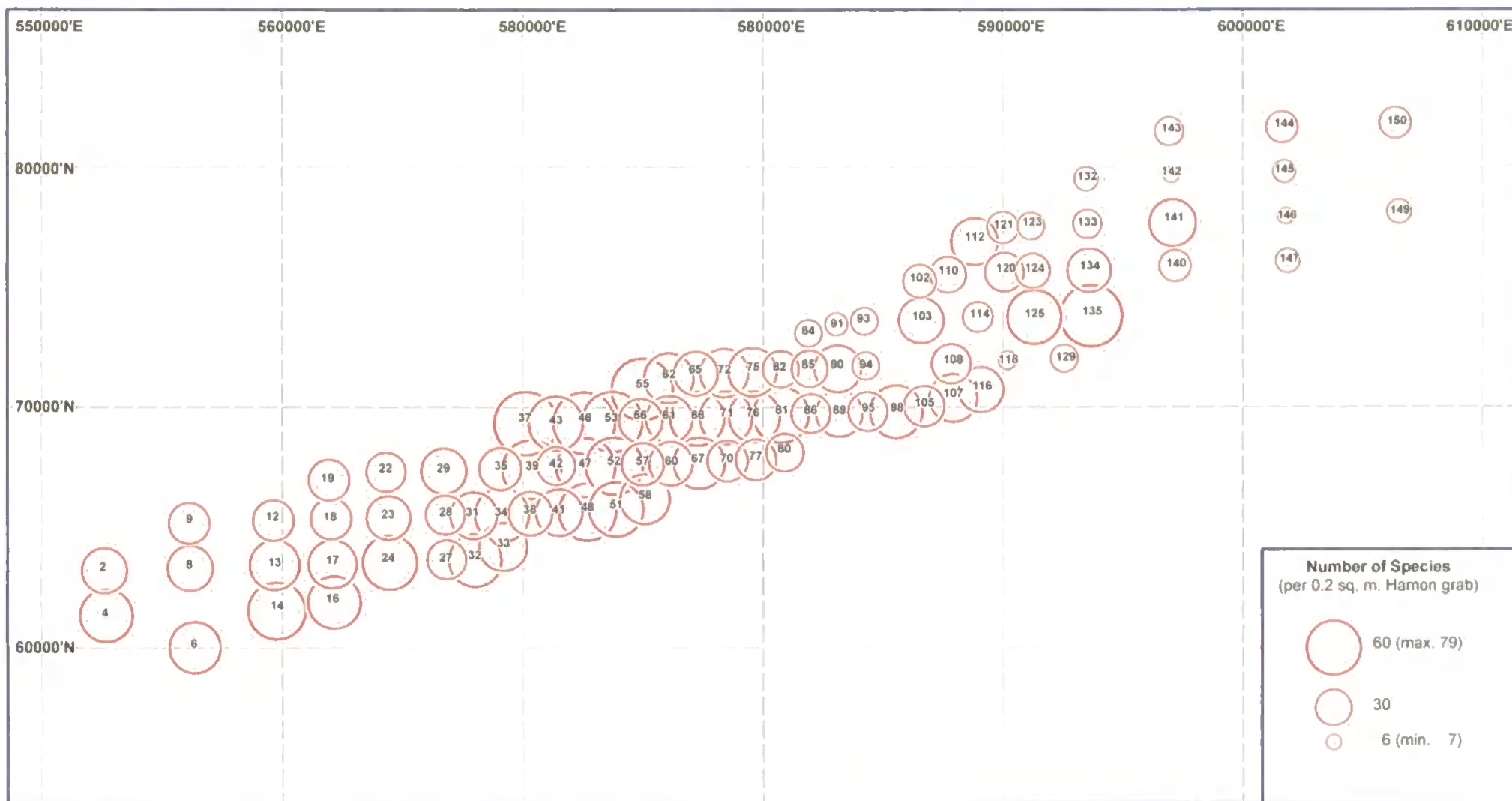


Figure 29. Thematic map showing the number of species of macrofauna (>1mm) per 0.2 m² based on Hamon grab samples at benthic stations sampled in the West Bassurelle study area.

Polychaetes dominated the population density (59.5% of all individuals recorded), followed by crustaceans (31.1%), molluscs, echinoderms and miscellaneous groups (Figure 28). Figure 30 shows a maximum of 506 individuals was reached at station 52, the lowest macrofaunal numbers were found at station 146 (20 individuals). As for species diversity, density decreased at the easternmost stations where sands represented the dominant substratum type. Numbers of individuals were greatest in coarse deposits, possibly as a result of suitable attachment surfaces for epifaunal species being more readily available for colonisation. Density showed a similar pattern to that of species diversity, with the highest densities of macrofauna being recorded in southwest and central stations, coinciding with the coarser substratum. In contrast, the sandy deposits of the northeast were inhabited by fewer individuals per 0.2 m² Hamon Grab sample.

Biomass (AFDW) that reached a maximum at station 120 where over 17 grams (AFDW) of invertebrate material was measured; the lowest biomass value (0.04 gm /AFDW) was recorded at station 132. Figure 28 reveals that molluscs accounted for approximately 29% of the total biomass (large specimens of *Glycymeris glycymeris* were frequently encountered). Other significant contributors to the biomass included echinoderms (22.5%) and crustaceans (21.5%). Polychaetes contributed least to the total biomass despite being both the species and numerical dominants.

Macrofaunal biomass does not appear to follow the same pattern as species variety and population density, as peaks and troughs of biomass were distributed evenly throughout the area and did not appear to follow the gravel-sand gradient. Indeed, the clean sands of some eastern stations have high biomass values,

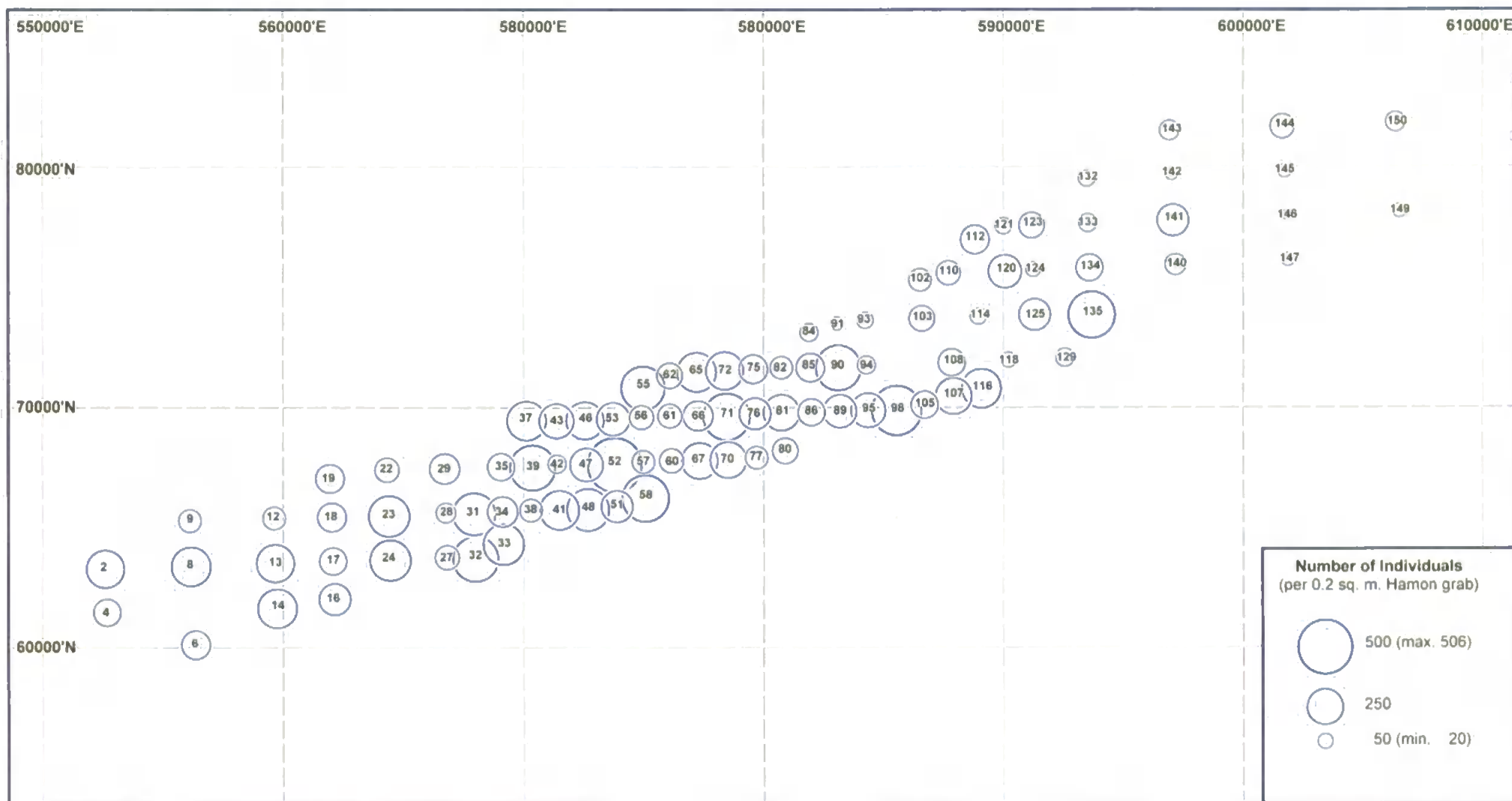


Figure 30. Thematic map showing the number of individuals of macrofauna (>1mm) per 0.2 m² based on Hamon grab samples at benthic stations sampled in the West Bassurelle study area.

attributable mainly to the presence of bivalve molluscs (*Ensis ensis*) and large, burrowing echinoids including *Spatangus purpureus* (Figure 31).

The MDS plot, based on the Bray-Curtis similarity of square root transformed abundance data, highlight the relationship between the sediment type and the benthos in the West Bassurelle region (Figure 32). It is apparent that alterations in the benthic community follow the sedimentary gradient described earlier, with the fauna changing in composition as the coarseness of the substratum lessens in a west-east direction.

Table 4. Summary of the species that contribute to the similarity of macrofauna recorded within the three sediment groups, identified by multivariate analysis of the samples taken in the West Bassurelle study area. The average abundance, the average similarity, the ratio of average similarity to standard deviation and the % contribution of each species to the similarity of each group is shown. For full results refer to Appendix XXVI.

Group 1						
Average similarity: 45.86						
Species		Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
<i>Galathea</i>	<i>intermedia</i>	20.3	3.63	2.24	7.92	7.92
<i>Eteone</i>	<i>longa</i>	16.3	3.03	2.06	6.6	14.52
<i>Syllis</i>	<i>gracilis</i>	7.94	2.3	2.76	5.01	19.53
<i>Scalibregma</i>	<i>inflatum</i>	8.94	2.16	1.81	4.72	24.25
<i>Pomatoceros</i>	<i>triqueter</i>	14.04	2.03	1.23	4.44	28.69
Group 2						
Average similarity: 36.26						
Species		Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
<i>Eteone</i>	<i>longa</i>	23.45	6.11	2.2	16.86	16.86
<i>Syllis</i>	<i>gracilis</i>	7.97	3.11	1.96	8.58	25.44
<i>Glycera</i>	<i>sp.</i>	8.42	2.21	0.87	6.11	31.55
<i>Echinocyamus</i>	<i>pusillus</i>	5.15	1.79	1.11	4.94	36.49
<i>Pionosyllis</i>	<i>sp. #1</i>	5.09	1.76	0.99	4.84	41.34
Group 3						
Average similarity: 25.64						
Species		Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
<i>Eteone</i>	<i>longa</i>	17.5	9.27	1.45	36.16	36.16
<i>Glycera</i>	<i>sp.</i>	5.12	3.12	0.91	12.15	48.31
<i>Aonides</i>	<i>paucibranchiata</i>	1.75	1.84	0.68	7.17	55.48
<i>Eurydice</i>	<i>sp.</i>	3.88	1.82	0.66	7.11	62.59
<i>Nemertea</i>	<i>spp.</i>	1.38	1.76	0.66	6.86	69.45

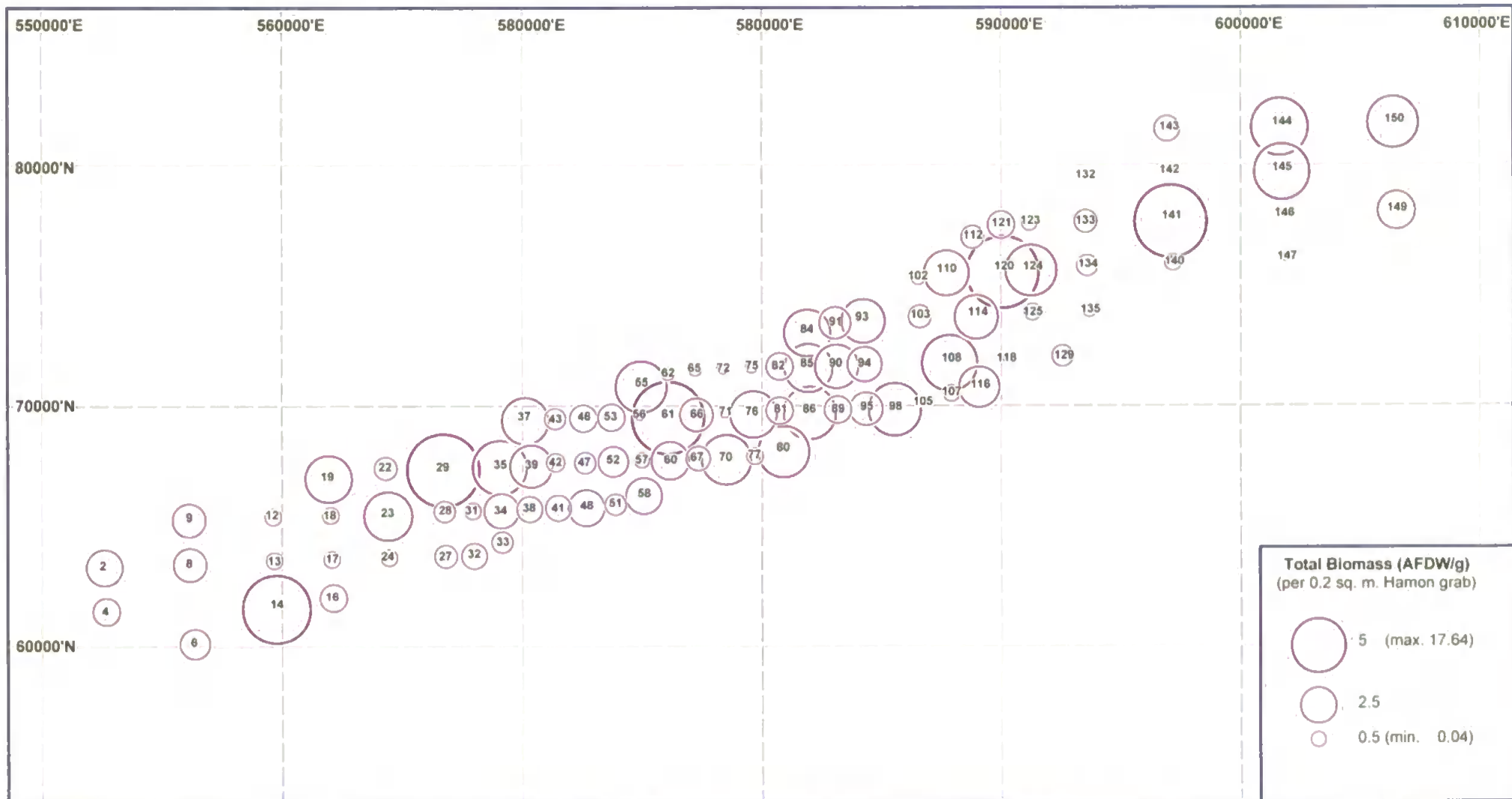
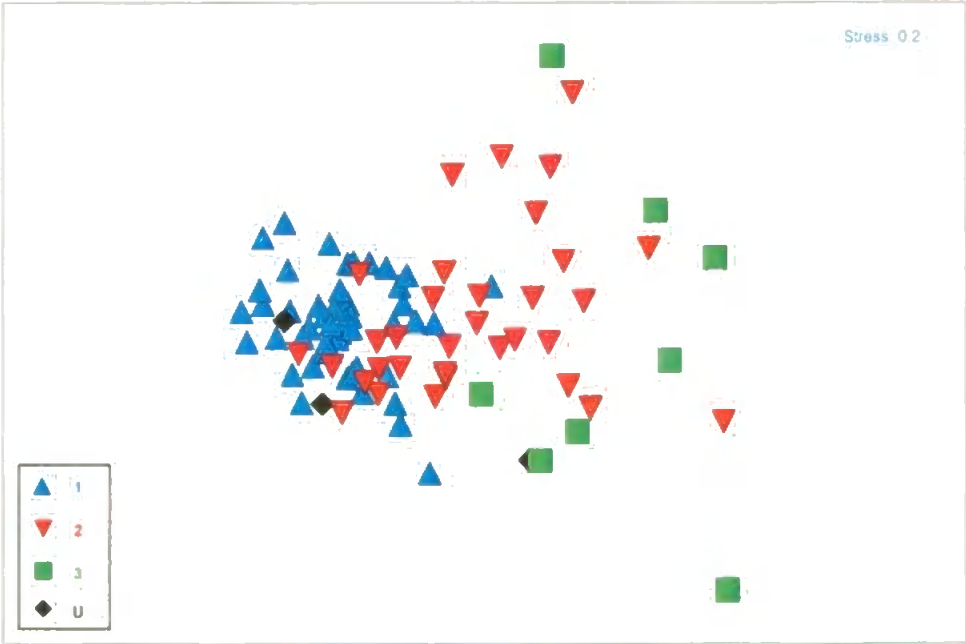
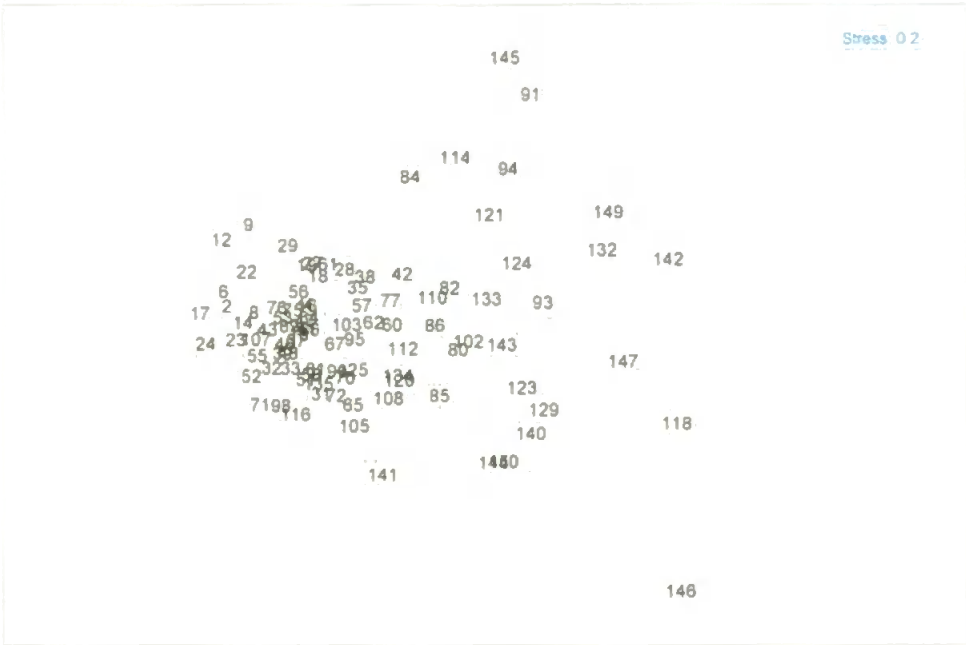


Figure 31. Thematic map showing the biomass (ash free dry weight) in grams per 0.2 m² based on Hamon grab samples at benthic stations sampled in the West Bassurelle study area.



From the SIMPER analysis, shown in Table 4 it can be seen that fauna associated with sediment group 1 comprised the decapod *Galathea intermedia* and polychaete species such as *Eteone longa*, *Syllis gracilis*, *Scalibregma inflatum* and *Pomatoceros triqueter*. Sediment group 2 had similar inhabitants to the first substratum type but the contribution of *G. intermedia* and *P. triqueter* to the internal similarity of samples in this grouping was much less. Indeed, these latter species were important contributors to the dissimilarity between the faunal assemblages of sediment groups 1 and 2 (Appendix XXVI). This is a probable reflection of these species favouring coarser substrata. The fauna associated with sediment group 3 was more similar to that of group 2 than to group 1. Again, this is likely to be due to the increased sand component and lowered coarse component of these two substratum types (groups 2 & 3). The lower internal similarity of the fauna allied with group 3 deposits points to higher variability in these samples. As with the sandy substrata described for other regions, this may be a result of low diversity leading to a lack of common species between the samples belonging to this sediment category. Large bivalves and echinoderms were present in many samples but did not feature in the SIMPER analysis. This is a likely consequence of under sampling of larger components of the macrofauna with the Hamon grab employed during sample collection.

4.3.4. North Nab

4.3.4.1. Substratum Composition

The deposits of this area were predominantly sands and gravels of varying coarseness. The central region of the survey area had been subjected to dredging since 1991 and consisted mainly of gravels; these gravels gave way to sands in the eastern sector of the survey grid. Western stations comprised gravels with variable amounts of sand and mud. Patches of bedrock were also noted in some

stations. Figure 33 shows a thematic map of the relative proportion of gravel recorded in samples taken from the North Nab survey area. It is apparent that the deposits situated in the central stations of the survey area comprised a high proportion of gravel (>60%). The low percentage of gravel recorded in the eastern stations reflected the sandy nature of the deposits in this region.

It can be seen from the dendrogram and corresponding MDS plot (Figures 34 & 35) that four definable groups of sediments existed in the North Nab survey area. One outlying station (sample 134) was situated in the centre of an anchor-dredged pit and was noted as being exceptionally coarse, possibly reflecting the removal of overlying finer sediments during the dredging process. Sediment groups 1 and 2 consisted of gravels with varying proportions of sand, whereas group 3 and 4 deposits were characterised by sands of varying coarseness.

Figure 36 shows that the sediments of the western and central stations belonged to the group 2 classification, with a much smaller group (sediment group 1) towards the east. The finer group 3 and 4 sediments were located in the north east of the region.

4.3.4.2. General Features of Benthic Community Structure

Polychaetes made up the greatest proportion of species, over 40% of all species recorded in the North Nab survey area belonged to this group (Figure 37). Crustaceans accounted for approximately 29% of all recorded taxa, followed by miscellaneous groups (20%) and the Mollusca (9%). Echinoderm species were conspicuously absent and only comprised around 1% of the identified species. Crustaceans made the greatest contribution to overall abundances, due most to the exceptionally high numbers of *Balanus* sp. at some stations. Polychaetes,

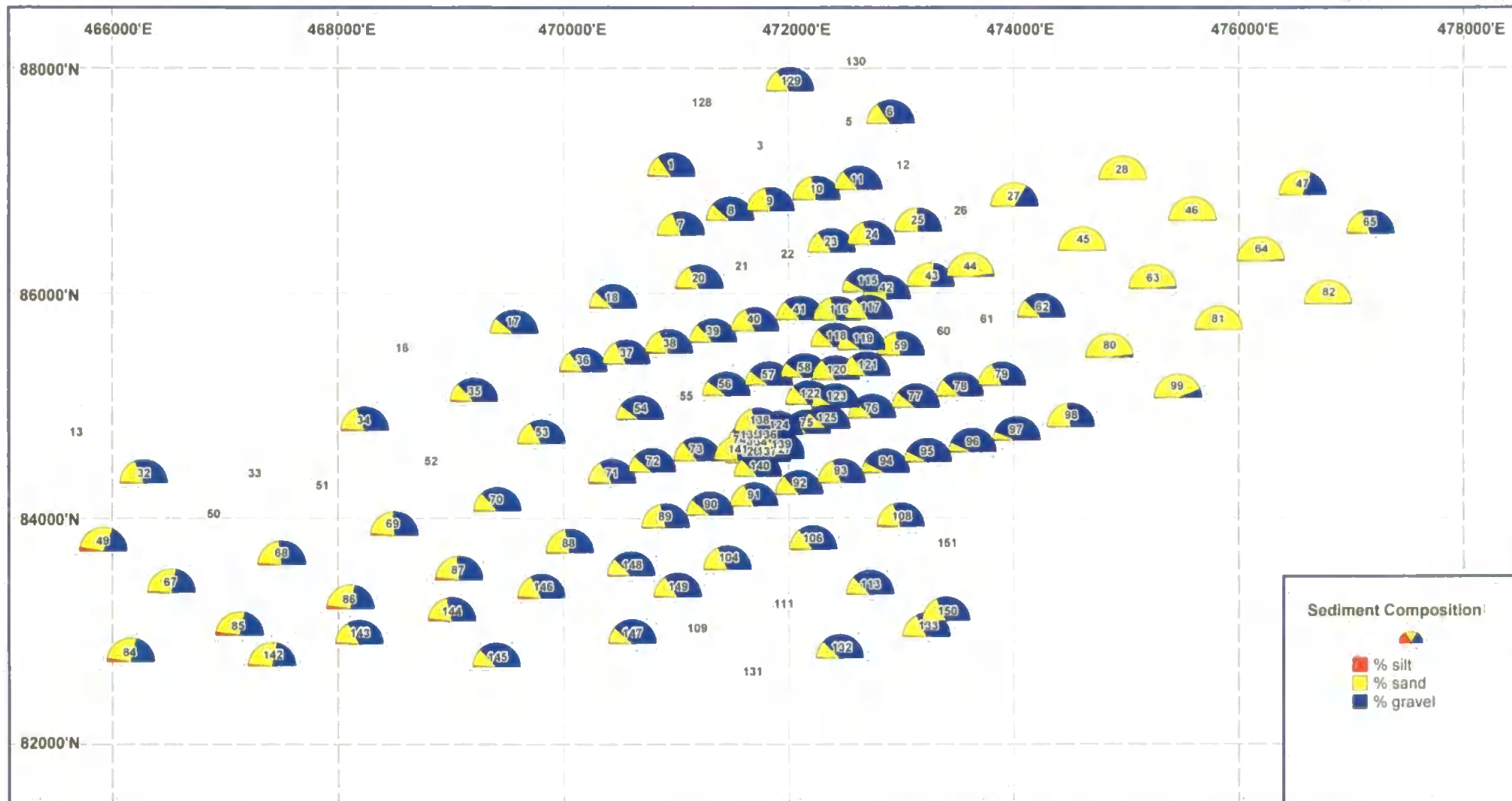
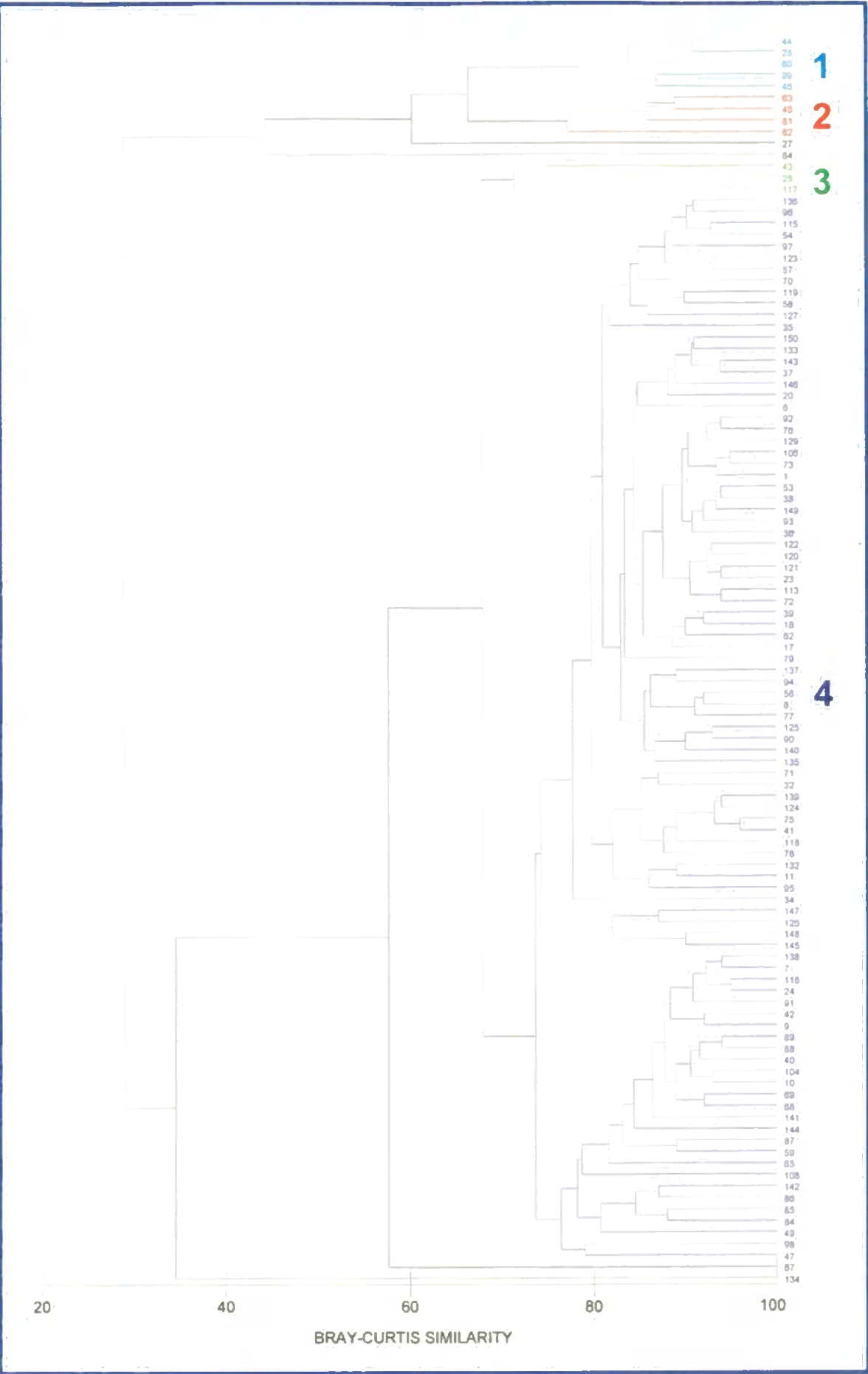
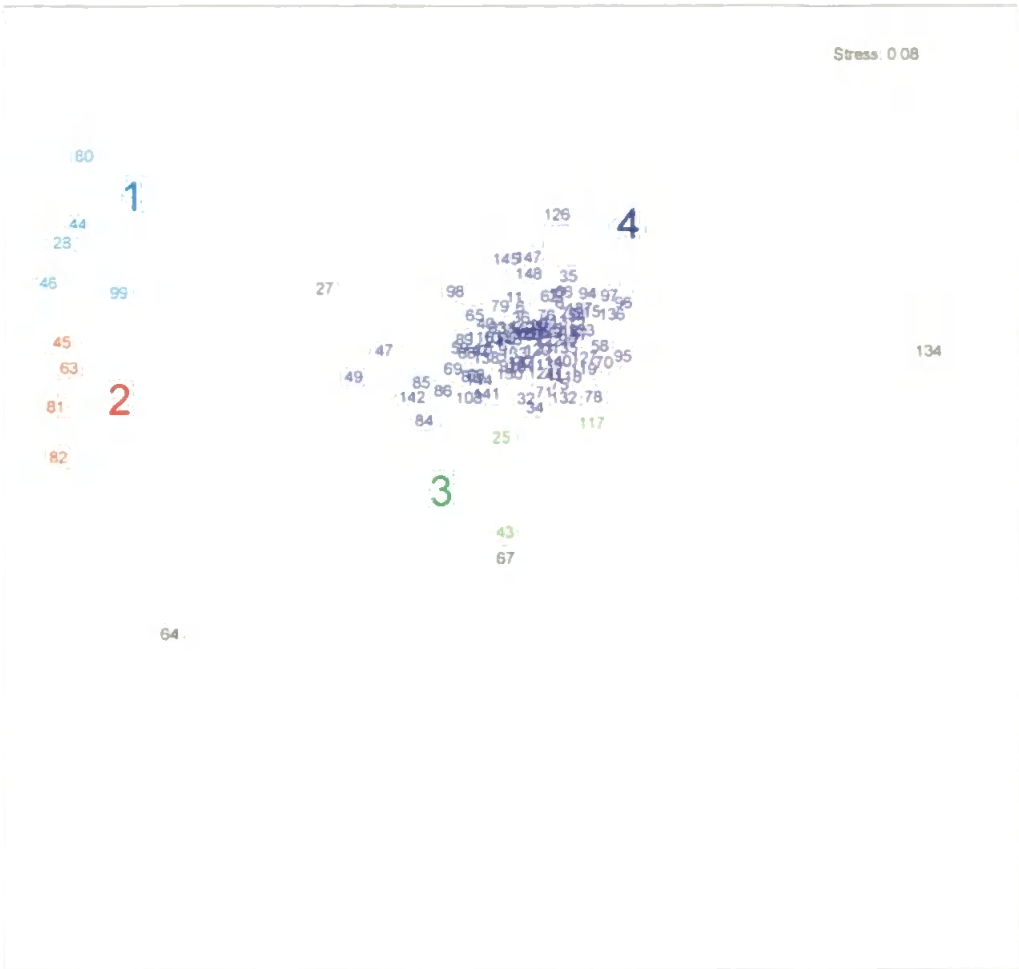


Figure 33. The relative percentage of silt (<0.036mm), sand (0.036-1mm) and gravel (2-32mm) at benthic stations sampled in the North Nab study area.





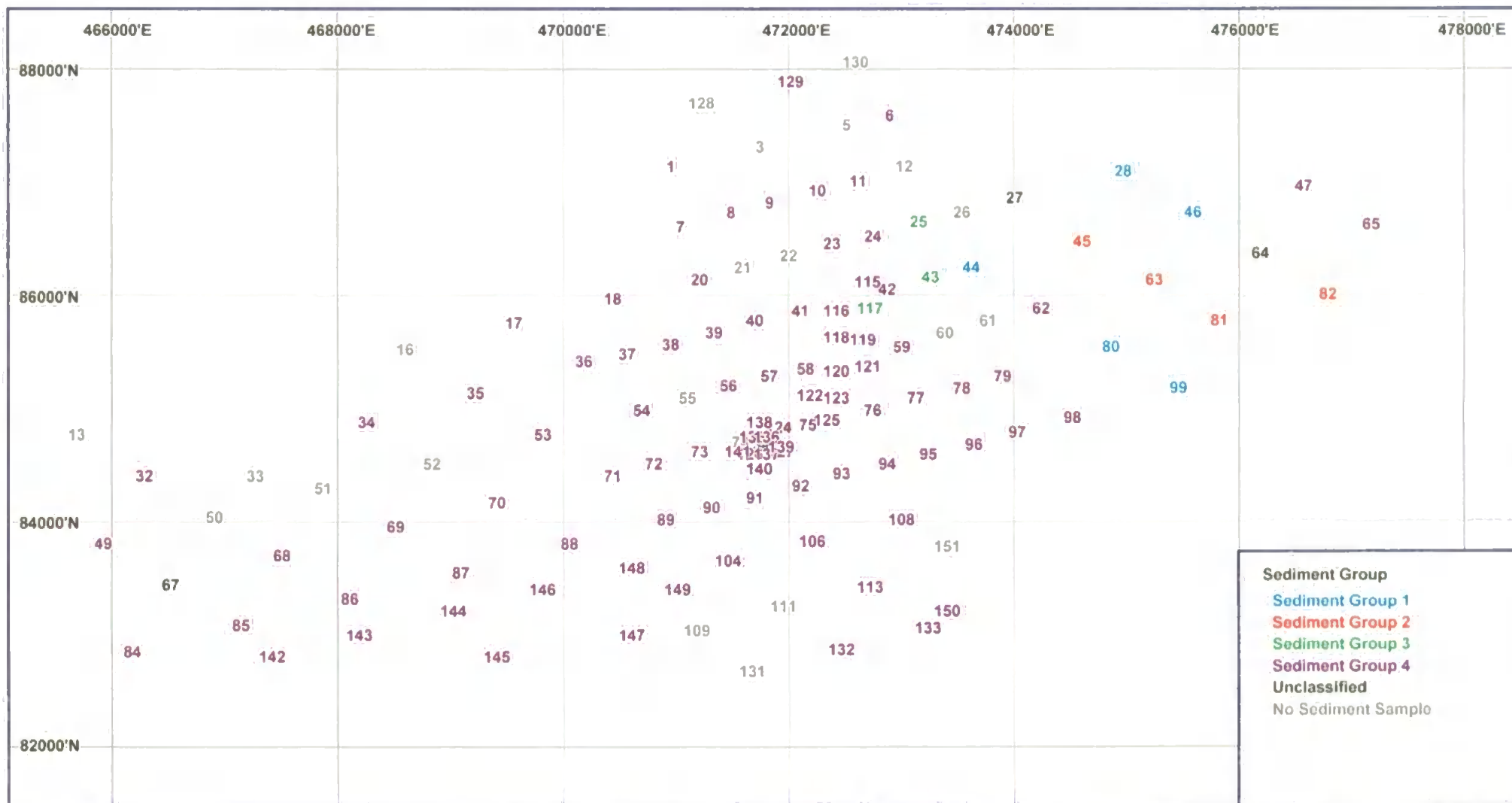


Figure 36. The distribution of the sediment groups identified by non-parametric multivariate analysis of particle size composition of benthic stations sampled in the North Nab study area.

molluscs and miscellaneous groups all shared a similar abundance contribution. Biomass was dominated by miscellaneous groups (57%), particularly, the bryozoan, *Flustra foliacea*, although other epifaunal taxa, such as *Distomus variolosus* and a number of hydroid species, were also important. Molluscs accounted for approximately 17% of the overall biomass, due largely to high numbers of relatively large-bodied *Crepidula fornicata*.

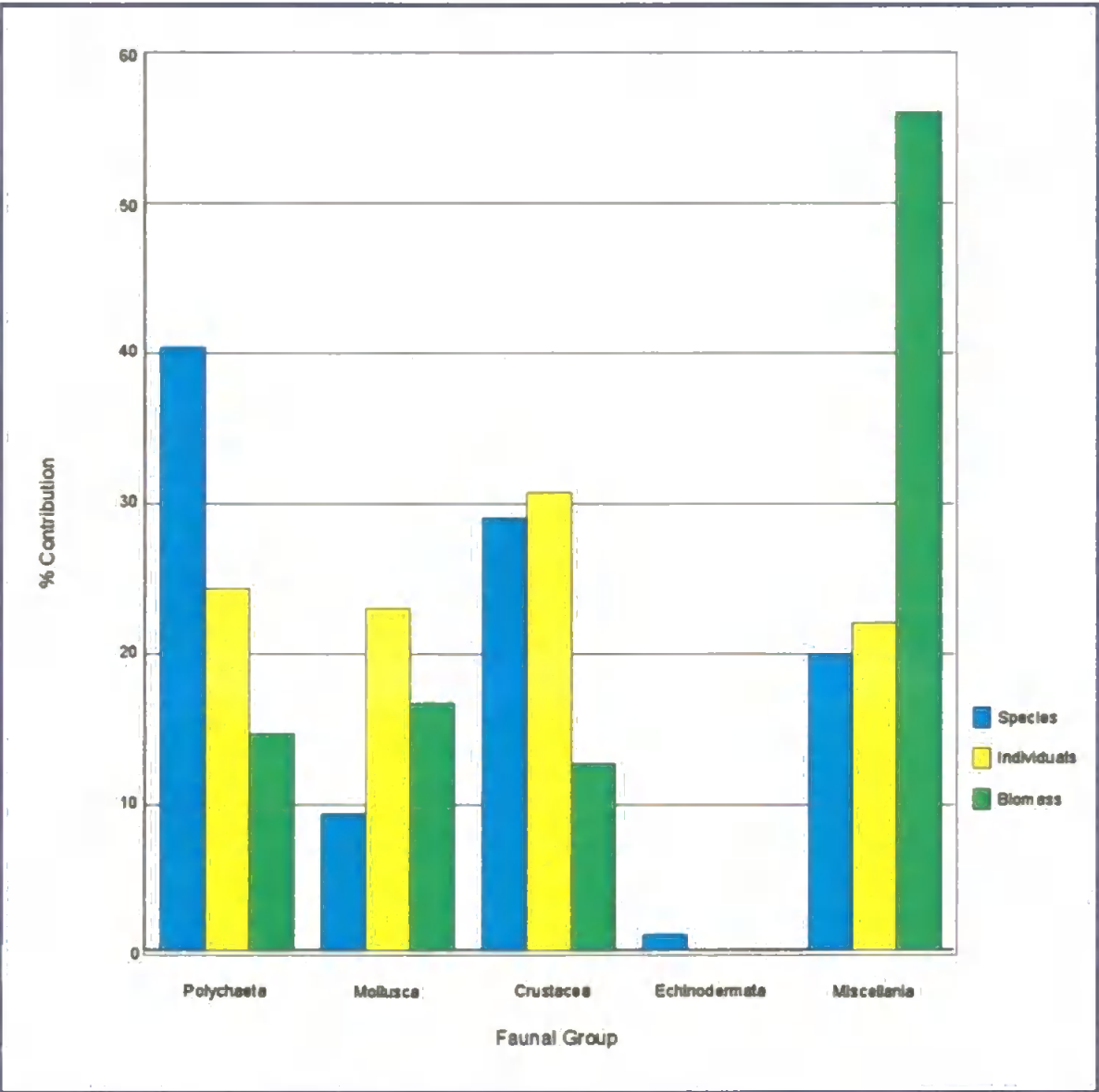


Figure 37. The relative contributions of the main faunal components to species diversity, population density, and the biomass of benthos recorded in the North Nab study area. The abundance of 'miscellania' is not comparable with that for other groups due to difficulties in the quantification of colonial organisms (such taxa have been recorded as 'present' and assigned a nominal value of 1).

A total of 312 species was recorded in the North Nab survey area. Overall, the community was dominated by polychaetes and crustaceans, with hydroids, molluscs and bryozoans prominent faunal components at some stations. Echinoderms were relatively infrequent and minor components of the biomass. Species variety reached a maximum at station 142 where 65 taxa were noted. Low species variety was encountered at stations 80-82 where 1-4 taxa were recorded. The mean number of species recorded per 0.2 m² Hamon Grab sample in the North Nab survey area was 22.9 (30.3 in control stations only). Figure 38 shows a west/east diversity divide, high diversity is observed in stations in the west and central areas with decreasing diversity to the east. This change in species numbers reflects a shift from coarse gravels in western and central stations to clean sand and sandy gravels in the east.

Macrofaunal population density varied considerably throughout the survey area. A maximum of 1423 individuals was discovered at station 144. In contrast, a single specimen was recorded at station 82. The mean population density per 0.2 m² in the North Nab survey area was 192.1. As for species diversity, the fine deposits of the east supported lower population densities than stations in central and western sectors (Figure 39).

Total biomass was highest at station 144, whereas stations 43 and 82 had very low biomass values (<0.01 g). Contributions to the biomass were generally highest from taxa belonging to the phyla Bryozoa, Mollusca and Arthropoda (Crustacea). Biomass appeared to be lowest in the sandy sediments of the east; elevated biomass levels were present in stations to the north west of the dredged area and in samples taken from a group of stations in the south west of the survey grid (Figure 40).

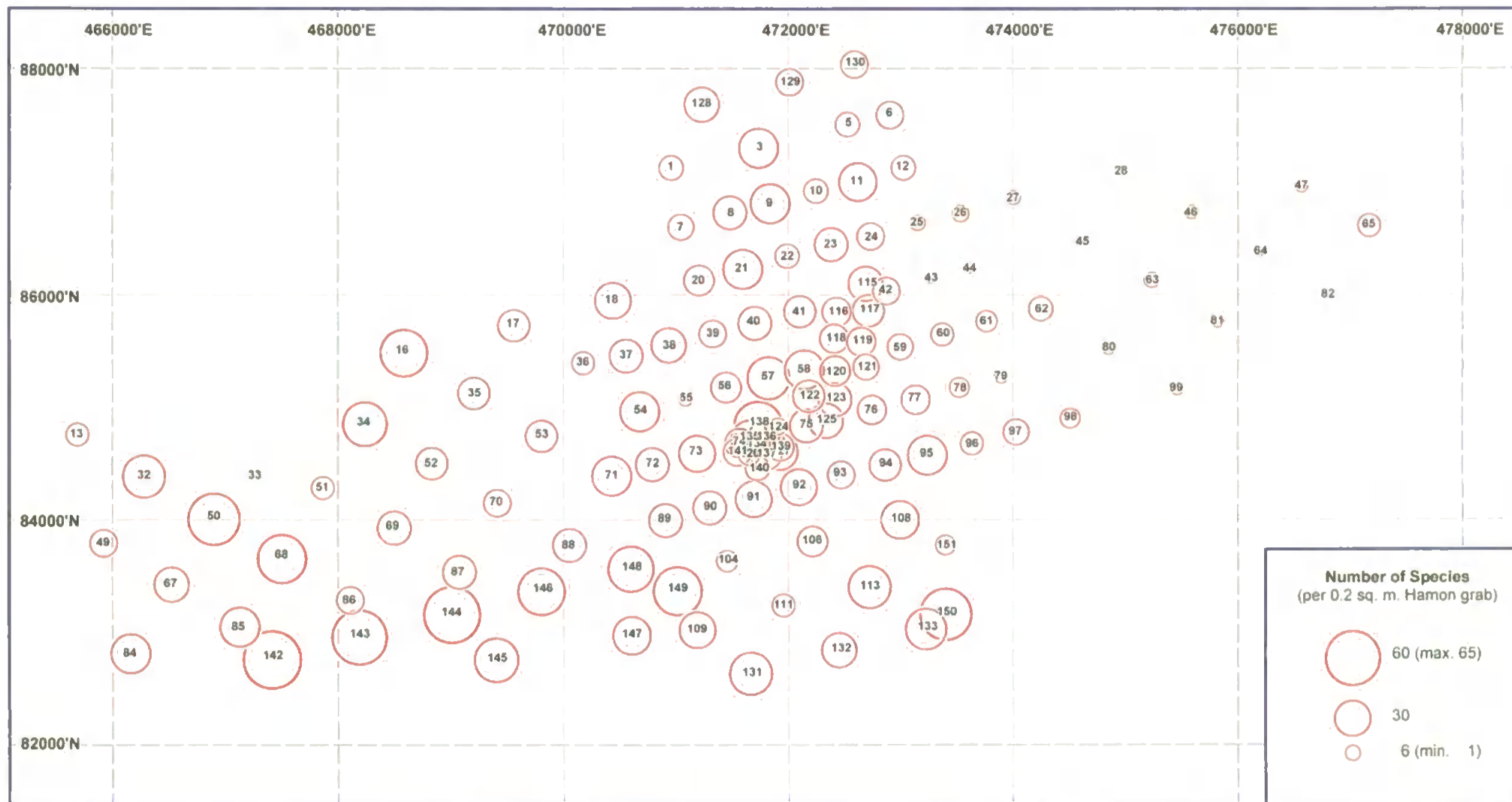


Figure 38. Thematic map showing the number of species of macrofauna (>1mm) per 0.2 m² based on Hamon grab samples at benthic stations sampled in the North Nab survey area.

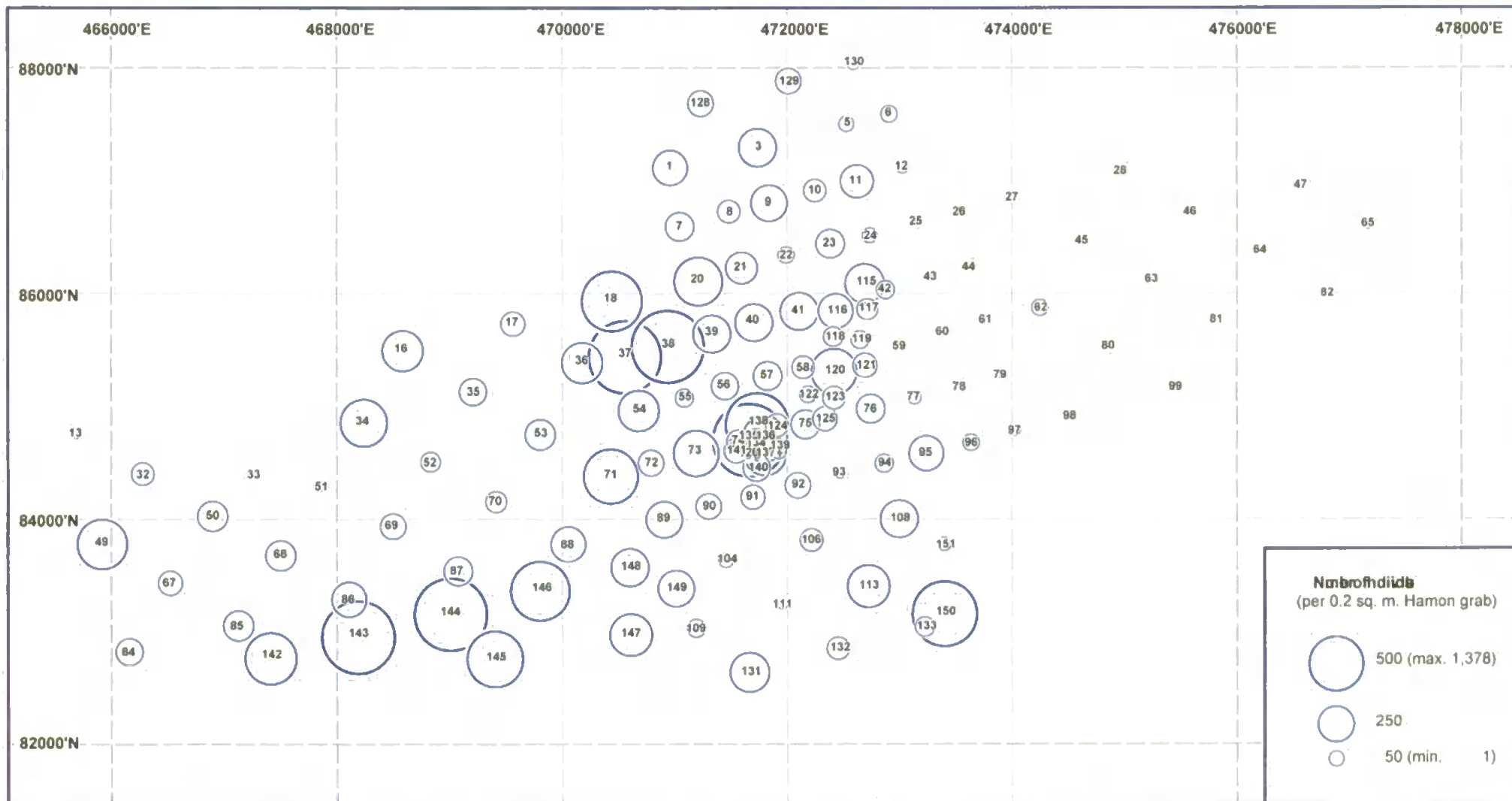


Figure 39. Thematic map showing the number of individuals of macrofauna (>1mm) per 0.2 m² based on Hamon grab samples at benthic stations sampled in the North Nab study area.

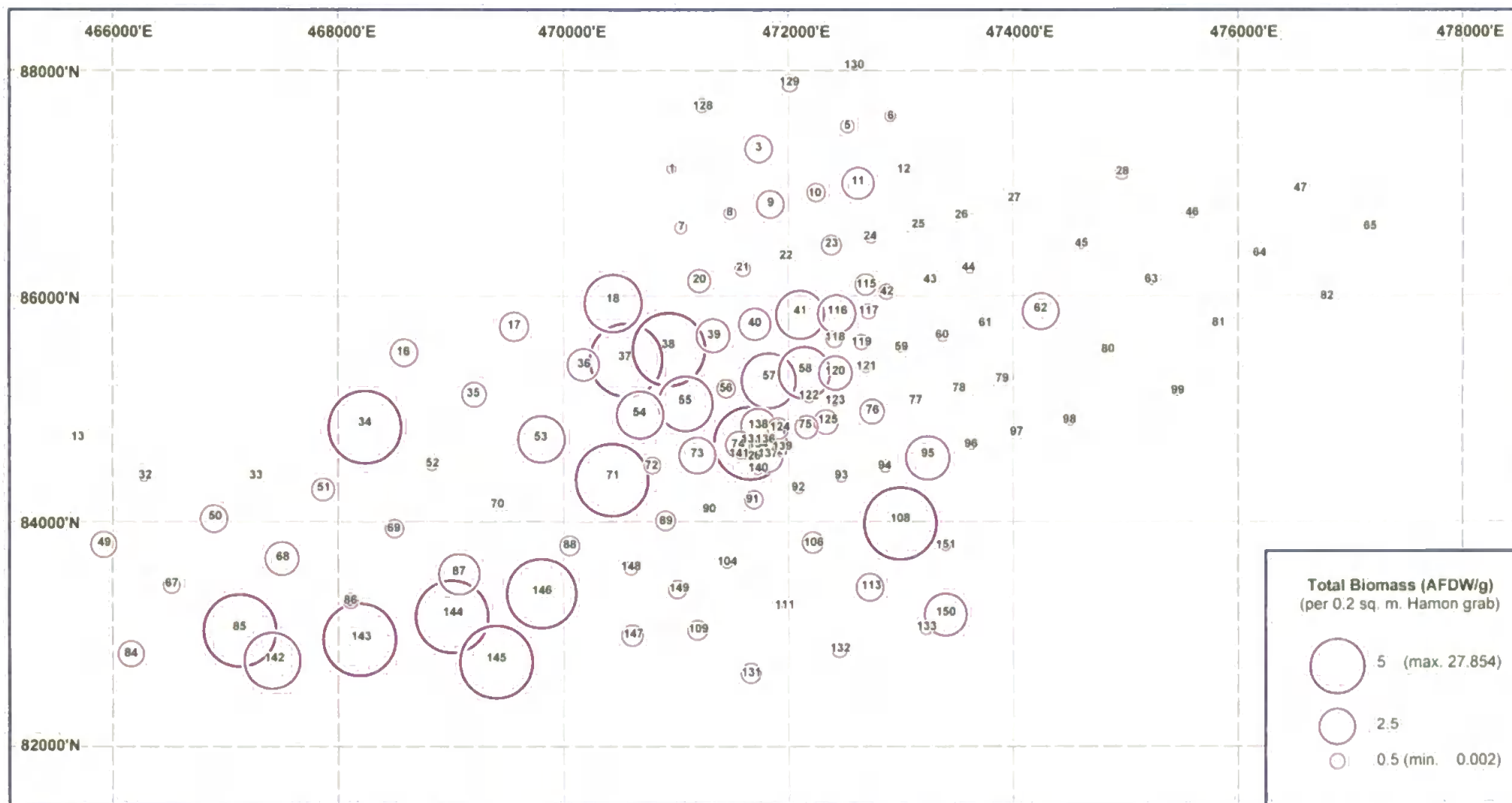


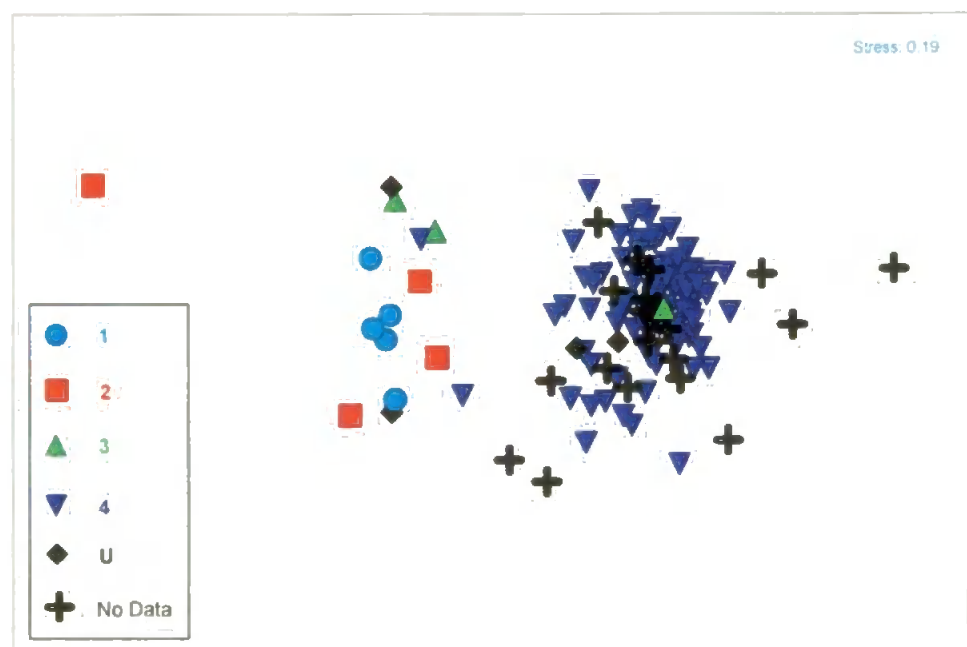
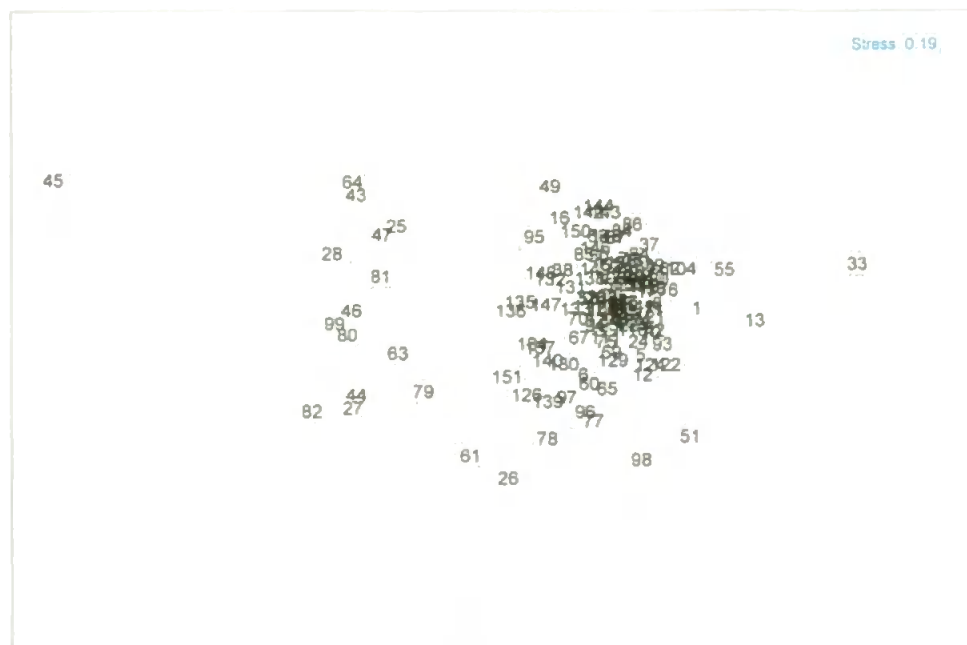
Figure 40. Thematic map showing the biomass (ash free dry weight) in grams per 0.2 m² based on Hamon grab samples at benthic stations sampled in the North Nab study area.

There was a distinct separation of the fauna associated with the coarse sediment groupings (3 & 4) and those taxa characteristic of the finer deposits of the area (sediment groups 1 & 2) (Figure 41). The MDS plot (Figure 41) also highlights the inherent difficulties of sampling very coarse substrata with conventional Hamon Grab deployment, shown by the prevalence of 'no data' symbols (where a sufficient sediment sample could not be obtained after several attempts).

Table 5. Summary of the species that contribute to the similarity of macrofauna recorded within the four sediment groups, identified by multivariate analysis of the samples taken in the North Nab study area. The average abundance, the average similarity, the ratio of average similarity to standard deviation and the % contribution of each species to the similarity of each group is shown. For full data refer to Appendix XXVII.

Group 1						
Average similarity: 38.24						
Species		Av. Abund	Av. Sim	Sim/SD	Contrib%	Cum. %
<i>Nephtys</i>	<i>caeca</i>	4	18.31	1.13	47.89	47.89
<i>Scoloplos</i>	<i>armiger</i>	1.6	7.76	0.6	20.3	68.19
<i>Nephtys</i>	<i>hombergii</i>	1.2	6.48	0.61	16.95	85.13
Group 2						
Average similarity: 16.76						
Species		Av. Abund	Av. Sim	Sim/SD	Contrib%	Cum. %
<i>Travisia</i>	<i>forbesii</i>	2	10.96	0.88	65.36	65.36
<i>Nephtys</i>	<i>caeca</i>	1.75	3	0.41	17.9	83.26
Group 3						
Average similarity: 11.23						
Species		Av. Abund	Av. Sim	Sim/SD	Contrib%	Cum. %
<i>Travisia</i>	<i>forbesii</i>	3	9.89	0.58	88.06	88.06
Group 4						
Average similarity: 29.16						
Species		Av. Abund	Av. Sim	Sim/SD	Contrib%	Cum. %
<i>Distomus</i>	<i>variolosus</i>	39.44	4.64	0.82	15.92	15.92
<i>Crepidula</i>	<i>forficata</i>	50.76	3.69	0.91	12.65	28.56
<i>Pomatoceros</i>	<i>sp.</i>	7.48	2.31	1.12	7.92	36.49
<i>Sabellaria</i>	<i>spinulosa</i>	11.69	2.11	0.98	7.23	43.72
<i>Notomastus</i>	<i>sp.</i>	4.7	2.08	1.05	7.13	50.85

Table 5 shows the contributions to the similarity of species associated with the sediment groups identified by multivariate analysis. It can be seen that the largest sediment group (2) has important contributions from species such as *Distomus variolosus*, *Crepidula forficata*, *Pomatoceros* spp. and *Sabellaria spinulosa*. These species infer the presence of a coarse substrate as these taxa require a stable



attachment surface and have been reported in similar conditions by Vallet *et al.* (2001). *Crepidula fornicata* has also been described as a species that can increase habitat complexity. By removing large quantities of suspended organic matter *C. fornicata* then deposits the filtered material on the seabed as pseudofaeces thus encouraging colonisation by surface deposit feeders. Where this species is found in dense aggregations, as is the case at many sites in the North Nab study area, it can prevent burrowing of certain benthic taxa (Vallet *et al.* 2001). Its shells can also provide habitats for decapods and attachment surfaces for encrusting taxa such as *Sabellaria spinulosa* (de Montaudouin & Sauriau 1999). In contrast sediment groups 1, 3 and 4 are characterised by *Travisia forbesii*, *Nephtys caeca* and *Scoloplos armiger*, all of which are described by JNCC (1997) as being associated with habitats possessing a high sand content.

4.3.5. Coal Pit

4.3.5.1. Substratum Composition

The percentage composition of gravel, sand and silt at each sample location within the survey area is shown in Figure 42. Central stations have a large proportion of gravel with western-most and eastern regions possessing a greater sand and silt component. A group average sorting dendrogram showing the percentage similarity of the sediments (based on the Bray-Curtis similarity of untransformed percentage retained particle size data) at each of the stations sampled in July/August 2000 is shown in Figure 43. The corresponding two-dimensional MDS ordination is also shown (Figure 44). The sediments of the survey area comprised three, closely related sediment types. These are identified as Group 1 sediments (coded blue) comprising sandy gravels, Group 2 sediments (coded red) consisting of sands, and Group 3 sediments (coded green) sands with some gravel and silt. The distribution of these deposit types indicated a large central region,

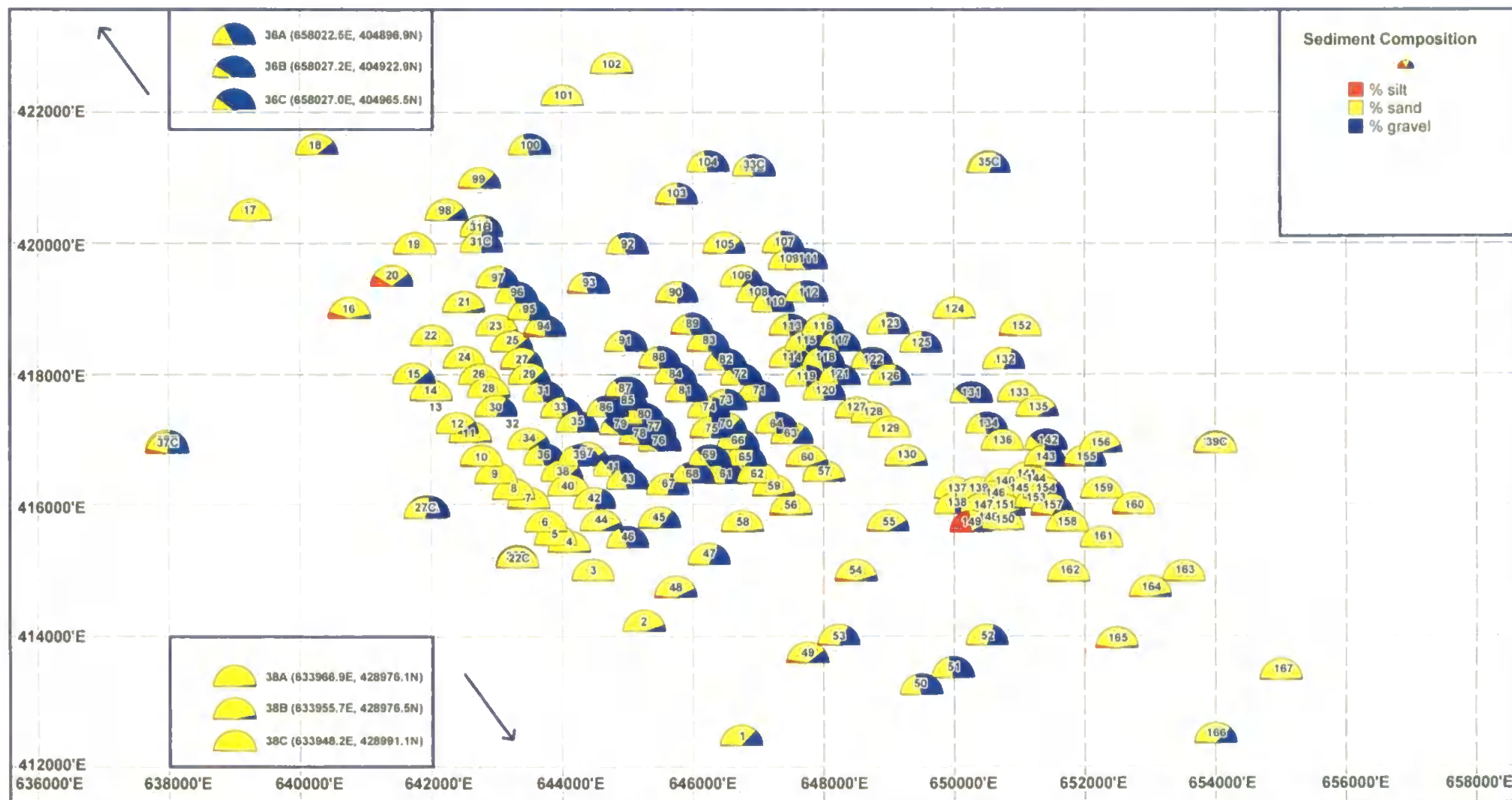
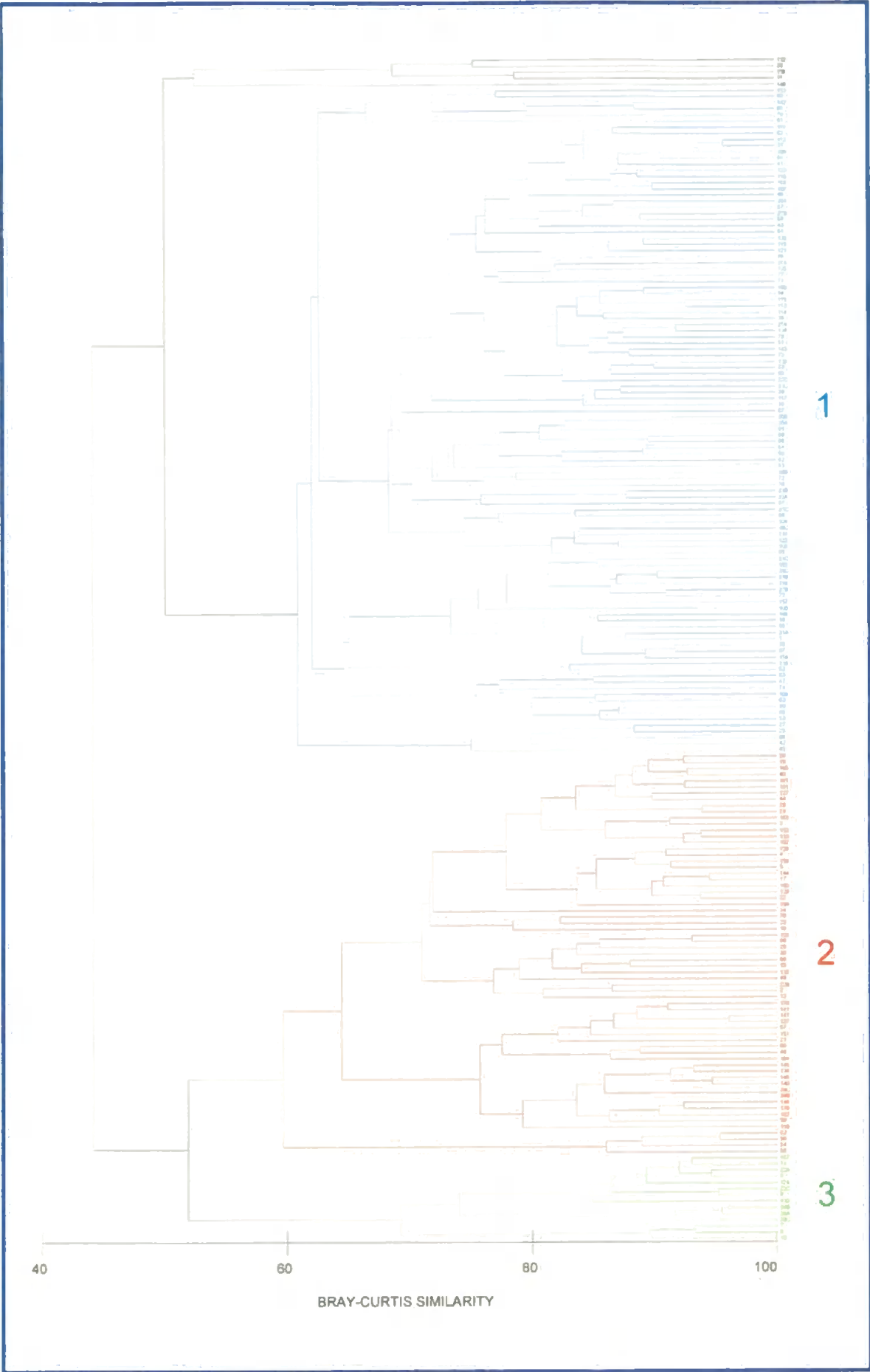
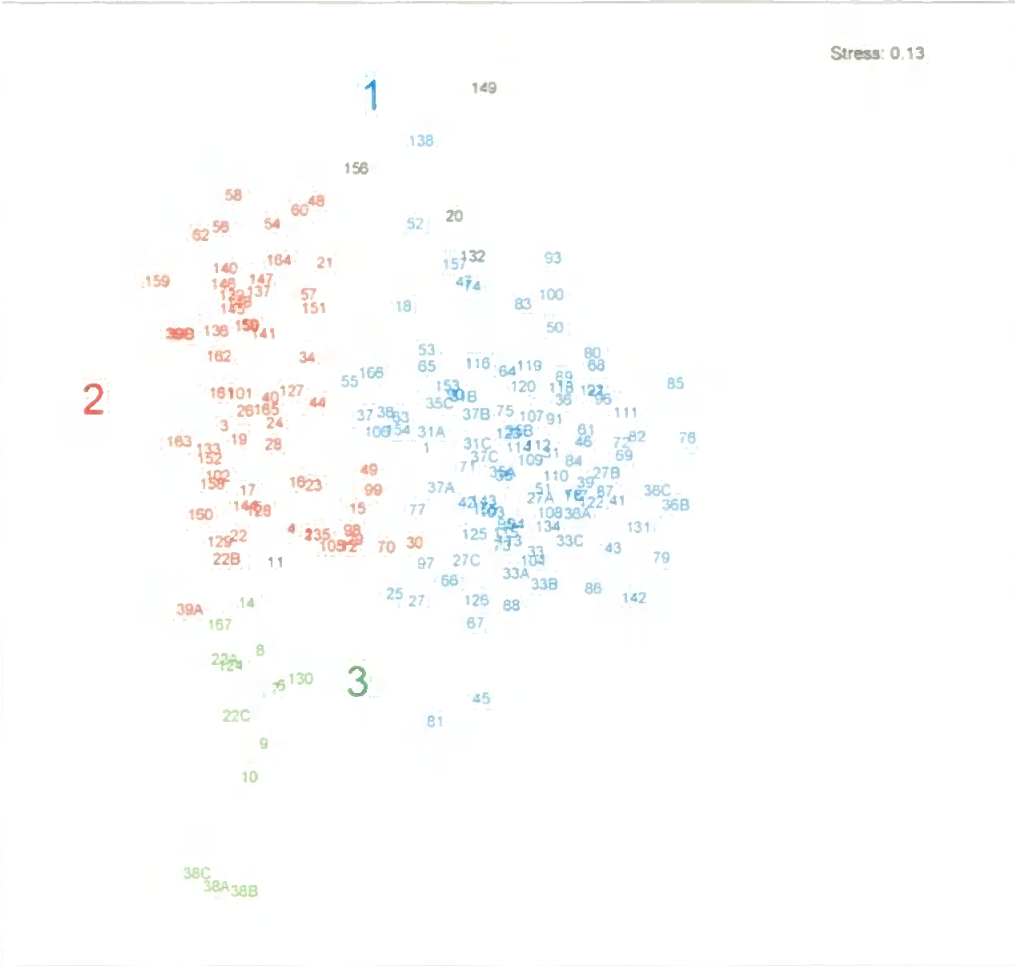


Figure 42. The relative percentage of silt (<0.036mm), sand (0.036-1mm) and gravel (2-32mm) at benthic stations sampled at the Coal Pit study area.





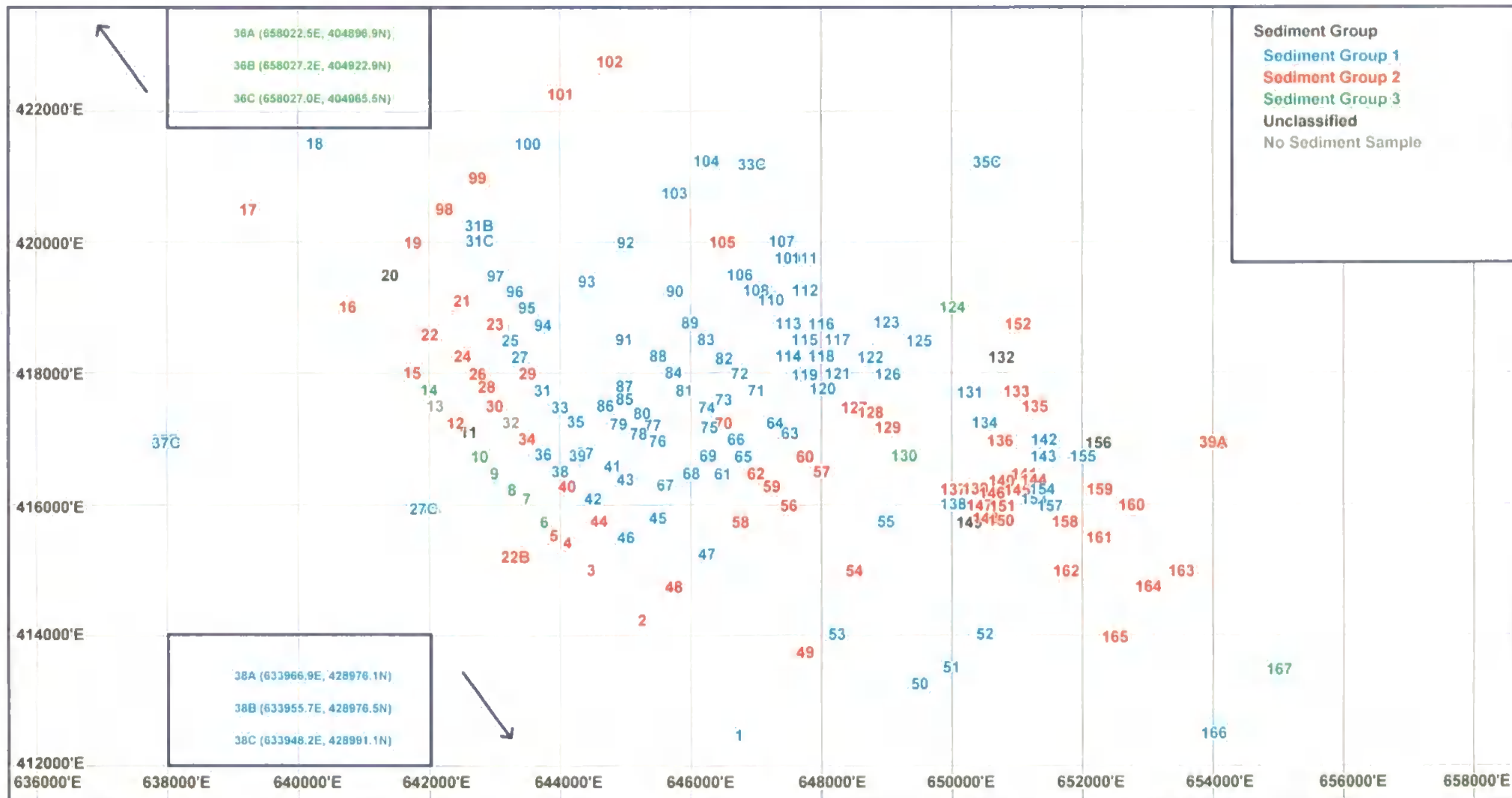


Figure 45. The distribution of sediment groups identified by non-parametric multivariate analysis of particle size composition of benthic stations sampled in the Coal Pit study area.

characterised by group 1 sediments, group 2 substrata occurred in eastern and western stations, and group 3 deposits were noted only in relatively isolated pockets (Figure 45).

4.3.5.2. General Features of Benthic Community Structure

As with many other sites in the North Sea, Polychaeta dominated the benthic macrofauna (Figure 46). They comprised almost 66% of the number of individuals and 32% of the species recorded, but only 13% of the biomass.

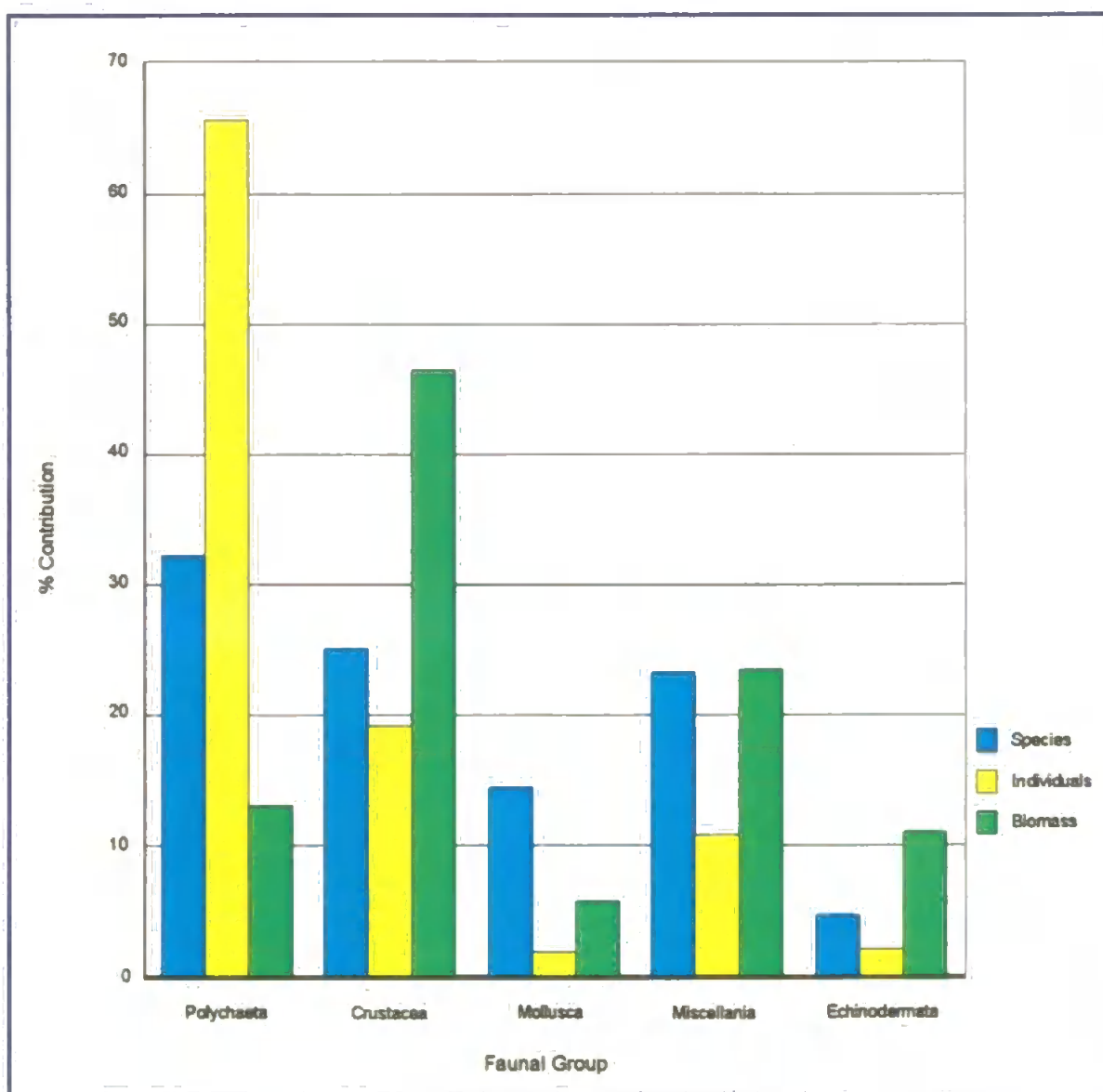


Figure 46. The relative contributions of the main faunal components to species diversity, population density, and the biomass of the benthos recorded in the Coal Pit study area. The abundance of 'miscellania' is not comparable with that for other groups due to difficulties in the quantification of colonial organisms (such taxa have been recorded as 'present' and assigned a nominal value of 1).

Therefore, polychaetes constituted numerous small organisms compared with other faunal groups. In contrast, the Crustacea comprised only 19% of the number of individuals, 25% of the species variety, but 47% of the total biomass, reflecting the relatively large size of many of the Crustacea in the survey area compared with the polychaete worms. Mollusca comprised only a small component of the macrofauna, whilst Echinodermata and miscellaneous groups contributed 11% and 24% respectively of the total biomass recorded from the sediments.

A total of 238 species of macrofauna was recorded from the Coal Pit survey area. Species variety reached a maximum of 69 at station 82. The lowest species count was observed at station 19 where 13 taxa were noted. Population density peaked at station 84, where 1549 individuals were recorded; in contrast, the sandy substratum of station 19 was found to contain 27 individuals. Species richness exhibits a generally constant level throughout the survey area with peaks in the central regions (Figure 47). Diversity appeared to drop, albeit very slightly, in the clean sands of the south east.

Population density was highest in the central regions with low numbers recorded in the extreme southeast coinciding with a shift towards finer sediments (Figure 48). Peaks of population density occurred in stations characterised by coarse deposits and in sediments with elevated silt content. High biomass levels occur at stations with coarse substrata and were attributable mainly to the large population of the barnacle *Balanus crenatus* (Figure 49).

The fauna display a definable relationship with the sediments, along a gradient of changing particle size composition, and therefore, displayed a large degree of overlap (Figure 50). This was a likely reflection of the high uniformity of the

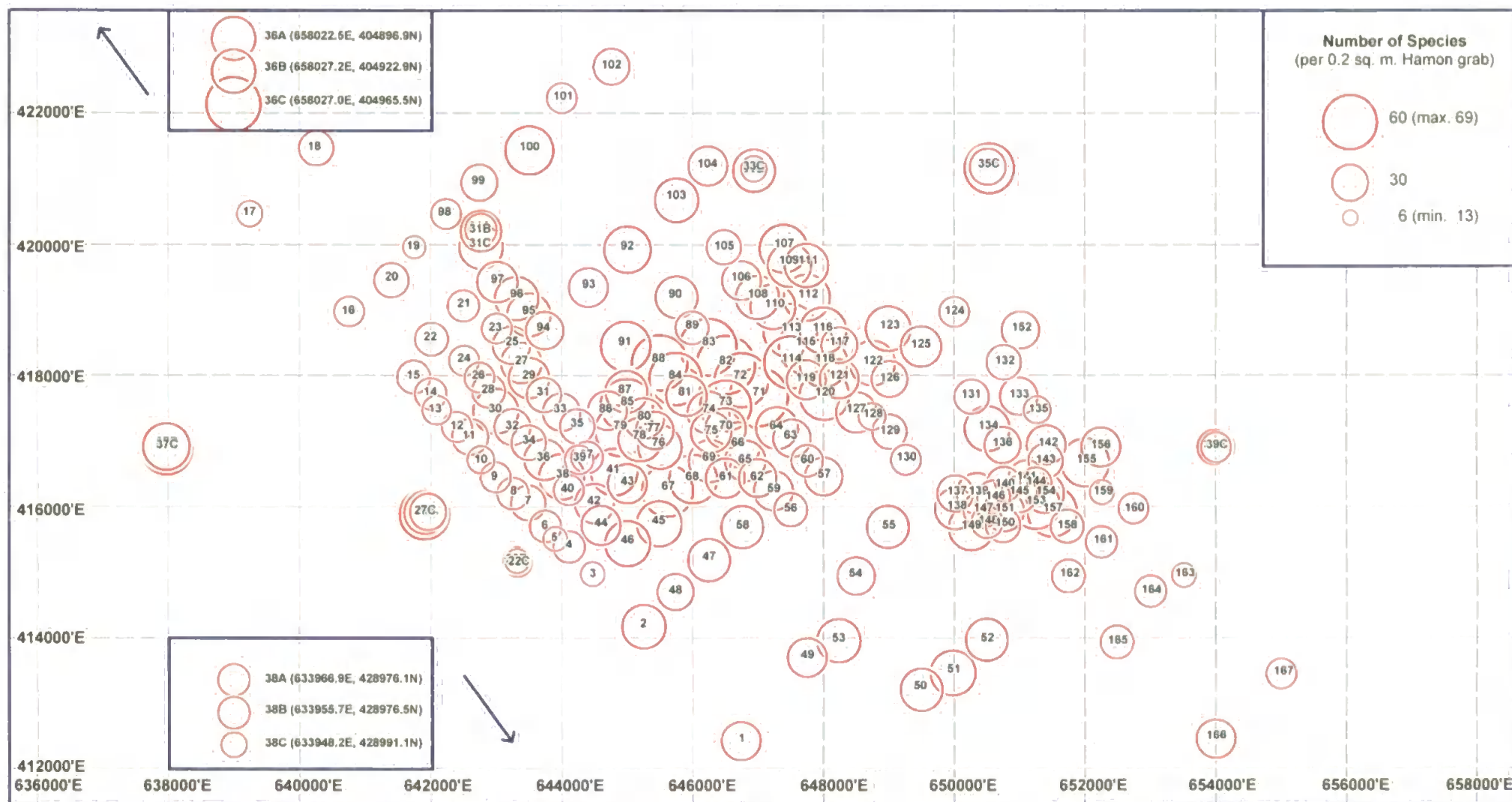


Figure 47. Thematic map showing the number of species of macrofauna (>1mm) per 0.2 m² based on Hamon grab samples at benthic stations sampled in the Coal Pit study area.

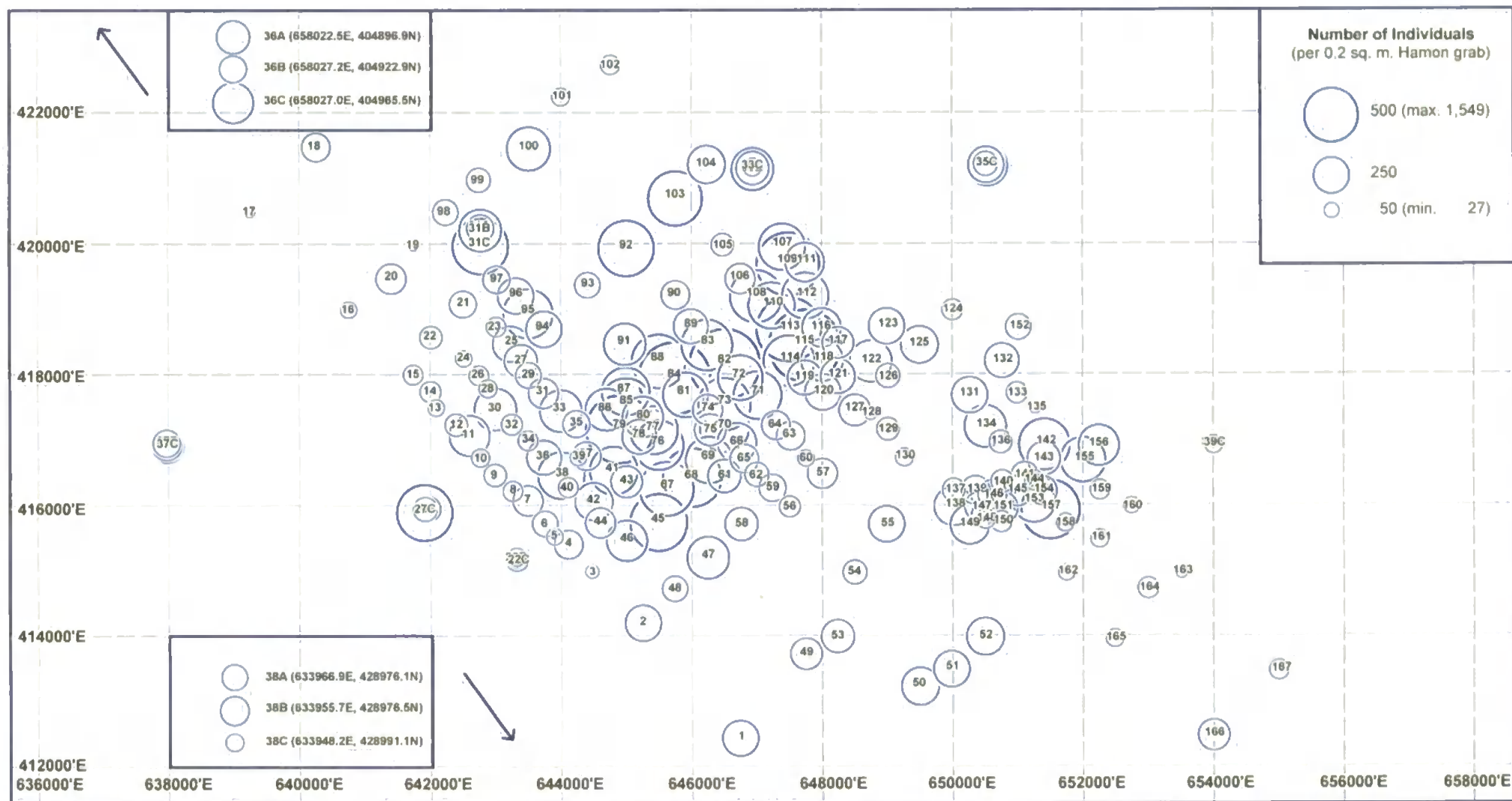


Figure 48. Thematic map showing the number of individuals of macrofauna (>1mm) per 0.2 m² based on Hamon grab samples at benthic stations sampled in the Coal Pit study area.

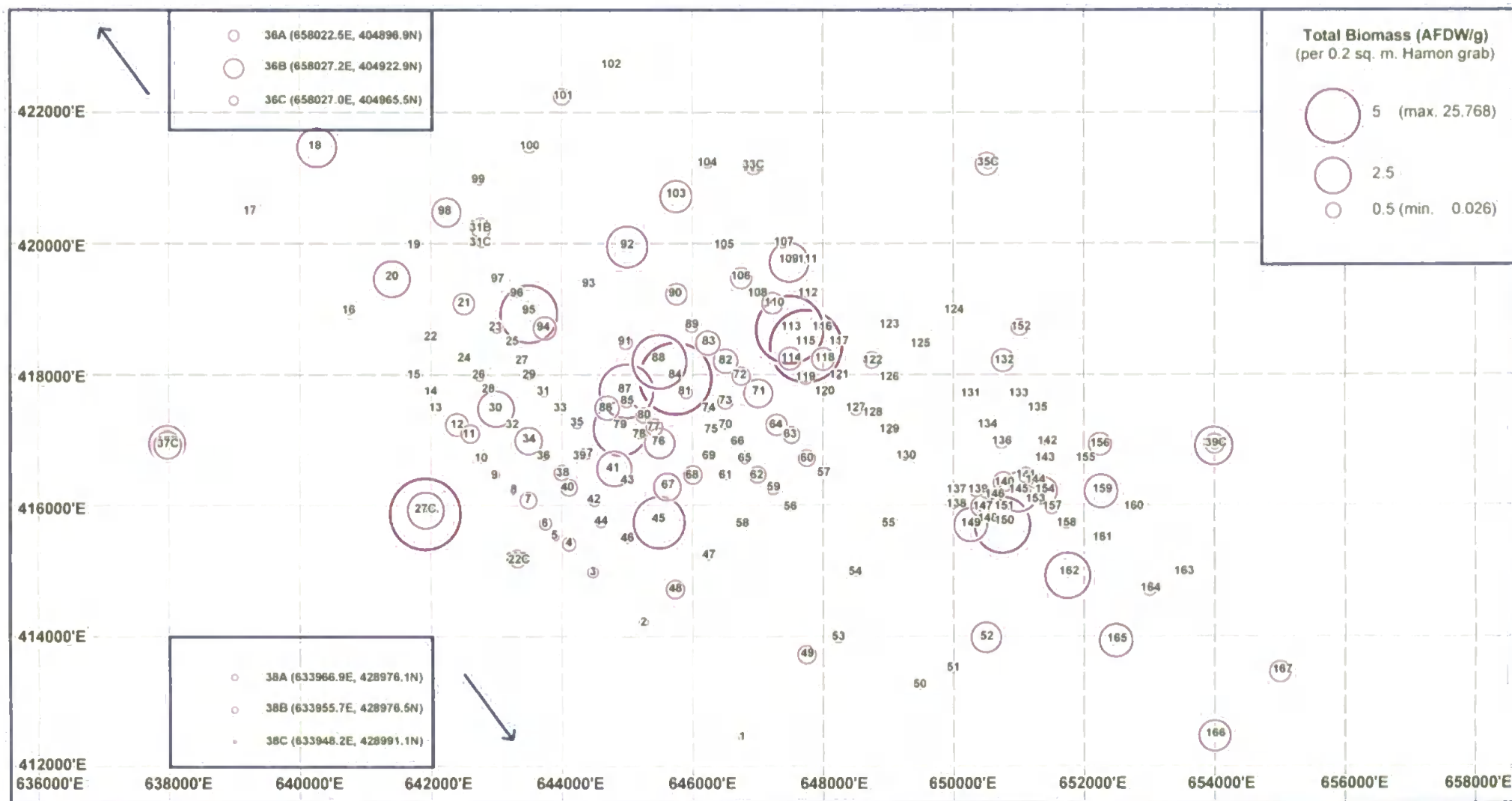


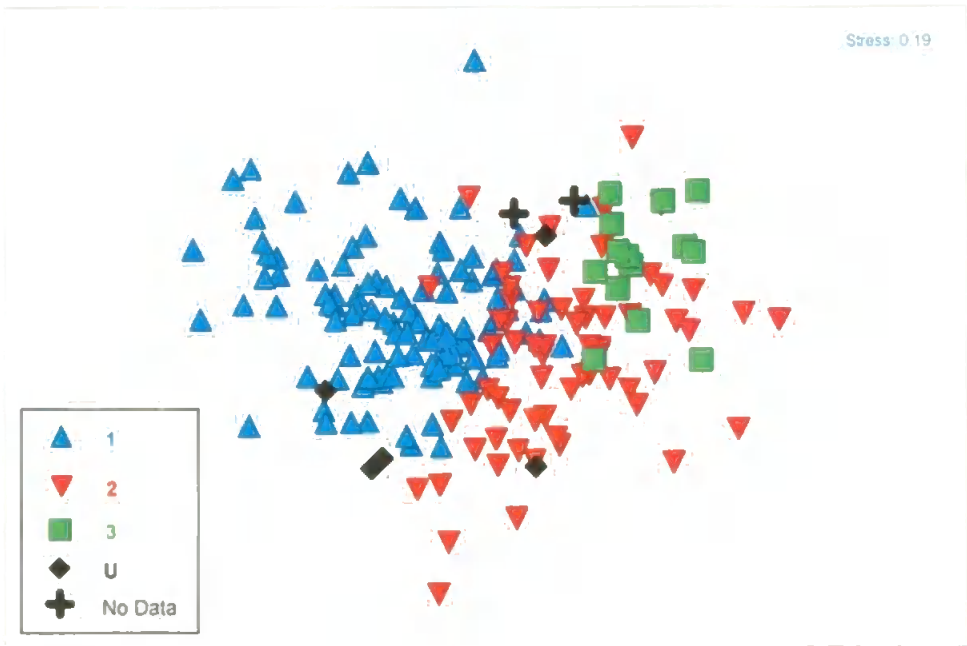
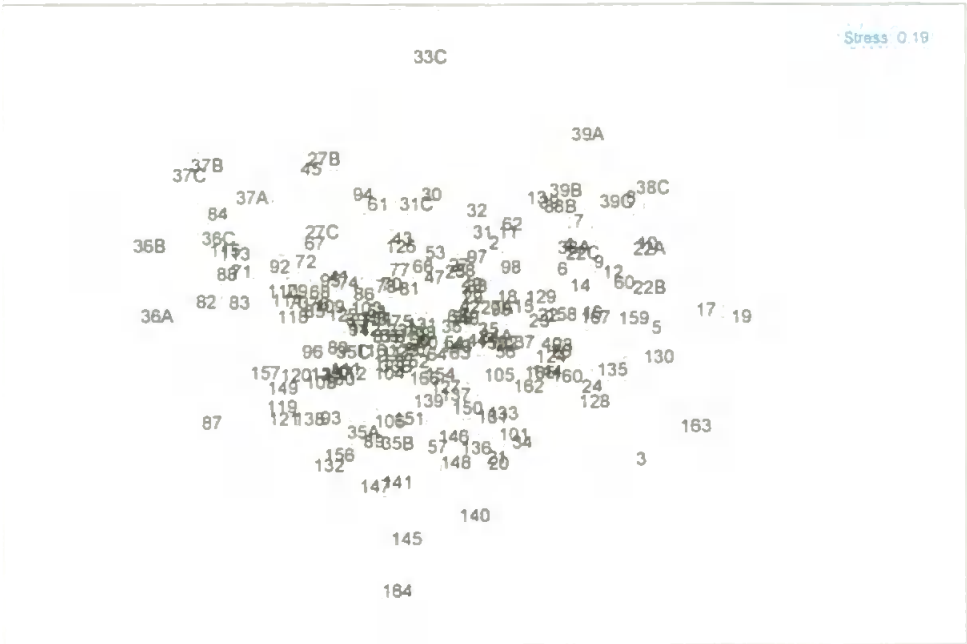
Figure 49. Thematic map showing the biomass (ash free dry weight) in grams per 0.2 m² based on Hamon grab samples at benthic stations sampled in the Coal Pit survey area.

sediments in the area. Differences between the fauna do appear to reflect the differences between the sediment types, although it should be noted that such differences are indeed subtle.

Table 6. Summary of the species that contribute to the similarity of macrofauna recorded within the three sediment groups, identified by multivariate analysis of the samples taken in the Coal Pit study area. The average abundance, the average similarity, the ratio of average similarity to standard deviation and the % contribution of each species to the similarity of each group is shown. For full data refer to Appendix XXVIII.

Group 1						
Average similarity: 45.63						
Species		Av. Abund	Av. Sim	Sim/SD	Contrib%	Cum. %
<i>Ophelia</i>	<i>borealis</i>	65.35	6.5	1.67	14.25	14.25
<i>Spiophanes</i>	<i>bombyx</i>	18.25	3.26	2.09	7.15	21.4
<i>Nematoda</i>	<i>spp.</i>	19.6	3.11	1.75	6.81	28.21
<i>Exogone</i>	<i>hebes</i>	13.2	2.28	1.24	4.99	33.2
<i>Nemertea</i>	<i>spp.</i>	6.48	2.05	2.04	4.49	37.69
Group 2						
Average similarity: 43.33						
Species		Av. Abund	Av. Sim	Sim/SD	Contrib%	Cum. %
<i>Ophelia</i>	<i>borealis</i>	22.4	6.45	2	14.88	14.88
<i>Nephtys</i>	<i>caeca</i>	5.38	3.54	1.6	8.16	23.05
<i>Spiophanes</i>	<i>bombyx</i>	7.25	3.16	1.29	7.28	30.33
<i>Pseudocuma</i>	<i>longicornis</i>	4.31	2.36	1.34	5.45	35.78
<i>Nematoda</i>	<i>spp.</i>	5.31	2.18	1.13	5.02	40.8
Group 3						
Average similarity: 49.94						
Species		Av. Abund	Av. Sim	Sim/SD	Contrib%	Cum. %
<i>Bathyporeia</i>	<i>sp.</i>	14.29	7.59	5.08	15.2	15.2
<i>Ophelia</i>	<i>borealis</i>	13.21	6.38	2.74	12.78	27.98
<i>Nephtys</i>	<i>caeca</i>	10.64	6.17	2.02	12.35	40.33
<i>Spiophanes</i>	<i>bombyx</i>	6.57	5.04	3.42	10.09	50.42
<i>Pseudocuma</i>	<i>longicornis</i>	4.71	2.65	1.31	5.31	55.73

Changes in the composition of the fauna associated with the three sediment groups are revealed by inspection of Table 6. All three groups share similar constituent taxa such as *Ophelia borealis*, *Spiophanes bombyx*, *Nephtys caeca* and *Bathyporeia* sp. However there are differences in the relative abundances of these (and other) species among the samples from the 3 identified sediment types. Taxa such as *O. borealis*, *S. bombyx*, *N. caeca* and *Bathyporeia* sp. that are amongst the most important contributors to the internal similarities of faunal



samples from the three sediment groups, are typical of sand dominated biotopes (JNCC 1997).

4.3.6 Summary

The Shipwash Gabbard site, located off the coast of Suffolk, was largely made up of sandy gravels with silt and rock patches distributed sporadically. The fauna of this region consisted of the amphipod *Ampelisca spinipes*, and the polychaetes *Lumbrineris gracilis* and *Lanice conchilega* in the gravel-dominated deposits; sandy sediments had the polychaetes *Glycera lapidum*, *Pisione remota* and *Ophelia borealis* as major members of the benthic invertebrate assemblage.

Deposits in the Lowestoft study area comprised mainly sand overlying patches of gravel. Key species recorded included the polychaetes *Lagis koreni*, *Ophelia rathkei* and *Glycera rouxii*, and the mysid shrimp *Gasterosaccus spinifer*.

The West Bassurelle site, situated in the eastern English Channel, possessed gravel and sand overlain with significant quantities of shell. Eastern stations had greatly reduced gravel content and consisted largely of fine sands. The benthic community was characterised by many polychaete taxa including *Eteone longa*, *Syllis gracilis* and *Glycera* sp. with the decapod *Galathea intermedia* recorded frequently from coarse deposits.

The North Nab site, characterised by coarse sediment largely comprised of gravels with cobbles and silt at western and central stations and finer, sandy deposits in the east, had a fauna typified by epibenthic taxa such as the gastropod *Crepidula fornicata*, the bryozoan *Flustra foliacea*, polychaetes such as

Pomatoceros sp. and *Sabellaria spinulosa*, and the colonial ascidian *Distomus variolosus*.

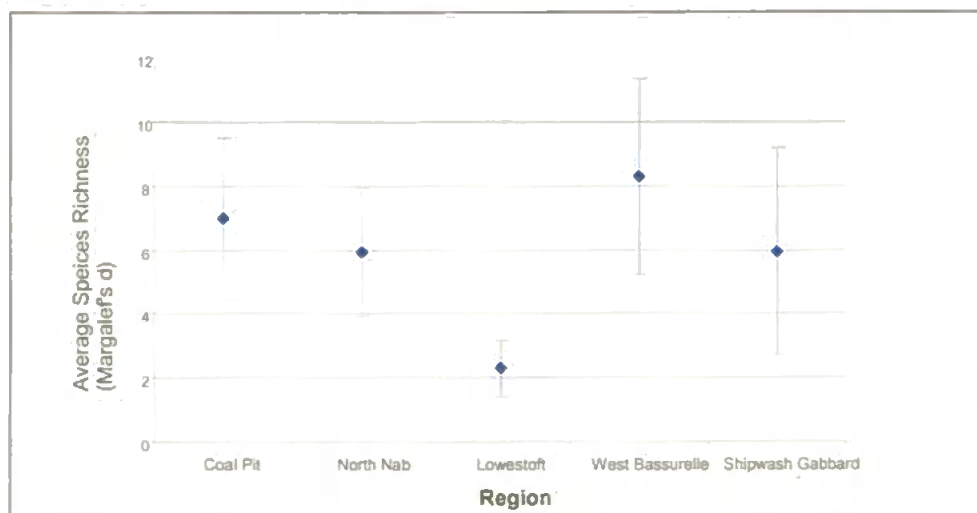
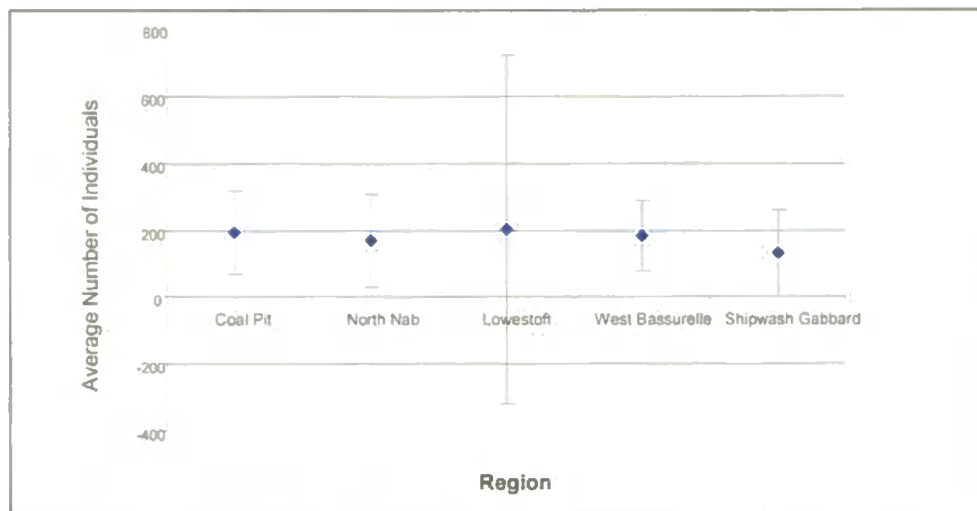
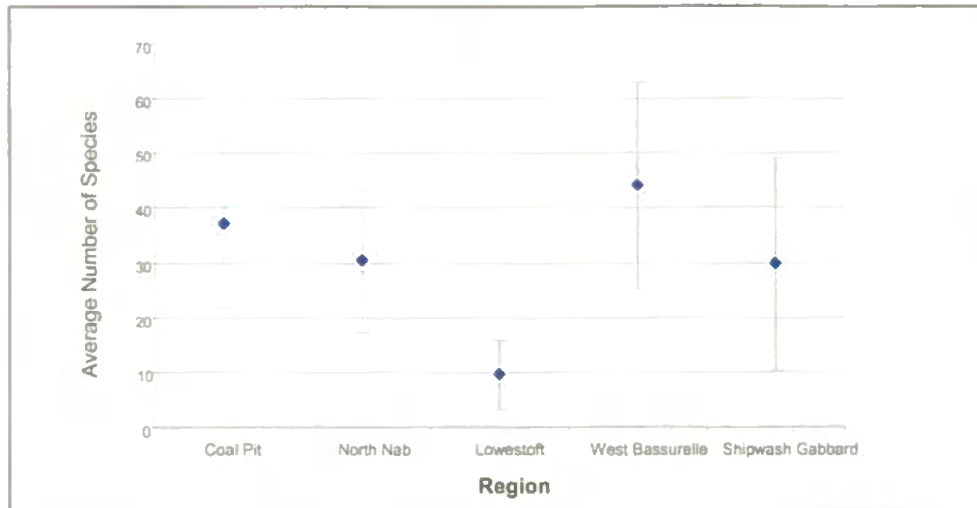
The sediments at Coal Pit were made up of sandy gravels that increased in sand and silt content in both western and eastern directions. The benthos was dominated by *Ophelia borealis*, other polychaetes frequently encountered included *Spiophanes bombyx* and *Notomastus latericeus*.

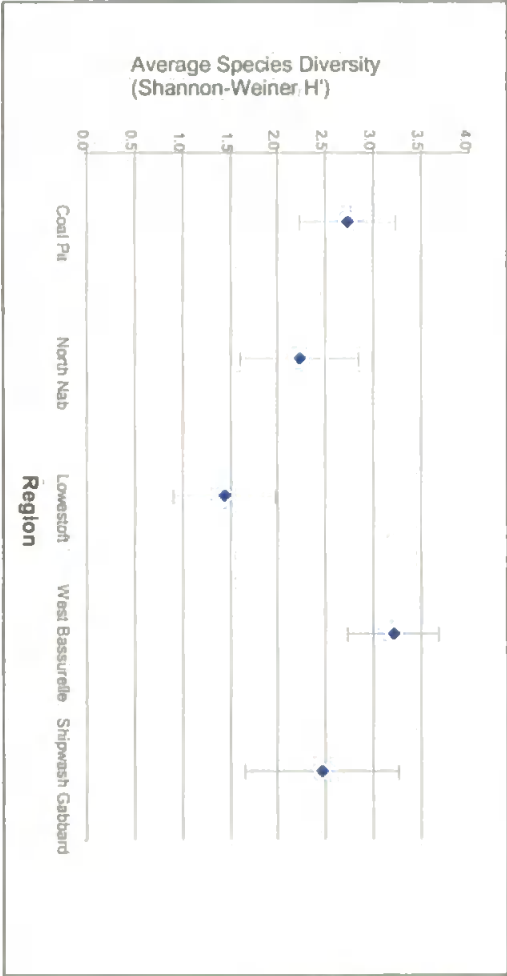
4.4. Results of Regional Comparisons

4.4.1. Univariate Measures

Figures 51 & 52 show plots of the mean number of species and individuals, the mean species richness (Margalef's d), evenness (Pielou's J'), species diversity (Shannon-Weiner H') and dominance (Simpson's $1-\lambda$) at the five study areas. A one-way ANOVA test was carried out for each of the indices listed above (Table 7). Significant differences were observed between the sites, however, a posteriori power analysis revealed that, at the level of sample replication used, many of the tests had insufficient statistical power to detect differences where they occurred and, subsequently, had an increased risk of type II error being encountered. This lack of statistical power of the tests used here means that it is not possible to reject the null hypothesis that there is no significant difference between a range of diversity indices calculated for the five study areas. (for full ANOVA tests refer to Appendix XXIX).

Generally, mean number of species was between 30-40, except for the Lowestoft region where the mean number of species fell to around 10. Population density was at its lowest at Shipwash Gabbard where the mean number of individuals was 130 (excluding sample 33A where approximately 28000 *Lagis koreni* were





recorded). Lowestoft had the highest mean number of individuals, with over 200 per 0.2 m²; similar numbers were recorded in the remaining locations ranging from 168 (North Nab) to 192 (Coal Pit)

Table 7. One-way ANOVA on a range of ecological indices, also shown are the results of a posteriori power analysis.

Index	F	P	Power
Number of species (S(log))	48.38	4.75×10^{-29}	0.23
Number of individuals (N(log))	15.88	2.04×10^{-11}	0.14
Species Richness (d)	34.66	2.06×10^{-22}	0.13
Evenness (J)	15.92	1.93×10^{-11}	0.64
Shannon-Weiner Species Diversity H'	64.68	4.91×10^{-36}	0.28
Simpson's dominance (1-λ)	38.56	2.06×10^{-24}	0.06

Species richness, evenness and diversity were highest at West Bassurelle; conversely, dominance was lowest at this location, possibly as a reflection of the environmental stability of the area. Coarse, heterogeneous deposits provide attachment surfaces for epifaunal taxa, thus increasing further the habitat complexity. Relatively deep water at this location, and the lack of anthropogenically-derived impacts such as aggregate extraction, waste dumping and heavy fishing effort, are all likely to be contributory factors in the development of a diverse community. In contrast, the Lowestoft study area exhibited the lowest species richness and diversity, and displayed the highest dominance. The latter combination is a possible indication of a dynamic, unstable environment. The deposits of the region comprised sandy gravels and have been reported by Kenny *et al.* (1991) to be subject to a high degree of natural disturbance, mainly in the form of scouring by sands shifting in the powerful tidal conditions. Such disturbances preclude the development of significant colonies of epifaunal

species, therefore, maintaining the benthos in an intermediate successional state (Boesch & Rosenberg 1981).

4.4.2. Multivariate analysis

An ANOSIM test revealed the global value of R for the data of all areas was >0.775 regardless of the level of transformation employed, this indicates that a null hypothesis of; there are no differences between the benthic assemblages of the 5 study areas, is rejected, at the 0.1% (0.001) level (for data see Appendix XXX). illustrates the test comparisons between the five locations. All study sites can be seen to be significantly different from one another in terms of their macrobenthic assemblages (based on genera). In all cases a significance level of 0.1% was observed. The R -values shown for the site comparisons give an absolute measure of how separated the benthic assemblages of the five locations are from one another. It is apparent, at all levels of transformation, from the large R -values that Coal Pit is well separated from both West Bassurelle ($R = >0.9$) and North Nab ($R = >0.8$). Comparison between the Coal Pit assemblages and those of Lowestoft and Shipwash Gabbard gave R -values of >0.85 and >0.86 . The West Bassurelle study area is clearly distinct from all North Sea regions with calculated R -values in the range 0.775-0.96. Recorded genera from the North Nab region exhibit a small degree of overlap with Lowestoft and Shipwash Gabbard taxa but remain clearly distinguished ($R = >0.645$ & >0.454 respectively). The Lowestoft and Shipwash Gabbard faunas, whilst readily identifiable as distinct assemblages, have the lowest pairwise R -value (0.299-0.415) and therefore exhibit a greater degree of community overlap than the other regions across the range of transformation used.

Table 8. Average abundance and percentage contribution to the internal similarity of the top five ranked genera recorded from each of the study areas.

Coal Pit		Average Similarity: 36.03
Genera	Average abundance	Contribution to similarity %
<i>Ophelia</i>	37.22	12.69
<i>Spiophanes</i>	8.44	9.08
<i>Nephtys</i>	5.96	6.74
<i>Bathyporeia</i>	9.78	5.85
<i>Exogone</i>	10.93	4.89
West Bassurelle		Average Similarity: 39.25
Genera	Average abundance	Contribution to similarity %
<i>Eteone</i>	18.68	10.95
<i>Glycera</i>	9.1	7.88
<i>Syllis</i>	7.6	6.44
<i>Galathea</i>	14.84	5.07
Spionidae (unidet.)	5.23	4.45
North Nab		Average Similarity: 31.35
Genera	Average abundance	Contribution to similarity %
<i>Distomus</i>	50.65	19.51
Serpulinae (Juv)	7.29	10.21
<i>Crepidula</i>	22.82	8.56
<i>Sabellaria</i>	7.12	6.91
Capitellidae (unidet.)	5	6.08
Lowestoft		Average Similarity: 20.44
Genera	Average abundance	Contribution to similarity %
<i>Nephtys</i>	1.63	20.03
<i>Ophelia</i>	2.42	18.94
<i>Gastrosaccus</i>	2.84	14.29
<i>Lagis</i>	93.63	13.15
<i>Glycera</i>	0.92	7.76
Shipwash Gabbard		Average Similarity: 22.91
Genera	Average abundance	Contribution to similarity %
<i>Glycera</i>	4.47	17.07
<i>Lumbrineris</i>	5.95	6.97
<i>Ampelisca</i>	11.79	5.81
<i>Lanice</i>	5.71	3.94
<i>Pisone</i>	4.18	3.58

Table 8 shows the results of the SIMPER analysis performed on the genera for all locations (for full results refer to Appendix XXXI). Samples obtained from the Coal Pit region have an internal similarity of 36%; taxa responsible for the between sample similarity include *Ophelia*, *Spiophanes* and *Nephtys*. Comparable within-site similarity was shown for samples collected from West Bassurelle, however, this site had a somewhat different community with the polychaetes *Eteone*, *Glycera*, *Syllis* and the decapod *Galathea* being important genera. The macrobenthos of the North Nab region also exhibited a similarity of over 30% between samples. Again, the community was in contrast with those of Coal Pit and West Bassurelle both in terms of important taxa (*Distomus*, Serpulinae, *Crepidula* and *Sabellaria*) and the life-history traits of the key genera concerned. The North Nab community appeared to be dominated by epifaunal, filter-feeding invertebrates as opposed to the largely infaunal, mixed feeding guild communities of the other regions. Lower internal similarities were noted at the Lowestoft and Shipwash Gabbard sites (20% and 22% respectively). The Lowestoft site was characterised by polychaetes including *Nephtys*, *Ophelia*, *Lagis* and *Glycera*, and the mysid shrimp *Gastrosaccus*. Important components of the community at the Shipwash Gabbard location included the amphipod *Ampelisca* and polychaetes such as *Glycera*, *Lumbrineris*, *Lanice* and *Pisione*.

Table 9. Pairwise comparisons of each of the five study locations and the average dissimilarity between each pair of sites. Average abundance and percentage contribution to the dissimilarity between each of the study areas made by key taxa is also shown.

Coal Pit & Lowestoft		Average dissimilarity: 88.16	
Genera	Coal Pit Average abundance	Lowestoft Average abundance	Contribution %
<i>Ophelia</i>	37.22	2.42	5.58
<i>Lagis</i>	1.3	93.63	4.65
<i>Bathyporeia</i>	9.78	0.76	3.65
West Bassurelle & Lowestoft		Average dissimilarity: 96.04	
Genera	West Bassurelle Average abundance	Lowestoft Average abundance	Contribution %
<i>Eteone</i>	18.68	0.34	4.97
<i>Lagis</i>	0	93.63	3.85
<i>Galathea</i>	14.84	0.16	3.07
North Nab & Lowestoft		Average dissimilarity: 94.72	
Genera	North Nab Average abundance	Lowestoft Average abundance	Contribution %
<i>Distomus</i>	50.65	0	9.88
<i>Sabellaria</i>	7.12	69.26	5.04
<i>Crepidula</i>	22.82	0	4.97
Coal Pit & West Bassurelle		Average dissimilarity: 85.84	
Genera	Coal Pit Average abundance	West Bassurelle Average abundance	Contribution %
<i>Ophelia</i>	37.22	0.06	4.28
<i>Eteone</i>	2.74	18.68	2.79
<i>Galathea</i>	0.04	14.84	2.43
Coal Pit & North Nab		Average dissimilarity: 89.78	
Genera	Coal Pit Average abundance	North Nab Average abundance	Contribution %
<i>Distomus</i>	0	50.65	6.11
<i>Ophelia</i>	37.22	0	5
<i>Crepidula</i>	0	22.82	3.23
West Bassurelle & North Nab		Average dissimilarity: 86.56	
Genera	West Bassurelle Average abundance	North Nab Average abundance	Contribution %
<i>Distomus</i>	0.1	50.65	5.81
<i>Eteone</i>	18.68	0	4.05
<i>Crepidula</i>	0	22.82	3.1

Coal Pit & Shipwash Gabbard		Average dissimilarity: 85.42	
Genera	Coal Pit Average abundance	Shipwash Gabbard Average abundance	Contribution %
<i>Ophelia</i>	37.22	0.82	5.11
<i>Bathyporeia</i>	9.78	0	3.11
<i>Exogone</i>	10.93	0.26	2.71
West Bassurelle & Shipwash Gabbard		Average dissimilarity: 85.21	
Genera	West Bassurelle Average abundance	Shipwash Gabbard Average abundance	Contribution %
<i>Eteone</i>	18.68	0	4.43
<i>Galathea</i>	14.84	0.11	2.82
<i>Syllis</i>	7.6	0	2.65
North Nab & Shipwash Gabbard		Average dissimilarity: 88.51	
Genera	North Nab Average abundance	Shipwash Gabbard Average abundance	Contribution %
<i>Distomus</i>	50.65	0	7.93
<i>Crepidula</i>	22.82	0.05	4.06
Serpulinae	7.29	0	3.1
Lowestoft & Shipwash Gabbard		Average dissimilarity: 87.98	
Genera	Lowestoft Average abundance	Shipwash Gabbard Average abundance	Contribution %
<i>Lagis</i>	93.63	9.95	7.08
<i>Sabellaria</i>	69.26	10.63	4.52
<i>Glycera</i>	0.92	4.47	3.58

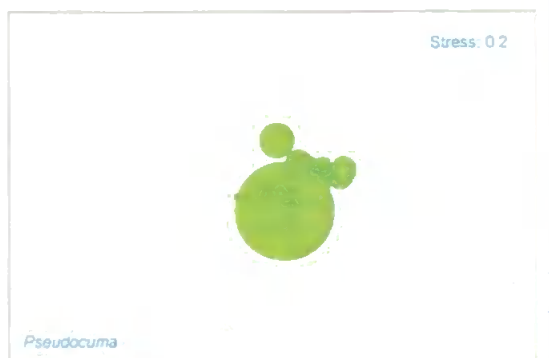
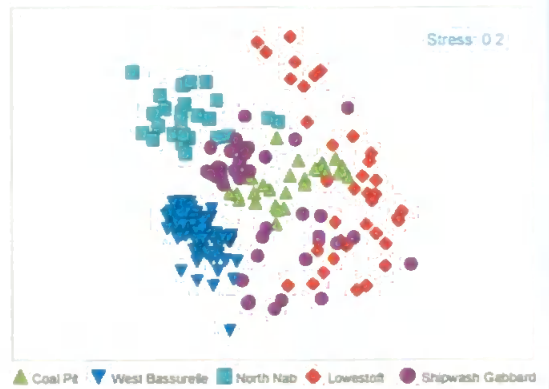
Comparison of Coal Pit and West Bassurelle showed that the average dissimilarity between the two sample groups was 85.8, with *Ophelia*, *Eteone* and *Galathea* being the largest contributors to the dissimilarity (Table 9 see Appendix XXXI for full data). The average dissimilarity between Coal Pit and North Nab was 89.8, indicating that the assemblages of the two areas were very different. The major contributors to this difference between Coal Pit and North Nab included *Distomus*, *Ophelia* and *Crepidula*. Average dissimilarity between Coal Pit and Lowestoft was high (88.2). Taxa largely responsible for this disparity included *Ophelia*, which is abundant at Coal Pit, and *Lagis*, which was found at high average densities in the

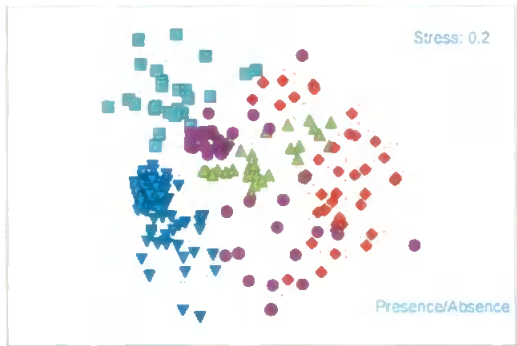
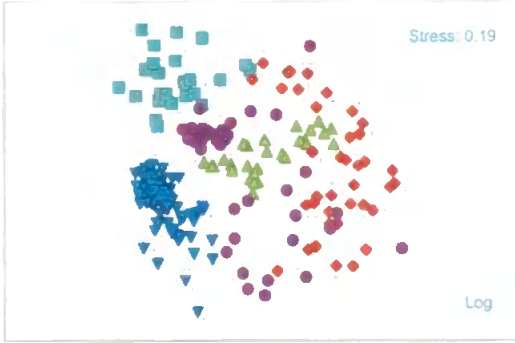
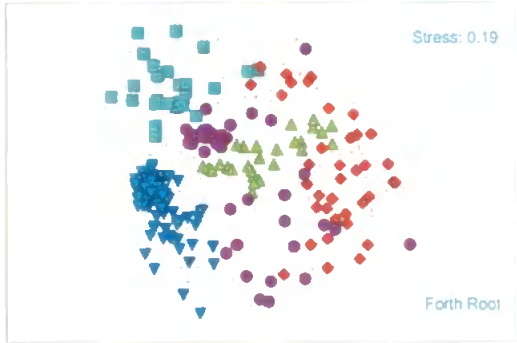
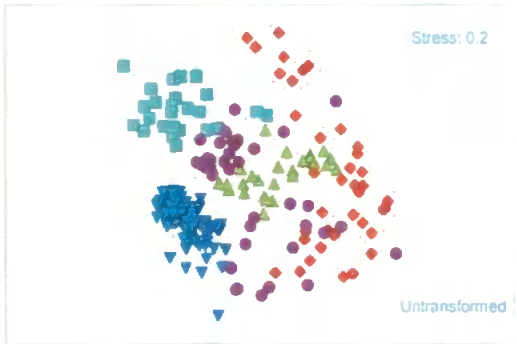
Lowestoft region. The two English Channel locations also exhibited a large average dissimilarity (86.6), the differences being attributed largely to the presence of *Distomus* and *Crepidula* at North Nab, and *Eteone* and *Galathea* at West Bassurelle. A value of 96% average dissimilarity between West Bassurelle and Lowestoft indicated very little correspondence between the genera recorded at the two locations. *Eteone* and *Galathea* were encountered regularly in the deposits of West Bassurelle but infrequently at Lowestoft; *Lagis* was totally absent from samples obtained from West Bassurelle but made an important contribution to the fauna of the Lowestoft site. Lowestoft was also found to be highly dissimilar to North Nab (the other English Channel study area) where average dissimilarity between the two sites was 94.7. *Distomus* and *Crepidula* were not recorded at Lowestoft, but were abundant at North Nab; conversely, *Lagis* was abundant at Lowestoft but absent at North Nab. Comparison of the genera recorded at Shipwash Gabbard and Coal Pit revealed an average dissimilarity of 85.4. This differentiation is best explained by high numbers of *Ophelia* at Coal Pit, with *Ampelisca* being noted frequently in the sediments of Shipwash Gabbard. An average dissimilarity of 85.2 was found when Shipwash Gabbard genera were compared to the samples from West Bassurelle. Major sources of variation between the faunas of the two regions can be attributed to *Ampelisca* at the North Sea site, and high abundances of *Eteone* and *Galathea* at the Channel location. Shipwash Gabbard was found to have an average dissimilarity of 88.5 when compared with North Nab (English Channel). The main contributors to this value were *Distomus*, *Crepidula* and serpulids, frequently found at North Nab, and *Sabellaria* that was found at both locations. Finally, the remaining North Sea sites, Lowestoft and Shipwash Gabbard, had an average dissimilarity of 88%. The major contributing genera to this value were *Lagis* and *Sabellaria*, found at both locations

but in higher densities at Lowestoft, and *Ampelisca*, that was more frequent in the samples taken from Shipwash Gabbard.

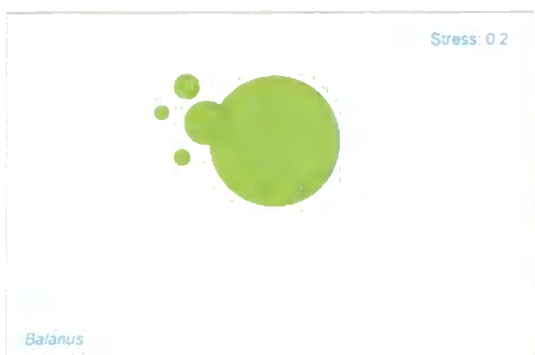
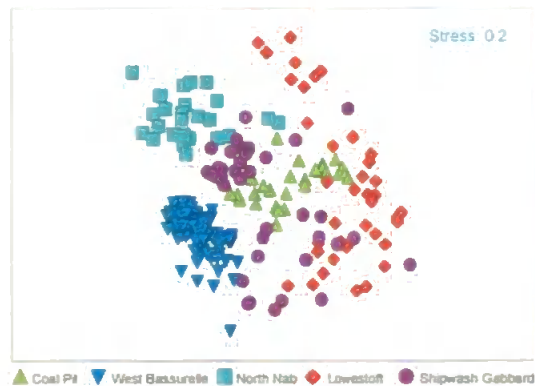
Subjecting the data to a range of transformations had an apparently negligible effect on the separation of the five study areas shown in the MDS plots (Figure 53), this is confirmed by the multivariate dispersion (MVDISP) indices shown in Appendix XXXII. Separation of the samples into their respective areas, even when severe transformations such as presence/absence were employed, indicated that the species assemblages characterising each location were readily definable to that particular locality. This pattern was shown was by ANOSIM tests (Appendix XXX), using data subjected to the following transformations: untransformed, square root, 4th root, log, and presence/absence. These tests revealed that the communities of each location remained identifiable when the more abundant species were given more weighting (no transformation), when the intermediate and rare taxa were given more influence (square root, 4th root and log transformations), and when overall species composition (excluding colonial taxa) was compared (presence/absence).

Figures 54-57 highlight some of the key genera identified in the SIMPER analysis and their relative abundance at the stations. It is evident that certain taxa have an affinity with a particular site. For example, *Ophelia*, *Pseudocuma* and *Bathyporeia* occurred at Coal Pit, *Crepidula*, *Balanus* and *Distomus* at North Nab, *Lanice* and *Lumbrineris* at Shipwash Gabbard, and *Gastrosaccus* at the Lowestoft site. The bivalve *Glycymeris* and the decapod *Galathea* featured in samples obtained from West Bassurelle. In contrast, taxa such as *Nephtys*, *Spiophanes*, *Scalibregma* and *Glycera* were widely recorded across the different study areas.





- ▲ Coal Pit
- ▼ West Bassurelle
- North Nab
- ◆ Lowestoft
- Shipwash Gabbard



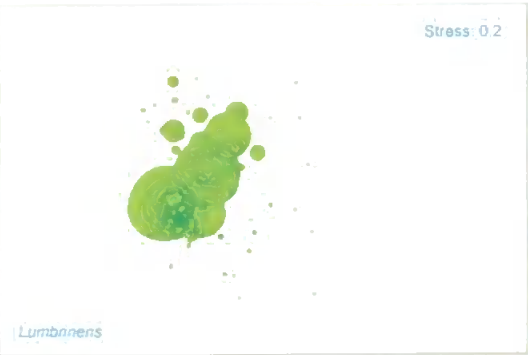
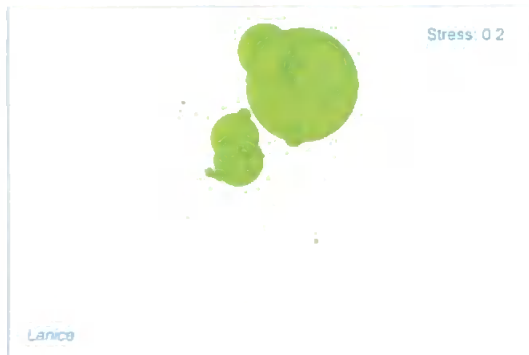
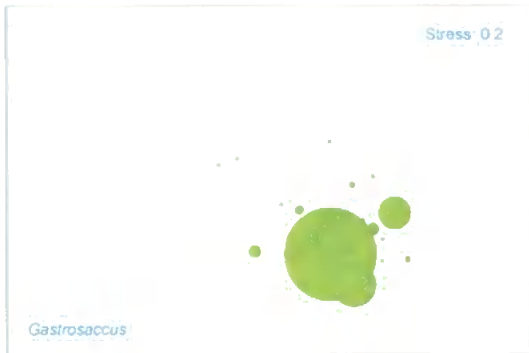
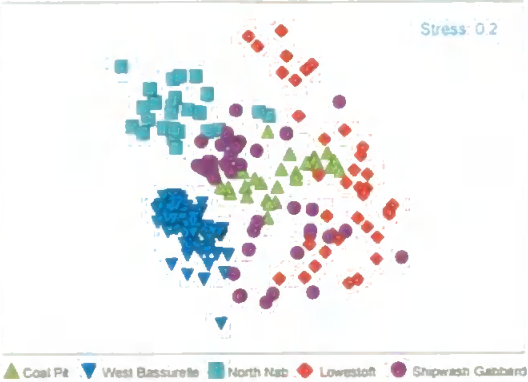




Figure 58 shows the cumulative dominance of benthic taxa (genera) recorded in each of the five study locations. Generally, a single genus accounted for between 10-25% of the total faunal dominance, however, the Lowestoft site exhibited a higher level of dominance of around 45%.

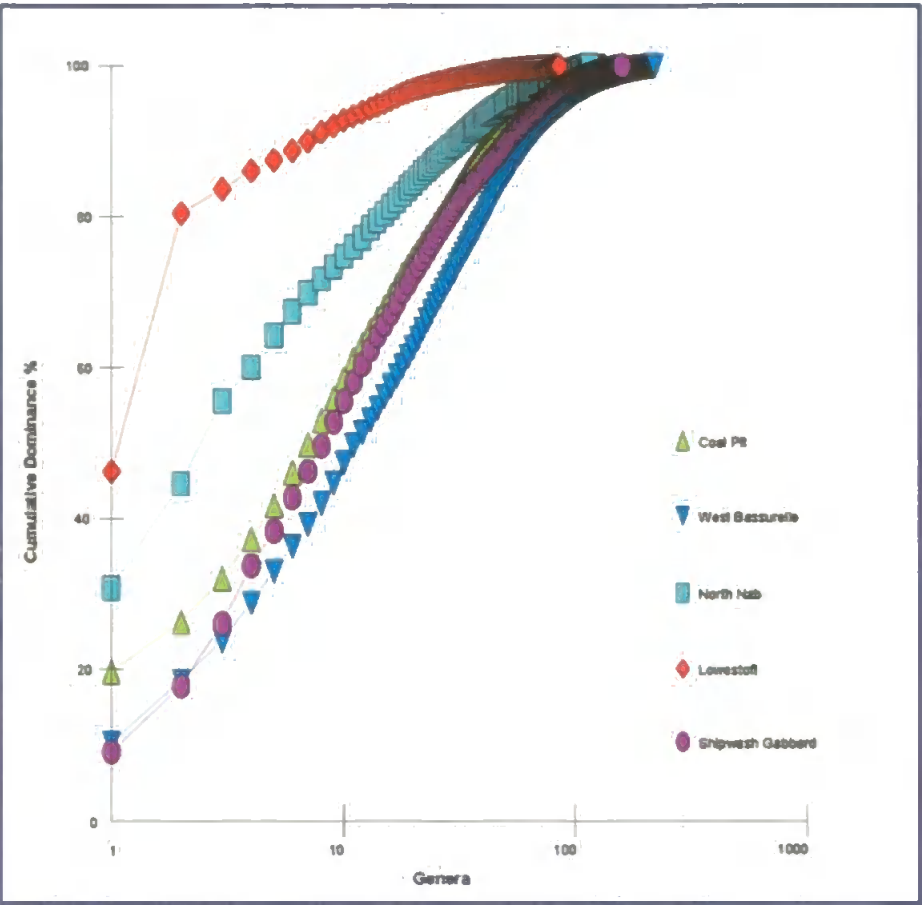


Figure 58: Cumulative dominance curves plotted for genera recorded in each of five study areas from different locations in the North Sea and English Channel.

4.4.3. Feeding groups

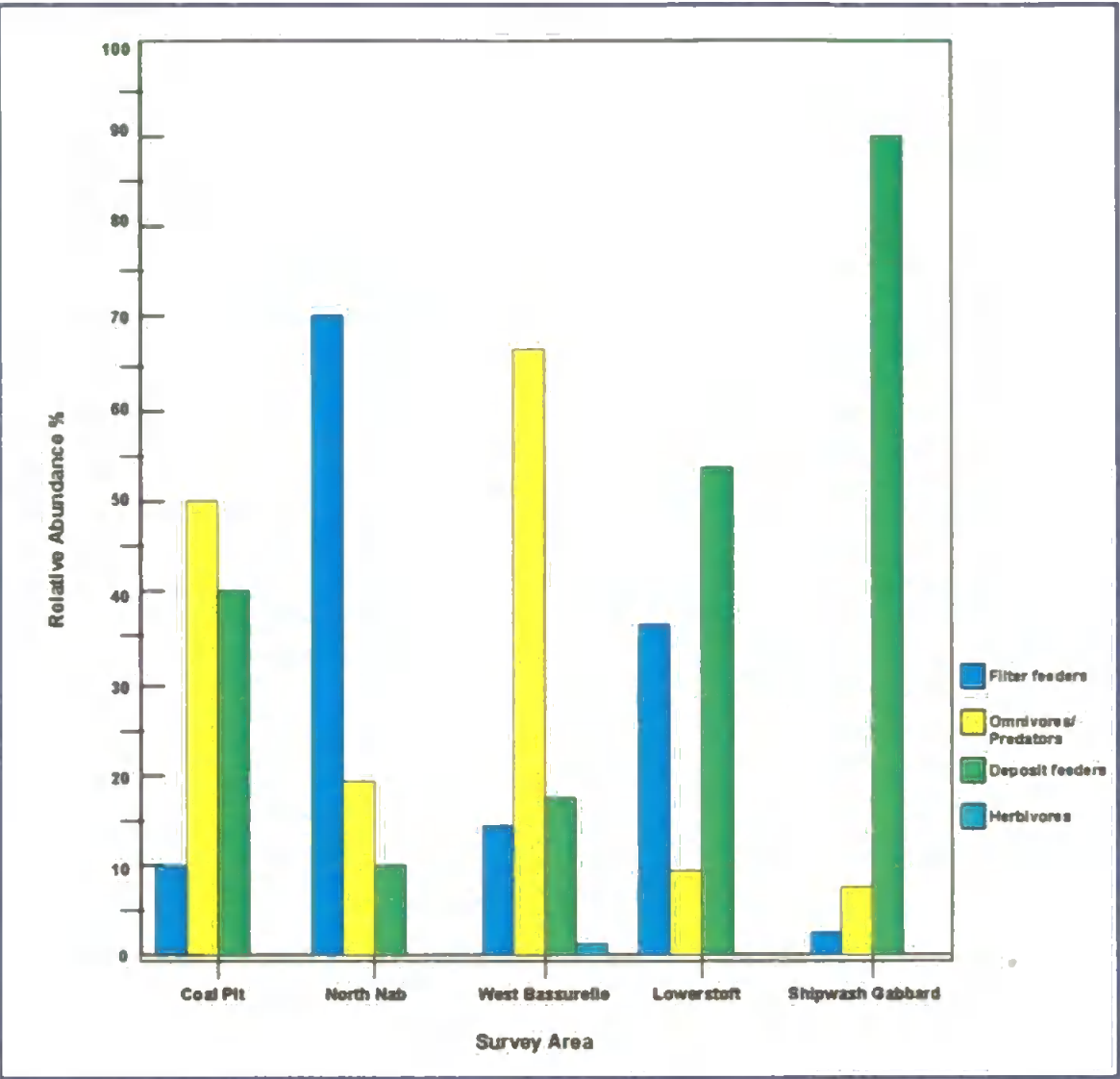


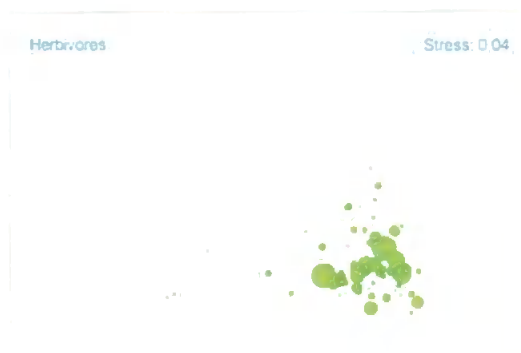
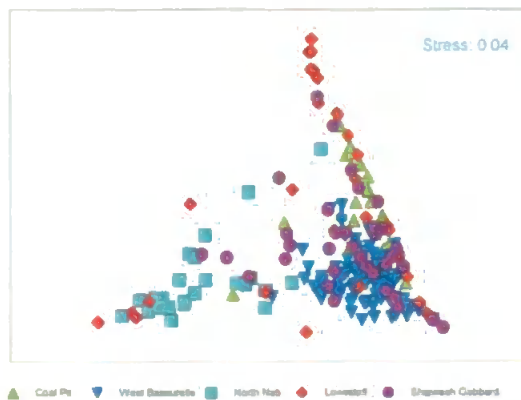
Figure 59. The relative abundance (%) of different macrofaunal feeding groups recorded in each of five study areas from different locations in the North Sea and English Channel.

Figure 59 shows the percent contribution to the overall abundance of four feeding groups. Deposit-feeding taxa accounted for over 90% of the abundance at Shipwash Gabbard, approximately 55% at Lowestoft and 40% at Coal Pit. West Bassurelle and Coal Pit were characterised by relatively high numbers of omnivores/predators, with contributions of approximately 70% and 50% respectively; at North Nab, omnivores/predators category accounted for around 25% of the total abundance of fauna. Filter feeders accounted for the greatest share of the abundance at North Nab (65%) and Lowestoft (35%). This group was

a comparatively minor contributor to the abundances of the other areas. The results of ANOVA tests (Appendix XXXIII) indicated that the differences between the abundance of filter feeders and omnivores/predators at the five sites were significant at the 5% and 1% level respectively. As colonial taxa (many of which are filter feeders) were not included in this analysis due to difficulties in quantification, a somewhat different pattern may be expected for an analysis, which included such species. To overcome such difficulties of quantification, species biomass data should be used as opposed to species abundance values. However, in the present study, this was not possible due to the lack of suitable data (i.e. biomass was recorded for major taxonomic groups only, or in the case of the two historical datasets for total sample biomass only).

The results of the ANOSIM test on the untransformed, standardised data for feeding groups based on Bray-Curtis Similarity (Appendix XXXIV) revealed that the calculated global R-value was 0.507 and was significant at the level of 0%, thus implying that, in general, the areas can be regarded as distinct in terms of the trophic composition of the resident fauna. The majority of the pair-wise regional comparisons showed significant differences, although some are more closely related than others. No significant differences were observed between the feeding group structure at Coal Pit and Lowestoft or between Coal Pit and Shipwash Gabbard ($R = 0.016$ $p = 25\%$, and $R = 0.037$ $p = 11.1$ respectively).

Samples from West Bassurelle formed a relatively tight cluster, as did samples from the Coal Pit site (Figure 60). In contrast, stations from Shipwash Gabbard and Lowestoft were widely spread, indicating low internal similarity. The latter was confirmed by analysis of multivariate dispersion (Appendix XXXV). MDS ordinations with feeding groups superimposed illustrate the relative importance of



the groups at a given location. Omnivores/predators were present in significant quantities at a number of locations, with filter feeders and deposit feeders increasing in their importance at North Nab and Lowestoft respectively. Herbivorous taxa played a minor role in the make up of the community structure at these locations. Contributions to the similarity of samples between and within sites in terms of feeding groups are shown in (Appendix XXXVI).

4.4.4. Discussion

Whilst some similarities exist between the study areas, they generally possess contrasting faunal characteristics. West Bassurelle possesses a far more uniform community, shown by the higher internal similarity value given in Table 3 and the tight clustering shown in the MDS ordination (Figure 53), than Shipwash Gabbard and Lowestoft. Reasons for these differences relate possibly to the environmental conditions at each area. For example, there is a correlation between diversity and environmental stability (Sanders 1968; Boesch 1972). Systems characterised by heterogeneous and stable substrata, such as those found at West Bassurelle, have a somewhat more predictable and uniform array of key faunal components, however, their stability is dictated by the prevalence of undisturbed (and uniform) environmental conditions (Boesch & Rosenberg 1981). Conversely, faunal samples obtained from areas subjected to fluctuating environmental conditions, such as those at Lowestoft, often exhibit lower inter-sample similarity because the community is altered frequently by disturbance events. Therefore, the benthic community is never able to reach an advanced successional state. This can result in a mosaic of successional states leading to considerable spatial heterogeneity and hence sample variability (Levin & Paine 1974).

Evidence for the relationship between the composition of the biological resources and environmental conditions were reported by Rees *et al.* (1999) who described similar infaunal assemblages for areas on both eastern and western UK coasts where environmental conditions were comparable. Perhaps surprising was the lack of similarity between samples from the North Sea sites, despite sharing many key genera such as *Ophelia*, *Lagis*, *Bathyporeia* and *Ampelisca*, average dissimilarities between the locations remained high. The differences between the locations appear to be due to the dominance of certain genera in one area, whilst the same taxa being of only moderate importance in another.

It appears that the conditions of certain areas favour particular genera. Probably, this is a reflection of environmental circumstances, although life-history traits of the individual species and species interactions are also of importance in shaping benthic distributions. While some species are characteristically associated with certain environments, their distributions are rarely confined to that habitat alone (Snelgrove & Butman 1994). As shown in this study, faunas of different sedimentary conditions show a degree of overlap.

Taxa such as *Ophelia* are usually indicative of mobile sandy substrata (Vanosmael *et al.* 1982), conditions that were prevalent at the North Sea locations. Such taxa were largely absent in the sediments at both Channel sites. Possible mechanisms for such site fidelity among certain members of the benthic fauna are discussed below.

One possible explanation for the apparent association between certain species and a particular sediment type could be that their larvae behave like the sediment particles themselves. For example, if the larvae of a given species possesses

characteristics similar to the transported sediment grains (i.e. specific gravity, and gravitational fall velocity), then the larvae themselves may be subject to hydrodynamic sorting akin to that of the sediments characterising a particular region (Butman 1989).

The coarse grain size at the North Nab site may be a factor in the decreased importance of deposit feeders compared to the other regions. Substrata of this type may present considerable mobility problems for species possessing an interstitial lifestyle, unless their body size is sufficiently small to be accommodated between the coarse particles. Conversely, the increased availability of stable attachment surfaces, created by coarse gravel and cobbles, is likely to favour colonisation by filter feeding taxa such as those recorded from North Nab.

Certain deposit feeding species may have their distributions shaped by a further mechanism that of optimum ingestible grain size. Self & Jumars (1988) have demonstrated preferential grain size selection in a number of taxa. This situation may well explain the high numbers of *Lagis koreni* that occurred in many samples from the North Sea sites but not in the coarser sediments of the Channel regions. A related species, *Cistenides (Pectinaria) gouldii*, was shown by Whitlatch & Weinberg (1982) to ingest specific particle sizes.

It is evident from the analysis of the feeding groups that even at such a high level of data aggregation (i.e. from genera to feeding group), differences in the overall assemblages were maintained. This is further evidence that the findings of Warwick (1988a, 1988b, 1988c), Warwick *et al.* (1990) and Somerfield & Clarke (1995) can be applied to functional groups (such as feeding method) as well as lower levels of taxonomic definition (i.e. phyla). One possible criticism of

aggregating data to feeding group level is the inaccuracy of categorisation. The work of Fauchald & Jumars (1979), Dauer *et al.* (1981), and Levington (1991) provides evidence that, under certain flow and sediment transport conditions, some species of deposit feeder switch to a suspension-feeding mode. Thus, this plasticity in feeding mode indicates that categories assigned to feeding groups in this study may be over simplistic.

General comparisons between the areas indicate variation in assemblage type. Possible explanations for this variation are likely to involve environmental factors such as sediment type and hydrodynamic conditions. The following chapter attempts to clarify the importance of measured environmental factors from this study, namely sediment particle size, in controlling the distribution of the benthos. The possible roles of further physical factors, not measured in the present study, are considered also.

4.4.5. Summary

The faunal assemblages of the five study sites show some species in common but are largely dissimilar in terms of community type. Dynamic, frequently-disturbed areas with sediments fine enough to be mobilised regularly, such as that found at the Lowestoft site, have been shown to be inhabited by taxa suited to such conditions. Tubiculous, sessile and epilithic forms were not evident in large numbers under such conditions. However, mobile taxa with high fecundity and rapid regeneration periods were found in large numbers. In contrast, the relative stability of the West Bassurelle study site, characterised by deeper water and coarse sediments, supports a diverse range of sessile, tubiculous, epilithic, and slow growing taxa.



**CHAPTER 5. ANIMAL: SEDIMENT
RELATIONSHIPS. THE ROLE OF
ABIOTIC FACTORS IN SHAPING
BENTHIC DISTRIBUTIONS**

The aim of this chapter is to investigate the role of sediment particle size in the determination of benthic distribution patterns.

CHAPTER 5. ANIMAL: SEDIMENT RELATIONSHIPS. THE ROLE OF ABIOTIC FACTORS IN SHAPING BENTHIC DISTRIBUTIONS

5.1. Introduction

A central question to the exploitation of marine aggregate resources is what is the nature of recovery of the benthos following dredging? It is a widely held view that recovery of the benthos to pre-dredging levels is dependent largely on the ability of the seabed deposits to return to that which existed prior to extraction (van Moorsel 1994; Kenny & Rees 1996; Boyd *et al.* 2003). Therefore, it is necessary to investigate the importance of the substratum in determining the structure of benthic macrofaunal assemblages.

The relationship between benthic invertebrates and the substratum in and/or on which they live is essential to the understanding of the extent to which marine macrobenthic assemblages are shaped by the physical characteristics of their surroundings. Consequently, studies of animal-sediment associations have attracted much attention. Early investigations (Petersen 1913; Ford 1923; Stephen 1933) indicated that certain communities were associated with particular substratum types. More recent work (Warwick & Davies 1977; Rees *et al* 1999; Parry *et al* 1999; and Newell *et al* 2001) (for review see Snelgrove & Butman 1994) have emphasised a range of sedimentary characteristics, as well as other abiotic factors, are involved in shaping the distributions of benthic fauna. Many studies have demonstrated a correlation between the benthos and the composition (either by particle size analysis or descriptive techniques) of deposits (Buchanan *et al* 1978; Shackley & Collins 1984; Rhoads *et al* 1985; Eleftheriou & Basford 1989; van Dalssen *et al* 2000). In contrast, other investigations have found little correspondence between benthic macro-invertebrate assemblages and the substratum (Tyler & Banner 1977; Larsen 1979; Duineveld & van Noort 1990;

Newell *et al* 2001). In this chapter, relationships between macrobenthic community structure and sediment composition (particle size) at the five locations featured in this study are investigated.

5.2. Analytical Methods

To determine which subset of sediment particle sizes, if any, provided the closest match to the distribution of the benthos, the BIOENV routine contained in the PRIMER software package was employed to calculate correlation values (Weighted Spearman Rank) between samples of benthic macrofauna and corresponding subsets of sediment particle size data (Clarke & Ainsworth 1993; Clarke & Warwick 1994; Clarke & Gorley 2001). At North Nab and Coal Pit the two actively dredged areas, only control stations were analysed in order to eliminate any possible 'interference' attributable to dredging operations. Nine stations from the West Bassurelle data were omitted from the analysis as they consisted of a very fine sandy substratum, located at the eastern extremity of the sample grid, and were therefore considered to be unrepresentative of the general assemblage of this region.

For all areas, faunal data were subjected to a square root transformation to decrease the weighting of highly abundant taxa. Due to the inherent difficulties in quantification, colonial taxa were excluded also from the analyses, except for the tunicate *Distomus variolosus*, (where single zooids are conspicuous and were treated as individuals) this species was an important member of the North Nab fauna.

Using 'Draftsman' plots, the particle size data (percentage retained) were shown to exhibit a skewed distribution which, according to the work of Clarke & Ainsworth

(1993), reduces the effectiveness of the use of Euclidean distance for the environmental variables. Sediment particle size data were subjected to a log transformation prior to the construction of the similarity matrix of Euclidean distance. This was performed to achieve approximate normal distribution of the data.

5.3. Results

Correlations between sediment composition and the benthos are shown in Table 10. The correlation value, and the subset of sediment variables which 'best explain' the distribution of the benthic fauna, are given as well as the single variable with the highest correlation as identified by the BIOENV analysis. Animal: sediment relationships appear to be strongest at the Shipwash Gabbard site (with a correlation value of 0.652). The group of sediment fractions which most closely match this pattern comprised fine gravel, sand and silt. Sediment grain size also appeared to have a reasonably strong relationship with the faunal distributions at the Coal Pit site. In contrast, a weak relationship was evident at the North Nab survey area (0.239) with silt contributing to this best match. Lowestoft (0.353) has a faunal distribution best explained by the presence of fine sand and silt. The West Bassurelle region has a best correlation (0.293) contributed to by gravel and sands. Table 10 also gives the best correlation of the benthic data with a single sedimentary variable; it can be seen that the faunas of Lowestoft and Coal Pit have moderate associations with silt and gravel respectively. Silt is also the single particle fraction with the best match to the benthos in the North Nab samples. A weak relationship is observed for the macrobenthos and fine gravel in the West Bassurelle region, the single particle size fraction in the Shipwash Gabbard area with the best match was found to be sand of 1 mm diameter.

Table 10. The optimal combinations and single sedimentary variables responsible for the best match, as measured by Weighted Spearman Rank correlation, with the benthos recorded at five locations.

Location	Variable	Correlation
North Nab	<0.063mm	0.239
West Bassurelle	8mm, 2mm, 0.125mm, 0.063mm	0.293
Coal Pit	37.5mm, 2mm, 0.063mm, <0.063mm	0.561
Shipwash Gabbard	8mm, 4mm, 1mm, 0.5mm, 0.125mm, 0.063mm, <0.063mm	0.652
Lowestoft	0.063mm, <0.063mm	0.353
Location	Best Single Variable	Correlation
North Nab	<0.063mm	0.239
West Bassurelle	8mm	0.259
Coal Pit	10mm	0.404
Shipwash Gabbard	1mm	0.470
Lowestoft	<0.063	0.320

5.4. Discussion

From the calculated correlation values (Table 10), sediment particle size accounted for a significant amount of the apparent variability in species richness and abundance at Shipwash Gabbard and Coal Pit sites. However, in other locations, it appears that sediment particle size has a far less significant role in shaping the benthic communities. Interestingly, the dynamic, less stable North Sea regions showed that sediment particle size may be an important shaping factor, whereas the more stable, equilibrium communities of both English Channel study sites did not.

An explanation as to why there was little agreement between sediment particle size and the benthos of the English Channel sites may lie in the type of community inhabiting these locations. Both the North Nab and West Bassurelle sites were characterised by a highly diverse fauna comprising many species, such as *Crepidula fornicata* and *Galathea intermedia*, fitting the classic 'k-selected' description (Pianka 1970; Zajac *et al.* 1998). Such species are indicative of stable habitats and are thus not subjected to significant physical disruption. Therefore, their distributions may be expected to be less dependent on physical parameters and more on biological interactions such as competition for food resources and/or space (Table 11).

Table 11. Characteristics of opportunistic (*r*-selected) and equilibrium (*k*-selected) taxa (adapted from Zajac *et al.* 1998).

Opportunist species (<i>r</i> -selected)	Equilibrium species (<i>k</i> -selected)
Many reproductions per year	Few reproductions per year
High recruitment	Low recruitment
Rapid development	Slow development
Early colonisers	Late colonisers
High death rate	Low death rate
Small body size	Large body size
Sedentary	Mobile
Deposit feeders (mostly surface feeders)	Deposit and suspension feeders
Brood protection; lecithotrophic larvae	No brood protection; planktotrophic larvae

A further interaction occurs between space providing structurally-complex taxa, such as hydroids and bryozoans and a wide range of species that utilise the epifauna, refuges. However, correlation analysis of the abundance of benthic species with the biomass of miscellaneous groups (including hydroids and bryozoans) at both locations, gave low values, illustrating the occurrence of these organisms may not be important in shaping the distributions of other (non-sessile)

benthic taxa. The presence of many of these epiphytic species requires the presence of coarse particles, or some other attachment surface, and therefore, it is likely that the presence of these epifaunal species is highly correlated with particle size particularly with those particle sizes at the coarser end of the spectrum. However, other structure-forming organisms, such as the polychaete *Sabellaria spinulosa*, may play a key role in habitat creation leading to increased species diversity. At both North Nab and West Bassurelle, this latter species formed small patches on the surface of large particles as opposed to the extensive reefs described by Vorberg (2000) (Figure 61).



Figure 61. *Sabellaria spinulosa* (arrowed). Note the non-reef forming growth form present at North Nab and West Bassurelle.

Lack of correlation between sediment particle size data and the benthos, as shown in this study, has been reported previously for sites in the English Channel (Holme & Wilson 1985). Holme & Wilson (1985) reported that the faunal distributions were

determined largely by the degree of disturbance by tidal scour and seasonal events such as sediment resuspension during storms. Snelgrove and Butman (1994) stated that as the sediment generally reflected the boundary-layer flow and sediment transport regime, correlations between animal and sediment distributions may not be caused by any particular aspect of the sediment itself, rather by the physical forces that created that particular sedimentary environment. Clearly, if this is the case, measurement of a far greater suite of environmental variables would be necessary to explain the factors controlling benthic faunal distributions. Webb (1968) postulated that porosity and permeability could be more appropriate measures of sediment structure than sediment particle size.

Thiebaut *et al.* (1997) conducted an investigation spanning a five-year period in the muddy, fine sands of the eastern Bay of Seine. They reported that a community characterised by *Abra alba* (Mollusca) and *Cutellus pellucidus* (Mollusca) varied strongly from year to year, whereas another assemblage, largely comprising the polychaetes *Lagis* (*Pectinaria*) *koreni* and *Owenia fusiformis* remained relatively stable. The distribution of the fauna was shown to possess a poor correlation with sediment grain size and it was suggested that community structure was more likely to be dependent on salinity and post-settlement processes related to food availability (Thiebaut *et al.* 1997).

The Shipwash Gabbard and Coal Pit study site exhibited a relatively strong relationship with particle size fractions of gravel, sand and silt. In contrast to the study sites located elsewhere, the species assemblage from these regions comprised largely polychaetes such as *Lagis* and *Ophelia*, with the amphipods *Ampelisca* and *Bathyporeia* also being important members of the fauna. Such organisms (*Lagis* and *Ophelia*) could be assigned reasonably to the *r*-selected, or

opportunistic, category as described by Pianka (1970). Sediment particle size is, perhaps unsurprisingly, important in areas where fine particles predominate. Despite the presence of gravels, finer particles tended to cover the coarse deposits. Therefore, many epilithic species are unable to settle/attach to what is an essentially unstable, and often mobile, substratum. The physical structure of such a substratum may be unsuitable for the establishment of a community that includes most competitively superior 'k-selected species' requiring stable environmental conditions. Hence, biological interactions, such as competitive exclusion, are of lower significance in such areas. The benthos in such a region may be held in a permanent state of early to intermediate succession with the fauna comprising a somewhat transitory community (Pearson & Rosenberg 1978).

The sediment particle size: macrobenthos relationship was weaker at the Lowestoft site compared with Shipwash Gabbard and Coal Pit. At Lowestoft, fine sand and silt appear to be the major factors contributing to the correlations. As noted previously, high silt content appears to coincide with increased species diversity (relative to other samples in this study area) at this site. As found at both other North Sea locations, Lowestoft had a fauna characterised by opportunistic species. Indeed, most were typical of a sandy offshore, North Sea community (JNCC 1997) with *Ophelia* spp, *Ampelisca spinipes*, *Lagis koreni* and *Nephtys* spp making significant contributions to inter-sample similarities. However, the lower correlation between the benthos and sediment particle size indicated that further, unmeasured, variables were of importance.

One obvious difference between the two Channel sites and the three North Sea locations was the prevalence of invertebrates that reside on the surface of the substratum in the Channel sites (*Galathea intermedia* and *Crepidula fornicata*) and

those that live within it in the North Sea (*Ophelia* spp and *Spiophanes bombyx*). Ellis & Rogers (2001) reported that organisms that live within the deposits appear generally to be more dependent on substratum type than those which live on the surface. This may explain the apparent lack of correlation between the fauna and particle size in the Channel sites and the relatively strong relationship in the Shipwash Gabbard and Coal Pit areas. However, this explanation does not explain the rather low values obtained for Lowestoft.

A major consideration in linking benthic community patterns with sediment particle size lies in the importance of other physical parameters that determine the nature of the substratum itself. Due to the complex interrelationships between the sediment profile and the hydrodynamic regime, measuring sediment grain size alone may not represent the key physical features at a given location (Parry *et al.* 1999). To achieve a better understanding of which physical/environmental factors control benthic distributions, a far wider range of parameters would need to be measured. Whilst sediment composition is likely to play a key role in determining benthic community structure, it is far more likely that benthic faunal assemblages are controlled by a combination of abiotic (both naturally occurring and anthropogenic) and biotic influences. It is also probable that factors such as bottom boundary layer flow hydrodynamics, and sediment transport both by physical movement and scouring, play important roles in shaping the benthos (Miller & Sternberg 1988; Palmer 1988a; Cacchione & Drake 1990).

Although numerous studies are cited to support the theory that infaunal distribution patterns are correlated with sediment grain size (Longbottom 1970; Eagle 1973; Eleftheriou & Basford 1989; Duineveld *et al.* 1991; Mackie *et al.* 1995), most studies provide little insight into the mechanism(s) responsible for such

associations (for review refer to Snelgrove & Butman 1994). Most of the evidence for grain size directly influencing community structure comes from laboratory-based experiments. For example, Gray (1974) showed that, when larvae or adults of several species of benthic invertebrates were given a choice of sediment types in small dishes, the organisms selected preferentially the sediment closely resembling that of the natural adult habitat. Similarly, Botero & Atema (1982) found that juvenile lobsters, *Hommarus americanus*, favoured a rocky substratum when faced with a choice of rock, mud, or sand. However, neither of these studies provides proof that the substratum choices were dictated by grain size alone. Parry *et al.* (1999) demonstrated that coarser sediments had higher numbers of species and beta diversity compared to finer grained deposits. However, these authors did conclude that invertebrate body size variability was not determined by grain size alone but was controlled, presumably, by local, short-term factors such as disturbance, sediment resuspension during storms and variability in organic flux. Newell *et al.* (2001) established that community structure of the benthos showed a poor correlation with particle size distribution of sand and gravels from the English Channel, despite certain species clearly being associated with certain substratum types.

Perhaps a controlling factor of greater importance than sediment particle size is the organic content of the substratum, purely on the basis of being a major food source for many deposit-feeding invertebrates. Evidence for the importance of carbon content in shaping benthic communities can be seen in many pollution studies (see Pearson & Rosenberg 1978). Thiebaut *et al.* (1997) reported that food limitation (sediment organic matter concentration) was likely to be an important factor in the survivorship of juvenile *Owenia fusiformis* (Annelida) and hence of importance in structuring the adult populations. Many studies of this type have

highlighted an array of species that often show significant responses to organic loading either by increasing or decreasing in abundance, thus illustrating the importance of organic matter. However, current issues concerning the quality, as well as the quantity, of organic matter as a food source for both deposit and suspension feeders, plasticity in feeding mode, the specificity of feeding types to sediment type and boundary-layer flow regime (as a covariate of sediment type) render premature the unifying principle of organic content being a major causal factor in the shaping of infaunal distributions (Snelgrove & Butman 1994). The use of sediment grain size as an explanatory factor for macrofaunal distributions has been questioned by Buchanan & Kain (1984) who stated that the treatment of sediments for particle size analysis results in the removal of natural aggregates such as faecal pellets. Hence, the resulting disaggregated sediment is no longer representative of the deposit encountered by an organism in the natural environment and may, therefore, no longer be of use in determining its role in the structuring of the benthos (Dernie *et al.* 2003).

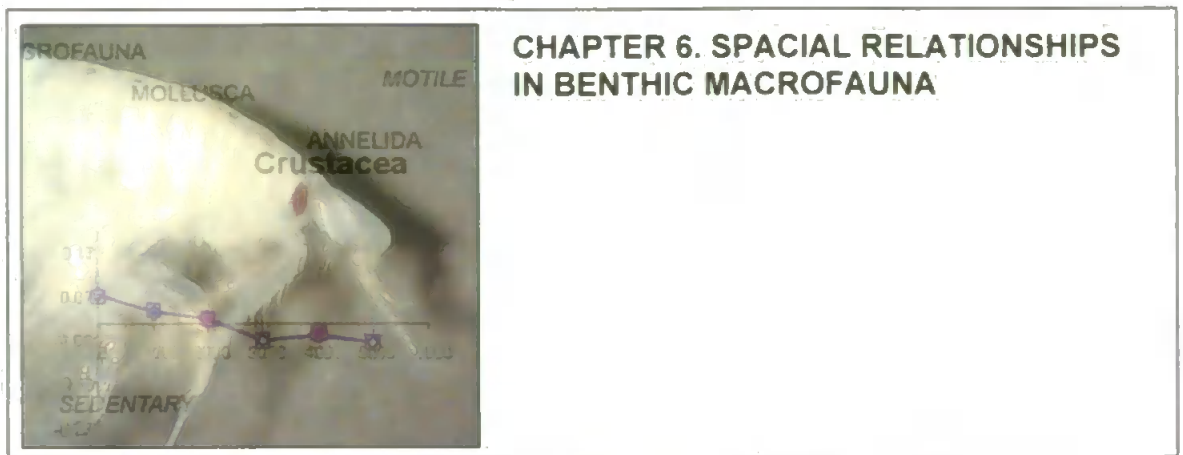
The contribution of waves and currents in determining the distributions of the benthic inhabitants is indeed a significant one. Although storm-induced disturbance is likely to be most intense in shallow habitats, Drake & Cacchione (1985) showed that even at depths of 100m, winter storms could transport approximately $1000\text{kg m}^{-1}\text{day}^{-1}$ of resuspended sediment across the continental shelf. Cacchione *et al.* (1987) stated that winter storms may be a major controlling factor in the distribution of surface sediments in deeper water and that it would be wrong to discount the effects of waves on benthic assemblages in deep waters. Work carried out by Holme & Wilson (1985), in an area of the English Channel subjected to high tidal stream velocities, demonstrated that the patchiness of benthos on a local scale (100m) was dictated by the scouring action of shifting

sands. They described a number of community types whose structure was determined by the frequency and extent of disruption caused by the movement of sand and pebbles (Holme & Wilson, 1985).

The previous discussion highlights the requirement for a range of physical parameters to be recorded if meaningful and predictive explanations for faunal distributions are to be made in the future. A useful closing statement highlighting this need is provided by Warwick & Uncles (1980) who stated: *"correlations between sediment type and faunal communities are restricted usually to analysis of the static pattern of sediment granulometry, largely ignoring the hydrodynamic environment in which the animals live. However, it is this hydrodynamic regime (mainly the tidal currents) that largely determines the sedimentary characteristics of an area and can be considered as the ultimate cause of the broad-scale community patterns observed"*.

5.5 Summary

The role of sediment particle size in shaping the distribution of the benthic macrofauna appears to be less important at the more stable habitats of the English Channel than in the more readily disturbed North Sea locations. A possible explanation for this apparent dissimilarity is that in stable regions, where sediment transport does not impede the development of a community to an advanced successional stage, biological interactions such as competition for food and /or space become increasingly important. In unstable habitats, physical disruption of the sediments, and therefore any organisms inhabiting it, along with other factors such a sand scour, preclude the development of an advanced invertebrate assemblage. Thus sediment particle size, and the factors of which it is a function, are primary forcing factors in the shaping of the benthos in dynamic environments.



CHAPTER 6. SPACIAL RELATIONSHIPS IN BENTHIC MACROFAUNA

The aim of this chapter is to detect spatial patterns such as patch size/separation in the macrobenthic invertebrate assemblages of five environmentally contrasting locations and to provide possible explanations for these patterns.

CHAPTER 6. SPATIAL RELATIONSHIPS IN BENTHIC MACROFAUNA

6.1. Introduction

Scale is an important consideration in ecological research. Conclusions based on processes or interactions at one scale may not be valid at another, therefore, knowledge regarding scaling effects must be obtained by the investigator to avoid misinterpretation (Eberhardt & Thomas 1991; Legendre *et al.* 1997). The heterogeneous nature of the benthic environment dictates that biotic and abiotic variables are scale-dependent (Eggleston *et al.* 1998). Environmental forcing factors, population and community dynamics and stochastic events, such as storms, promote heterogeneity (Dutilleul & Legendre 1993). This variability of biological and physical factors leads to patchiness both in terms of habitats and inhabitants. Weins (1976) defined a patch as: "a surface differing from its surroundings in terms of size, shape, habitat type and heterogeneity, and boundary characteristics". A patch can also be defined relative to the organism or community being studied and the questions being asked (Weins 1976; Eggleston *et al.* 1998). Eggleston *et al.* (1998) recommended that patches be defined using objective criteria to identify their boundaries; for example, the relative rate of change in a variable of interest (i.e. a particular species) per unit of space. Past investigations of spatial patterns have focused on the distribution of individual species (Hewitt *et al.* 1997; Legendre *et al.* 1997; Eggleston *et al.* 1998) and benthic communities (Somerfield & Gage 2000; Paiva 2001). The following section attempts to identify spatial relationships in communities (and categories within communities) from locations in the English Channel and North Sea.

6.2. Analytical Methods

A Spearman rank correlation coefficient (ρ), was used to examine relationships between similarity and inter-sample distances. This was calculated between matrices of species similarity and distance (see Somerfield & Gage 2000). To highlight the contributions made by various faunal groups to the observed patterns, similarity matrices were constructed using the abundance data for all species (where present) belonging to the following categories: total macrofauna, Polychaeta, Crustacea, Mollusca, and sedentary and motile fauna. For data from North Nab and Coal Pit, matrices for amphipod species were also assembled. Prior to the calculation of Bray-Curtis similarity, abundance data were subjected to a fourth root (Root Root) transformation, thus increasing the influence of the intermediately abundant taxa. Untransformed data were also used to show trends attributable to the numerically abundant organisms. Each similarity matrix of abundance was correlated with a second matrix of distances (metres) between sample locations. Significance of the correlations was determined by a Monte Carlo permutation procedure contained in the RELATE routine, found in the PRIMER software package (Clarke & Warwick 1994; Somerfield & Clarke 1995; Somerfield & Gage 2000).

Rank correlograms (Somerfield & Gage 2000) have been used to highlight spatial autocorrelation in the various faunal categories. Autocorrelation is defined by Legendre & Legendre (1996) as "the property of random variables which take values, at pairs of locations a certain distance apart, that are more similar (positive autocorrelation) or less similar (negative autocorrelation) than expected for randomly associated pairs of observations". Spatial autocorrelation applies to autocorrelation over geographic space.

The construction of correlograms first involves the aggregation of the distance data into distance classes. Two different ranges of distance class were assigned to the data due to large variation in maximum and minimum distances between the areas. Smaller distance classes were used at North Nab and Coal Pit. Therefore, relationships between the biotic and distance matrices have been investigated at two separate scales, a small scale (North Nab and Coal Pit) and large scale (Shipwash Gabbard, Lowestoft and West Bassurelle). Matrices were constructed for each distance class, values relevant to a given distance class were coded as 1, the remaining entries were coded as 0. Sequential Mantel tests (Mantel 1967; Legendre & Legendre 1996) were performed between the abundance matrices and the distance matrix of a given distance class; the 1 or 0 coding means positive correlations correspond to positive autocorrelation (Somerfield & Gage 2000). The resulting correlation values were then plotted against the corresponding distance class to create a multivariate correlogram (Appendices XXXVII - XLI). The use of a Spearman rank correlation coefficient (ρ) in this study led to the creation of plots known as rank correlograms. Correlograms can be employed to reveal multivariate patchiness (Somerfield & Gage 2000). In this instance the term multivariate patchiness is used as the variable of interest (Bray-Curtis similarity) and is calculated from more than one variable namely species abundance. Approximate patch diameter is indicated on the correlograms by the point at which the line crosses the x-axis; the estimated distance between the patches corresponds to the width between where the line first crosses the x-axis and the point where the line crosses the x-axis for a second time.

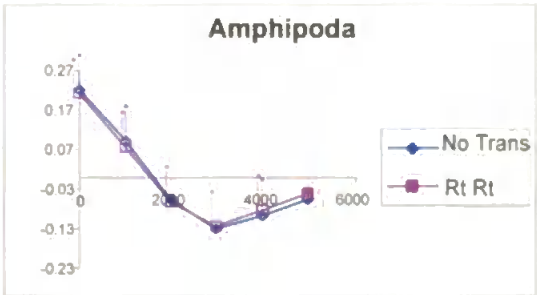
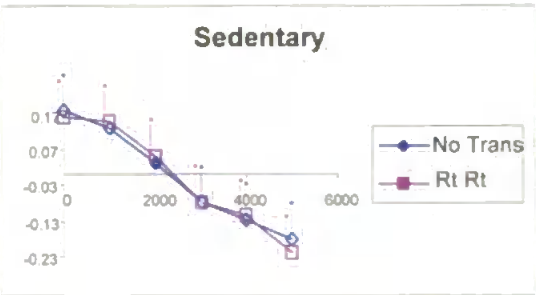
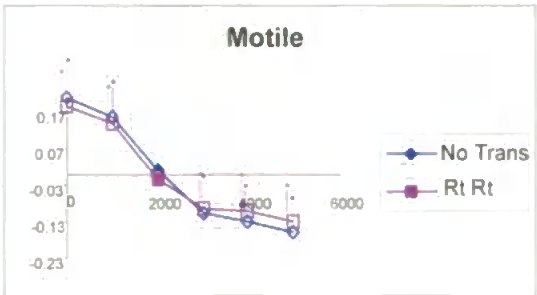
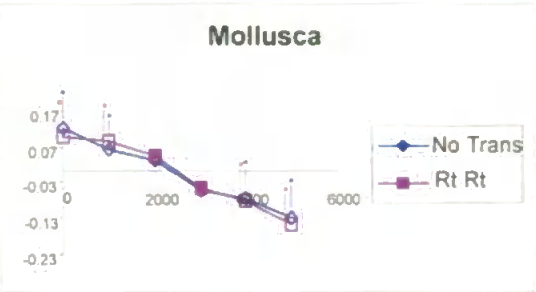
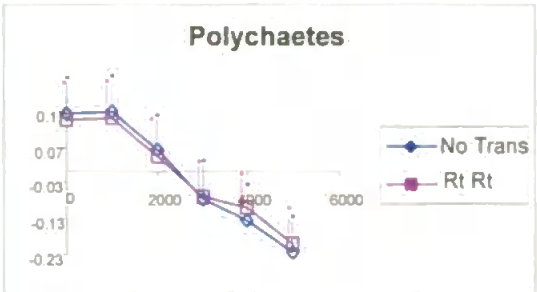
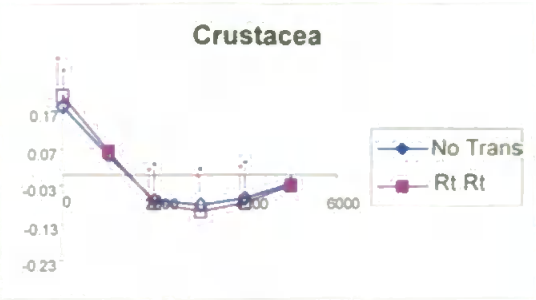
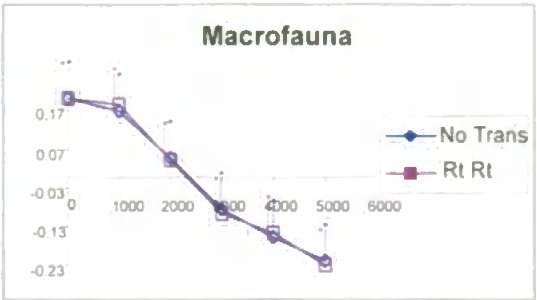
6.3. Results

6.3.1. Relationship with distance

Distance-related spatial structure (Table 12), defined as a significant correlation between faunal and distance matrices (Somerfield & Gage 2001), was evident in all faunal groupings at both of the English Channel sites (North Nab and West Bassurelle). At the Coal Pit study site, simple spatial structure, i.e. decreasing similarity with increasing distance, was evident for all groupings with the exception of numerically-dominant crustacean species. Spatial structure at the Shipwash Gabbard site appeared to reflect variation between both the numerically-dominant species and the species composition of the Mollusca. At Lowestoft, spatial structure of the benthos was a result of variation in abundant annelids and differences in the species composition of this group.

6.3.2. Relationship with distance classes (small scale)

At the North Nab site, the overall fauna, polychaete, mollusc, sedentary and mobile components all exhibited above average similarities in samples up to 2000-2500m apart (Figure 62). These similarities indicate the existence of a simple relationship between sample similarity and distance (i.e. samples further apart in distance are less similar to one another, in terms of the composition of fauna/faunal groups, than samples that are closer together). At distances greater than 2500m, the samples become dissimilar regardless of the level of transformation. Assemblages of Crustaceans were similar in samples up to 1500m apart, less similar in samples >1500-4000m apart and then returned to the average similarity expected for all samples at 5000m+. This pattern, in the Crustacea, is possible evidence of patchiness, the patches being around 1500m across but separated by distances of up to 3500m. Amphipods displayed a similar trend to that of crustaceans in general and are, therefore, likely to be responsible



* Indicates significance at that particular point.

Table 12. Summary of the Spearman Rank correlations between corresponding elements of similarity matrices calculated from abundance of major groups using untransformed and fourth root transformed data and inter-sample distances within the five study areas (* $p < 0.05$, ** $p < 0.01$).

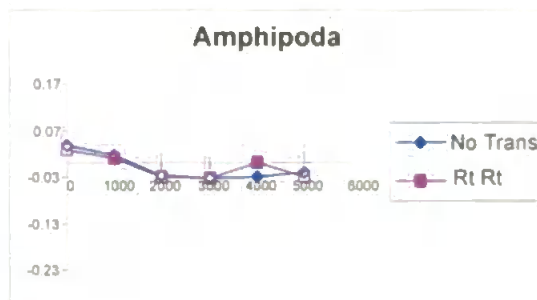
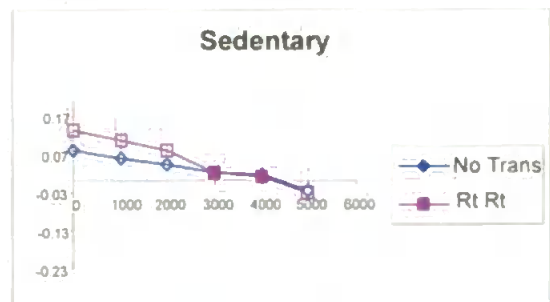
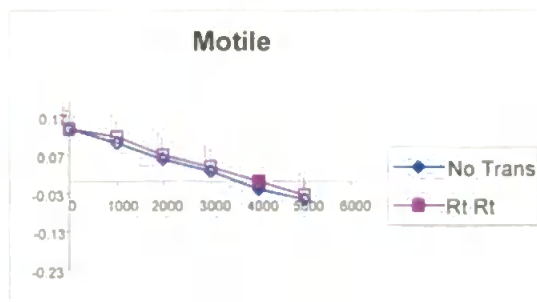
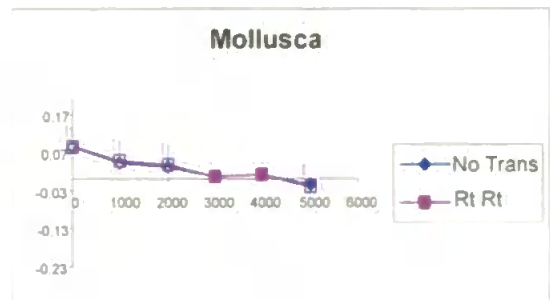
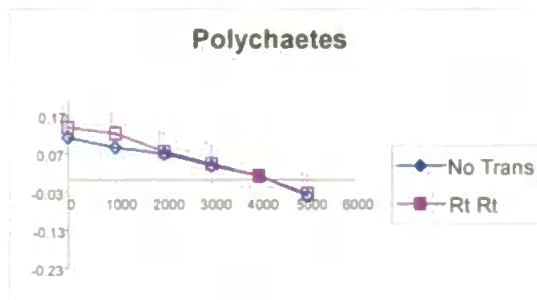
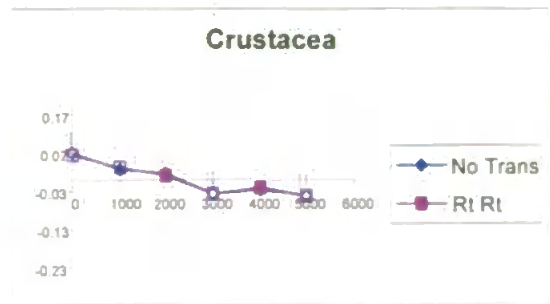
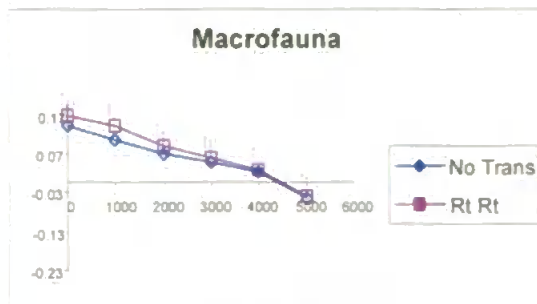
Study Area	Group	Transformation	
		None	Fourth Root
North Nab	Macrofauna	0.366**	0.377**
	Annelida	0.336**	0.296**
	Crustacea	0.181**	0.209**
	Mollusca	0.189**	0.194*
	Sedentary	0.366**	0.377**
	Motile	0.342**	0.323**
Coal Pit	Macrofauna	0.211**	0.311*
	Annelida	0.249**	0.301**
	Crustacea	0.047	0.061*
	Mollusca	0.14**	0.149**
	Sedentary	0.184**	0.296**
	Motile	0.238**	0.286**
West Bassurelle	Macrofauna	0.424**	0.419**
	Annelida	0.399**	0.401**
	Crustacea	0.333**	0.36**
	Mollusca	0.235**	0.257**
	Sedentary	0.334**	0.323**
	Motile	0.393**	0.432**
Shipwash Gabbard	Macrofauna	0.181*	0.203*
	Annelida	0.144*	0.187*
	Crustacea	0.159	0.254*
	Mollusca	0.288**	0.307**
	Sedentary	0.241**	0.3**
	Motile	0.17*	0.2*
Lowestoft	Macrofauna	0.085	0.111*
	Annelida	0.121*	0.134*
	Crustacea	-0.06	-0.051
	Mollusca	-	-
	Sedentary	-0.003	0.035
	Motile	0.077	0.1*

for the patterns observed for the group as a whole. Indeed, amphipods accounted for approximately 64% of all crustacean species recorded from the area. Amphipods appeared to be present in patches of approximately 1700m in diameter separated by distances of around 3200m.

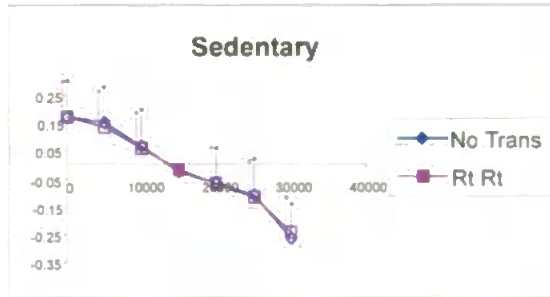
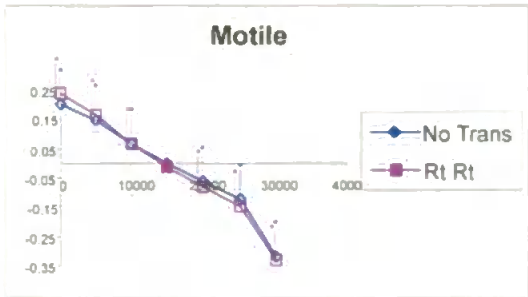
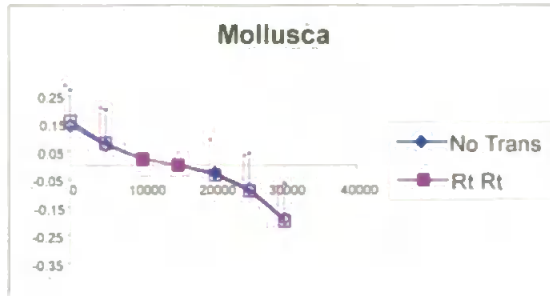
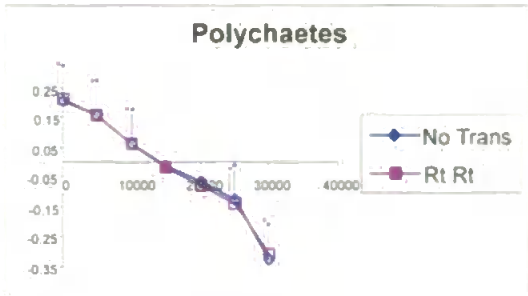
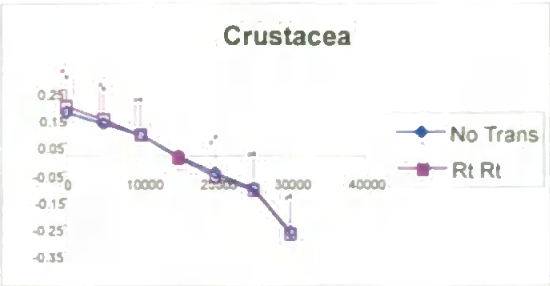
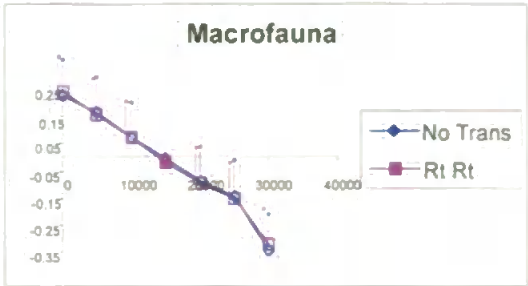
Samples of the total fauna obtained from the Coal Pit study area were generally similar to each another up to a distance of approximately 4500m although similarity decreased with increasing distance (Figure 63). This trend was present for the polychaetes, molluscs, and sedentary and mobile components of the fauna (Figure 63). A somewhat different pattern was displayed by the crustaceans in terms of species composition; samples were more similar up to 2400m but were significantly less similar at 3000m before returning to the average for all samples at 4000m. Beyond 4000m, samples once again become less similar. The absence of a non-linear pattern (i.e. samples becoming less similar with increasing distance) suggests multivariate patchiness with the crustaceans occurring in patches approximately 2400m in diameter separated by distances of around 1000m. A clearer demonstration of patchiness was found for the Amphipoda appeared to form patches of approximately 1400m across and at intervals of approximately 2500m.

6.3.3. Relationship with distance classes (large scale)

Most faunal components recorded from West Bassurelle (Figure 64) showed a simple similarity: distance relationship with distances of up to 15000m apart (i.e. decreasing similarity between samples with increasing distance). At distances of 20000m and greater, samples were significantly dissimilar to each other. This pattern applied to the dominant species and total species composition.



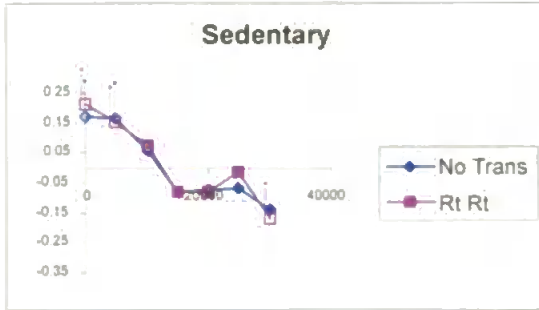
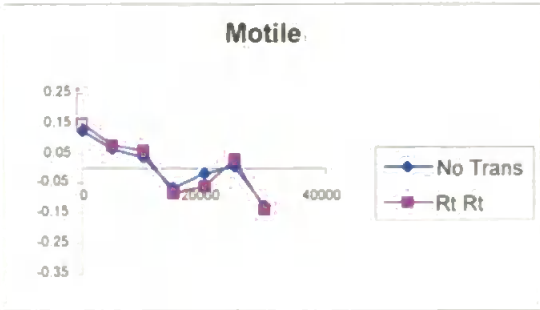
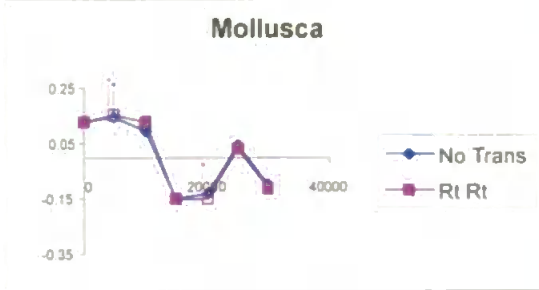
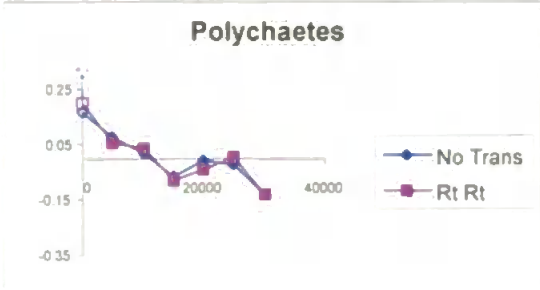
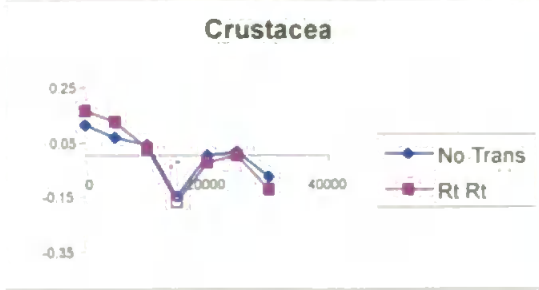
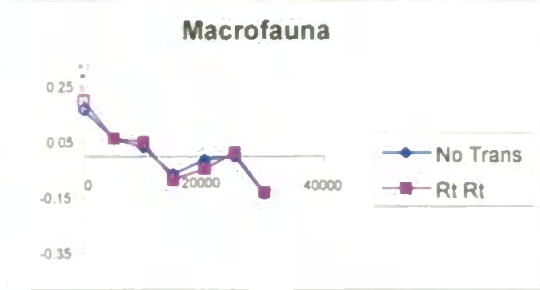
* Indicates significance at that particular point.



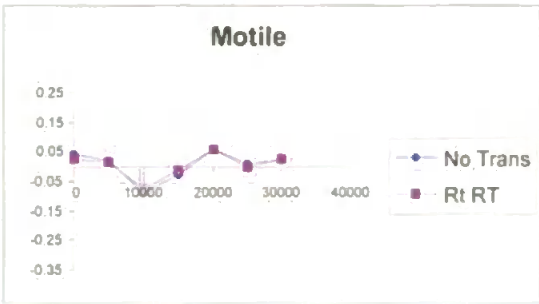
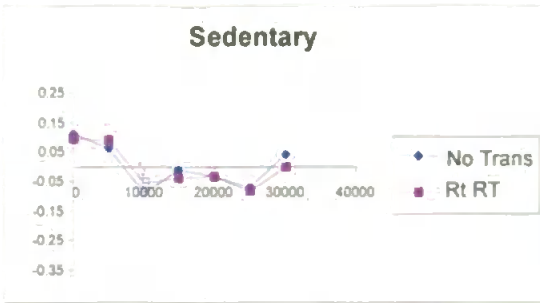
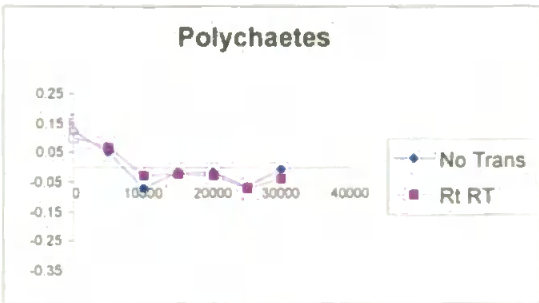
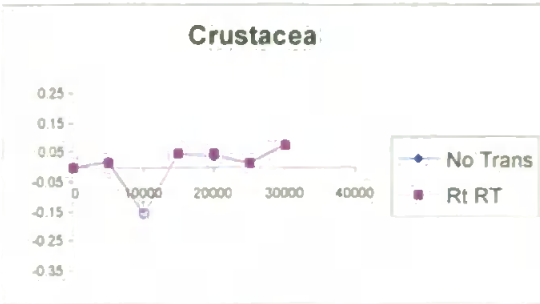
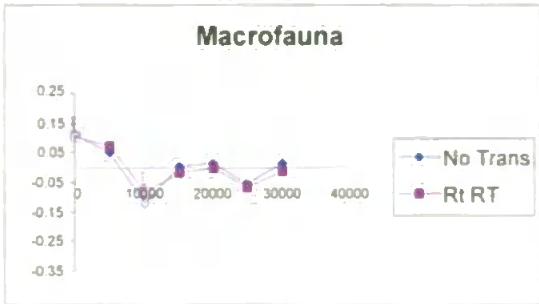
* Indicates significance at that particular point.

For the Shipwash Gabbard macrofauna, spatial structure was apparent and resulted from differences in species composition between samples 11000-12000m apart and those separated by 14000-15000m, which were less similar (Figure 65). This pattern, of samples being more similar in one distance class followed by samples of decreased similarity in the next distance class, was then repeated, with samples between 20000-25000m apart being close to the average for all samples whereas samples at 30000m were less similar. This trend, depicted on the correlogram by the line crossing the x-axis at the 11000-12000m distance class, the line then continues to fall up to the 14000-15000m before rising to the average similarity expected for all samples at 20000-25000m, is possible evidence of patchiness, the patches being approximately 11000-12000m across. This pattern was seen in the Polychaeta, Crustacea, and sedentary and mobile components of the fauna. The Mollusca followed a similar trend to the other faunal groupings with a comparable patch size; however, the patches appeared to be separated by greater distances.

The Lowestoft samples up to 8000m apart were more similar to one another (in terms of their faunal composition) than those separated by greater distances (Figure 66). Samples approximately 10000m apart were shown to be less similar to one another than samples separated by other distances. Samples from distances greater than 10000m showed a degree of similarity close to the average for all samples. This pattern of fluctuating inter-sample similarities implies multivariate patchiness, the patches being in the region of 8000m across for most faunal groups. An analogous trend was also evident for the Crustacea, although the patch diameter was smaller at around 5000m. For the Polychaeta this pattern of multivariate patchiness was only apparent in terms of the differences in dominant species (shown by the blue line in Figure 66).



* Indicates significance at that particular point.



* Indicates significance at that particular point.

6.4. Discussion

Inter-sample similarities were found between samples taken at greater distances at the Coal Pit site than at the North Nab study area. The more mobile sediments of the Coal Pit region was the probable forcing factor in the uniformity of the benthic fauna; 'equilibrium' species being precluded from establishment as a result of adverse environmental conditions. In contrast, the greater habitat complexity provided by a diverse sediment structure, and comparatively stable substratum, at North Nab favoured greater community variation, hence samples were less similar at shorter distances than at Coal Pit. Benthic distributional patterns at the North Nab site could also reflect the prevalence of epifaunal taxa at this location. Attachment surfaces, in the form of coarse gravel particles and cobbles, were in plentiful supply and utilised by a wide range of structurally-complex taxa such as hydroids and bryozoans, which in turn provide sanctuary for many other invertebrates. At both locations, the crustaceans exhibited signs of patchiness, the North Sea site having patches of a slightly greater diameter, possibly reflecting the environmental conditions of the region (i.e. suitable benthic habitats spanning much larger areas due to the relatively homogeneous sediment composition). Patchiness was more clearly demonstrated for the Amphipoda than for other groups. Interestingly, the patch size for amphipods was similar at both locations as was the distance at which the patches were separated. The differences in hydro-physical regime at the two sites suggested that this pattern (assuming it is not merely coincidental) for amphipods was likely to be independent of environmental factors and may, therefore, reflect the behavioural traits of the Amphipoda as a group. Thrush *et al.* (1997) noted the importance of information relating to natural history in the integration of studies across spatial scales. Reproductive rates, life cycles, scales of movement, behaviour and resource requirements for different life stages all played key roles in how species respond to environmental

heterogeneities on different scales. Dauvin (1987) noted that recolonisation of the amphipod *Ampelisca* following an oil spill was retarded as a result of the demographic strategy of the amphipods (i.e. absence of pelagic larvae, low capacity of dispersion and low fecundity). Therefore, the patch size of the amphipods at both locations is a probable result of their propensity, due to life history traits, to form insular populations. Patch size of the amphipods may also be mediated by behavioural characteristics of certain genera such as *Ampelisca* (frequently recorded at both North Nab and Coal Pit) as this genus is tubiculous and therefore may display increased site fidelity compared with those taxa that do not possess a tubiculous habit (Lincoln 1979).

Both North Sea sites (Lowestoft and Shipwash Gabbard) exhibited signs of multivariate patchiness for the fauna as a whole and for the groupings. In contrast to the site from the English Channel (West Bassurelle), where inter-sample similarity decreased with distance in a linear pattern, the North Sea sites shared similar hydro-physical characteristics both in terms of substratum composition and current/tidal regime. These relatively important environmental stressors, therefore, could account for the patchy distributions of the fauna in samples obtained at these sites. Patch size may be dictated by the unstable nature of the environment due to the increased likelihood of larvae encountering unfavourable conditions and, therefore, being unable to become established. The apparent patchiness of the fauna is likely to be directly related to the variability (patchiness) of the seabed conditions. Rapid colonisers, suited to the dynamic conditions, were dominant at these sites and included polychaete such taxa as *Ophelia rathkei* and *Lagis koreni* (Lowestoft) or *Ampelisca spinipes* and *Lanice conchilega* (Shipwash Gabbard). The overall pattern at Shipwash Gabbard appeared to correspond very closely to that of the polychaetes, which accounted for approximately 40% of the total

recorded species. Tube-building species such as *Lanice conchilega* were recorded frequently. The presence of this latter species may represent an example of a biological factor exerting control over a wider scale than other biotic factors such as behaviour or scale of organism movement. Polychaete tube mats has been reported by Thrush *et al.* (1996) to be important in the prevention of erosion by the stabilising of sediments. It is probable that the abundant occurrence of *Lanice conchilega* in the Shipwash Gabbard region may act as a 'physical' variable in altering local sedimentary makeup. The patch size for molluscs recorded at Shipwash Gabbard was somewhat larger than that of other groupings and is a probable reflection of the relatively large body size of this group. At the Lowestoft site, crustaceans exhibited a smaller patch size than the overall fauna or that of the Polychaeta. *Ampelisca* was the dominant genus of this group, supporting the pattern identified at North Nab and Coal Pit. These findings accord with those of Zajac *et al.* (1998) who stated that environmental factors exert control of macrofaunal distributions over multiple scales. Life history and population processes are critical for benthic distributions at primarily meso-large scales, whereas biotic interactions have the most effect at small scales (Figure 67). This small-scale pattern was observed for the amphipods in the present study with high similarities being exhibited regardless of environmental conditions, implying that life-history/behavioural traits may be controlling factors. Hewitt *et al.* (2002) provided an example of a scenario in which the distributions of certain members of the benthos were controlled by the density and behaviour of one, locally abundant species (in this case the suspension-feeding bivalve *Atrina zelandica*). Thrush *et al.* (1997) noted that local processes interacting with extremes in physical variables often generated heterogeneities in ecological systems. For example, although sediment disturbance by individual deposit feeders is very localised, the sediment structure resulting from such behaviour can be altered and become

predisposed to resuspension and erosion during storms with the associated loss of fauna on much larger scales (Rhoads *et al.* 1977). Conversely, Thrush *et al.* (1996) reported that polychaete tube mats can reduce erosion by providing greater sediment stability. McArdle *et al.* (1997) found that local biological processes were not responsible for large-scale spatial patterns and that biologically-generated large-scale spatial patterns were unlikely to appear unless recruitment and/or mortality were strongly aggregative.

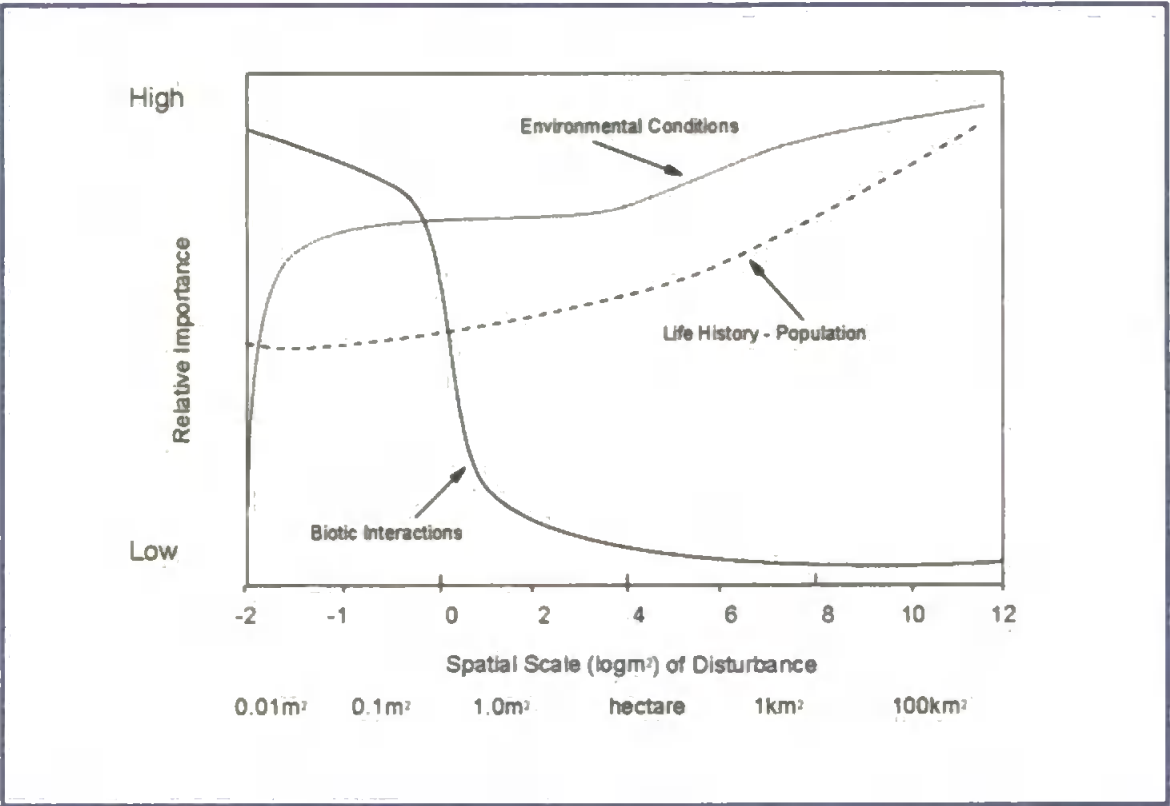


Figure 67. Predicted relative levels of importance of factors controlling successional dynamics over different scales of disturbance (redrawn from Zajac *et al.* 1998).

Results from the North Sea sites, examined at a far larger scale, imply that environmental conditions, and possibly suitable related life-history traits (illustrated by the dominance of '*r*-selected taxa'), were major controlling factors. These larger scale investigations imply, that in regions of environmental instability (i.e. mobile substrata and current-swept localities), patches were approximately 8-12km in diameter. Therefore, accurate 'baseline' assessments of the benthic resources

inhabiting such an area could be obtained by locating sample stations by up to 12km apart. This would eliminate the need for a sampling grid comprising large numbers of stations, and therefore, reducing time and labour costs. Indeed, Armonies (2000) suggested that community monitoring in the North Sea required a wide spatial sampling scale to reduce variability of abundance estimates. Over the past decade, spatial structure of ecological landscapes has been recognised increasingly as a functionally important component of ecological systems. The presence of spatial structuring in benthic communities, as demonstrated in the present study, has implications for the location of controls and replicate samples, particularly in large-scale investigations (Lindegarth *et al.* 2000; Hewitt *et al.* 2001). Investigations of spatial patterns such as the present study could be employed as an exploratory tool in designing sampling strategies that take into account the scale (of sampling) appropriate to sample the overall/common species characterising a given region (Hewitt *et al.* 1997). Impact assessment studies should take the factor of scale into account by noting that as spatial scale increases the number of processes and their importance will change, thus increasing the variability encountered by the study (Hewitt *et al.* 2001). Sampling strategies should ensure that arrays of variables, both biotic and abiotic, are monitored enabling a range of analytical methods to be used.

6.5. Summary

Multivariate patchiness was evident in the samples obtained from the North Sea sites (Shipwash Gabbard and Lowestoft) but not in the samples from the English Channel site (West Bassurelle). Environmental heterogeneity in the North Sea sites is a possible cause of patchiness, whereas, the uniformity of the substratum at West Bassurelle is a likely explanation for the absence of patchiness in the benthic invertebrate communities of this region.

Spatial patterns in different components of the benthos were evident in the benthic communities at North Nab and Coal Pit. Inter-sample differences occurred at smaller distances at the English Channel site (North Nab) compared with the samples obtained from the North Sea area (Coal Pit), such differences in patch size may reflect a greater habitat complexity provided by a heterogeneous substratum and the presence of epibenthic taxa such as hydroids and bryozoans which in turn provide refugia for other invertebrates. Such complexity was absent at Coal Pit leading to increased sample uniformity. Amphipods were present in similar patch sizes at each location and patches were separated by similar distances. The contrasting environmental conditions at these sites make it probable that patch size and separation in the amphipoda is mediated by behavioural/life history traits of this group.



CHAPTER 7. IMPACTS OF AGGREGATE EXTRACTION ON THE BENTHOS OF TWO ENVIRONMENTALLY CONTRASTING DREDGING AREAS

The aim of this chapter is to compare and contrast the macrobenthic responses following dredging at two environmentally distinct aggregate extraction areas.

CHAPTER 7. IMPACTS OF AGGREGATE EXTRACTION ON THE BENTHOS OF TWO ENVIRONMENTALLY CONTRASTING DREDGING AREAS

7.1. Introduction

With the increase in environmental pressure on land-based sources of aggregates and the lack of suitable recyclable resources, the demand for these materials must be satisfied by an alternative source, namely the seabed (BMAPA 2000). An estimated 70% of the continental shelf area is covered by mineable, relict sediments (Charlier & Charlier 1992). These deposits were laid down during the quaternary glaciations when fast-flowing rivers poured out across the, then dry, shelf (sea levels were approximately 100m lower than today). Following the last ice age (around 8000 years ago), these ancient river valleys were submerged as the sea level rose during the following warm interglacial period. With the continuing rise of the sea level, the deposits were concentrated in the vicinity of these former watercourses and, by the action of the sea, have been sorted. The gravel, being essentially immobile, has become locked in the old river terraces; conversely, the sands have been transported forming, in many cases, large banks. Charlier & Charlier (1992) noted that sands were predominant in areas where current speed ranged from 2.8-3.7 km/h (1.5 to 2 knots), and gravel deposits were most frequently located in regions of less than 50m depth with current speeds ranging from 3.7-7.4 km/h (2 to 4 knots). In many such areas, the gravel deposits are extensive enough to make large-scale extraction a viable economic proposition.

The origins of dredging for stones, sand and gravel date back to when sailing ships required ballast to lower their centre of gravity to improve stability. Sand and gravel was dredged from the beds of harbours and loaded into the unladen vessels, thus simultaneously clearing the channels. The dredging of sand and

gravel was first practised in the Thames in the 1550s; by the 17th and 18th centuries, it had developed into a significant maritime activity (BMAPA 1995). By the 1930s, the dredging industry served two main roles that persist today. Firstly the extraction of sand and gravel, generally in offshore areas, for supply to the construction industry, export and beach fill schemes. Secondly, for the maintenance of safe navigation channels in harbours. Harbour dredging produces an essentially worthless product that is disposed of at sea or, increasingly, used in 'beneficial use' schemes. In the United Kingdom today, sand and gravel reserves are mined from licenced areas mainly on the east and south coasts and from the Humber and Thames Rivers, with exploitation on a lesser scale occurring in the Bristol Channel, Liverpool Bay and the Mersey (BMAPA 2000). At the time of writing this thesis (2004), dredging licences cover approximately 1700 km² (0.12%) of the UK continental shelf (BMAPA 2000).

Approximately 21% of the sand and gravel used in England and Wales is now supplied by the marine aggregates industry. A large proportion of this material is utilized by the construction industry, predominantly in ready-mixed concrete and associated products. Around one third of the marine aggregates extracted from UK waters are exported to the continent, mainly to the Netherlands. Indeed, Amsterdam's Schiphol airport was constructed from aggregate mined in Norfolk (Pearce 1996). Aggregates are also sold to France, Belgium and Germany; the remainder is implemented in beach nourishment schemes aimed at replacing material lost to coastal erosion (Figure 68).

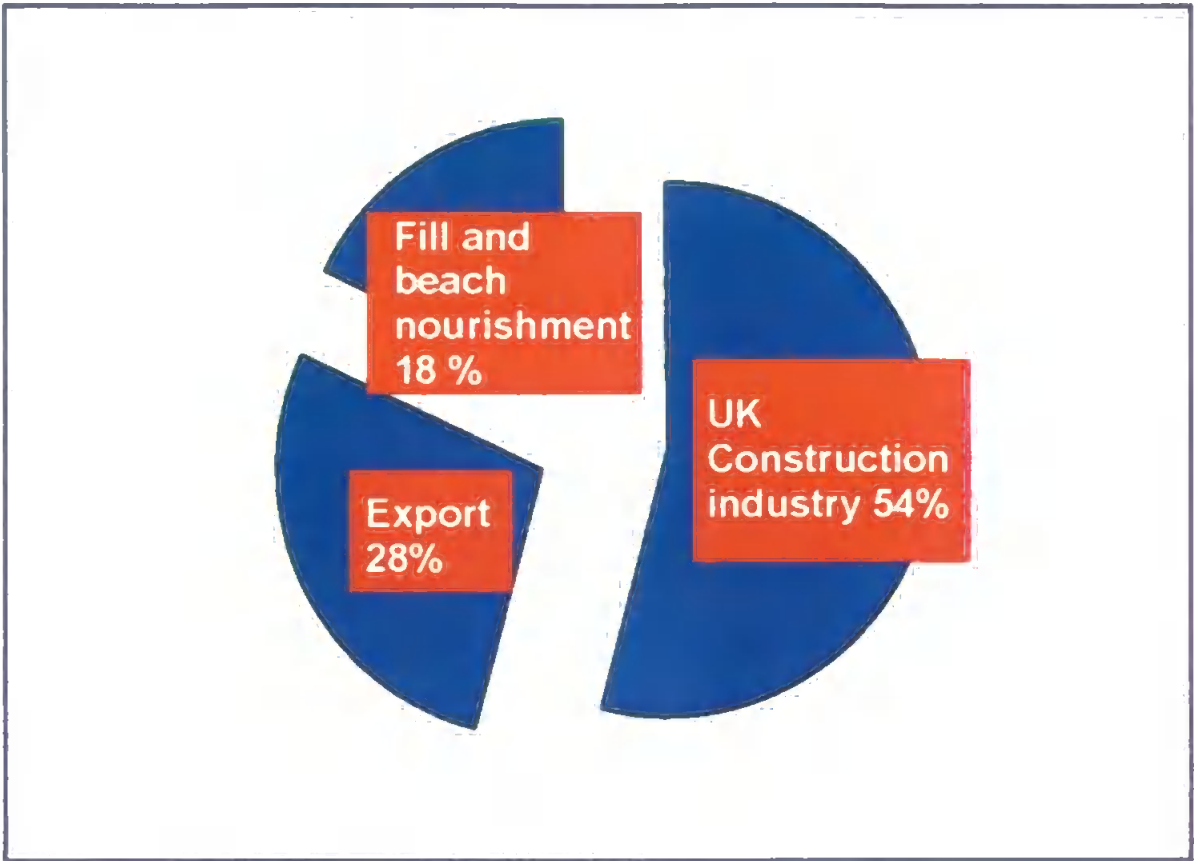


Figure 68. Uses of marine aggregates mined in UK waters (BMAPA 2000).



Figure 69. Trailer dredger (Arco Humber) operating in the North Sea, showing material rejected overboard after screening.

Currently, a fleet of around fifty dredgers operates in the UK. Larger vessels can load around 5000 tonnes of sand and gravel in approximately three hours but the actual quantities processed are far higher. In areas where overboard screening takes place, approximately 8000 tonnes of fine material is rejected for a cargo of this size (Hitchcock & Drucker 1996) (Figure 69). There are two major techniques employed by the dredgers working in UK coastal deposits: anchor dredging and trailer dredging. Usually, the anchor dredger is employed in areas where the deposits are thick and localised. The vessel remains stationary other than periodic movements to winch itself a few meters to maintain the flow of gravel. The trailer dredger operates whilst underway at speeds of up to 3 knots and a pipe is trailed along the sea floor. This method is suited to shallower, more evenly distributed, deposits (Figure 70).

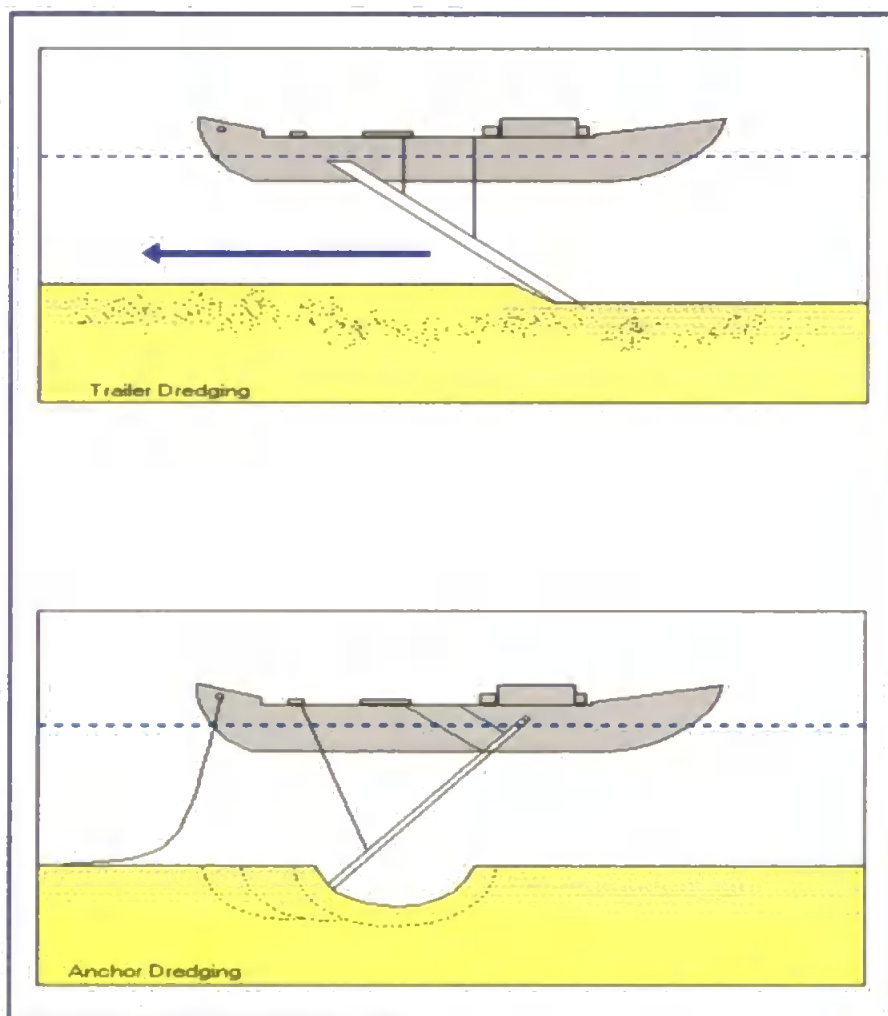


Figure 70. Diagrams showing Trailer and Anchor dredgers (redrawn from BMAPA 2000).

In recent times, The Global Convention on Biodiversity, the European Union's Habitats Directive and updates to the Oslo and Paris Convention have led to an increased focus on the conservation of marine biodiversity (Posford Duvivier Environment & Hill 2001). This increased attention has led to the development of policies designed to protect marine resources from increasing environmental pressures, including the extraction of marine aggregates. Current Government policy is designed to ensure continued use of marine dredged minerals to the extent that sustainable development is not compromised. The policy must provide sufficient access to resources to allow the dredging industry to meet market demands, whilst simultaneously preventing environmental damage attributable to dredging operations. The government aims to meet this objective (ODPM 2002) through the following actions:

- minimising the total area licensed/permitted for dredging;
- the careful location of new dredging areas;
- considering all new application in relation to the findings of an Environmental Impact Assessment (EIA) where such an assessment is required;
- adopting dredging practices that minimise the impact of dredging;
- requiring operators to monitor, as appropriate, the environmental impact of their activities during, and on completion of, dredging; and
- controlling dredging operations through the use of conditions attached to the dredging licence or dredging permission.

Essentially, the environmental impacts of marine aggregate extraction depend partly on the method of dredging employed and partly on the amount of material rejected by overboard screening and overspill from the hopper. In terms of physical disturbance, anchor dredging affects deposits at a series of points on the sea floor leaving pits or depressions. The dredged pits can attain 20 m or more in depth and measure up to 75 m in diameter. The effect of trailer dredging on the seabed topography is characterised by a series of dredged tracks usually around 2-3 m wide and up to 0.5 m deep, however, in areas subjected to intense trailer dredging, these tracks intersect one another and the seabed can be lowered by

2 m or more. Such physical impacts attributed to dredging will therefore lead to unavoidable impacts on the biological resources of the mined area as no benthos is likely to occur below the dredged depth.

A further concern is the possible impact of the settlement of temporarily suspended fine material in the vicinity of the worked area. Early predictive models estimated that, under worst-case conditions, these sediments could remain in suspension for 4-5 tidal cycles and be transported as far as 20 km to each side of a point source of discharge (Newell *et al.* 1998). The environmental implications of these suspensions are not well known. Hitchcock & Drucker (1996) demonstrated that most sand-sized particles settled within 500m of the dredger. Thus burial of benthic invertebrates occupying this zone is a primary concern. Conversely, other studies have provided evidence that the release of organic material in such 'sediment plumes' can lead to an enhancement of the benthos (Jones & Candy 1981; Poiner & Kennedy 1984).

7.2. Materials & Methods

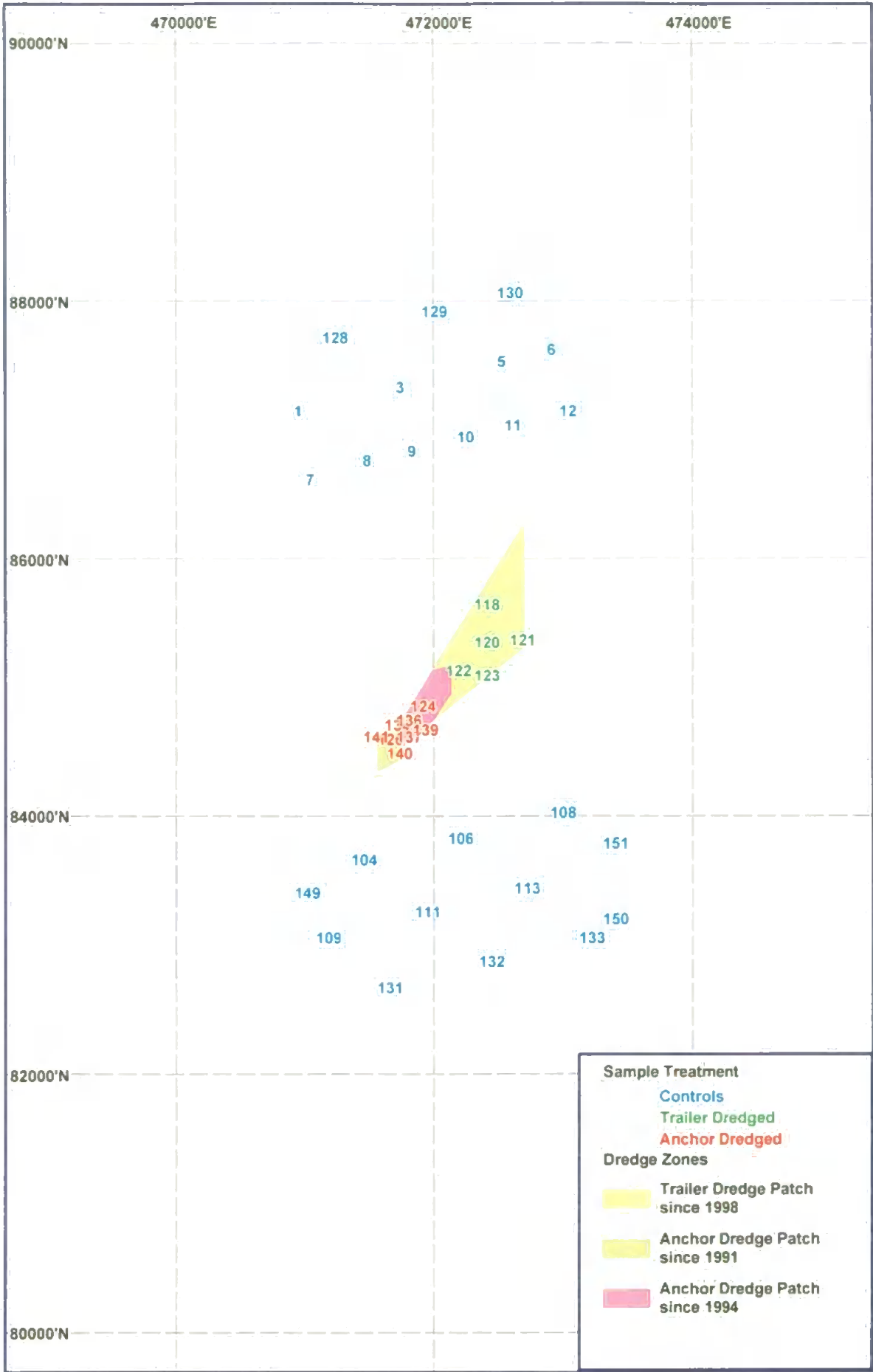
7.2.1. Sample Positions

7.2.1.1. North Nab

Stations were sampled to allow comparison between anchor dredged, trailer dredged and control stations (Figure 71).

7.2.1.2. Coal Pit

Samples were positioned to allow comparison between trailer dredged, abandoned and control areas (Figure 72).



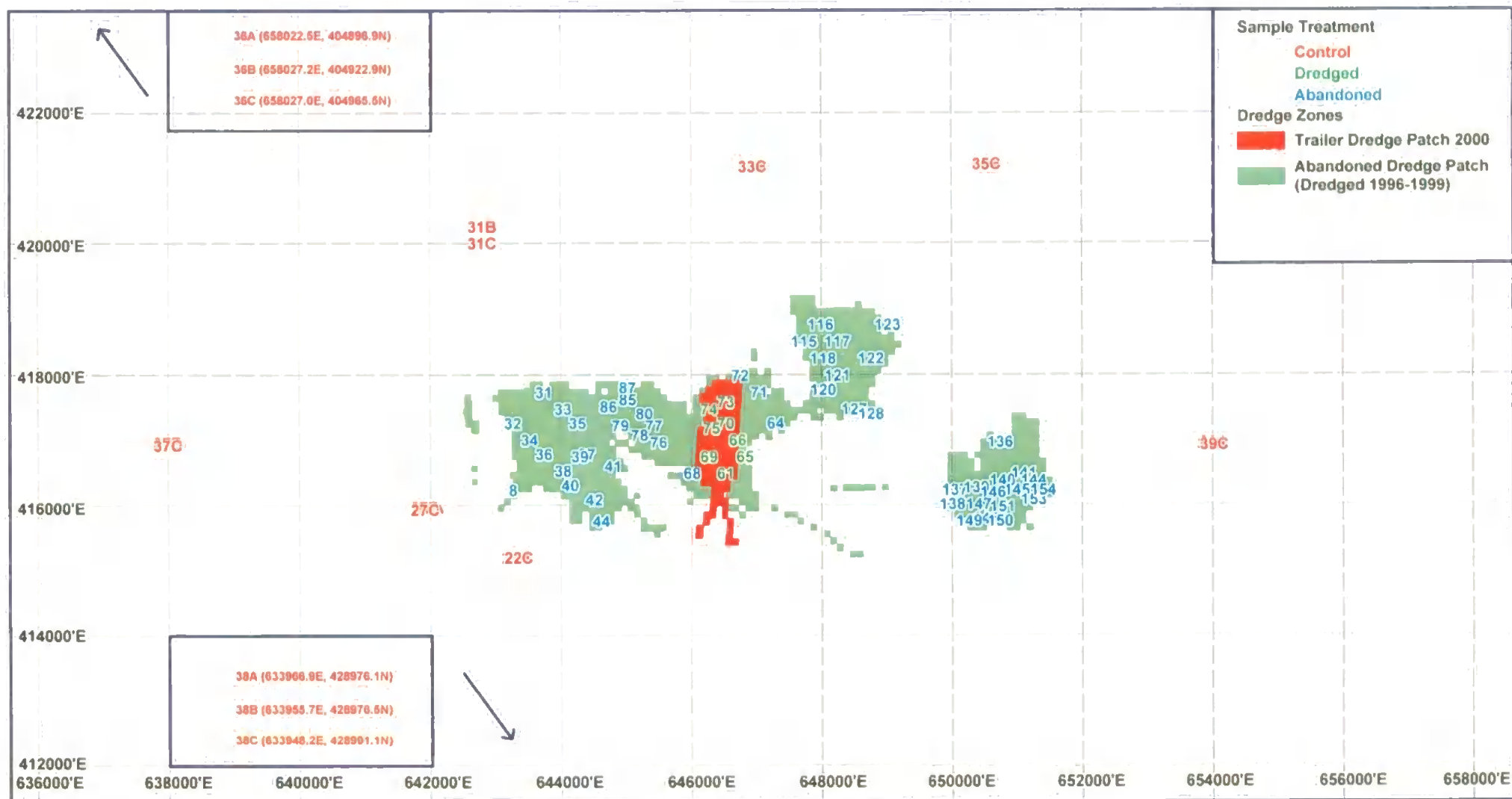


Figure 72. Detailed view of Control, Dredged and Abandoned stations sampled in the Coal Pit study area.

7.2.2. Analytical Methods

7.2.2.1. Univariate Analysis

Histograms were plotted of biomass and body-size. One-way ANOVA (analysis of variance) tests were performed in order to check for significant differences between control, Anchor dredged and trailer dredged sites at North Nab and between control, trailer dredged and abandoned sites at Coal Pit. These tests were performed for total biomass (log B) and body size (log B/N) and for a range of diversity indices including; number of species (log S), number of individuals (log N), species richness (d), evenness (J) and species diversity (H'). A *posteriori* power analysis (Clarke & Green 1988) was carried out for each test. This was done to calculate the statistical power of the ANOVA test or, conversely, to detect the likelihood of a type II error i.e. the probability of failing to detect a difference (in this case a 10% difference) when one actually exists.

7.2.2.2. Multivariate Analysis

Pooled species abundance data (excluding colonial taxa), recorded in samples collected from control, anchor dredged and trailer dredged sites (North Nab) and from control, trailer dredged and abandoned sites (Coal Pit), was used to construct data matrices to be analysed by multivariate techniques. Multivariate analysis was carried out using the PRIMER software package (Clarke & Warwick 1994, Clarke & Gorley 2001) and the following routines were employed.

(a) Multidimensional Scaling Ordination (MDS): this technique allows the construction of a 'map' or configuration of the samples in multi-dimensional space. This configuration attempts to position the samples as accurately as possible to reflect the similarity between the samples. For example, if sample 1 has a greater similarity to sample 2 than it does to sample 3 then sample 1 will be positioned

more closely to sample 2 than it is to sample 3. This 'map' of the relative similarities between samples is plotted in two dimensions. It is important to remember that this two-dimensional plot is a representation of a multi-dimensional picture. When numbers of samples are analysed, or datasets that include samples that are very different from other samples, the accuracy of the two-dimensional plot may be reduced, a measure of how accurate the two-dimensional representation is (stress) is given on the MDS plot. Stress values <0.1 correspond to a good ordination, values <0.2 gives a useful two-dimensional picture but too much reliance on the fine details of the plot should not be placed, stress >0.3 indicates that the samples are close to be positioned in an arbitrary manner and should be treated with caution, particularly in the upper half of this range.

(b) ANOSIM (Analysis of Similarities): this procedure was used to test the null hypothesis that there were no differences in community composition between the five study sites featured in the present investigation. This routine goes through three main stages in the examination of H_0 , these are:

(i). The calculation of the ANOSIM statistic from the dataset: this reflects the observed differences *between* sites, contrasted with differences *within* sites. The test is based upon the corresponding (rank) similarities between samples. The test statistic R is calculated using the equation:

$$R = (\bar{r}_B - \bar{r}_W)/(M/2)$$

Where \bar{r}_W is the average of all rank similarities among replicates *within* sites, \bar{r}_B is defined as the average of rank similarities from all pairs of samples from *between* sites, and $M = n(n-1)/2$, where n is the total number of samples in the dataset. R usually lies between 1 and 0; $R = 1$ if all replicates within a particular site are more similar to one another than any sample from a different site. If R is approximately 0, there are no differences between the sites, therefore, between and within site similarities

are the same on average. Technically, R can be as low as -1 although this is unlikely, as it would indicate that within site similarities were less than between site similarities. The R statistic can be used also to give an indication of the degree of separation between the sites. Well-separated sites exhibit R values greater than 0.75, sites that are clearly separable but show some overlap display R values in the region of 0.5, and sites that are barely separable give R values of 0.25 and below. It should be noted that R can be very small and still be significantly different from zero.

(ii). Recalculation of the R statistic under permutations of the sample labels: Using the null hypothesis of no differences between sites, there is likely to be little effect on average to the R value if the sample labels belonging to each site are arbitrarily reshuffled, because if H_0 is true then all samples are merely replicates of a single site. Therefore, the permutation tests basically consist of rearranging the sample labels, recalculating the R statistic and repeating a large number of times.

(iii). The final stage is the calculation of the significance level: This is carried out by referring the observed value of R , calculated from the 'original' data, to the distribution of R as simulated by permutation. If the observed value of R is unlikely to have come from this distribution, this is evidence to suggest that H_0 should be rejected.

(c) SIMPER (Species Contributions to Similarity): this routine allows comparison between groups of samples from one site to another. Species responsible for the dissimilarity between the two sites are listed in decreasing order of importance in the discrimination of the two regions. Values relating to the contribution to the dissimilarity between the sample groups, and the average abundance of a given taxon at each site, are shown. This routine also provides information on which species are responsible for the within-site similarities and their contribution to the internal similarity of the group.

(d) *k*-dominance curves: these plot cumulative ranked abundance against species rank, or log species rank. This procedure has the effect of smoothing the curves. The more elevated the starting point of a curve, the more dominated the group of samples it represents and, therefore, the greater the level of biological 'stress' associated with the sample group.

(e) Production Meta-analysis: A multivariate method described by Warwick & Clarke (1993) and Warwick (1993) allows comparisons between samples obtained from regions with contrasting environmental conditions. At the Phylum level, meta-analysis eliminates the large variability found between data sets of species level data (by using data aggregated to the phyletic level). Such data sets, which exploit the increase in similarity of benthic communities when examined at higher levels of taxonomic resolution, have been shown by Warwick & Clarke (1993) to be amenable to multivariate analyses. Both abundance and biomass data are first aggregated to the phylum level and are then treated to represent the approximate production of each phylum using the equation:

$$(B/A)^{0.73} \times A$$

where B = biomass (AFDW/g), A = abundance (0.73 being the average exponent of the regression of annual production for macrobenthic invertebrates) (Warwick 1993; Clarke & Warwick 1994). These data [(Appendix L for North Nab (March/September 1999); Appendix LI for Coal Pit (July/August 2000))] can be used to construct a similarity matrix, which in turn forms the basis for MDS plots. For the present study from North Nab and Coal Pit survey regions were combined to determine if any changes in the production levels of different phyla can be attributed to the mining of marine aggregates.

7.3. Results

7.3.1. North Nab

Figure 71 shows in detail the positions of samples in relation to the exploited areas. Stations subjected to anchor dredging possessed fewer species than trailer dredged sites, non-dredged stations and controls (Figure 38), although not as low as the sandy stations in the non-dredged eastern region. Macrofaunal abundances at stations subject to different levels of dredging disturbance were comparable (Figure 39). The actively dredged region was characterised by reduced biomass at most sites but no more so than many non-dredged stations (Figure 40). High biomass levels were recorded at many stations surrounding the dredged area, particularly to the northwest and southwest. Trailer-dredged regions had comparable numbers of species, numbers of individuals and biomass values to control stations. A suppression of biomass was evident at anchor dredged sites, when pooled averages were compared a 76% reduction in biomass and a 32% reduction in number of species was noted.

One-way ANOVA tests (using pooled samples from control ($n=25$), anchor ($n=8$) and trailer dredged areas ($n=5$)) were performed for number of species ($\log S$), number of individuals ($\log N$), species richness (d), evenness (J), species diversity (H'), total biomass ($\log B$) body size ($\log B/N$) and biomass of major phyletic groups. Significant differences were identified for species richness (d) and species diversity (H'), however, a posteriori power analysis revealed that both tests did not have sufficient statistical power to accurately test for such differences (see Table 13). Therefore, the null hypothesis that there was no significant difference between the treatments cannot be rejected. In addition, it is not possible to state if there is no significant difference between the sites, as the tests performed did not have sufficient power to detect any differences that may or may not exist.

Table 13. Table summarising the results of a one-way ANOVA of number of species, number of, species richness (d), species evenness (Pielou's J), Shannon-Weiner diversity (H(log_e)), Simpson's dominance, total biomass and biomass of major groups. Comparisons are made between individuals control, anchor and trailer dredged stations for the North Nab study area. Results of a posteriori power analysis are also shown.

Log Species

Source of variation	df	SS	MS	F	P-value
Among groups	2	0.1349637	0.0674818	2.1677271	0.1295591
Within groups	35	1.0895579	0.0311302		
Total	37	1.2245216			
Power = 0.25					

Log Abundance

Source of variation	df	SS	MS	F	P-value
Among groups	2	0.0006879	0.0003439	0.0028714	0.997133
Within groups	35	4.1922404	0.1197783		
Total	37	4.1929283			
Power = 0.55					

Richness

Source of variation	df	SS	MS	F	P-value
Among groups	2	21.44762	10.72381	3.4333706	0.0434991
Within groups	35	109.3192	3.1234058		
Total	37	130.76682			
Power = 0.1					

Evenness

Source of variation	df	SS	MS	F	P-value
Among groups	2	0.1389146	0.0694573	2.222828	0.1233687
Within groups	35	1.0936541	0.0312473		
Total	37	1.2325687			
Power = 0.13					

Species Diversity

Source of variation	df	SS	MS	F	P-value
Among groups	2	3.0125423	1.5062711	4.1208781	0.0247106
Within groups	35	12.793266	0.3655219		
Total	37	15.805808			
Power = 0.12					

Simpson's dominance

Source of variation	df	SS	MS	F	P-value
Among groups	2	0.1915565	0.0957783	2.6617932	0.0839278
Within groups	35	1.2593914	0.0359826		
Total	37	1.4509479			
Power = 0.14					

Log Biomass

Source of variation	df	SS	MS	F	P-value
Among groups	2	0.9972853	0.4986427	1.545395	0.2274265
Within groups	35	11.293225	0.3226636		
Total	37	12.290511			
Power = 0.06					

Log Body Size

Source of variation	df	SS	MS	F	P-value
Among groups	2	0.9512353	0.4756177	3.2279938	0.0516908
Within groups	35	5.1569548	0.1473416		
Total	37	6.1081902			
Power = 0.08					

Log Polychaeta Biomass

Source of variation	df	SS	MS	F	P-value
Among groups	2	0.9931789	0.4965894	1.3437696	0.2739868
Within groups	35	12.934234	0.3695496		
Total	37	13.927413			
Power = 0.05					

Log Mollusca Biomass

Source of variation	df	SS	MS	F	P-value
Among groups	2	0.2415418	0.1207709	0.1623244	0.8508017
Within groups	35	26.040342	0.7440098		
Total	37	26.281884			
Power = 0.04					

Log Crustacea Biomass

Source of variation	df	SS	MS	F	P-value
Among groups	2	1.1403972	0.5701986	0.7930426	0.4604281
Within groups	35	25.165042	0.7190012		
Total	37	26.305439			
Power = 0.04					

Log Miscellania Biomass

Source of variation	df	SS	MS	F	P-value
Among groups	2	3.2407488	1.6203744	2.5862599	0.0896256
Within groups	35	21.928618	0.6265319		
Total	37	25.169366			
Power = 0.04					

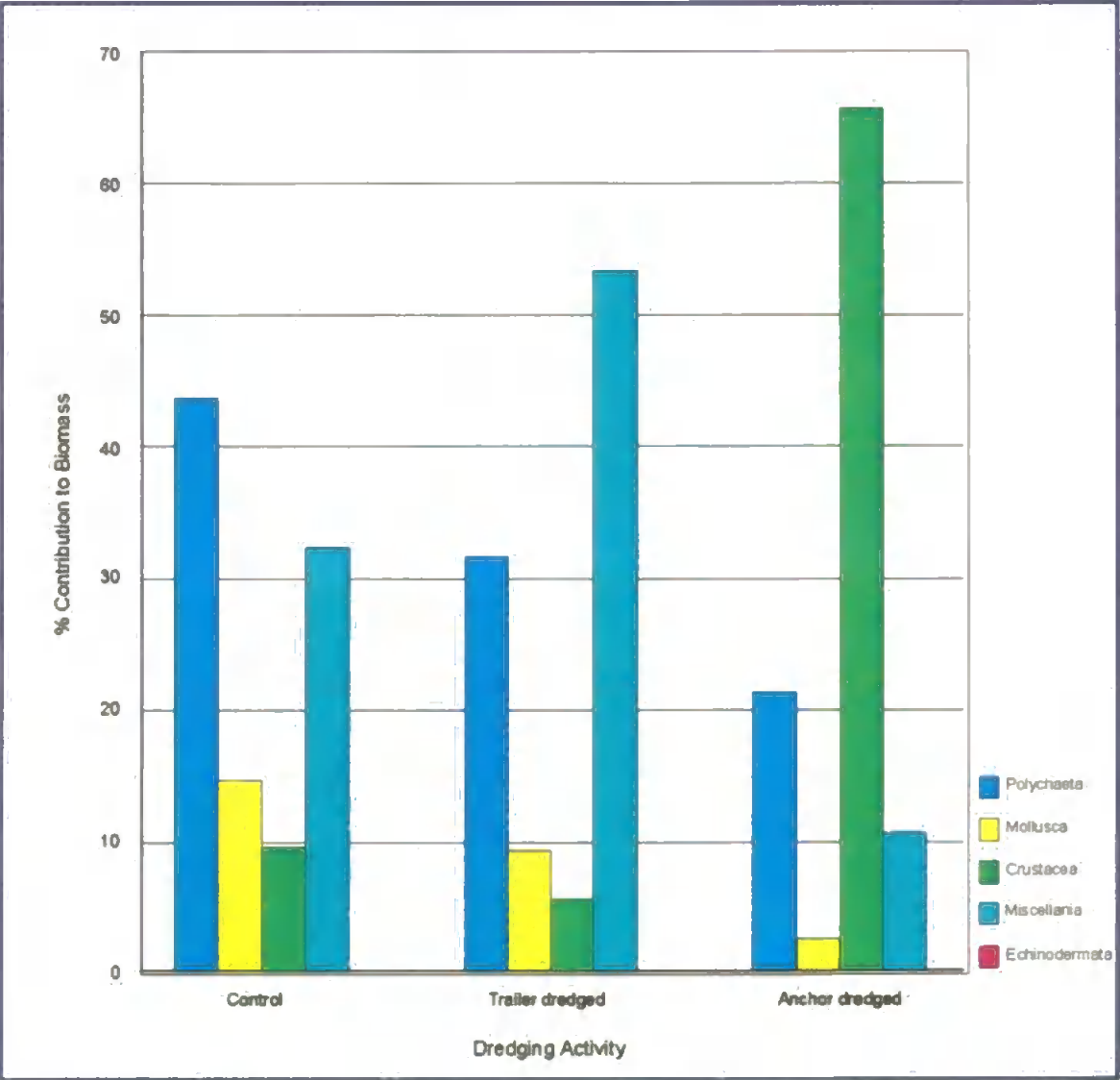


Figure 73. Percentage contribution to the total biomass by various taxonomic groups in stations subjected to different dredging pressures in the North Nab study area. Control stations are those assumed to be unaffected by dredging activity.

Figure 73 shows the relative contributions of the faunal groups to total recorded biomass at stations subject to various levels of disturbance attributable to aggregate mining. Analysis of variance (ANOVA) on the log-transformed biomass of the different faunal groups revealed that the differences were not significant. The low power of the ANOVA test does not allow acceptance of the null hypothesis (Table 13).

There was a reduction in the mean size of individuals in anchor-dredged samples (Figure 74). Body size in trailer-dredged stations also appeared to be reduced compared to control samples. ANOVA (Table 13) revealed these differences were not significant. Once again, this test did not have sufficient power to detect differences and, therefore, the null hypothesis could not be accepted.

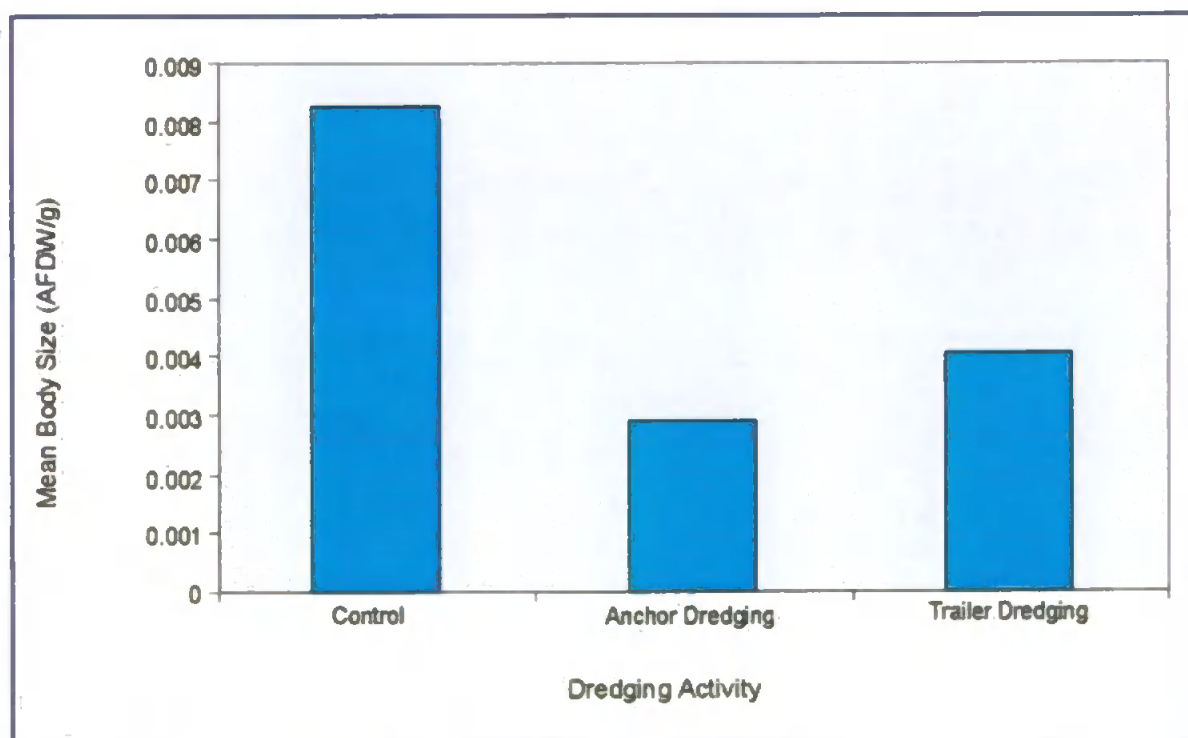


Figure 74. The mean size of individuals (AFDW/g) recorded at stations subjected to different dredging pressures in the North Nab study area. Control stations are those assumed to be unaffected by dredging activity.

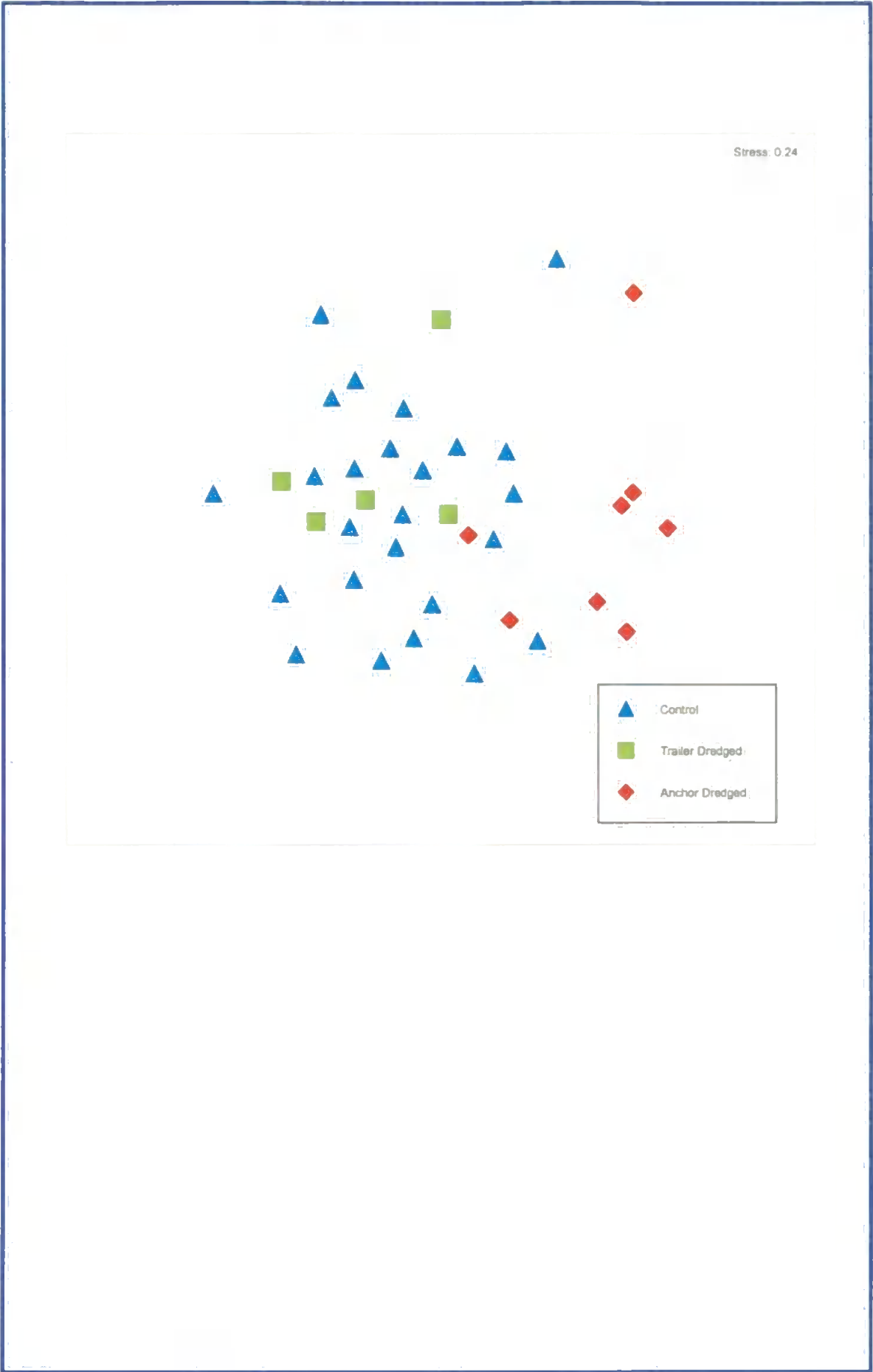
The low statistical power of the ANOVA tests (carried out on a range of univariate measures), demonstrated that to detect any apparent impacts of dredging on such

indices a greater number of samples would need to be taken in each treatment. The data were therefore more amenable to analysis by multivariate techniques.

From the MDS plot (Figure 75) it can be seen that trailer-dredged sites were not readily separable from control stations. Therefore, it may be assumed that trailer dredging has had no significant effect on the community structure (based on species abundances) of the benthos. Some stations subjected to anchor dredging appeared to pull away from the main group of control and trailer-dredged samples, which may be possible evidence of an impact on community structure attributable to anchor dredging. The index of multivariate dispersion (see Appendix XLIII) did not show the anchor-dredged samples to have a higher value than the other treatments, indicating that the former samples did not exhibit increased variability and, therefore, elevated stress (see Warwick & Clarke 1993).

A one-way ANOSIM test between controls, trailer dredged and anchor dredged samples (for full results refer to Appendix XLIV) gave a global R-value of 0.256 (significance level of 0.5%) which indicated an overall significant difference. Pairwise test comparisons between the groups revealed significant differences between anchor dredged and control stations ($R=0.421$ with a significance level of 0%) and also between anchor and trailer dredged samples ($R=0.516$ at the 0.2%) level. No significant difference was observed between samples from control sites and those obtained from trailer-dredged areas ($R = -0.096$, $P = 74.3\%$).

k-dominance curves demonstrated that stations subjected to anchor dredging were highly dominated (approx. 72%) by a single species; in contrast, trailer-dredged sites were less discriminated by one species (around 53%) and control stations exhibited lower dominance values (Figure 76).



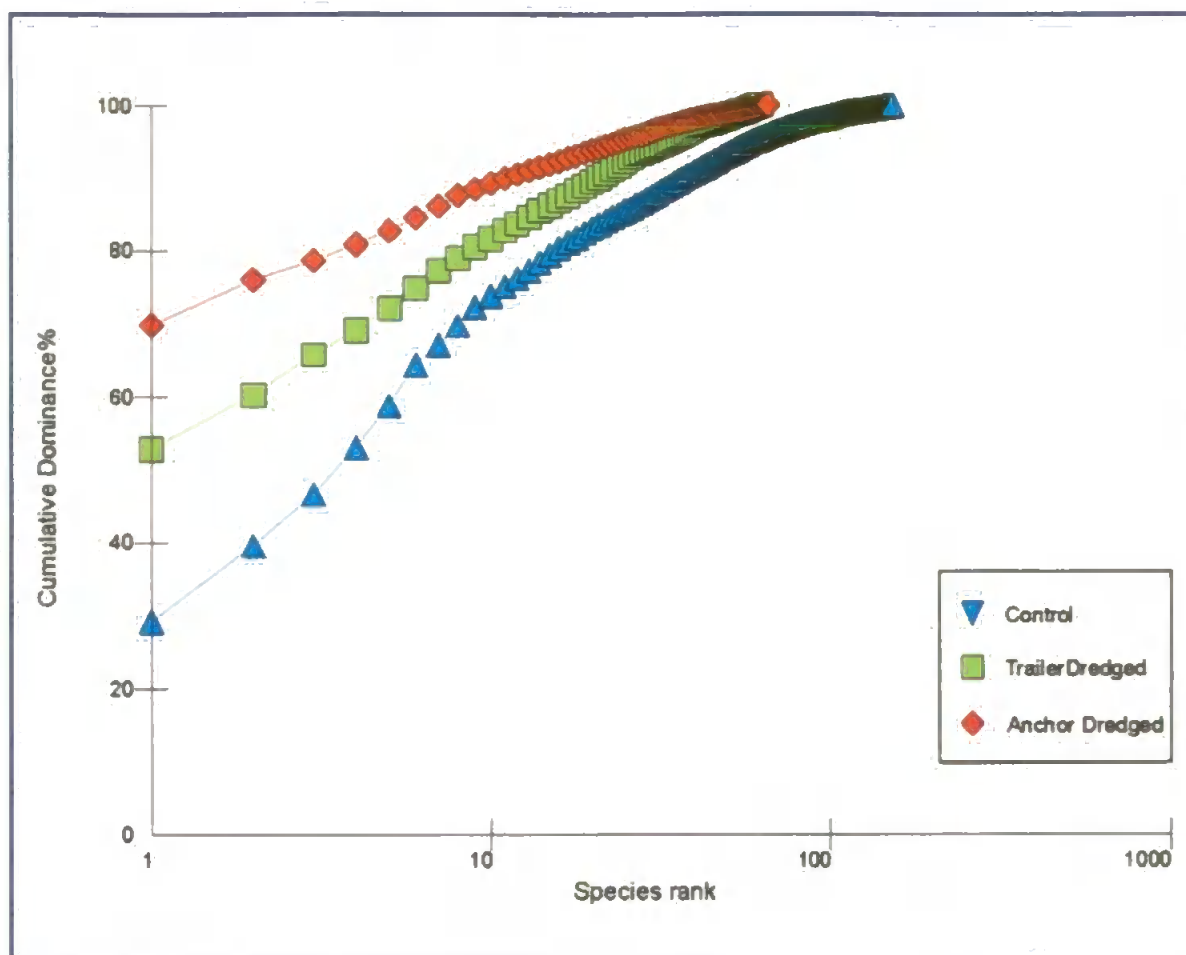


Figure 76. Cumulative dominance curves plotted for the benthic macrofauna in stations subjected to different dredging pressures in the North Nab study area. Control stations are those assumed to be unaffected by dredging activity.

The percentage contribution of species in three categories of stations grouped according to dredging activity was calculated by SIMPER analysis (for full results refer to Appendix XLV). It can be seen clearly that the component species within the anchor-dredged category was significantly altered compared to the other groups. Anchor-dredged sites were characterised by the presence of large numbers of the barnacle *Balanus balanus*, whereas control stations had *Distomus variolosus* (Chordata) and *Pomatoceros* sp. (Annelida) as the main contributors to the internal similarity of the stations within the group. Trailer-dredged stations, as indicated by the MDS plot (Figure 75) and ANOSIM test (Appendix XLIV), were characterised by a similar contributing species to the control sites.

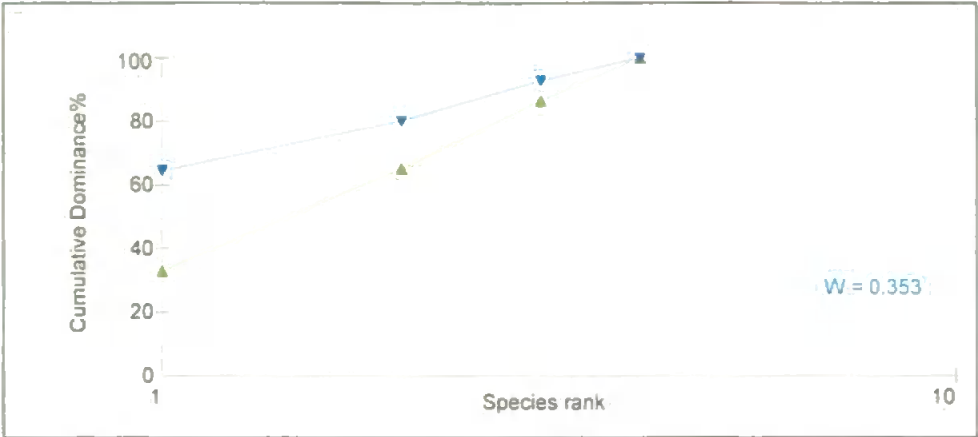
For control samples, the biomass curve (coded blue) was above the abundance curve (coded green), indicating the presence of an 'undisturbed' community (Figure 77). In contrast, the plots depicting the samples from anchor- and trailer-dredged sites showed the abundance curve to intersect the biomass curve, portraying a 'moderately disturbed' situation. The slightly negative W values confirmed this subtle disruption to the benthos.

Figure 78 depicts the number of individuals belonging to various phyletic groups in relation to a transect running from north to south through two stations known to have been dredged within one week of sample collection. It can be seen that, with the exception of annelids (polychaetes), the number of individuals of crustaceans, molluscs and chordates (mainly comprising *Distomus variolosus*) decreased dramatically in the dredged zone before rising once more to the south of the extraction area.

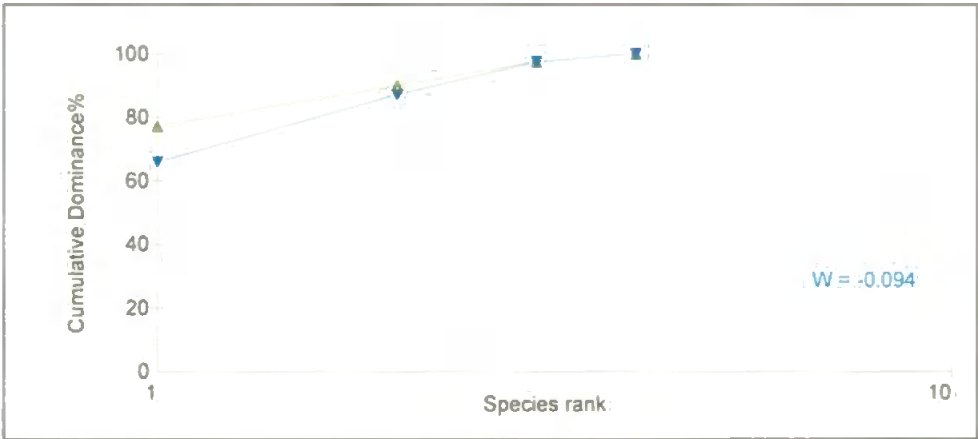
7.3.2. Coal Pit

Figure 72 shows in detail the positions of samples in relation to the exploited areas. Inspection of Figures 47 & 48 highlights the lack of any dredging impact on the number of species and the number of individuals, with dredged and abandoned areas showing similar values to control samples. In contrast, biomass values were low in the vicinity of the dredging area and in many parts of the abandoned sectors; however, suppressed biomass levels were noted in many areas situated well away from any potential dredging impact (Figure 49).

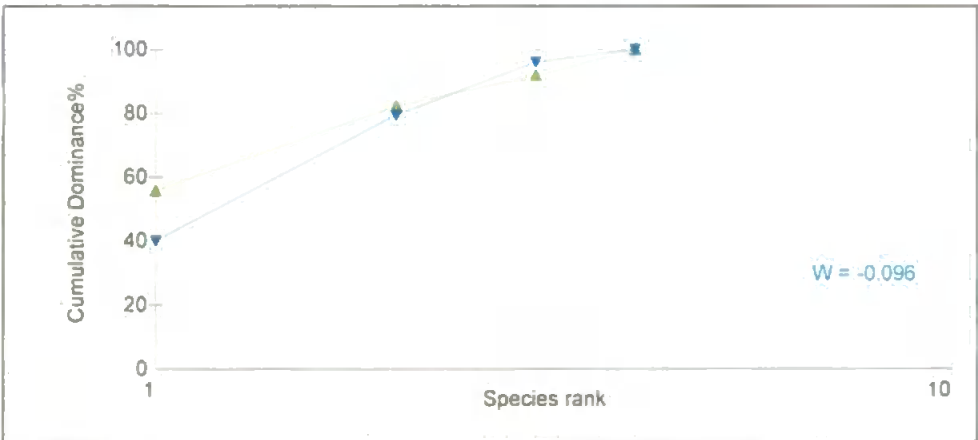
Table 14 gives the results of one-way ANOVA tests, using samples from control (n=27), abandoned (n=52) and dredged (n=8) areas, that were performed for numbers of species (log S), numbers of individuals (log N), species richness (d),



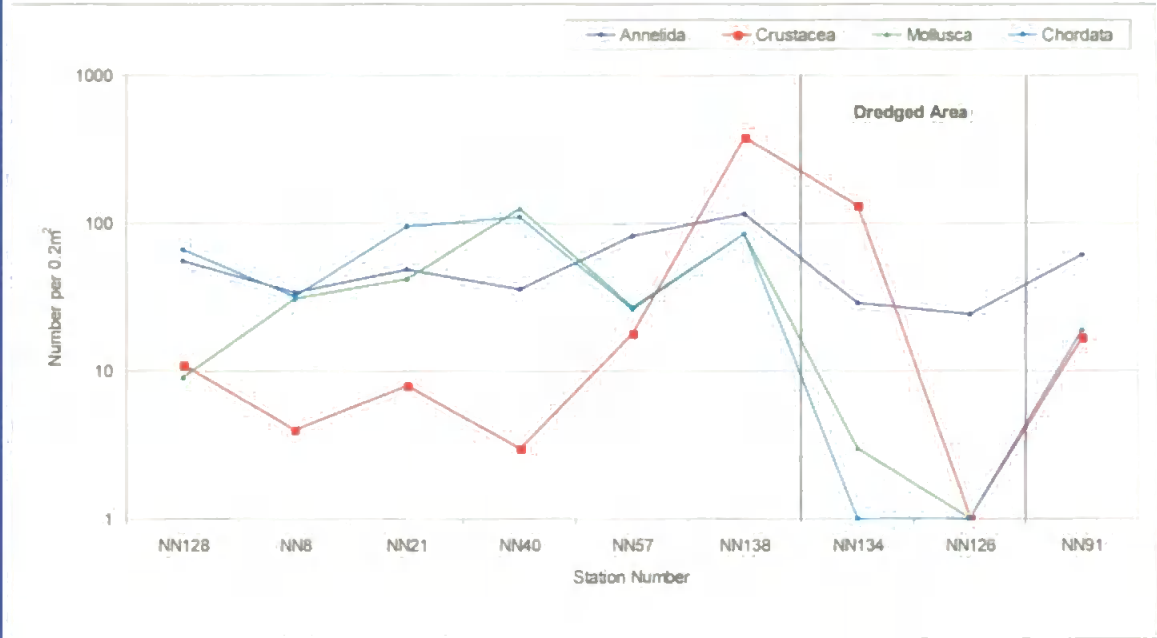
Control



Anchor Dredged



Trailer Dredged



evenness (J), dominance (λ) and species diversity (H'). Significant differences were not observed. A Power analysis showed that the statistical power of the ANOVA tests performed for log S, log N, d, J and H' was not sufficient to accept the null hypothesis. The ANOVA test for dominance had sufficient power to detect a 10% difference 96% of the time, therefore, in terms of dominance; the null hypothesis of no difference between the groups can be accepted.

Table 14. Table summarising the results of a one-way ANOVA of number of species, number of individuals, species richness (d), species evenness (Pielou's J), Shannon-Weiner diversity ($H(\log_e)$), Simpson's dominance, total biomass and biomass of major groups. Comparisons are made between control, anchor and trailer dredged stations for the Coal Pit study area. Results of a posteriori power analysis are also shown.

Log Species

Source of variation	df	SS	MS	F	P-value
Among groups	2	0.0670965	0.0335482	1.5827057	0.2114824
Within groups	84	1.7805278	0.0211968		
Total	86	1.8476243			
Power = 0.68					

Log Abundance

Source of variation	df	SS	MS	F	P-value
Among groups	2	0.4177467	0.2088733	2.8339586	0.0644129
Within groups	84	6.1911138	0.0737037		
Total	86	6.6088604			
Power = 0.17					

Richness

Source of variation	df	SS	MS	F	P-value
Among groups	2	1.4608867	0.7304434	0.2496637	0.7796388
Within groups	84	245.75959	2.9257094		
Total	86	247.22047			
Power = 0.3					

Evenness

Source of variation	df	SS	MS	F	P-value
Among groups	2	0.0367728	0.0183864	1.7509978	0.1798771
Within groups	84	0.8820443	0.0105005		
Total	86	0.9188171			
Power = 0.77					

H'

Source of variation	df	SS	MS	F	P-value
Among groups	2	0.1203308	0.0601654	0.3488868	0.7064904
Within groups	84	14.485768	0.1724496		
Total	86	14.606099			
Power = 0.48					

Simpson's

Source of variation	df	SS	MS	F	P-value
Among groups	2	0.0088081	0.0044041	0.5710913	0.5670865
Within groups	84	0.6477777	0.0077116		
Total	86	0.6565858			
Power = 0.96					

Log Biomass

Source of variation	df	SS	MS	F	P-value
Among groups	2	1.0027073	0.5013537	1.6533876	0.1975678
Within groups	84	25.471164	0.3032281		
Total	86	26.473871			
Power = 0.07					

Log Size

Source of variation	df	SS	MS	F	P-value
Among groups	2	2.139904	1.069952	3.5253556	0.0338711
Within groups	84	25.494157	0.3035019		
Total	86	27.634061			
Power = 0.09					

Log Polychaeta Biomass

Source of variation	df	SS	MS	F	P-value
Among groups	2	0.4853601	0.24268	1.817358	0.1687836
Within groups	84	11.2169	0.1335345		
Total	86	11.702261			
Power = 0.15					

Log Crustacea Biomass

Source of variation	df	SS	MS	F	P-value
Among groups	2	3.453735	1.7268675	1.7744329	0.1758766
Within groups	84	81.748298	0.973194		
Total	86	85.202033			
Power = 0.05					

Log Miscellania Biomass

Source of variation	df	SS	MS	F	P-value
Among groups	2	5.7043192	2.8521596	3.4408203	0.0366211
Within groups	84	69.629153	0.8289185		
Total	86	75.333472			
Power = 0.05					

Log Mollusca Biomass

Source of variation	df	SS	MS	F	P-value
Among groups	2	1.2904903	0.6452452	0.5325918	0.5890506
Within groups	84	101.76761	1.2115192		
Total	86	103.0581			
Power = 0.05					

Log Echinodermata Biomass

Source of variation	df	SS	MS	F	P-value
Among groups	2	13.238926	6.6194631	4.4931393	0.0140003
Within groups	84	123.75198	1.4732379		
Total	86	136.99091			
Power = 0.04					

Control stations (mainly sandy deposits) appeared to have fewer molluscs than the coarser substrata of the abandoned sites but were otherwise comparable in the contribution of the other faunal groupings (Figure 79).

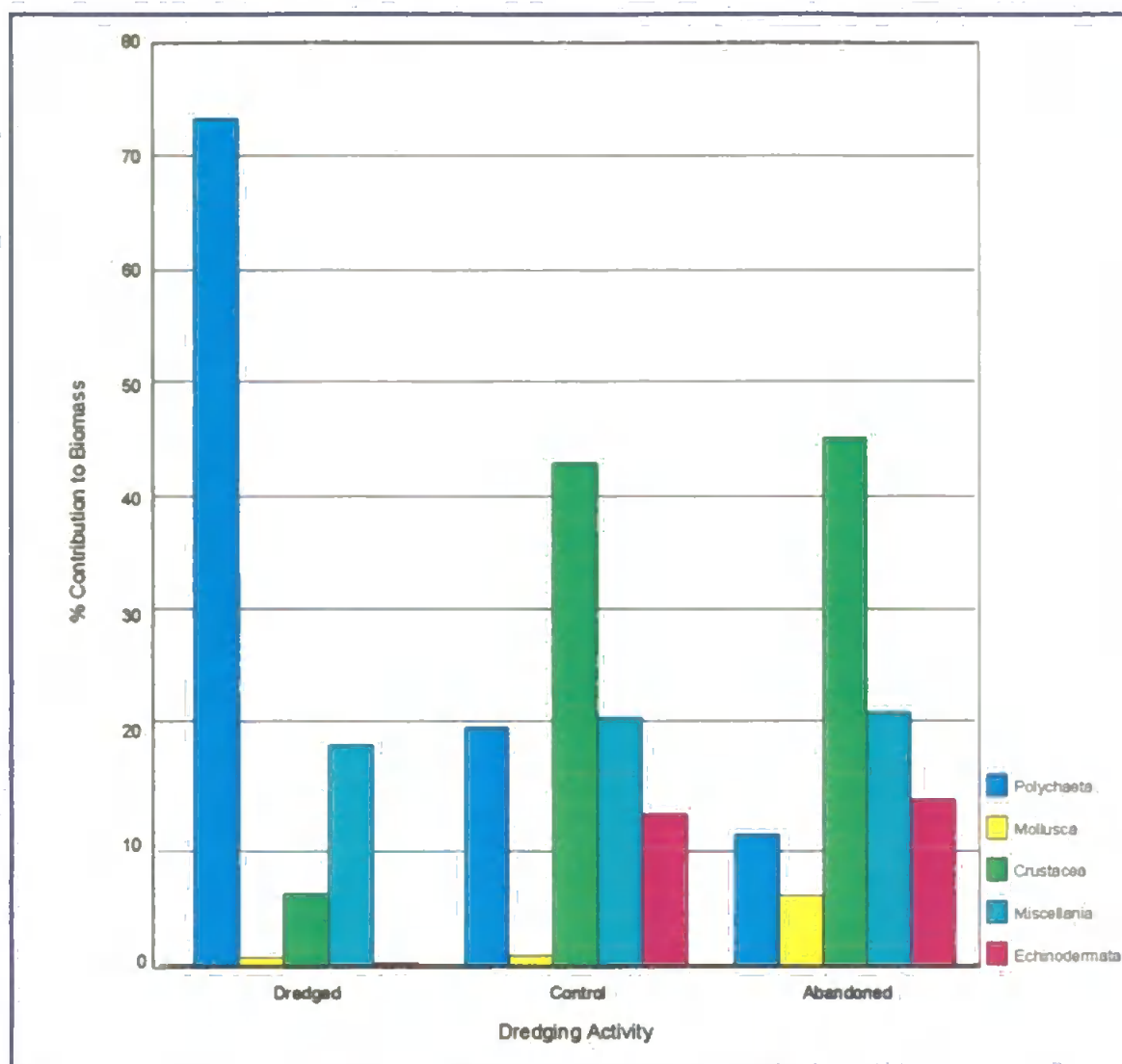


Figure 79. Percentage contribution to the total biomass by various taxonomic groups in stations subjected to different dredging pressures in the Coal Pit study area. Control stations are those assumed to be unaffected by dredging activity.

The dredged area had a higher polychaete biomass than the other sites. However, crustacean, molluscs and echinoderm biomasses were substantially lower in the dredged stations than at other sites. Analysis of variance (carried out for log total biomass, log polychaete biomass, log crustacean biomass, log mollusc biomass, log echinoderm biomass and log biomass of miscellaneous groups) identified significant differences for log echinoderm biomass ($F = 4.49$, $P = 0.01$) and for the log transformed biomass of miscellaneous groups ($F = 3.44$, $P = 0.04$). However, subsequent power analysis revealed that these, and the non-significant tests, were unable to detect differences with the required degree of certainty (Table 14).

The mean size of individuals in control stations and abandoned (i.e. previously dredged) stations were similar (Figure 80). Stations subjected to dredging at the time of sample collection had a much-reduced value for mean size of individual. An ANOVA test showed that differences in mean body size between the groups were significant ($F = 3.53$, $P = 0.03$). However a posteriori power analysis revealed that the ANOVA test lacked sufficient power to accurately detect differences, therefore the null hypothesis cannot be rejected (refer to Table 14).

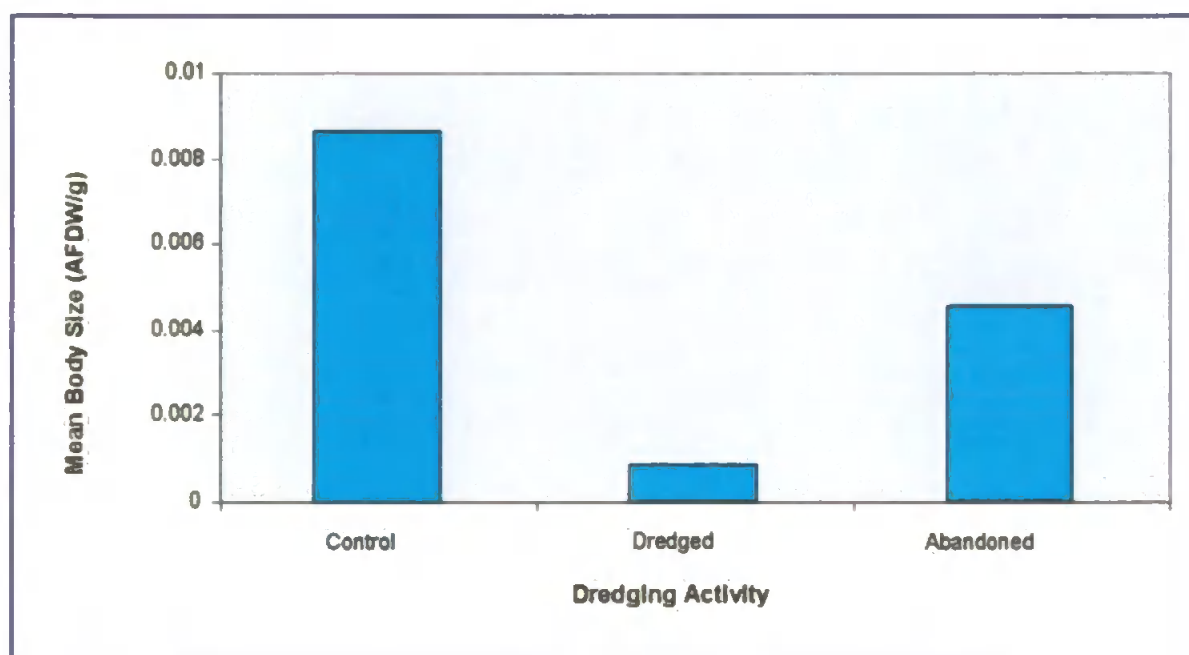
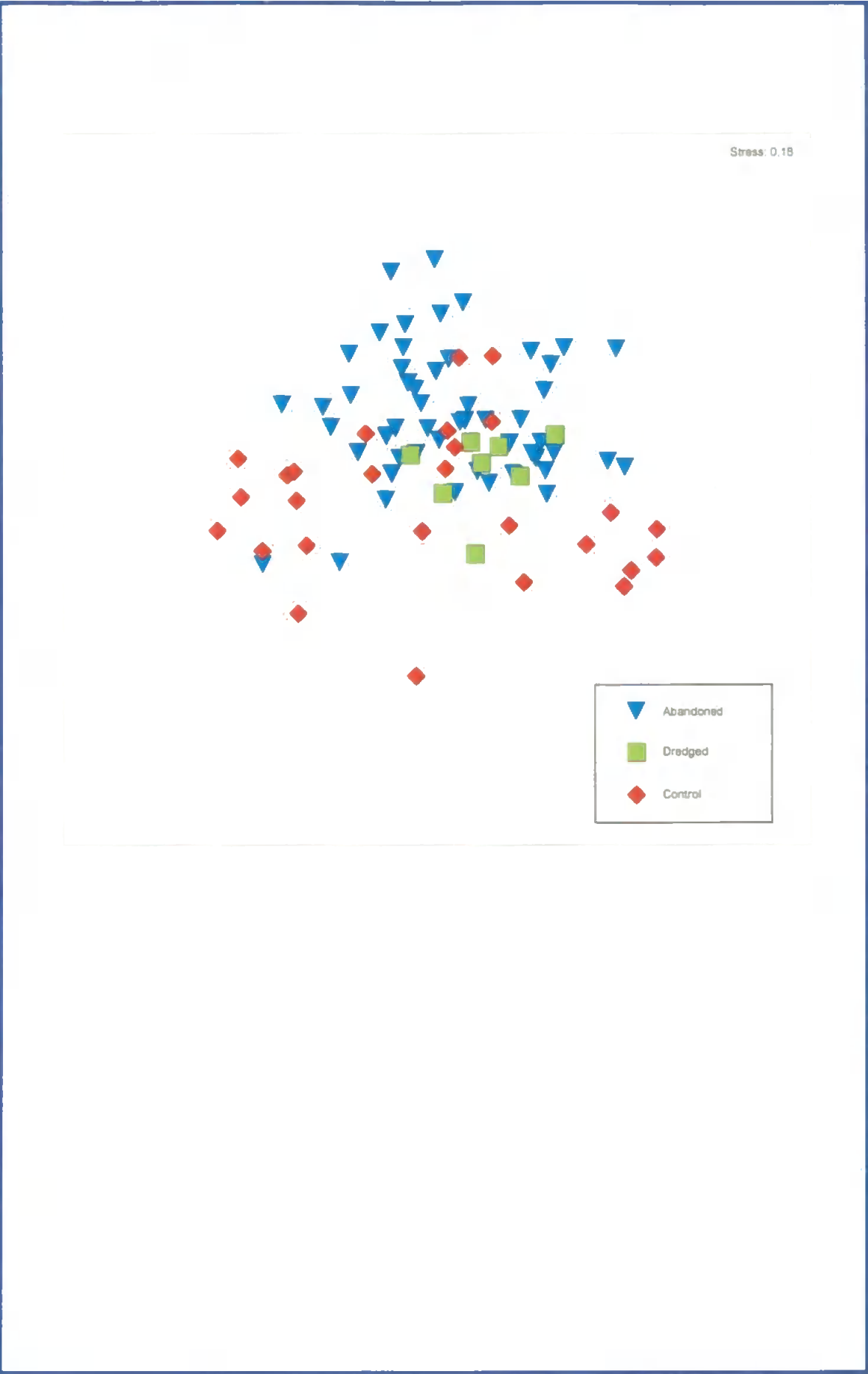


Figure 80. The mean size of individuals (AFDW/g) recorded at stations subjected to different dredging pressures in the Coal Pit study area. Control stations are those assumed to be unaffected by dredging activity.

A two-dimensional MDS ordination showed the dredged stations formed a group superimposed over the stations from abandoned and control areas (Figure 81). This shows that, within the context of the variability of benthic biological samples, the community composition of the dredged sites was similar to that of the other sites. Abandoned and control samples also showed a degree of overlap. Values calculated for multivariate dispersion can be seen in Appendix XLVII and do not show increased variability in dredged samples.



The results of an ANOSIM test are given in Appendix XLVIII, the global value for R (0.212) was significant at the 0.1% level, indicating that the between-group and within-group similarities were not approximately equal and therefore samples from the different dredging history treatments are significantly different overall, although the low R -value suggests the groups do not readily separate. Inspection of the pairwise test comparisons between the three groups revealed no significant difference between abandoned versus dredged and dredged versus control sites. Significant differences were found between abandoned and controls ($R = 0.311$, $P = 0\%$).

Inspection of the k -dominance plots suggests some increase in dominance in the macrofaunal communities of dredged areas compared to 'control' areas; abandoned stations displayed intermediate dominance (Figure 82). One species (*Ophelia borealis*) accounted for 30% of the population density in the dredged stations, whereas the same species accounted for 20% in the 'control' stations.

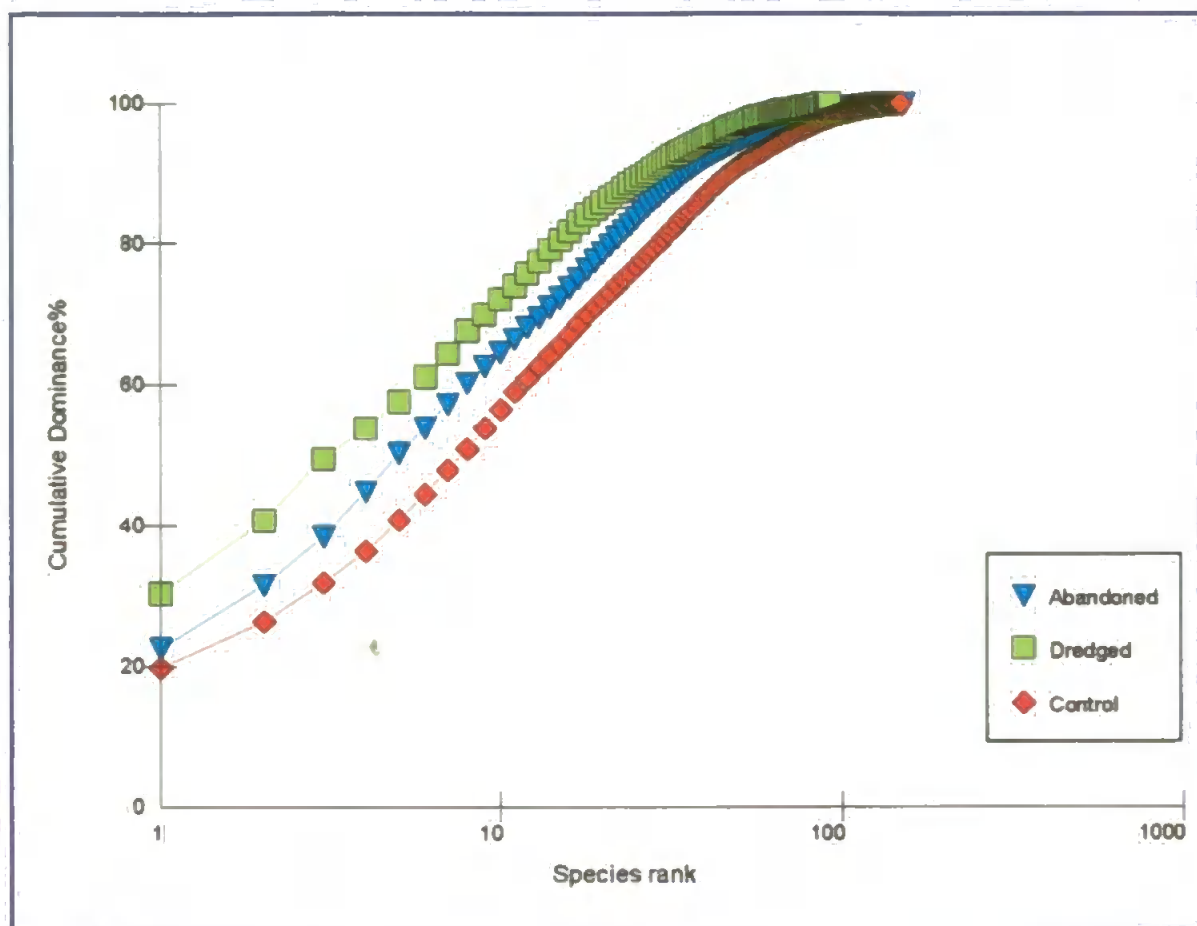


Figure 82. Cumulative dominance curves plotted for the benthic macrofauna at stations subjected to different dredging pressures in the Coal Pit study area. Control stations are those assumed to be unaffected by dredging activity.

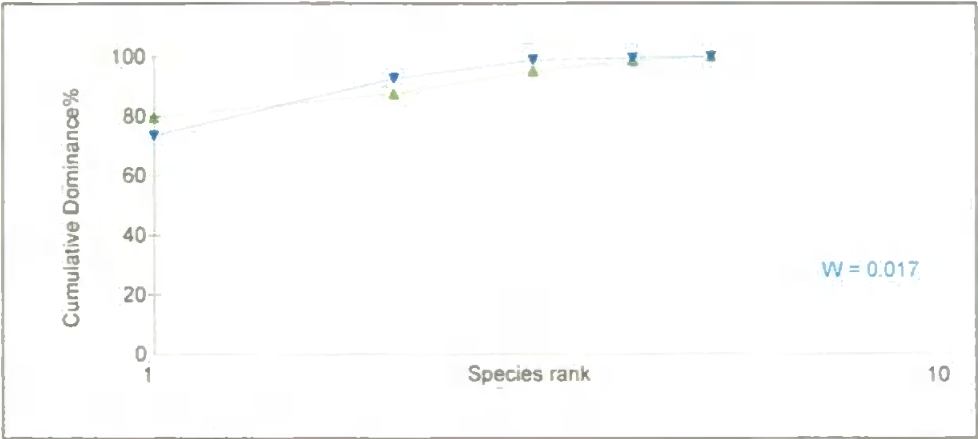
The relative contribution of the most important species contributing 60% of the within group similarity for dredged sites, previously dredged (abandoned) sites and 'control' sites situated beyond the boundaries of potential impact are summarised in Appendix XLIX. It can be seen that the 'control' stations well outside any likely impact of dredging were dominated by a similar species assemblage to that in dredged stations and in deposits where dredging had ceased (abandoned). The polychaete *Ophelia borealis* is the highest contributor to the internal similarity of all categories. The main species accounting for the difference between controls and abandoned sites (shown in the ANOSIM test) is attributable to differences in the relative abundances of *Ophelia borealis*, nematodes and *Lagis koreni*.

In most sample categories, the abundance curve (shown in green) was above the biomass curve, indicating 'disturbed' conditions (Figure 83). All sample groups showed the 'disturbed' condition as defined by the ABC criteria set out by Warwick (1986). However, the dominance of polychaete species, particularly *Ophelia borealis*, in this entire area, regardless of apparent dredging activity levels, was the likely cause of this pattern. The ABC plots do not reveal any impact associated with dredging, rather, they reflect an area dominated by polychaete taxa (high abundance/low biomass). Furthermore Warwick & Clarke (1994) demonstrated that there was no apparent change in body size of individuals within a species with increasing levels of perturbation, which could explain why a clear pattern of increased 'disturbance' between the treatments was not evident.

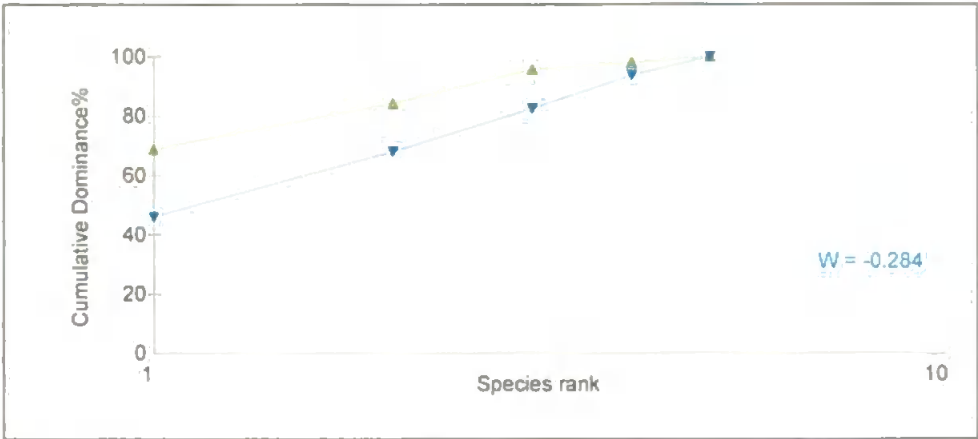
Figure 84 depicts the number of individuals belonging to various phyletic groupings recorded in a series of samples taken on a transect running in a south-north direction through the dredged zone. Crustaceans and molluscs exhibited a decrease at the dredged stations compared with those outside the extraction area. In contrast, nematodes and annelids appeared to increase in abundance in the dredged samples. There was no clear response for the echinoderms.

7.3.3. Production Meta-analysis (North Nab and Coal Pit data combined)

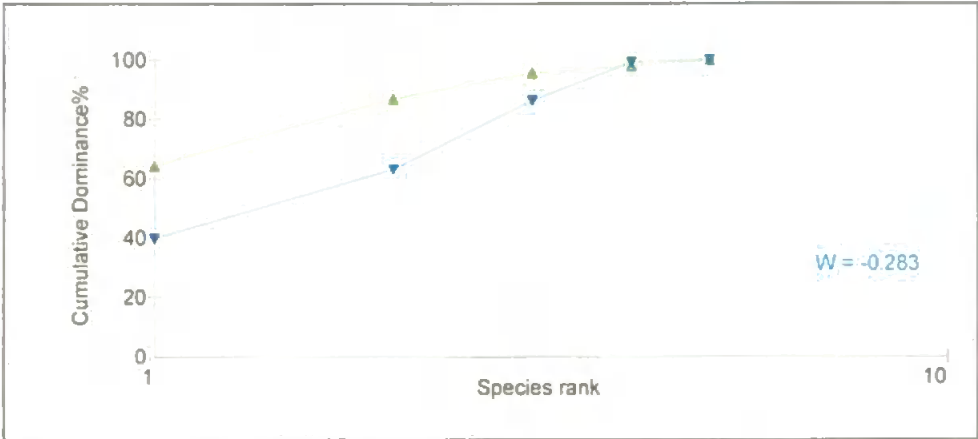
There was no obvious separation of samples subjected to direct removal of sediment or to the settling of rejected material (Figure 85). With the exception of two anchor dredged samples from the North Nab study area (NN137 and NN140), the majority of potentially impacted sites cannot be differentiated from the control stations. Similarly, when individual phyla were highlighted, no pattern of impact associated with the disturbance was evident.



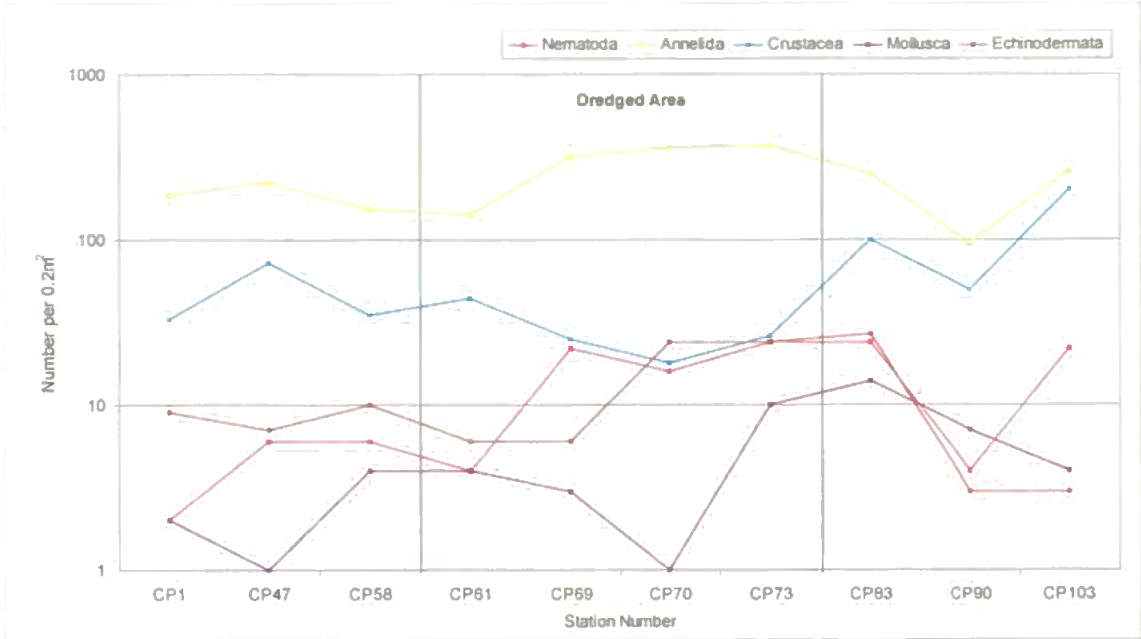
Dredged

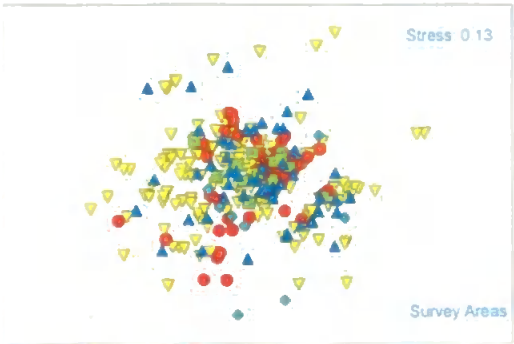
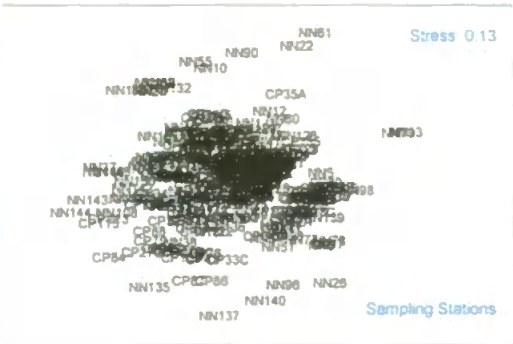


Abandoned

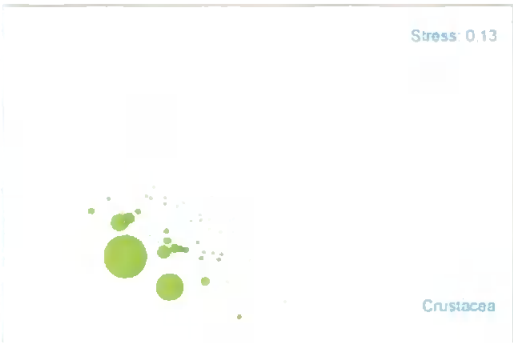
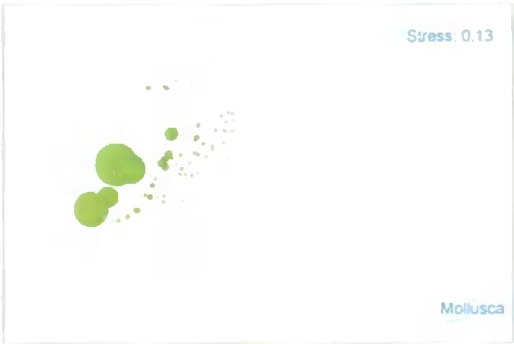
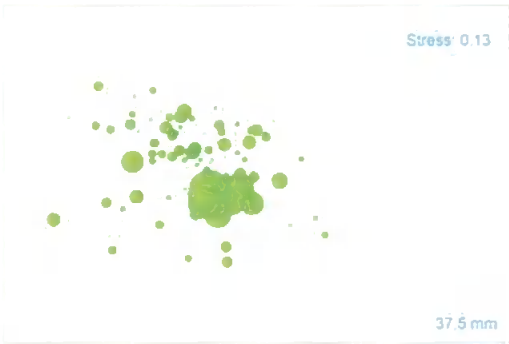


Control





▲ Control ▼ Non-dredged ● Abandoned ■ Trailer Dredged ◆ Anchor Dredged



7.4. Discussion

7.4.1. North Nab

Overall comparisons of the species diversity (S) and population density between stations subjected to anchor dredging, trailer dredging and controls did not reveal any obvious impacts. However, average biomass did show a reduction of 76% between anchor dredged and control stations. Similar values for invertebrate biomass, were reported by Van Moorsel (1994) and Kenny & Rees (1994) at dredged sites in the North Sea. Due to the time elapsed, since aggregate extraction at most of the stations in the anchor-dredged area had ceased, recovery may have already begun (the average time elapsed since dredging at these stations was 8 months). Therefore, different samples used in the present study may reflect different successional stages in the recovery process (Pearson & Rosenberg 1978). This may explain the absence of any clear impact on numbers of species and individuals, as these elements are likely to reach pre-dredging levels before biomass values (Desprez 2000). An investigation by Desprez (2000), in a dredged area off the French coast of the English Channel, revealed biomass had not reached pre-dredging levels after 16 months whereas species richness had returned to pre-extraction levels.

Present data on the impacts of anchor dredging on the contributions of different faunal groups to the total biomass appear to confirm general principles of macrobenthic community response to disturbance (see Pearson & Rosenberg 1978; Newell *et al.* 1998). Suppression of molluscan biomass and miscellaneous groups such as hydroids and bryozoans, in favour of crustaceans, in anchor-dredged areas suggests some, albeit subtle, alteration in community type. The abundance and biomass dominants (Crustacea) in the area subject to aggregate removal by means of anchor dredge largely consisted of juvenile barnacles

(*Balanus balanus*), and the amphipods *Ampelisca brevicornis* and *Leptocheirus hirsutimanus*. Contributions to biomass and abundance by these species in such disrupted zones possibly represent an early colonization community (Pearson & Rosenberg 1978; Newell *et al.* 1998). In contrast to other areas (controls and trailer dredged), major biomass and abundance contributors included the gastropod *Crepidula fornicata*, the tunicate *Distomus variolosus* and the tube-dwelling polychaete *Pomatoceros* spp. Such taxa require stable attachment surfaces and are, therefore, unlikely to persist in an environment subjected to heavy dredging intensity. Juvenile *B. balanus*, whilst comparatively minor faunal components in other treatments, were recorded in large quantities in the anchor-dredged area. In this regard, this barnacle species may be acting opportunistically by rapidly colonising areas denuded of space competitors such as *D. variolosus*, *Pomatoceros* spp. and *F. foliacea*.

In terms of the community structure (identified by multivariate analysis of species abundance data), a difference was apparent between the areas exposed to different levels of dredging intensity. The anchor-dredged samples appeared to separate from the other samples, albeit only slightly, implying possible disturbance-induced community alteration. Trailer-dredged samples were more closely allied to control samples. Greater internal similarity between trailer-dredged samples, compared to anchor-dredged samples, perhaps indicates a decrease in stress attributable to this form of aggregate extraction. However, values calculated for multivariate dispersion revealed comparable values for these two sample treatments.

A further, sensitive measure of the uniformity of species composition in natural communities in relation to environmental gradients are so-called 'k-dominance'

curves, which show the percentage of a population which is represented by each of the species within the community (see Lambshead *et al.* 1983; Clarke 1990; Warwick 1993). For the North Nab area, an impact of anchor dredging in terms of dominance was apparent with high (>70%) dominance by one species (juvenile *Balanus balanus*). In contrast, trailer-dredged sites had only a slightly higher dominance than control or non-dredged areas, providing possible evidence for the decreased ability of trailer dredging to adversely affect the benthic communities of this region in comparison to anchor dredging. As a general rule, communities not subjected to stress often have a *k*-dominance curve that shows no evidence of a major dominance by one or a few species. In contrast, where there is an environmental stress imposed on a community, sensitive species are replaced by large numbers of those (resistant) species that are capable of survival. This process leads to a reduction in species variety and a numerical dominance by one or a few resistant species. In these cases, the community may show as much as 70-90% dominance by one or two (resistant) species. This is the probable scenario for the anchor-dredged sites examined in the present study where the recolonising *Balanus* and *Ampelisca* were present in elevated abundances. Furthermore, the ABC curves (Figure 76) showed sites subjected to both anchor and trailer dredging exhibited abundance curves that intersect the curve plotted for biomass. This intersection reflects the presence of numerically-dominant species in these disturbed areas, (i.e. small-bodied opportunists or colonisation by new recruits in this case juvenile *Balanus*). Sites not affected directly by substratum removal were somewhat different, with the biomass curve lying above that of the abundance curve. This is typical of an undisturbed community characterised by large-bodied taxa that do not display numerical dominance but are the dominant biomass components (Warwick 1993).

7.4.2. Coal Pit

Impacts of trailer dredging in the Coal Pit study area were not apparent based upon number of species and individuals. Some reduction in biomass levels occurred in the dredged and abandoned areas but not to such an extent that it can be linked implicitly to aggregate extraction, as other stations also showed similarly low biomass values. However, contributions to total biomass by various categories of fauna did highlight some discrepancies between dredged samples and the other treatment groups. Polychaetes represented the largest contributors to the biomass of dredged samples, whilst echinoderms, crustaceans and molluscs represented minor components. In contrast, control and abandoned regions had significantly greater inputs to the total biomass from these groups, particularly crustaceans and echinoderms. These findings are in accordance with the general community responses to stress (in this case the removal of aggregates) (Boesch & Rosenberg 1981; Newell *et al.* 1998). The substitution of larger bodied, slower growing (*k*-selected) taxa (a description that may be applied to many echinoderms) with small, rapidly colonising opportunists (for example, many polychaete species) is a successional community shift akin to that described by Pearson & Rosenberg (1978) in response to organic enrichment. This response to dredging is confirmed if mean individual body size is considered. Non-dredged, control and abandoned sites all demonstrated similar mean body sizes, whilst the dredged samples exhibited a significantly reduced value. Echinoderms appeared to represent large-bodied individuals thus validating their status as '*k*-selected' taxa. Conversely, polychaete species recorded in this area characterized small-bodied individuals and can, therefore, be assigned to the '*r*-selected' (opportunistic) category (Pianka 1970; Rees & Dare 1993).

Despite the dredged samples forming a weak grouping in the two-dimensional MDS ordination, the fact that they were not readily separable from the other treatment groups indicated the community structure of the sites had not been altered greatly by the dredging process. Indeed, when the species responsible for greatest contribution to the internal similarities were scrutinised, *Ophelia borealis* was the most important species in all dredging intensity groups. Dissimilarities between the various treatment types were highly dependent on the relative abundance of this polychaete, indicating that the distribution of this species throughout the Coal Pit region was unaffected by dredging activity. Due to the time elapsed since the last dredging episode in this area (<60 days), it is unclear whether *O. borealis* was a pioneering coloniser in areas of aggregate removal or if it arrived at a later successional stage, possibly out-competing some other, unidentified initial settler. *Ophelia borealis* is a species indicative of mobile, sandy substrata (Vanosmael *et al.* 1982), which explains its prevalence in the deposits of the area. It is likely that such a species would be well adapted to physical disturbance, hence its persistence in areas subjected to dredging. Relocation of this species into the cleared (dredged) furrows may possibly come from narrow, non-dredged 'strips' between the dredge tracks, either by active migration to such areas or by the slumping of the sides of the tracks themselves (Kaplan *et al.* 1975).

Dominance was shown to be higher at stations subjected directly to aggregate extraction, however, the species responsible for this dominance (*Ophelia borealis*) was also the numerical dominant in the other treatment groups, indicating no major benthic community readjustment (in terms of dominance by a species characterising dredged samples) in response to the dredging operations. Interestingly, abundance biomass curves (Figure 82) suggested a grossly

disturbed community in most sample groups, except for the dredged grouping where the apparent disturbance level was reduced (Warwick & Ruswahyuni 1987). Whilst the latter seems to confirm the status of the Coal Pit region as a frequently disrupted area, it is not possible to say whether this seemingly 'grossly disturbed' (Warwick 1993) scenario was attributable to natural factors or an anthropogenic factor aside from aggregate extraction (i.e. trawling). The fact that the dredged samples appeared to be subjected to a lower stress level than the other treatments may be an example of the intermediate disturbance hypothesis (Connell 1978). In such a naturally dynamic environment, it is possible that the impact attributable to aggregate extraction could be regarded as 'intermediate'. The disturbance may have reduced the numbers of the dominant taxa (competitive superior i.e. *Ophelia borealis*) to a level that provided the remaining benthic inhabitants a better chance of establishment, thus a 'healthier' community was depicted by the ABC plots. The pattern observed may also highlight a weakness of the assumptions made by the ABC method (i.e. that undisturbed communities are characterised by one or a few very large, biomass-dominant species, each represented by few individuals, while the numerical dominants are small in terms of their biomass (Warwick 1993)). This scenario was not the case throughout the Coal Pit area, with the apparent lack of large-bodied taxa likely to be a reflection of the dynamic environmental conditions rather than to some anthropogenically-derived input.

The apparent lessening of the disturbance level depicted in the ABC plot of dredged samples (Figure 83) may reflect a substratum alteration or a change in the hydrodynamics associated directly with the dredging process at these sites. Perhaps the removal of an upper layer of sediment has led to the presence of a more stable substratum. Particle size analysis revealed no obvious modification in

sediment characteristics at the dredged locations, however, such a change may be in the form of sediment cohesiveness or penetrability. A further possible explanation may lie in the dredge tracks themselves. These areas of 'lowered seabed' may be more sheltered from the direct force of wave action or the effects of scouring by mobilised sediments and, therefore, may represent a habitat more amenable to colonisation (between dredging events) than surrounding deposits subjected to more sustained disruption.

7.4.3. Comparison of Dredging Impacts at North Nab and Coal Pit

Trailer dredging at the two environmentally contrasting regions led to no detectable impact on macrobenthic community structure. Anchor dredging at the North Nab study site led to an alteration in community type with the deposits of the dredged zone being dominated by juveniles of the barnacle *Balanus balanus*.

Generally, the responses of both dredged areas conform to the findings of previous investigations regarding macrobenthic response to disturbance, although initial impacts may have been obscured by the time lapse between sample collection and the last dredging episode. As described in chapter 4, clear differences between the two regions existed in terms of their benthic community characteristics and their hydrophysical regimes. Despite these differences, community responses in both regions were comparable with regard to the apparent elimination of larger bodied, long-lived species from the dredging zones. A feature of the dredged region of the Coal Pit survey area was the post-dredging community. In terms of species abundance, it was very similar to that of the control, abandoned and non-dredged sites. In contrast, the anchor-dredged region in the North Nab area exhibited a slightly altered fauna compared to trailer dredged and control stations. Therefore, the dredging method employed is likely to

determine the level of impact, with anchor dredging resulting in a more severe alteration in macrobenthic community structure when compared to trailer dredging (at least at the intensity of trailer dredging in these two areas).

A further explanation for contrasting macrobenthic responses may lie in the respective hydrophysical regimes of the two regions. Sediments characterised by sandy deposits are likely to experience greater sediment transport rates and subsequent scour/resettlement than regions comprising a coarser substratum. Therefore, the resident fauna is likely to be well suited to life in such unstable conditions. Conversely, a region of relative environmental stability is likely to be inhabited by a fauna consistent with such an environment, and is, therefore, not well adapted to survive intense physical disruption associated with dredging. It should be noted that the community responses of Coal Pit and North Nab, when the effect of trailer dredging in the two areas is examined, was comparable (the community structure was not significantly altered by this form of dredging). The apparently less severe nature of the trailer dredging process (i.e. sediment removal down to a lesser depth than in anchor dredging) and the presence of non-dredged 'strips' between the dredge tracks to facilitate subsequent recolonisation, may be such that even in an environmentally stable region such as North Nab the fauna is able to recover.

The apparent lack of impact at the Coal Pit survey area may be a reflection of using species abundance data. Using this type of data does not necessarily account for responses to stress below the level of a certain species. Figure 86 highlights the relationship between environmental stress and taxonomic variability (Boesch & Rosenberg 1981). It may be that the stress attributable to dredging in such an area does not represent a level of disturbance sufficient to cause a

change in the species complement. However, this does not imply that a given stressor has not in some way, affected the fauna of such a region. It may be that those genetically more capable of surviving under such conditions have replaced less hardy individuals. A useful description to bear in mind when considering responses to disturbance is provided by Boesch & Rosenberg (1981) who stated: *"The changes in response to stress can be regarded as a continuum interrupted by steps occurring at points where the level of adaptability exceeds the capacity of that level of organisation."*

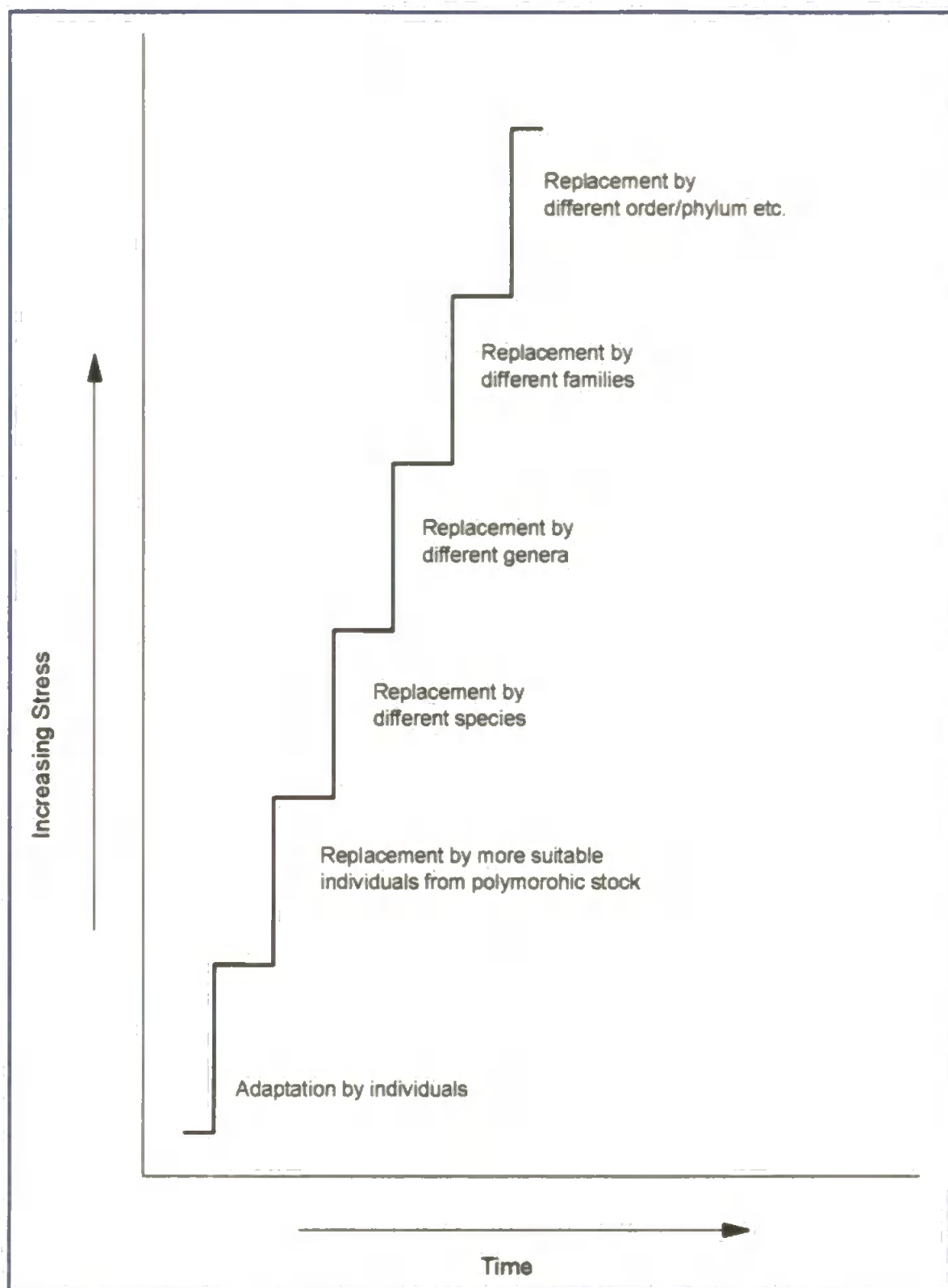


Figure 86. Schematic diagram of the relationship between environmental stress and taxonomic variability (adapted from Boesch & Rosenberg 1981).

Past investigations have attempted to highlight the differences in community response to dredging in environmentally contrasting regions. Van Dalfsen *et al.* (2000) compared the impacts of dredging on fauna in the North Sea and the western Mediterranean. Both North Sea sites and the Mediterranean site showed

a loss of equilibrium species after dredging, particularly echinoderms in the North Sea and molluscs in the Mediterranean. At all three sites, an increase in opportunistic (polychaete) species was recorded. Alterations to the sediments in the dynamic, current-driven North Sea sites after dredging were found to be negligible. In contrast, the hydrodynamic conditions of the Mediterranean study area precluded the redistribution of sediment and a significant shift in sedimentary conditions was observed, with sand being overlain by a 50-200mm thick layer of very fine particles. Recovery of the community structure in North Sea locales was found to be completed within 2-4 years. Conversely, at the Mediterranean site recovery had not taken place within this time period. Indeed, the recovery of commercially exploitable bivalve species was predicted to take up to a decade. The authors cite the alteration to the sediments in the Mediterranean site as the primary factor governing recovery of the benthos (Van Dalfsen *et al.* 2000). The findings of this investigation provide a possible explanation for the apparently rapid recovery of the abandoned sites in the Coal Pit region. Abandoned sites had similar deposit types to other non-dredged stations in the area one year after cessation of dredging activities. Therefore, infilling of the dredge tracks, if it had taken place, is likely to have been with sediments analogous to those present prior to aggregate extraction, thus promoting the establishment of a similar species assemblage. Further evidence implicating substratum alteration in the recovery rate of the benthos after dredging is provided by Van der Veer *et al.* (1985) who demonstrated that, in regions of low tidal stream velocities, sediment composition was altered dramatically and recovery of the benthos was retarded (recovery took 5-10 years). In contrast, tidal channels with strong currents were filled-in and recolonised after 1-3 years. Use of the term 'filled-in' may be misleading in describing the sequence of substratum recovery following dredging. In dredged areas characterised by a particularly coarse substratum, no natural mechanism is

likely to exist for the remobilisation of particles over approximately 6mm (Hall 1994). Therefore, infilling of dredge tracks with a deposit type akin to that which existed prior to dredging is not likely to occur in such areas. In instances where 'recovery' of seabed deposits in areas characterised by coarse sediments has been observed, perhaps a more realistic scenario is that fine material rejected during the dredging process initially fills the dredge tracks, and is then subsequently winnowed away by natural currents, thus exposing the 'cleaned' underlying coarse deposits.

General conclusions regarding recovery rates following dredging are difficult to make due to the highly varied impacts reported throughout the literature. Pagliai *et al.* (1985) investigated the post-dredging recovery of the macrobenthos in a coastal area off Sardinia. Immediately following dredging, the benthic inhabitants of the silty deposits that characterised the area were completely eliminated. However, after six months, the benthic communities were substantially recovered and were similar in composition to assemblages recorded prior to dredging. Similarly, a study at Tampa Bay, Florida (Conner & Simon 1979) showed almost complete recovery of the benthos six months after dredging. Longer recovery periods were reported by Van Der Veer *et al.* (1985) who found that, on a tidal flat in the Dutch Wadden Sea, no recovery of the benthos had occurred after 16 years. Taylor & Saloman (1968), working in Boca Ceiga Bay, Florida, reported that recovery of the benthic fauna of dredged, shell-sands took approximately ten years to complete.

7.5. Summary

Impacts of aggregate extraction at the North Nab site were found only in stations subjected to anchor dredging. Anchor dredging was shown to be associated with a benthic community shift from one characterised by *Distomus variolosus*, tubicolous polychaetes and the gastropod *Crepidula fornicata* to a community dominated by juvenile *Balanus balanus*. Trailer dredging was not shown to have any significant impacts on macrobenthic community structure.

Impacts of trailer dredging on the benthic invertebrate community at the Coal pit site were not demonstrated. The polychaete *Ophelia borealis* was found to be the dominant species in all sample categories.

It was therefore concluded that trailer dredging (at the level intensity employed at these sites) was not associated with any significant disruption to benthic community structure, despite the obvious differences in abiotic and biotic characteristics of the two dredged areas.

RECOVERY OF BENTHIC MACROFAUNA

HABITAT

DREDGING

SEDIMENT TYPE

DREDGING METHOD

DREDGING INTENSITY

RECOVERY RATE

RECOVERY TIME

CHAPTER 8. A QUANTITATIVE META-ANALYSIS OF DREDGING IMPACTS ON BENTHIC MACROFAUNA

The aim of the meta-analysis carried out in this chapter was to detect general recovery trends/rates in contrasting habitats subjected to aggregate removal.

CHAPTER 8. A QUANTITATIVE META ANALYSIS OF DREDGING IMPACTS ON BENTHIC MACROFAUNA.

8.1. Introduction

The impacts of marine aggregate extraction on the benthos have been reported widely in the literature and are reviewed by Newell *et al.* (1998). However, quantitative analysis of the biological responses related to dredging has not been attempted and general conclusions regarding impacts in different habitats have been based largely on observation. Collie *et al.* (2000) quantitatively reviewed the benthic responses to disturbance from fishing activities. A quantitative analysis of impacts related to marine aggregate mining, and subsequent recovery, would be beneficial for the conservation of potentially sensitive areas. Following the approach of Collie *et al.* (2000), a meta-analysis of the summary data extracted from 17 published dredging impact studies and data from the current study (see Appendix LII) forms the basis for this chapter. The aim was to highlight benthic responses and recovery in contrasting habitats subjected to dredging.

8.2. Materials and Methods

A total of 33 manipulations/observations of the effects of dredging disturbance on benthic fauna and communities, was extracted from 17 published dredging impact studies and data from the present study (Appendix LII). Studies were classified according to habitat type, (gravels, sands and deposits containing significant quantities of mud). Estimates of the effects of dredging on species richness, abundance and biomass were extracted, along with effects at the genus level where possible, and at higher taxonomic levels otherwise. Specific taxa were treated as independent observations to define macrobenthic responses, regardless of the taxon in question. The level of response to dredging, of each variable, was then calculated from the mean values in controls (including 'before'

samples in some studies) and the mean values in dredged samples, using the following equation:

where A_d = abundance in dredged samples and A_c = abundance in controls. To

$$\% \text{ Difference} = \left(\frac{A_d - A_c}{A_c} \right) \times 100$$

observe the recovery process following dredging, time series data (where available) were utilised. In order to account for the occurrence of zero time intervals i.e. taken immediately after dredging, $\log(1+x)$ to the base e was used, therefore zero times come out as $\log_{\text{time}} = 0$ and samples taken after 1 day would equal $\log_{\text{time}} = 0.693$. Analysis of variance (ANOVA) for unbalanced designs was used to test for significant differences between the responses in different habitats (for all regions and for northern European studies only), and between different time categories within habitats (Appendix LIII).

Plots depicting macrobenthic responses to dredging over time (Figures 87 - 89) are fitted with a cubic spline smoother, this has the effect of smoothing out interpolated values between data points. A cubic spline is created for each interval between data points using the formula:

$$Y = a + bx + cx^2 + dx^3$$

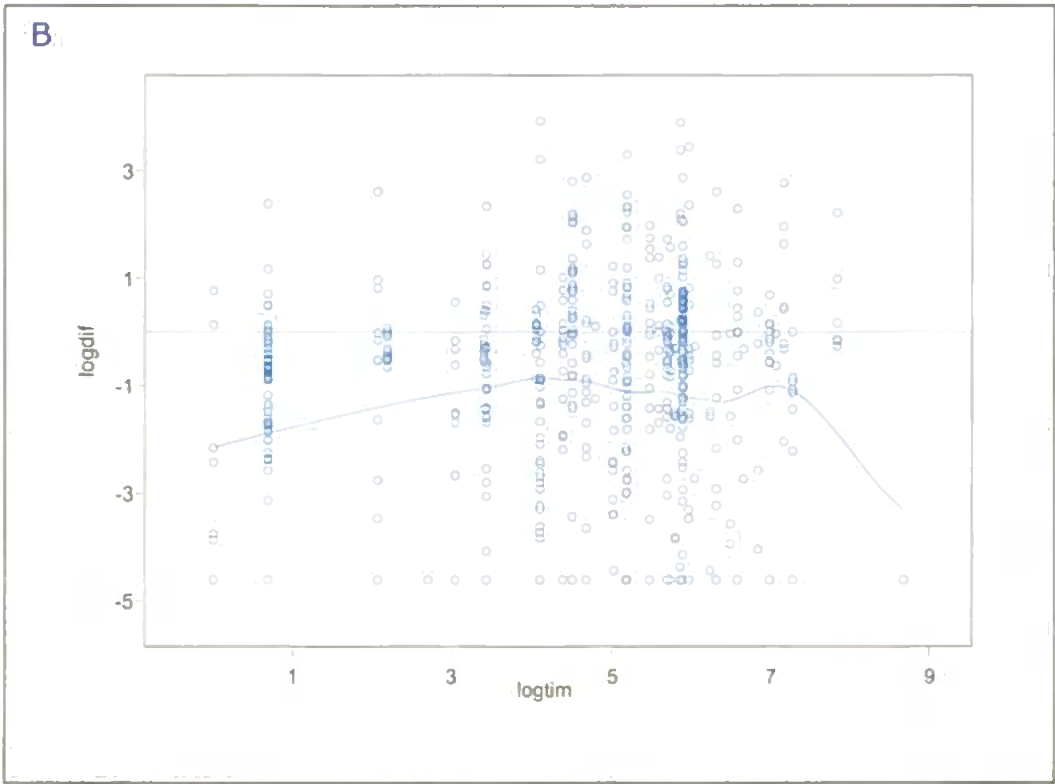
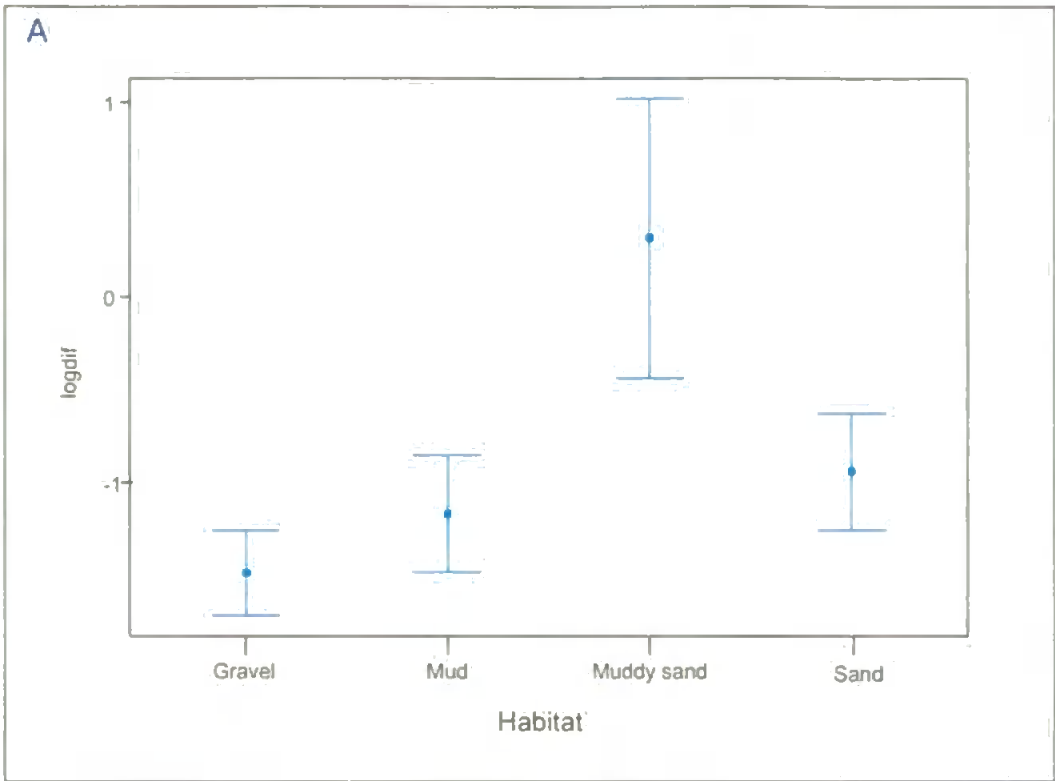
Using iteration the slopes of the lines at the beginning and end of each interval are then matched to give a smooth line through the data.

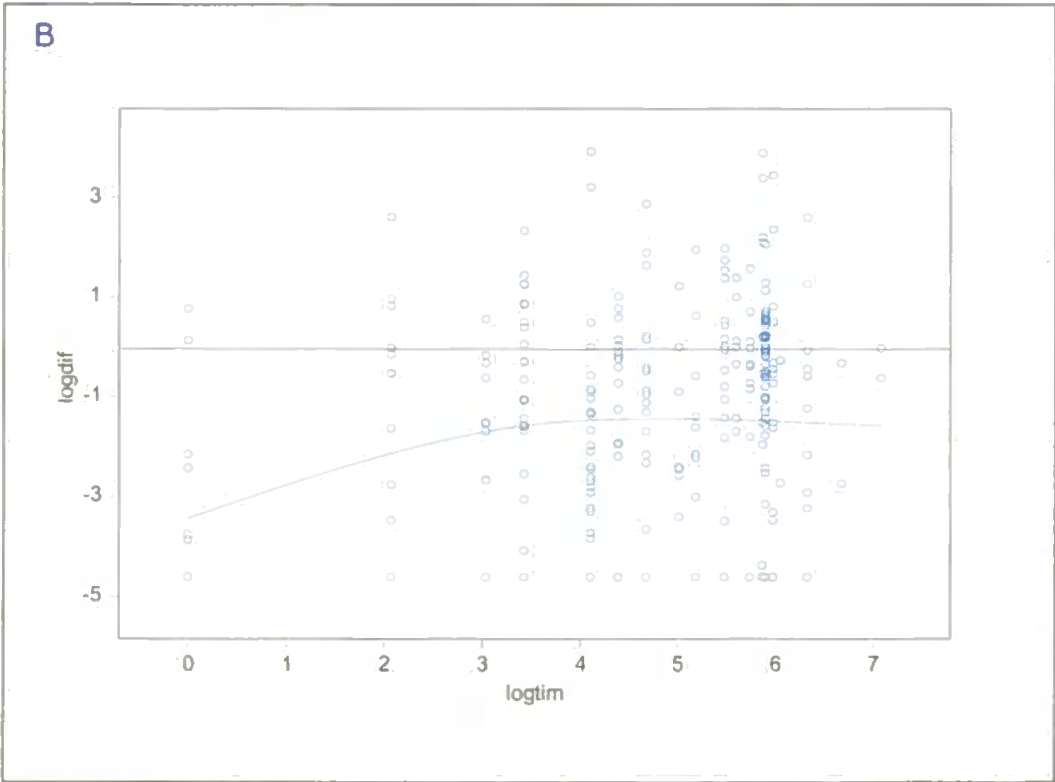
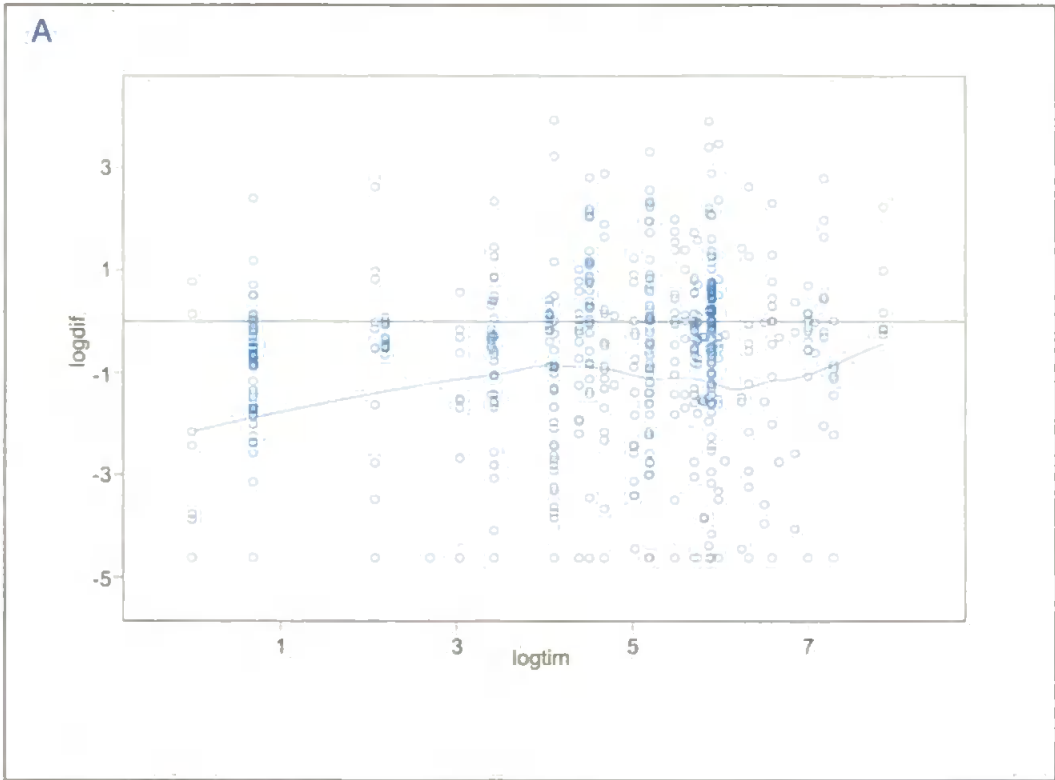
8.3. Results

The response to dredging in the different habitat types is shown in Figure 87a. Environments characterised by muddy sand exhibited a positive response, likely a result of fewer suitable studies being identified in such a substratum, as indicated by the large error bars on Figure 87a. Response levels (Appendix LIII) were significantly different between all habitat types. The response following dredging over time is summarised in Figure 87b. The recovery trajectory initially increases with time, followed by a period of fluctuation and then by a dramatic fall. The sharp drop in apparent recovery status was attributable to one study in which no recolonisation occurred after a 15-year interval (Van der Veer *et al.* 1985). The latter study was, therefore, deemed atypical and was omitted from further analysis to avoid skewing the result unrealistically. Figure 88a shows the same response (as log difference) without the data from Van der Veer *et al.* (1985). The revised plot produces an almost linear recovery trend with a degree of fluctuation, although complete recovery is not demonstrated after a period of ten years. Analysis of variance, (Appendix LIII), demonstrated that the levels of response over time ('recovery') were significantly different at all time intervals, again highlighting the lack of complete recovery.

Table 15. Summary of ANOVA of time categories (since dredging) in three contrasting habitat types. * $P < 0.05$, ** $P < 0.01$, *** $P < 0.001$.

Habitat	F-value
Northern European Gravel	4.63**
Northern European Sand	1.49*
Mud and Muddy Sand (All Regions)	6.69***





Mean differences (log) for the five time categories (Appendix LIII) showed the greatest negative response occurred in the time category immediately following the dredging event (0-1 days since dredging). Table 13 gives the results of an analysis of variance using data from studies based in northern Europe (sand and gravel) and world-wide (muddy substrata), and demonstrates that significant differences existed between different time categories in gravel and sands.

Examination of northern European gravels (Table 14), revealed a gradual reduction in the negative response of the benthos over the five time categories. In contrast, the recovery trend in sandy habitats (northern-Europe only) was highly variable. The initial response was less severe than in gravels, and the subsequent recovery after 2-180 days was also of a higher rate; following this time interval, fluctuations occurred to give a negative response lower than the initial response after 703+ days (Table 14).

Table 16. Mean macrobenthic response over time in northern European gravel and sand habitats.

Northern European Gravel					
Grand Mean: -1.81979			Standard Error: 0.25465		
Time Category	1	2	3	4	5
Mean	-3.7063	-1.6403	-1.4941	-1.3281	-0.9302
Standard Error	0.5136	0.1829	0.2768	0.2452	1.0869
Northern European Sand (van der Veer <i>et al.</i> 1985 removed)					
Grand Mean: -1.05936			Standard Error: 0.20482		
Time Category	1	2	3	4	5
Mean	-0.9721	-0.4274	-1.6125	-1.0765	-1.2083
Standard Error	0.5159	0.264	0.6985	0.4033	0.2496

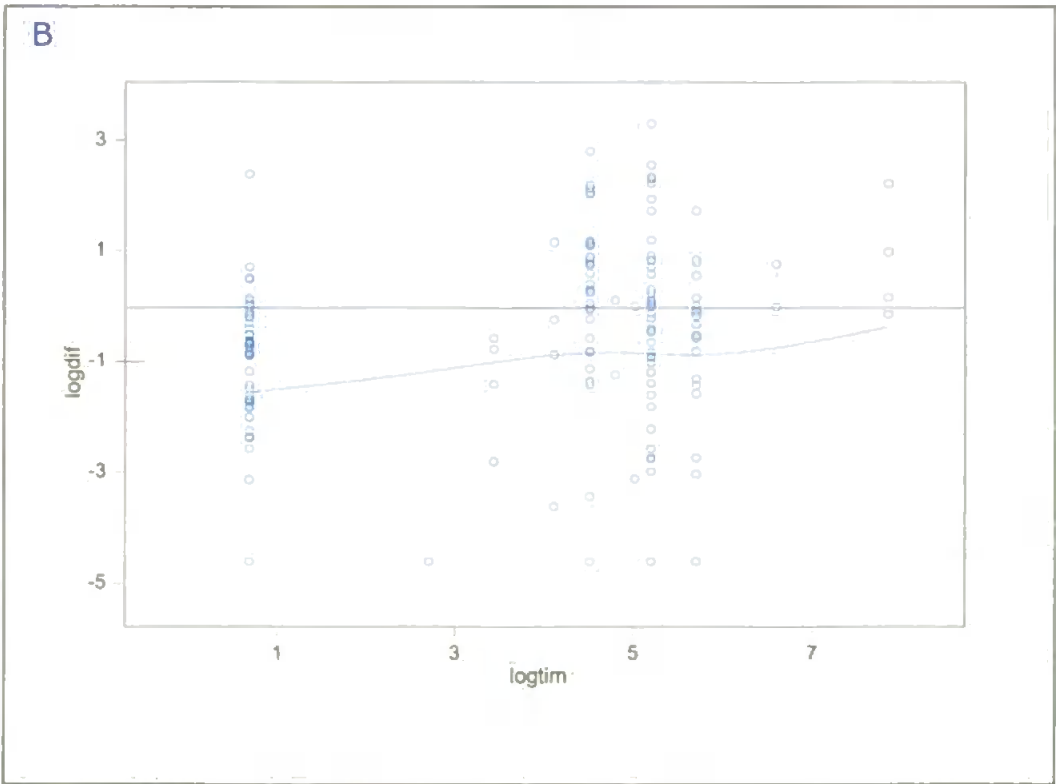
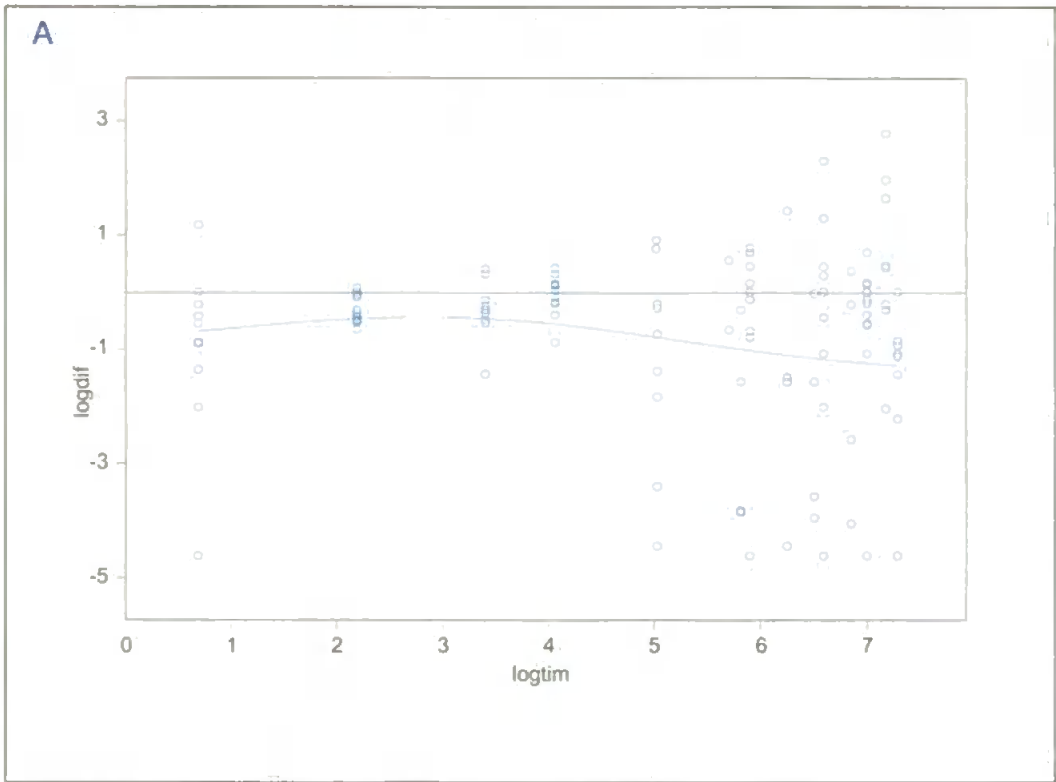
Figures 88b & 89a illustrate the general trends in gravels and sands from northern European studies. In gravels, initial recovery was linear, followed by a gradual 'tail-

off. Similarly in sands, the initial recovery phase (between 2-180 days) was linear, but levelled off after 181-364 days, before steadily falling for the remaining time periods.

In areas subjected to dredging with a muddy substratum (mud and muddy sand from all regions) recovery appears to be incomplete after two years, as in the other habitat types. Recovery was largely constant over time and was closer to attaining pre-dredging status than in gravels and sands (Figure 89b). Analysis of variance confirmed that significant differences existed between the responses at the different time intervals in muddy environments (Appendix LIII).

8.4. Discussion

Patterns revealed by meta-analysis of the global data (all habitats and all regions) suggested benthic recovery following the cessation of dredging operations was a long-term process that was unlikely to be completed up to two years after a dredging incident. Further analyses indicated that impacts attributable to aggregate extraction and subsequent recovery trajectories were likely to be heavily reliant on the habitat type in which the dredging occurred. Gravels appeared to show the expected pattern of a large initial impact, followed by a degree of recovery but not to pre-extraction levels. This lack of complete recovery may be a function of a physical alteration of the sediments (i.e. removal of coarse particles) that may preclude the settlement of epifaunal taxa (e.g. hydroids) and subsequently inhibit the repopulation of associated taxa (e.g. caprellids). Alternatively, more time may be required for complete recovery to be achieved, particularly if the pre-dredging assemblage was characterised by large-bodied/slow-growing species. Indeed, McCauley *et al.* (1977) reported that drastic changes to the benthos, induced by dredging, may alter ecological pathways,



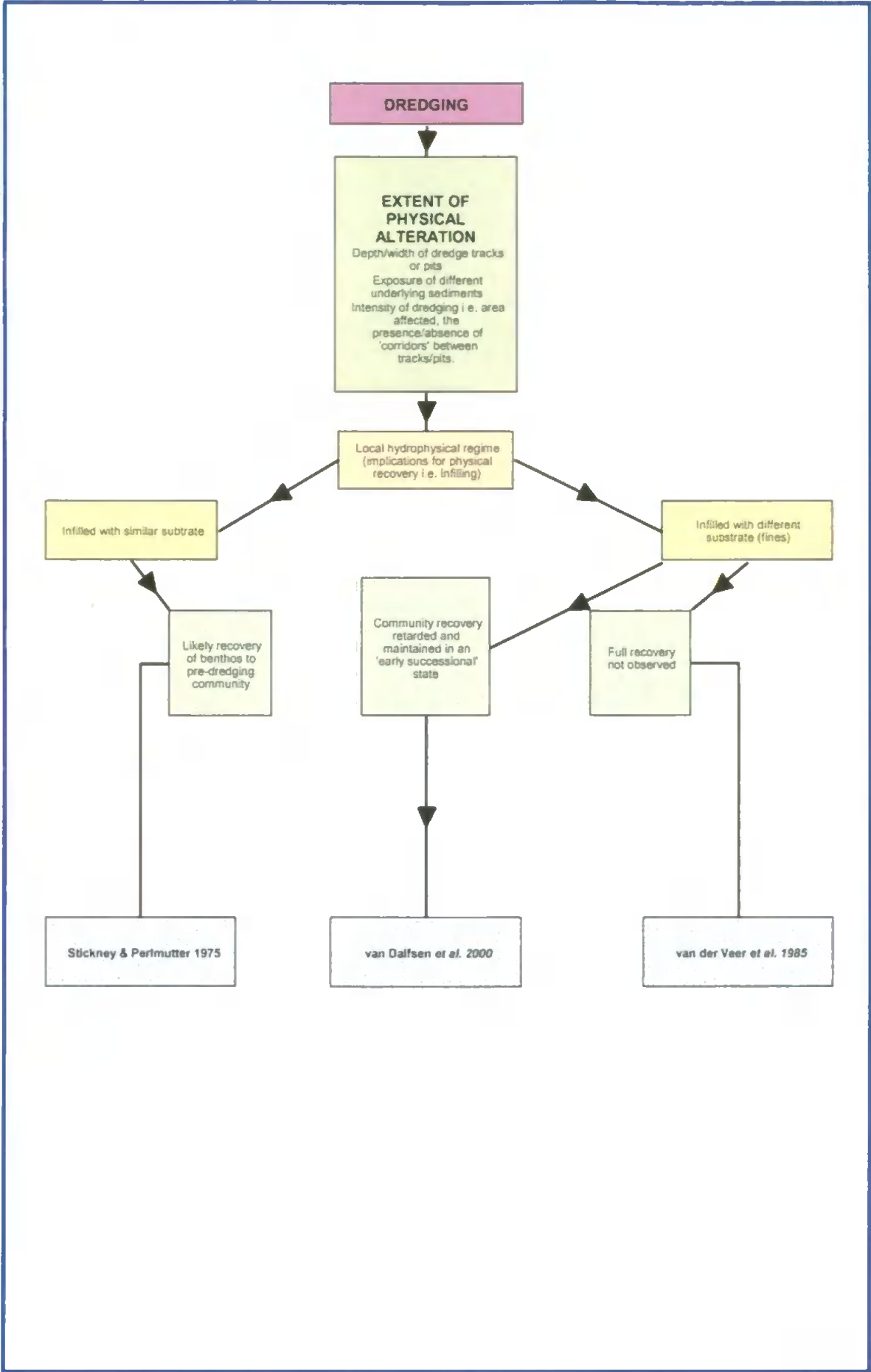
which may have taken many years to develop. Although 'recovering' communities may return to pre-dredging abundance levels, a return to full pre-dredging community structure may never occur (McCauley *et al.* 1977).

The situation in areas characterised by sandy substrata may reflect the unstable nature of such environments. It is impossible to say if the fluctuations shown in the recovery 'curve' were a reflection of the impact of aggregate extraction or simply the result of natural 'background' fluctuations attributable to other processes such as storm episodes. Monitoring of such areas for background fluctuation limits would highlight the impact of the dredging, whilst taking into account the effects of natural processes.

Response in muds and muddy sands could reflect the fact that most studies assigned to this habitat category were estuarine. A relatively limited range of species are able to live in this environment, thus providing a possible explanation for the more complete recovery. Also, if infilling with fines takes place in such situations, the sediment is unlikely to be altered significantly, as it was fine to begin with (although if the pit were infilled and subsequently became anoxic this may lead to the development of an impoverished community). Dense concentrations of infauna in neighbouring, undisturbed, deposits may also facilitate more rapid recovery in such situations. Guerra-Garcia *et al.* (2003) carried out an investigation (not included in the meta-analysis) of impacts resulting from the small-scale dredging of the muddy sediments of a harbour in the Strait of Gibraltar; the substrata and the benthos returned to pre-extraction status six months after the cessation of dredging.

Habitat type has been cited to be of key importance in terms of the extent and rate of recovery. Dernie *et al.* (2003) stated: "*it is generally accepted that the infaunal communities of dynamic, sandy habitats recover from physical disturbances more rapidly than those from depositional, muddy sediments* (Ferns *et al.* 2000; Schratzberger and Warwick 1999)". The ability for the benthos to recover following disturbance by dredging has been reported to be a function of the environment (for review refer to Newell *et al.* 1998). Adding to this viewpoint, Boyd *et al.* (2003) speculated that recovery was likely to be dependent on the ability of the dredged deposits to return to their original composition. Ultimately, however, subsequent 'recovery' of the substratum is dictated by the hydrophysical regime in any given location (Warwick & Uncles 1980).

Figure 90 outlines some general physical disruption scenarios of the seabed following dredging, their implications for impacts and the subsequent recovery of the benthos. By means of relevant published examples, it is evident that biotic recovery appears to be dependent on the extent of substratum alteration; namely, the ability of dredge tracks/pits to be infilled with a substratum similar to that which existed prior to the extraction, in terms of particle composition and stability. It has been demonstrated that, in situations where sediment particle size returned to the pre-dredged condition, recovery of the benthos was more rapid and complete (Stickney & Perlmutter 1975; Kenny & Rees 1994; 1996). However, alteration of sediment stability, as found in the investigations of Kenny & Rees (1994; 1996), can lead to sediments becoming more mobile than they were prior to dredging, which in turn promotes the maintenance of the community in an early state of development by local current, wave and storm conditions. Whilst the species composition returned to that of the pre-extraction assemblage, the individuals contributing to the community were eliminated intermittently before reaching adult



size, thus the community never attained full maturity. An alternative scenario following dredging is that of the dredge tracks/pits become infilled with a substratum different from that which existed before. This situation may persist if local current, wave and/or storm conditions are not of sufficient magnitude, or in the case of storms, only occur infrequently. This may result in an altered, and often impoverished, post-dredging community (van der Veer *et al.* 1985; van Dalftsen *et al.* 2000).

The condition of the deposits in an area subjected to aggregate extraction is dependent on a number of factors. Namely; (1) type and intensity of dredging, trailer and anchor dredging affect the seabed in different ways and with varying levels of severity related to the extent of the area dredged and the depth to which sediments have been removed; (2) habitat in which the extraction has occurred. This is largely determined by local hydrophysical regimes including current speed/direction, depth and seabed topography, which in turn determine the sediment composition. Evidently, recovery scenarios, both physical and biological, following aggregate extraction are determined by a combination of these factors.

The persistence of features, such as dredge tracks/pits, is a function of the sediment framework and the local hydrodynamic regime (Dernie *et al.* 2003). This relationship was highlighted by Eden (1975) who stated that, with the exception of regions characterised by mobile sands, backfilling of dredged seabed features was likely to be extremely slow. Further evidence of the persistence of dredge features was provided by Millner *et al.* (1977) who reported that dredged tracks or pits were likely to remain persistent features of seabed topography in areas of gravel substrata. Calculations have revealed that particles over approximately 6 mm are unlikely to be transported in even the strongest of tides (Hall 1994), suggesting

infilling of dredge furrows/pits with sediment similar to that which existed prior to dredging is unlikely in regions characterised by coarse deposits. Indeed, Dickson & Lee (1973a; b) revealed infilling with fine particles in a deep dredge pit located on a gravel bank. In such an area, under normal conditions, the settlement of fines was likely to be precluded by strong tidal currents, therefore, the depth of the dredge pit (3.5 m) and the resulting alteration of the hydrodynamic patterns inside the pit prevented the transportation of fine particles. Infilling with a contrasting sediment type, following dredging, has also been reported by Van Moorsel (1994) who noted extraction furrows became infilled with fine sands. However, one year on, the return of the pre-extraction community inferred a return to pre-dredging abiotic conditions, in this case, the alteration in sediment type following aggregate extraction was only a temporary situation (Van Moorsel 1994). The shallow tracks enabled erosion (winnowing) of the fine particles by tidal currents. A return to pre-dredging sedimentary conditions, in terms of particle size composition alone, leading to the return of the pre-dredging community may be an over simplistic scenario. If consolidation of the sediment is altered, the substratum may no longer be suitable for inhabitation by an assemblage akin to that present prior to dredging, even if the composition, as revealed by particle size analysis, infers the sediments are unchanged. Clearly, the recovery of the sediments must be viewed in a wider perspective than mere particle size composition alone, and other habitat characteristics (such as sediment cohesion/penetrability) should be considered. For example, if a muddy (estuarine) habitat was dredged exposing underlying clay, particle size analysis would show the dredged sediments and surrounding deposits to be similar, however, the difference between the two deposit types, in terms of compaction, water content and penetrability, (and, as a consequence, the biota they can support) would be profound.

Substratum structure is likely to be important in terms of complete recovery in certain situations. However, in regions where the hydrophysical regime prevents the permanent settlement of a 'new' deposit type, as in van Moorsel 1994 (i.e. fine materials swept away by tidal currents), recovery may be reliant on a number of other factors. The extent of the impacted area, and the intensity of the dredging operations at a given site, has implications for any subsequent recovery. Olafsson *et al.* (1994) stated that the scale of disturbance influences the type of initial colonisers and, therefore, possibly the future stable population. For example, if a location has been subjected to intense trailer dredging over a large area, as is the case for many North Sea licence areas, recovery of species with a planktotrophic larval stage may be favoured, as the distance required to reach the centre of such an area may preclude the repopulation by taxa possessing a benthic larval form, therefore, possibly leading to a prolonged period of succession before the pre-dredging benthic assemblage is attained. Peripheral locations may be more likely to be repopulated by fauna from neighbouring non-dredged sites. Beukema (1982) reported that coastlines defaunated by the formation of winter ice were often recolonised by bivalves with planktotrophic larvae. Areas subjected to large-scale disturbances attributable to storms, oil spills and organic enrichment exhibited a similar pattern of initial recolonisation by planktotrophs (Pearson & Rosenberg 1978; Arntz & Rumohr 1982). Contrastingly, as the spatial scale of disturbance shrinks, so the advantage to planktotrophic taxa diminishes (Scheltema 1986; Frid 1989; Karlson & Levitan 1990). Such a scenario, of a reduced area of disturbance, may be found in a region subjected to light trailer dredging, resulting in undisturbed bands between the dredge furrows. Repopulation by fauna representative of these undisturbed 'corridors' may be favoured, possibly promoting a shorter recovery period. An example relevant to such a pattern of recolonisation is provided by Levin (1984) who showed that recruitment into

settling trays, and defaunated plots at a short distance from undisturbed communities reflected a reasonably unbiased sample of the surrounding fauna.

A further aspect that may negatively effect recovery following physical disruption of the substratum was highlighted by Dernie *et al.* (2003) who hypothesised that disturbance of sediments may lead to the disruption of the redox potential discontinuity layer and to the remobilisation of sulphides that may deter prospective colonists from settling. Recovery following aggregate extraction may be influenced both negatively and positively by the 'processing' of benthic invertebrates as a result of dredging activities. Avoidance of disturbed areas by new recruits, in response to the release of defensive secretions by injured animals, has been observed by Ferns *et al.* (2000). An alternative recovery scenario was highlighted by Groenewold & Fonds (2000) who reported that benthic scavengers, such as hermit and swimming crabs, migrated to areas recently disrupted by beam-trawling in response to the presence of moribund benthic infauna damaged by the fishing gear. In view of the quantities of 'processed' invertebrate matter generated during dredging works (Newell *et al.* 1998), these factors may be of significance in relation to recovery, or even enhancement, of the benthos following aggregate extraction.

The effects of fluctuations in recovery trajectories on the recovery of the fauna may be clarified by the findings of Burd *et al.* (2000) who demonstrated recovery after the disposal of mine tailings was punctuated by the slumping of the waste material, thereby, further smothering an adjacent area. This scenario may also occur in the dredge tracks/pits. Steep-sided pits are prone to slumping (Kaplan 1975). Also, tracks/pits may be infilled with different sediment than the original deposit (Dickson & Lee 1973a; b). Therefore, full 'recovery' may never take place

as environmental alteration prevents it. This change in seabed conditions may provide a possible explanation for the lack of complete recovery in any of the plots based on the meta-analysis in the present study. Jewett *et al.* (1999) also noted that the recovery process was interrupted following several severe storm events. Recovery of the benthos may also be tempered by exposure to other sources of disturbance such as trawling.

Comparisons between the impact attributed to dredging and fishing are not straightforward as both activities disrupt the sediment-water interface in different ways. Whilst dredging removes the substratum, along with the organisms that inhabit it, fishing usually (with the exception of suction dredging techniques) removes very little but still involves physically disturbing the substrata. Similarities between the magnitude of response in different habitats to trawling and dredging are not readily apparent. When the findings of the current study are compared to those of Collie *et al.* (2000), fishing appears to have the greatest initial impact in habitats that comprise muddy sands. In contrast, dredging operations appeared to have the greatest initial impact in gravel deposits, however, the initial response to dredging was, in general, of a greater magnitude than those associated with fishing activities. The latter is to be expected, owing to the tendency for complete defaunation following aggregate mining. Recovery trajectories, as opposed to initial impacts, are therefore more amenable to direct comparison between the two sources of disturbance. Recovery rates following fishing disturbance are far higher than those following dredging. Indeed, recovery in sandy deposits was shown to occur within 100 days. This is in sharp contrast to the recovery response to dredging in sand where full recovery was not demonstrated after a two-year period. An explanation of the contrast in recovery time between the two sources of disturbance may lie in the intensity/frequency of the disturbance events. Dredging

areas are likely to be visited regularly whereas fishing effort is likely to be more patchily distributed (Rijnsdorp *et al.* 1998). This repeated disturbance interrupts any recovery already underway and leads to the fauna being maintained in an altered state until the dredging operations are ceased. This scenario is particularly relevant to more stable habitats where newly-disturbed deposits are colonised by fauna suited to disturbed situations. Less stable environments are likely to have a fauna more readily adapted to repeated disruption and will, therefore, exhibit faster recovery rates.

8.5. Summary

Meta-analysis of a number of published dredging studies, together with data from the current study, has shown that the fauna of gravel deposits exhibit the largest initial negative response to dredging, with that of sand showing the least and that of that of muds being intermediately affected. Complete recovery was not demonstrated in any of the habitat types within the time period examined (approximately 7 years). In substrata characterised by gravels, recovery was slower than in sands or muds. It seems likely that a permanent alteration to the composition of the macrobenthos occurs in many gravel deposits. Recovery in sands was highly variable, a result, probably, of the unstable nature of such deposits. Impacts derived from dredging possibly fall within the range of impacts associated with the natural disturbances to which these deposits are subjected, i.e. storm episodes. Of the three habitat categories, muddy deposits displayed the most advanced point of recovery such that pre-dredging conditions were almost reached after approximately 7 years.

8.6. Recommendations for Dredging Best Practices

Impacts of dredging on the benthos, and its subsequent recovery, are likely to be dependent on a number of factors such as: (1) method of dredging employed; (2) intensity of dredging/spatial coverage of dredged zone; and (3) hydrophysical characteristics of the dredged site (substratum type, current speeds, net sediment transport and depth) that in turn determine the resident species assemblages. Therefore, conservation measures need to be put in place with a view to meeting site-specific criteria.

Anchor dredging is likely to lead to a greater negative impact on the benthos than trailer dredging. The deep pits left in the seabed following anchor dredging are likely to be long-term or permanent features. Recovery of the benthos in such areas is likely to be very slow, as regions where this form of dredging is practiced are often characterised by coarse substrata which support a diverse community of slow-growing, 'equilibrium' species. Therefore, the use of this dredging method should be limited.

Trailer dredging, at the level of intensity employed at the North Nab and Coal Pit sites, resulted in minimal impacts on the benthic invertebrate assemblages. Other investigations of impacts associated with this method of dredging, presumably in areas where greater quantities of aggregates were extracted, however, have revealed significant deleterious effects on benthic communities.

A possible measure to aid recovery following trailer dredging is to leave undredged 'corridors' between dredged furrows. This may facilitate the recolonisation process by preserving benthic colonies in neighbouring deposits. Such a measure may

also increase the likelihood of the post-dredging community returning to that which existed prior to aggregate extraction.

The 'rotation' of dredging sub areas, within the main licence area, may also be useful in reducing impacts on the benthos and is indeed already practiced by some dredging companies. This procedure may be of particular value in habitats of high biodiversity (equilibrium communities) by giving abandoned dredged zones a 'head start' in the, often protracted, recovery process. Whereby, at the end of the period of dredging, at least some of the sub areas are likely to have recovered or will have attained a more advanced stage of recolonisation.



CHAPTER 9. STUDY CRITIQUE

The following section aims to outline problems encountered in this study and to suggest ways to improve future investigations.

CHAPTER 9. STUDY CRITIQUE

A criticism of the current study is the relatively low numbers of samples obtained within the dredged regions compared to the other treatments. This situation would not have occurred had accurate data, regarding dredging intensities and actual locations of dredge tracks/pits, been made available prior to the time of sample collection. Such 'blind' sampling has to be approached with a certain amount of caution. Indeed, there is a 'trade-off' between obtaining sufficient samples in a dredge area to ensure some are located in an actual dredge track/pit, as opposed to in between them, and cutting back on samples that could have been taken elsewhere i.e. control locations. Problems such as these highlight the need for accurate data regarding areas currently dredged and those that are not, to be made available prior to investigation. The location of the licence area alone is seldom useful enough to plan sample locations as, in most cases, only a very small section of this area is actually exploited at any one time for aggregates. An example of this can be clearly seen in the Coal Pit study area (Figure 72). The central region of currently exploited sites was sampled originally with the intention of having two sectors comprising 15 samples each in an actively dredged region. When detailed data were provided later (post survey) regarding actual dredging levels, it was found that only the region shaded red in Figure 72 was subjected to aggregate removal at the time of the survey. However, if a more sporadically spaced sample grid had been employed, there may have been even fewer samples in the actively worked region.

A further problem is that of sample compatibility. The comparison of dredged samples with controls from areas some distance away may be affected by differences in natural environmental factors between the two areas or simply by the fact that samples situated closer to one another are more similar due to natural

population dynamics. A better approach would be to carry out two surveys taking samples inside the dredged area and control sites, before and after dredging. The issue of 'like with like' comparisons is then resolved. However, cost/time constraints, and the inherent complications in identifying an area not currently dredged but likely to be exploited in the near future, generally prohibits this type of investigation with respect to identifying dredging impacts. A dredging impact study carried out by Guerra-Garcia *et al.* (2003) followed the BACI (Before, After, Control, Impacted) approach. This investigation focused upon the impacts of small-scale dredging on the benthic macrofauna inhabiting the fine sediments of a harbour in the strait of Gibraltar. Dredged and control stations were sampled before dredging and on five occasions after dredging (3, 15, 30, 90, and 180 days). Such a sampling programme enabled the authors to monitor both the sequence of recolonisation and the time taken for the sediments and biota to return to their pre-dredging status.

Variation in elapsed time since the dredging event added to the problem of sample compatibility. This was particularly pertinent to the sampling of the anchor dredging regions. Usually, each dredge pit corresponds to a single aggregate extraction episode; therefore, each has a different time interval between the disturbance and the time of sampling. The diameter of these pits (up to 75 m, Newell *et al.* 1998) was such that, in order to ensure a sample had been taken from a particular pit, as opposed to another nearby (or overlapping) one, samples were required to be obtained from as near to the centre of the pit as possible. This required very accurate handling and positioning of the vessel, a task complicated by strong currents or adverse weather conditions.

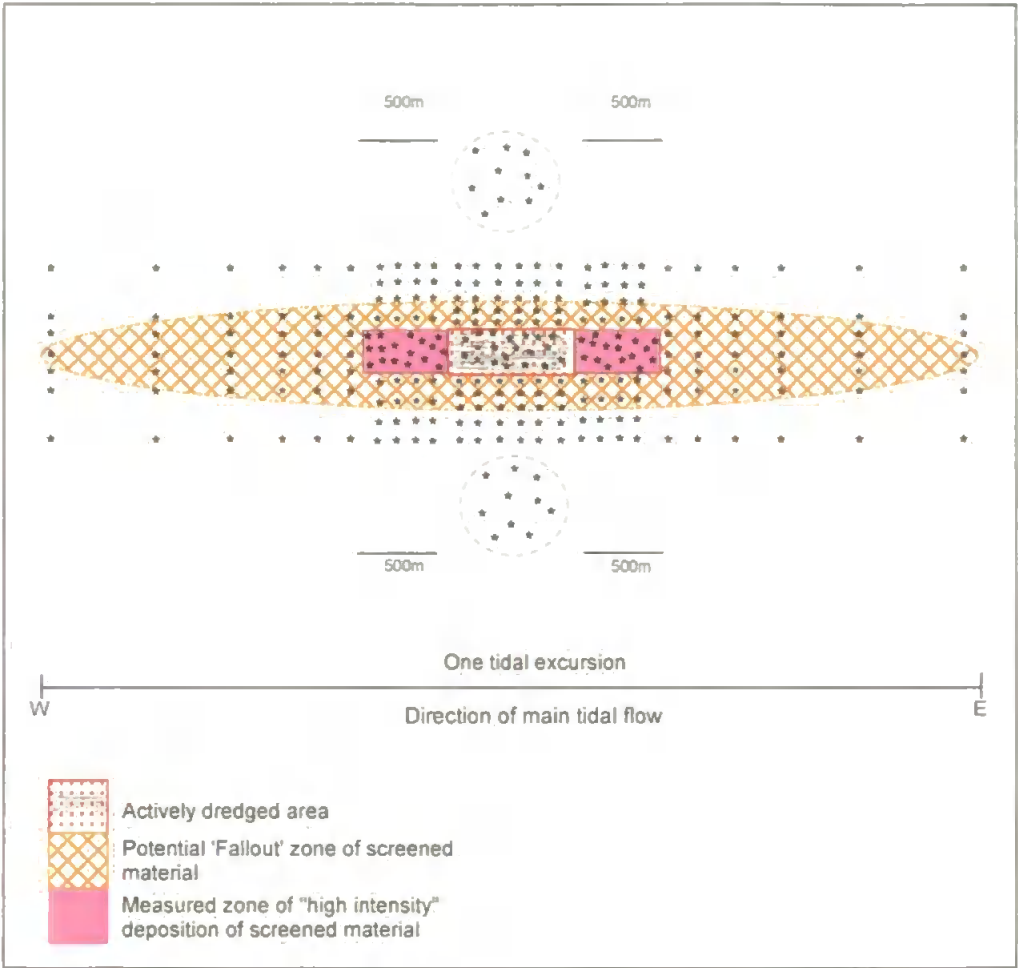
In trailer dredging scenarios, a sampling grid incorporating greater spatial separation is required due to the wide-ranging nature of the extraction process. Problems such as these highlight the need for accurate data regarding areas currently dredged and those that are not, being provided prior to an investigation. Such wide, potentially-impacted areas also create difficulties in the selection of control sites. Stations must be positioned across the axis of the main tidal flow in order to avoid possible influence from the settlement of material rejected or spilt during the dredging process. Suitable control sites must also be located outside the area previously dredged, which may span a considerable distance and, therefore, introduce the problem of natural spatial variation between samples that may be separated by considerable distances. The collection of control samples some distance from the impacted samples may potentially increase the likelihood of a contrasting habitat type being encountered and, therefore, leading to further sample incompatibility. Sampling multiple control sites is one way of combating this problem (Underwood 1992).

A model of an 'idealised' dredging impact survey requires samples to be taken not only in the area from which aggregates have been removed but also in the zone where 'far-field' effects attributable to resuspension of rejected (or disrupted, in the case of unscreened areas) sediments are likely. Such a sampling grid is heavily reliant on the provision of accurate data regarding the fate of material rejected by overboard screening and the location of the actual dredge tracks. Consequently, before the benthic sampling can be initiated, a preliminary survey must take place. This initial survey should be synchronised with an active dredging event in the chosen area. Using acoustic Doppler current profiling (ADCP) techniques accurate measurements of the zones covered by fallout material can be made (Gajewski & Uscinowicz 1993; Whiteside *et al.* 1995; Hitchcock & Drucker 1996; Hitchcock

1997). Side-scan sonar or swathe bathymetry can be employed to locate the dredge tracks or pits (Dickson & Lee 1973a,b; De Groot 1979b; Hitchcock 1997; Limpenny *et al.* 2002). Figure 91 shows a hypothetical sampling grid. For an investigation of dredging impacts of a screened area, this example assumes ADCP and side-scan sonar data has previously been amassed. Multiple samples are taken in the following treatments:

- (1) The directly dredged area, samples to be taken as closely as possible to dredged tracks, as identified by side-scan sonar/swathe bathymetry;
- (2) The zone of high intensity 'fallout' i.e. the region of highest potential deposition of screened material as measured by ADCP. Most of the material rejected by overboard screening has been shown to settle within 500 m of the dredge zone (Hitchcock & Drucker 1996);
- (3) Across the axis of main tidal flow, either side of the dredged zone and zone of 'high Intensity fallout', these samples are placed with a view to detecting the width of any potential effects of remobilised sediments. Gajewski & Uscinowicz (1993) demonstrate that the width of such a plume was approximately 50 m for a trailer dredger travelling on a straight course. However in most instances trailer dredging occurs along a meandering course, thus plume width can vary considerably;
- (4) Along the axis of potential dispersion of material rejected during the overboard-screening process, up to a distance of one tidal excursion in either direction of the main tidal flow. Again, samples should be positioned in transects across the direction of main tidal flow in order to identify the possible width of the plume;
- (5) Two 'blocks' of control samples, taken across the axis of the main tidal flow to ensure no influence of settling material rejected during the screening process

It should be noted that this approach necessitates the collection of large numbers of samples (281 in this example), however, the sample grid can be tailored to suit the objectives or time/labour cost constraints of a particular investigation. If the



investigator is only concerned with impacts within the dredged area, samples along the tidal excursion route can be omitted. Likewise, if a study is concerned primarily with impacts outside the dredge zone, samples within it can also be cut back. In order for such a study to yield sufficient information to detect even subtle alterations to benthic community composition, both biomass and numerical abundance of all species should be recorded from each sample, again this must be considered in terms of additional time and labour costs.

Complex nested sampling designs suited to techniques involving the response of a single variable (i.e. analysis of variance) are not suitable in terms of the number of replicates required for the investigation of impacts associated with the extraction of marine aggregates. Serious practical constraints on sampling designs and statistical analysis in real world situations include: randomised allocation of impact sites; assumptions of particular statistical distributions; data sets with many zeros, these are not amenable to an approach of this type. Univariate techniques (i.e. the response of a single variable) may not be accurate in defining impacts. For example, the number of species may stay the same but the constituent species themselves may be different and have therefore been altered by the disturbance event. Therefore, a multivariate approach (community level) (Clarke & Warwick 1994) is more suited to this type of investigation due to its robustness in detecting alterations in benthic assemblages in situations where the use of a univariate method may be precluded by the complexity of the study site. For example, a large-scale impacted area (both directly and indirectly) increases the likely spatial variation between controls and dredged sites. Also regions characterised by large numbers of moderately abundant species may lead to one carrying out large numbers of univariate tests (one for each species), further tests to detect error and power are also required, adding to the analytical load. Further to this, the use of

numbers of species and individuals are often misleading, as they do not reveal which species contribute to abundances.

The use of multivariate techniques such as those described in this study would appear to offer many advantages over more 'traditional' univariate analyses in the investigation of dredging related disturbance. Recently developed indices such as taxonomic distinctness (Warwick & Clarke, 1995, 1998; Rogers et al. 1999; Price 2002; Pérez-Losada & Crandall 2003) offer a unique approach to defining impacts and warrant thorough testing with regard to the detection and interpretation of changes in response to aggregate extraction.

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