Sandy beach monitoring to detect impacts against a background of long-term trends and variability in intertidal macroinvertebrate communities: an Orkney case-study

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Abstract

Orkney Islands Council Harbour Authority's (OICHA) long-term intertidal macroinvertebrate monitoring data from Scapa Flow, Orkney sandy beaches were reviewed, processed and analysed. Monitoring data for 13 sandy beaches were considered, and these are all characterised as Dissipative or Ultra-dissipative reflecting the sheltered nature of the sandy beaches. The impacts of variability and inconsistencies in macroinvertebrate sample identification and enumeration on data analysis were evaluated. In validation of recent data, it is found that abundance is reliably characterised, but with some inconsistencies in assignment of specimens to taxa are observed. The time series (1974-1990 and 2002-2016) of macroinvertebrate data were analysed for temporal (between year) and spatial (between site) variability; no Scapa Flow-wide temporal patterns are detected. At three sites temporal and spatial variability were investigated in detail and revealed shifts in macroinvertebrate time series in 2010/2011 due to extreme cold winters. Baseline macroinvertebrate data and Ecological Quality for the 13 Scapa Flow sites were described; the mean number of taxa (family level) is high (48) and in agreement with the expected number of taxa for sheltered sandy beaches. All sites are classed as having at least slightly disturbed ecological condition with one being classed as moderately disturbed in both recent (since 2002) and historical (1974-1990) time periods. Recommendations to OICHA regarding the future of the monitoring programme are given and include but are not limited to: continue the monitoring of ten sites in case of oil pollution; continue to monitoring of three sites for the effects of organic effluent discharge from Stromness waste water treatment facility; consider including the sandy beach monitoring as part of the OICHA non-native species monitoring programme; and reduce the sampling frequency at Dead Sand which is a moderately disturbed site.

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Table of Contents

Chapter 1 General introduction

In this chapter, a background to the Orkney Islands Council Harbour Authority (OICHA) sandy beach monitoring programme will be given with an overview of Scapa Flow as a monitoring location. The classification of sandy beaches and the use of benthic macroinvertebrates in monitoring are described and a thesis outline is specified.

1.1 History of the OICHA sandy beach monitoring programme in Scapa Flow, Orkney Islands

In early 1973 oil was discovered in the North Sea at the Piper field by the Occidental International Consortium (OXY) (ICOE 2016a). After the discovery of the oil and after the decision to build an onshore oil handling terminal, the OXY group explored eight options for their onshore oil handling terminal. The OXY group decided to build their onshore oil handling terminal on the island of Flotta in Scapa Flow, Orkney, including landing of a pipeline on the island (ICOE 2016a) (Figure 1.1). The island of Flotta was seen as an ideal location; the island is located in the sheltered, deep water of Scapa Flow, it is protected from severe wind, wave and current conditions, and it was the nearest sheltered harbour from the oil field suitable for an oil terminal (Howie et al. 1975). The oil handling terminal would receive, process, store and export crude oil and derived products (ICOE 2016b) and would therefore constitute a substantial undertaking in both construction and in operational phase. The development of the oil handling terminal was the first site in the UK to undergo an Environmental Impact Assessment (EIA) (ICOE 2016b) and this included several studies including a study on marine ecology (ICOE 2016b). The EIA was conducted in collaboration with Orkney Islands Council (then the Orkney County Council) and local and national stakeholders and concluded that an indepth analysis was required for two areas concerned: 1) Assessment of the Impact on the Marine Environment and 2) Assessment of the Visual and Landscape Impact (ICOE 2016b). In response to the first of these requirements the Orkney Marine Biology Unit (OMBU) was established in July 1974 by Dundee University on behalf of Orkney Islands Council (Jones 1974).

OMBU's aims were to design, establish and carry out baseline marine intertidal surveys in Scapa Flow, Orkney prior to the Flotta oil handling terminal becoming operational in 1976, and therefore providing extensive baseline data collected over a 2.5-year period (Jones 1974; Jones & Simpson 1976; Jones 1980). After the oil terminal became operational, an on-going marine intertidal monitoring programme continued the studies started during the baseline monitoring. Both the baseline studies and the on-going monitoring programme used quantitative methods (sampling along fixed transects) and population studies (gastropod population structure and growth studies; measurements for allometry (gastropod species shell length, breadth, height and weight, aperture length and soft tissue wet and dry weights)) acting as surrogate measures to determine the state of the marine environment (Jones 1974; Jones & Simpson 1976; Jones 1980). Sandy beach surveys formed part of the quantitative transect studies and were started in July and August 1974 at ten sites: Bay of Quoys, Waulkmill, Swanbister, Mill Bay, Longhope, Lyrawa Bay, Stromness, Scapa Bay, Roeberry Taing and Creeklands Bay (Figure 1.1) (Jones 1974; Jones & Simpson 1976). In 1982-1984 a further five sites were added to the sandy beach monitoring programme: Widewall (1982), Kirk Hope (1983), Congesquoy (1983), Cumminess (1984) and Dead Sand (1984) (Jones 1985) (Figure 1.1). Annual monitoring of the macroinvertebrate communities at the sites continued until 1989/90 when the arrangement between Orkney Islands Council and Dundee University was terminated (Jones et al. 1991).

In 1990 the Marine Environmental Unit (then the Environmental Unit) was set up as part of the Orkney Harbour Authority (then Harbours Department). This integration of the Marine Environmental Unit to the Orkney Harbour Authority, and therefore to Orkney Islands Council, was decided by the then Director of Harbours as a cost-effective solution to reduce the running costs of the Marine Environmental Unit and the costs of the ongoing intertidal monitoring programme. This change affected the monitoring programme severely; the sandy beach macroinvertebrate and other intertidal monitoring (rocky shore quadrat and population studies) ceased in 1990 and were replaced by other studies and work concentrating on different aspects of the Harbour Authority's activities.

The sandy beach monitoring was subsequently re-started in 2002 at seven sites (Scapa, Swanbister, Waulkmill, Widewall, Congesquoy, Cumminess and Dead Sand) and in 2006 at six sites (Creekland, Kirk Hope, Longhope, Lyrawa, Mill Bay and Quoys) (Figure 1.1), the monitoring at these thirteen sites is still on-going. No paper records or background information detailing the reasons behind the re-starting of the programme, methods used, or the site selection are available.

Information on the methods at both Historical and Current time periods is given in Chapter 2. A description of the sandy beach monitoring sites is given in Chapter 3.

Figure 1.1. Sampling locations in Scapa Flow, Orkney Islands in Historical and Current monitoring periods. Source: OIC.

A very similar soft-shore macroinvertebrate intertidal monitoring programme to the one in Scapa Flow was conducted in Sullom Voe, Shetland, where the Sullom Voe Oil Terminal is located (Jones & Jones 1981; Jones 1995). The intertidal macroinvertebrate monitoring at two sites, Dales Voe and Gluss Voe, in Shetland were carried out in 1977- 1984 alongside a sub-tidal monitoring programme which included 12 sub-tidal sampling stations within Sullom Voe (Jones & Jones 1981; Jones 1995). The sandy shore monitoring was instigated by the Shetland Oil Terminal Environmental Advisory Group (SOTEAG) and the work was contracted to Dundee University who had implemented and at the time were carrying out the Scapa Flow sandy beach macroinvertebrate monitoring programme (Jones & Jones 1981; Atkins et al. 1985; Jones 1995). The sandy beach macroinvertebrate monitoring at the two sites in Sullom Voe was terminated in 1984 (Jones 1995). In recent years (2014-2018) two soft sediment sites (Gluss Voe and Houb of Scatsta) have been included in the SOTEAG's sandy beach macroinvertebrate monitoring programme (SOTEAG 2014, 2016, 2018) but only samples for hydrocarbon analysis and grain size distribution have been collected at these two sites (SOTEAG 2014, 2016, 2018; R. Kinnear pers comm.).

1.2 Monitoring vs. surveillance

Monitoring is the systematic sampling and re-sampling (of e.g. an area) for a defined reason and for a defined end-point, compared to surveillance which is solely sampling for the observation of trends (Elliott 1993; De Jonge et al. 2006; Gray & Elliott 2009). Several types of monitoring for different purposes were discussed by Gray & Elliot (2009) and these are listed in Table 1.1. In Scotland compliance monitoring is carried out by the Scottish Environment Protection Agency (SEPA) in its regulatory role for licensing different types of discharges to the aquatic environment, including for water quality and biological monitoring assessment and classification in relation to the requirements of the EU Water Framework Directive (SEPA 2019). The OICHA sandy beach monitoring programme falls under the definition of operational monitoring as the monitoring is carried out by industry i.e. the Harbour Authority, however no clear end-point for this monitoring has ever been set. As no no-end point has been set for the OICHA monitoring programme it could also be classed as surveillance monitoring; the aim for the monitoring has been to detect trends with action then considered.

Type	Nature or reasons for	Benthic example
	monitoring	
Surveillance monitoring	A 'look-see' approach (i.e. what is there?), it may be started without determining the end points and relies on <i>post hoc</i> detection (a posteriori detection of trends with action then determined)	A wide-scale survey of an area, the primary and secondary community characteristics (species, diversity, abundance, $etc.$)
Condition monitoring	conservation bodies Nature (surveillance) to determine the present status of an area; it could be linked to biological valuation	If nature conservation area has been designated for its benthic community or for the presence of rare benthic species, then its condition needs to be monitored
Operational monitoring	Carried out by industry (e.g. dredging scheme) and may be linked to the aims of the management	To determine whether and area is silting and needs further dredging for deepening to allow vessel movements
Compliance monitoring	To determine if an area or an industry complies with a set of conditions laid down by licence; the licence could be for effluent discharge, disposal at sea, etc. As part of 'polluter pays principle', the industry will be required to fund the monitoring	An industry, e.g. a sewage or chemical works will be given a licence/permit (e.g. from an Environmental Protection Agency) to discharge which may contain a condition to monitor the bed community to ensure no harm is caused by the activity. A dredging company will be given a disposal licence

Table 1.1. Types of monitoring, from Gray and Elliott (2009).

1.3 Scapa Flow as a monitoring location

Scapa Flow is a large (324.5 km^2) naturally sheltered deep water area in the southern part of the Orkney Islands (Figure 1.2). It is sheltered in the east by the Orkney Mainland and the islands of Lamb Holm, Glimps Holm, Burray and South Ronaldsay, all of which are connected to Mainland Orkney by four Churchill Barriers which further increase the level

of shelter from wave and tidal movements. In the south west and west of Scapa Flow the islands of Hoy, South Walls and Graemsay provide shelter. Access to Scapa Flow from the south is through Sound of Hoxa which leads to the fast-moving tidal area of Pentland Firth, in the west Hoy Sound gives access to the west coast of Orkney and to the Atlantic Ocean (Figure 1.2).

Figure 1.2. Location of Scapa Flow, Orkney Islands in relation to Scotland and UK. Source: Open Street Map, ArcGIS.

The tidal movement within the Scapa Flow area is limited (Figure 1.3). Jones (1980) indicate a residence time in excess of one year for all the water within Scapa Flow, however, Woolf (2017 pers. comm.) states that less than one year is more likely for the "back waters within Scapa Flow". The area does not receive ocean swell due to its sheltered character and therefore the wave exposure of the coastline within Scapa Flow is low (Figure 1.4). All waves within Scapa Flow are wind generated and the shores have a maximum fetch of 20 km (Murray et al. 1999). The prevailing wind direction during the monitoring period has been from south-east (Figure 1.5a, and Appendix A), with the

storm events (Beaufort Scale 10 and above) approaching from west (Figure 1.5b, and Appendix B).

Figure 1.3. Tidal stream plot for Scapa Flow. Modelled at high water and 6 hrs after high water at Spring tides and references to Widewall Bay tidal information. Model based on The Orkney Islands Modelling System which is a 2-dimensional, high-
resolution hydrodynamic and water quality model of the Orkney Islands and their surrounding coastal wate Energy and Water Consultancy Services. It has been previously used for assessments of wastewater discharges and ballast water in the coastal environment and accepted as fit-for-purpose by the Scottish Environment Protection Agency (SEPA) Figure 1.3. Tidal stream plot for Scapa Flow. Modelled at high water and 6 hrs after high water at Spring tides and references to Widewall Bay tidal information. Model based on The Orkney Islands Modelling System which is a 2-dimensional, highresolution hydrodynamic and water quality model of the Orkney Islands and their surrounding coastal waters built by Intertek Energy and Water Consultancy Services. It has been previously used for assessments of wastewater discharges and ballast water in the coastal environment and accepted as fit-for-purpose by the Scottish Environment Protection Agency (SEPA) and Marine Scotland (Langley per. comm.) (Orkney Islands Council Harbour Authority 2017). and Marine Scotland (Langley per. comm.) (Orkney Islands Council Harbour Authority 2017).

Figure 1.4. Wave Exposure Index for Orkney Islands. Created using the Marine Scotland Science National Marine Plan interactive (NMPi) Atlas.

Figure 1.5. Sandy Hill, Orkney Islands. A. Mean wind speed in 2016, B. Maximum wind speed (Force 10+) in 2016. For 2002–2016 wind data see Appendix A and B. Created by K. Beaton using Orkney Islands Council Harbour Authority's wind data.

1.3.1 Shipping activity in Scapa Flow, 1977-2016

After the Piper Alpha field became operational in December 1976 a second oil field, Claymore, was discovered in the North Sea by the OXY group (ICOE 2016b). The Claymore field became operational in December 1977 and the combined production of oil from the two oil fields resulted in total of 323 ship movements in Scapa Flow in 1982 (Figure 1.6). The activity was sustained at this level until 1988 when the Piper Alpha disaster occurred with a loss of 167 people (Paté-Cornell 1993). After the disaster the OXY group sold both Piper Alpha and Claymore oil fields to Elf Aquitane (ICOE 2016b) and by 1994 the production was back to pre-disaster levels. Since 1999 the activity has been decreasing with an all-time low of only 13 ship movements in 2013. The Orkney Islands Council Harbour Authority's revised Ballast Water Management Policy (BWMP) for Scapa Flow was approved by the OIC in December 2013 (Orkney Islands Council Harbour Authority 2017) and the new Oil Transfer Licence was approved in 2015 by Maritime and Coastguard Agency (MCA). From 2014 onwards, the amount of oil products exported to Flotta Oil terminal from North Sea platforms has been on the increase. Concurrently the number of ship-to-ship transfers in Scapa Flow have risen since the Oil Transfer Licence approval, resulting in increased oil related shipping traffic in Scapa Flow (Figure 1.6).

Figure 1.6. Number of ships in Scapa Flow transporting crude oil, propane or ethane or carrying ship-to-ship (STS) transfers of oil or liquefied natural gas (LNG) in 1977-2016. Source: Orkney Harbour Authority.

1.4 Sandy beaches

The intertidal area of a sandy beach is defined by the tidal range which is marked by the low and high tide lines (Figure 1.7). Below the low tide is the sea and above the high tide is the splash zone, and in many sandy beaches, a sandy dune system prevails (McLachlan & Defeo 2018). The intertidal area is divided into high, mid and low zones, each supporting a distinct assemblage of macroinvertebrates (Dahl 1952; Armonies & Reise 2000; McLachlan & Defeo 2018).

Figure 1.7. Diagram of intertidal zones on a sandy beach. From: https://coast.noaa.gov/data/SEAMedia/Lessons/G3U1%20Overview%20Shorel ine%20Habitats.pdf

Sandy beaches vary from oceanic, to sheltered beaches and lagoons to estuarine sand flats (Brown & McLachlan 2002). In all types of sandy beaches their physical characteristics and biota are defined by waves, wind and sand (McLachlan & Defeo 2018). To describe a sandy beach, several physical parameters are required: the width of the intertidal area, wave height and frequency, tidal range, and the shore profile. These physical parameters influence the sediment grain size on a given sandy beach. Other physical parameters limiting the biota on a shoreline are seawater temperature and salinity. Biological features of a sandy beach can be described by the presence of meio- and macrofauna, macroalgae and by organic matter and nutrient cycling.

In 1983 Short and Wright proposed a classification system for microtidal sandy beaches (Short & Wright 1983). They categorised beaches into three broad types: reflective, intermediate and dissipative, with intermediate types further divided into four different types (longshore bar-through, rhythmic bar and beach, transverse bar and rip, and ridgerunnel or low tide terrace) giving a total of six beach types (Short & Wright 1983). This classification system was further developed by Wright & Short (1984) and by Masselink and Short (1993) to take into account the dimensionless sediment fall velocity (Deans Parameter) and Relative Tide Range (RTR) to characterise beaches into eight types, namely: Reflective, Reflective: low tide terrace with rip, Reflective: low tide terrace without rip, Intermediate, Intermediate: bar and rip channels, Dissipative: barred, Dissipative: non-barred and Ultra-dissipative (Short & Wright 1983; Wright & Short 1984; McLachlan & Defeo 2018) (Table 1.2). Deans Parameter is calculated using the wave height, wave frequency and sand fall velocity. Relative Tide Range is calculated using the tide range and wave height (Short & Wright 1983; Wright & Short 1984). The wave climate (height and frequency), tidal range and sediment grain size are the parameters which shape the beach and affect the macroinvertebrate community composition on a sandy beach (Defeo & McLachlan 2013). The Beach Type classification assists in the understanding of the beach state and the macroinvertebrate communities present.

Table 1.2. Beach Types as defined by Dean's parameter (Ω) and Relative Tide Range (RTR). (Short & Wright 1983; Wright & Short 1984; Masselink & Short 1993; McLachlan & Defeo 2018).

	Dean's parameter	Relative Tide Range (m)
Reflective	\leq 2	
Reflective: low tide terrace with rip	\leq 2	$3 - 7$
Reflective: low tide terrace without	$<$ 2	>7
Intermediate	$2 - 5$	
Intermediate: bar and rip channels	$2 - 5$	>7
Dissipative: barred	>5	
Dissipative: non-barred	>5	
Ultra-dissipative	$>$ 5	

Dissipative beaches are long, shallow beaches with fine sand and a large surf zone, reflective beaches are shorter with a steeper beach face and coarser sand, intermediate beach types fit between these two extremes (Gray & Elliot 2009; McLachlan & Defeo 2018). Wave exposure influences grain size of the sediment on sandy shores. The more energy a beach is exposed to the larger the grain sizes are, fine sand and mud tend to be found in areas with very little water movement (McLachlan & Defeo 2018). These parameters, grain size and exposure to wave action, are important factors for macroinvertebrate communities and determine the species distribution on a shoreline (Dexter 1984) and on different exposure types of beaches (Defeo & McLachlan 2013; McLachlan & Defeo 2018). The number of species on sandy beaches increases with the decreasing exposure to wave action (Dexter 1984; McLachlan & Defeo 2018).

Benthic macroinvertebrates are suited for long-term monitoring due to their size; most are retained on a 500μm mesh, which makes them easy to sample for monitoring purposes (Holme & McIntyre 1971). Macroinvertebrates are relatively sedentary and therefore unable to move away from pollution events or other stressors (Dauer 1993). Macroinvertebrates have frequent recruitment events (Giangrande et al. 1994) and have long life-cycles (≥1 year) (Ysebaert & Herman 2002). Marine benthic macroinvertebrates have been widely studied to describe community structures (Pearson 1970; Beukema 1979; Maurer et al. 1979); to detect pollution induced changes within macroinvertebrate communities (Pearson 1971, 1976; Gray & Mirza 1979; Rosenberg & Möller 1979; Gray & Christie 1983; Hargrave & Thiel 1983; Bilyard 1987; Warwick 1988; Warwick et al. 1990; Dauer 1993; Warwick & Clarke 1993; Kiyko & Pogrebov 1997); as indicators of water quality (Borja et al. 2000, 2004: Prior et al. 2004; Dauvin et al. 2007; Muxica et al. 2007; Borja et al. 2007, 2009; Josefson et al. 2009; Borja et al. 2011, 2012a) and they have been used to describe changes in the marine environment due to climate change (Schlacher et al. 2008; Schückel & Kröncke 2013).

The OICHA sandy beach monitoring programme was established (1) to detect and describe long-term changes in the marine environment of Scapa Flow which may result from industrial development of the region, and (2) to assess the effects of any major oil spills in terms of impacts and recovery rates (Jones 1980). Jones (1980) further explained that intertidal macroinvertebrates were chosen as study organisms as they are well researched and are readily available for on-going monitoring.

1.5 Sample collection, processing and identification

Infaunal benthic organisms are divided into four different class sizes; microfauna ($\leq 63\mu$ m), meiofauna ($63-500\mu$ m), macrofauna (500μ m-5cm) and megafauna ($>5cm$) (Gray 1981). Intertidal sandy beach macrofauna (macroinvertebrate) communities generally consist of polychaetes, amphipods and bivalves (McLachlan & Defeo 2018). Details of sample collection, processing and identification for the Orkney monitoring programme are given in Chapter 3. Here a summary sketch of generalised approaches to sandy shore sampling and sample processing is given rather than setting out methods used in this thesis. Samples of macroinvertebrates from sandy beaches were collected using cores or quadrats at a set transect line from the top of the shore to the bottom of the shore during low tide (Atkins et al. 1985; McLachlan & Defeo 2018). The samples were sieved on site to remove sediment and the residual samples retained in a sample bag. Once in the laboratory the samples were preserved in 4% buffered formalin prior to further processing (Barnett 1984; Atkins et al. 1985; Hemery et al. 2017). Once the samples were placed in the fixative for the minimum required time (Start et al. 1992) they were processed further: the samples were rinsed with freshwater to remove the formalin solution, hand sorted, identified and enumerated. Once the identification and enumeration were completed the data were entered into spreadsheets or into a database (Worsfold & Hall 2010). Each stage of this process is liable for errors and operator variability (Ranasinghe et al. 2003; Haase et al. 2006; Jones et al. 2007; Haase et al. 2010). Ellis (1988) details how without a sufficient Quality Control in place for each stage of a monitoring programme, and especially for identification, the data from the said monitoring programme can become meaningless. To assess the errors in sorting and identifying macroinvertebrates from river surveys Haase et al. (2006) analysed data from 10 different countries. The authors concluded that errors were detected at both sorting and identification stages and that the errors could have been reduced by implementing a Quality Control Programme. Figure 1.8 outlines the elements required for a comprehensive Quality Control (QC) Programme (Elliott 1993; Gray & Elliot 1997; Stribling et al. 2003). The elements are; standardised operating procedures for macroinvertebrate sample collection, processing and for data entry and management; the presence of adequate laboratory equipment and facilities to perform the tasks, e.g. fume hood for rinsing samples and suitable microscopes for the identification of macroinvertebrates. For macroinvertebrate sample processing, experience and training are vital elements and all personnel should be trained in all procedures and supervised as required. After an analyst has completed sample sorting, identification or data entry, a second analyst should QC the same sample or data entry to ensure the sorting has been carried out thoroughly, all species have been identified precisely and accurately and all data entry has been filled correctly (Elliott 1993; Gray & Elliot 1997; Stribling et al. 2003). The QC for the identification (ID) of macroinvertebrates comprises six parts (Figure 1.8) each of which is vital in ensuring producing good quality macroinvertebrate data and maintaining it (Elliott 1993).

The potential for variation in a data set is further increased if the data are collected by different people or in different monitoring periods (Frid et al. 2009; Schooler et al. 2017). The absence of Quality Control Programme in the processes of the OICHA sandy beach monitoring programme in 2002-2016 could potentially introduce variability and errors to the data, and subsequently affect the statistical analysis of the data. To understand these issues a comprehensive investigation of the data, potential errors and variability, was undertaken and presented in Chapter 4.

Figure 1.8. Flowchart of a Quality Control Programme.

1.6 Thesis aims and objectives

Research aim: To assess the state of the long-term (1974-1990 and 2002-2016) macroinvertebrate community data from Scapa Flow, Orkney in order to set the baseline community data and ecological health of the sandy beaches.

Following research objectives would facilitate the achievement of this aim:

- 1. Review and process sandy beach macroinvertebrate data available at OICHA (Chapters 2-7).
- 2. Describe the 13 Scapa Flow macroinvertebrate monitoring sites with specific details on sandy beach location, morphology and site-specific anthropogenic impacts (Chapter 3).
- 3. To investigate the Current time period macroinvertebrate data integrity prior data analysis; the Current time data were produced by several analysts with no Quality Control programme for macroinvertebrate sample processing, identification or enumeration in place. The macroinvertebrate data for three sites (Quoys, Congesquoy and Waulkmill) were re-identified and re-enumerated, providing 'Verified' data for the three sites. Using the 'Original' (as identified and enumerated in the Current time period) and 'Verified' data the impact of variability and inconsistency in macroinvertebrate sample identification and enumeration on data analysis will be quantified, while any errors in the data and issues with laboratory processes will be categorised, and together these will enable an assessment of the integrity of the macroinvertebrate data (Chapter 4).
- 4. To analyse the Scapa Flow macroinvertebrate data to determine any long-term temporal and spatial variability (Chapters 5 and 6). Temporal variability will be investigated in both between-year and between-time periods, spatial variability will be investigated at large scale (within Scapa Flow) and at small scale at sandy beach specific-level (within a sampling station). Large scale Scapa Flow-wide analysis will concentrate on eight sandy beaches (Chapter 5) the small-scale sandy beach-specific analysis will investigate the variability in three sites (Quoys, Congesquoy and Waulkmill) (Chapter 6).
- 5. To develop and test an approach towards the definition of the baseline macroinvertebrate community for the 13 Scapa Flow sandy beaches in the

Historical and Current time periods using dominant taxa as a descriptor against which any future changes or perturbations can be measured (Chapter 7).

6. To define the ecological quality status of the 13 Scapa Flow sandy beaches (using the macroinvertebrate community composition) in Historical and Current time periods against which any future changes or perturbations can be measured (Chapter 7).

1.7 Thesis layout

This thesis describes the monitoring sites, evaluates the methods employed in both the Historical and Current time periods and assesses the impact of variability and inconsistency in macroinvertebrate sample identification and enumeration on data analysis. Long- and short-term spatio-temporal variability in the macroinvertebrate communities at the Scapa Flow sandy beaches are analysed. A baseline macroinvertebrate community structure is described for each of the monitoring sites and the ecological quality status are set, against which any future impacts can be measured. A critical review of the monitoring programme was carried out with a set of recommendations presented to Orkney Islands Council Harbour Authority.

Each data chapter (Chapters $4 - 7$) includes an introduction with background literature relevant to that chapter. Chapter 8 is a discussion chapter, presenting conclusions from the data chapters and a critical review of the monitoring programme with a set of recommendations for Orkney Islands Council Harbour Authority.

Chapter 2 Methods

In this chapter the history of past sandy beach surveys in Orkney is briefly summarised before describing the survey and laboratory methods currently used. Statistical approaches to identifying pattern in macroinvertebrate community composition are also described.

2.1 Sandy beach sampling

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2.1.1 Historical surveys, 1974-1990

When the sandy beach monitoring programme was started in 1974 (Jones 1980) it encompassed ten sandy beach sites: five sites on Hoy (Bay of Creekland, Bay of Quoys, Lyrawa Bay, Mill Bay and Longhope Bay) and five sites on Mainland Orkney (Stromness Bay, Swanbister Bay, Waulkmill Bay, Scapa Bay and Roeberry Taing) (Chapter 1 Figure 1.1).

At each site a fixed transect was established down the centre of the beach with sampling stations at 30 cm vertical intervals from the level of highest astronomical tides down to low water spring tides, using a level and a staff (Atkins et al. 1985). Transects had up to 13 sampling stations which were labelled from Station 0 (Highest Astronomical Tide) to Station 13 and transects varied in length from approximately 76 m to over 400 m (Atkins et al. 1985). Distances between the sampling stations were measured and, together with the vertical heights, were used to characterise shore profiles for each transect.

During the Historical time period sampling over the years varied between sites, with five sites covered most years from 1974 to 1989 (Table 2.1.A) (Jones 1974; Jones & Simpson 1976, 1977; Jones et al. 1978, 1979; Jones 1980; Jones et al. 1981, 1982; Jones 1983, 1985; Jones et al. 1986-1991). Samples were collected annually during the months of June – October (Jones 1974; Jones & Simpson 1976, 1977; Jones et al. 1978, 1979; Jones 1980; Jones et al. 1981, 1982; Jones 1983, 1985; Jones et al. 1986-1991).

In 1974-1977 at each sampling station for macroinvertebrate determination, five 0.1m^2 quadrat samples were collected to a depth of 100 mm (Jones 1974; Jones & Simpson 1976, 1977; Jones et al. 1978, 1979; Jones 1980; Jones et al. 1981, 1982; Jones 1983, 1985; Jones et al. 1986-1991), from 1978 onwards five $0.02m^2$ core (not stated but assumed cylindrical) samples were collected to a depth of 150 mm^{-1} (Atkins et al. 1985; Atkins et al. 1989). Each replicate macroinvertebrate sample was sieved using a 0.5mm

¹ The rationale behind the increased sampling depth is unknown. Given that the macrobenthic taxa considered are overwhelmingly likely to be concentrated in the upper few centimetres of sediment (Holme & McIntyre 1971) we assume that this has no influence on abundance estimates.

mesh sieve; the remaining sample was placed into a labelled container and subsequently fixed with 4% formalin solution (Atkins et al. 1985). On most of the sandy beaches (Bay of Quoys, Bay of Creeklands, Swanbister Bay, Waulkmill Bay, Scapa Bay and Widewall Bay) the upper stations were considered to be unsuitable for macroinvertebrate sampling as they consisted of shingle or bedrock. At these sites the upper stations were not sampled; across all sites, the highest shore stations varied from station 0 to down to station 7 (Atkins et al. 1985).

Table 2.1. Sandy beach surveys carried out during A. Historical and B. Current time periods. From OMBU reports (Jones 1974; Jones & Simpson 1976, 1977; Jones et al. 1978, 1979; Jones 1980; Jones et al. 1981, 1982; Jones 1983, 1985; Jones et al. 1986-1991) and datasheets held at Marine Environmental Unit, Scapa.

Site	A. Historical Time Period																
	1974		1975 1976 1977 1978 1979				1980	1981		1982 1983 1984			1985 1986	1987	1988		1989 1990
Congesquoy										X	X	X	X	X	X	X	
Cumminess											X	X	X	X	X	X	
Dead Sand											X	X	X	X	X	X	
Scapa	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	
Swanbister	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	
Waulkmill	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X		
Wide wall									X	X	X	X	X	X	X	X	
Creekland	X	X	X	X	X	X		X	X								
Kirk Hope										X	X	X	X	X	X		
Longhope	X	X	X	X						X	X	X	X	X	X		X
Lyrawa	X	X	$\mathbf x$	X						X	X	X	X			X	X
Mill Bay	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X		
Quoys	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X		

In the laboratory the macroinvertebrate samples were hand sorted, identified to the highest taxonomic separation, and counted. During the historical sampling period the identification of macroinvertebrates was carried out by Dundee University personnel and students under the guidance of the university's taxonomic experts.

At each sampling station a rectangular $0.02m^2$ core sample was collected for granulometry analysis (no depth of the sample available). The granulometry samples were oven dried overnight at 70°C, analysed using a graded series of Endecott Test Sieves $(2000 \text{µm} - 63 \text{µm})$ at half-phi intervals on an Endecott Test Sieve shaker for 20 minutes and then weighed on a Mettler P163 electronic balance (Jones & Simpson 1977). Sediments left within each sieve were weighed and a sediment profile of the shore was created from these results. Granulometry data analysis is detailed in Section 2.2.5.3.

Organic carbon content was also recorded but these data will not be used in this thesis because no organic carbon content has been recorded for the surveys in the current period.

2.1.2 Current surveys, 2002-2016

After a period of 12 years when no sampling was carried out the sandy beach sampling programme was re-started in 2002 at selected sites (Table 2.1.B). The monitoring included four Mainland Orkney and South Ronaldsay sites: Scapa Bay, Swanbister Bay, Waulkmill Bay and Widewall Bay; and three Bay of Ireland sites: Congesquoy Bay, Cumminess Bay, Dead Sand. In 2006 the monitoring was re-started on seven Hoy sites: Bay of Creekland, Heldale, Kirk Hope Bay, Longhope Bay, Lyrawa Bay, Mill Bay and Bay of Quoys. Sampling at Heldale has been irregular and therefore it is not included in the data analysis; this site was removed from the on-going monitoring programme in 2014. From 2002 onwards, instead of sampling the full transects as they were set up in 1974, the macroinvertebrate sampling was limited to 1-3 stations per site with five replicates at each sampling stations (Figure 2.1). At the time it was decided that these sites and number of stations were suitable for the intertidal macroinvertebrate on-going monitoring programme.

The macroinvertebrate samples were collected using a $0.02m^2$ core (\varnothing 150mm) to a depth of 100mm and sieved using a 0.5mm mesh sieve. In the laboratory all macroinvertebrate samples were fixed in 4% formalin solution with Rose Bengal red stain and stored for at

least 10 days prior to rinsing and sorting. Macroinvertebrate samples were rinsed in a fume cupboard with copious amounts of water until no formalin residues were deemed to be present. Each replicate macroinvertebrate sample was hand sorted from the residual sediment in the laboratory on a large white tray. All macroinvertebrates were placed into

Figure 2.1. Diagram of two sandy shore sampling stations with five replicates core samples from each station. Not to scale. Drawing by E. Gerrie.

small sample tubes and preserved using 70% ethanol (2002-2007) or 1% propylene phenoxetol (2008-2016).

In the current monitoring period (2002-2016) the macroinvertebrate identification has been carried out in-house by the Marine Environmental Unit, Marine Services, Orkney Islands Council, using Leica stereo microscopes. Specimens with their head intact were counted. The samples have been identified to varied taxonomic levels from Phylum (Nemertea), Class (Oligochaeta) to species level when possible (see Section 2.2.4, below). Once identification was completed the results were entered into an Excel spreadsheet. From 2002 onwards, the Unit has had three different biologists and several technicians, which has inevitably led to different levels of in-house expertise (Table 2.4). In 2014 all samples from all sites were sent to a taxonomic laboratory (APEM Ltd) for identification. This was to verify the identification of all the species present and to create a voucher specimen collection to aid the identification of future samples.

At each sampling station a $0.003m^2$ core sample was collected to a depth 100 mm for granulometry analysis. The samples were collected at each sampling station at all sites in 1989, 2006 and 2014-2016. In 1989 and 2006 the analysis was carried out in-house at MEU. The granulometry samples were oven dried overnight at 95°C, analysed using a graded series of Endecott Test Sieves $(2000 \mu m - 63 \mu m)$ at half-phi intervals on an Endecott Test Sieve shaker for 15 minutes and then weighed on a Mettler P163 electronic balance. From 2014 onwards the granulometry samples were analysed by Thomson Ecology Ltd in Guilford using Malvern MS2000 laser diffraction particle size analyser following their TEN10 Particle Size Analysis standard procedure (Thomson Ecology Ltd 2015). Granulometry data analysis is detailed in Section 2.2.5.3.

In 2016 shore profiles for the thirteen sites were surveyed by Karl Cooper - Survey and CAD Services using Sokkia GSR2700ISX base and rover for RTK GPS surveying with logging of a 'static file' for post processing to obtain heights above and below Ordnance Datum and in turn heights above and below chart datum in Scapa Flow as referenced to Scapa pier on the Admiralty Chart (Karl Cooper pers. comm.). Data were logged using Carlson SurvCE v4.07 on a Juniper Allegro 2 data collector and the sections were drawn using AutoCAD 2005 (Karl Cooper pers. comm.). Each site was surveyed in a straight transect line fixed from the top of the shore (either ST0 (Highest Astronomical Tide) or at the top most sampling point) through the sampling stations. At Scapa Bay, Mill Bay and Kirkhope Bay measurements were only taken at the sampling stations, at Sands of Congesquoy, Cumminess Bay, Bay of Creekland, Dead Sand, Longhope Bay, Lyrawa Bay, Swanbister Bay, Waulkmill Bay, Widewall Bay and Bay of Quoys measurements were also collected along the profile (Chapter 3).

No co-ordinates were available for the Historical time period, the sampling stations along the transect line were established using a tape measure, starting from the HAT with set distances between the stations. Grid references were established for each sampling station in 2002, the method for this is not known. It is possible that there is discrepancy between the sampling station locations between the two monitoring periods.

2.2 Data management

2.2.1 Metadatabase creation

The Historical and Current sandy beach monitoring programmes have been on-going since 1974. To understand what data were available over this period the creation of a metadata base was of paramount importance. The metadata base specifies site details, including the type of site, type of data available and dates when samples had been collected from each site. Individual site metadata sheets have also been created which include more detailed information regarding each site, stations sampled, dates and if all Historical data were available.

2.2.2 Historical data, 1974-1990

Historical data for most sites were stored in paper format at the Orkney Islands Council Harbour Authority (OICHA) archives. The datasheets were photocopied and entered into Excel sheets before any data analysis took place. Data for the Bay of Ireland sites, Congesquoy, Dead Sand and Cumminess, from 1982-1990 were already in Excel sheet format.

Due to the Current macrobenthos data being mostly identified to family level, there was a requirement for the Historical data to be converted to family level to enable comparative analyses to be made. The processing of the data was done in several steps: the species names were changed into family names and unique sample identification numbers were created for each replicate sample. Where several species were in the same family, these separate rows of data were summed so that only one value for each family was derived. Once this process was repeated for each site for each year then the Historical data were in a suitable format for analysis.

2.2.3 Current data, 2002-2016

The Current data were stored in Excel data sheets. The sheets were first processed into format suitable for analysis, by creating one long species or family list and populating the data into columns. Unique sample identification numbers were formulated to enable this,

and this was followed by changing species and genus names into family names and summing the rows of data.

2.2.4 Terminology

The terms 'taxa' (plural) and 'taxon' (singular) are used throughout this thesis when referring to macroinvertebrate data that refer to anything higher than species. The identification of the OICHA sandy beach macroinvertebrate samples has always been to the lowest taxonomic level possible but due to the different levels of expertise of personnel over the years this has varied from species level identification to class in some taxa. Because of this it was decided to aggregate all data to family level or higher (e.g. order, class or phylum when appropriate) for the data analysis, taxonomic sufficiency is discussed in Chapter 4 Section 4.3 and Chapter 7 Section 7.1. Taxa aggregated, or only identified to phylum level are: Chordata, Hemichordata, Nemertea, Sipuncula, Phoronida, Platyhelminthes, Echinodermata; to class level: Oligochaeta, Enteropneusta, Sipunculidea; to order level: Brachyura, Cumacea, Decapoda, Mysida (Appendix C).

All names used in this thesis follow the guidance given by the World Register of Marine Species (WoRMS) [http://www.marinespecies.org/index.php.](http://www.marinespecies.org/index.php)

Named authorities for all taxa recorded in this work are listed in Appendix C, as inclusion in the main text would have made the thesis difficult to follow.

During this thesis (2012-2019) taxonomical changes and nomenclature changes have occurred in some of the taxa discussed. In most cases changing the data and thesis to reflect the changes has been possible but in one case the changes were unmanageable. The family name for the amphipod genus *Bathyporeia* at the start of the project was Pontoporeiidae (Hayward & Ryland 1995), near the end of the project this was revised to Bathyporeiidae (Hayward & Ryland 2017). In this thesis *Bathyporeia* species will be assigned to the family Pontoporeiidae with the knowledge that a revision of this genus has occurred.

2.2.5 Numerical analysis methods

Three software programmes were used for the analyses of the macroinvertebrate data; Plymouth Routines In Multivariate Ecological Research (PRIMER) (Clarke & Warwick 2001; Clarke & Gorley 2006), AZTI's Marine Biotic Index (AMBI) (Borja et al. 2012b) and R suite of software facilities for interactive data analysis (R: Core Team 2018).

2.2.5.1 PRIMER v6

The PRIMER v6 programme package is software developed for the analyses of a variety of data (biotic and abiotic) often associated with environmental studies; this includes the analyses of biological data such as arrays of taxa-by-samples data for community ecology (Clarke & Warwick 2001; Clarke & Gorley 2006). It is a well-developed software programme with a user-friendly layout. PRIMER has been widely used in benthic community analyses, for example, to analyse long-term natural variability in benthic macroinvertebrate communities (Kröncke & Reiss 2010); to analyse spatial and temporal differences in community structure within and between sites (Schückel & Kröncke 2013), studying temporal changes in North Sea benthos (Frid et al. 2009; Kröncke et al. 2011) to analyse the shifts in macrofaunal communities due to cold winters (Kröncke et al. 2013) and studying patterns using macroinvertebrate data aggregated across different taxonomic levels (Frid et al. 2009; Blanchard et al. 2010).

In this thesis all taxa were aggregated to family level (where possible) and abundances were standardised for 0.1m^{-2} and then analysed using multivariate routines available within PRIMER v6.

The data were standardised prior to analysis using fourth root transformation. The fourth root ($\sqrt{\sqrt{}}$) transformation is commonly used (Clarke & Warwick 2001) and has the effect of down-weighting the influence of abundant species that would otherwise dominate the analyses (Clarke & Warwick 2001).

The Bray-Curtis similarity coefficient was used to assess similarity in species composition across the different sampled stations. The Bray-Curtis similarity coefficient provides a measure of similarity between samples in terms of their species composition. The Bray-Curtis similarity coefficient gives values between 0-100, where 0 is given if two samples have no species in common, and 100 is given if two samples have exactly the same species composition (Clarke & Warwick 2001). Therefore, the closer the Bray-Curtis coefficient is to 100 the more similar the sites are in their species composition. The results of the Bray-Curtis coefficient are displayed in a triangular matric of similarities; it is this similarity matrix that is used as a starting point for the multivariate analyses of hierarchical clustering and non-metric multi-dimensional scaling (MDS).

In hierarchical clustering the samples are grouped on the basis of similarity and the groups are represented by a tree diagram or dendrogram where the branching structure represents the degree of similarity. Hierarchical clustering with group-average linkage was used as recommended by Clarke & Warwick (2001). With the hierarchical clustering routine in PRIMER it is possible to incorporate 'similarity profile' (SIMPROF) permutation tests, which test whether identified groupings are statistically significantly different from each other (Clarke & Warwick 2001). The results are represented in the cluster dendrogram by colour convention: red lines denote samples which cannot be significantly differentiated, black lines denote samples which are significantly different from each other (e.g. Chapter 4 Figure 4.2) (Clarke & Warwick 2001). The hierarchical clustering analysis groups the samples into discrete groups according to their similarity, rather than representing the inter-relationships of the samples on a spatial continuum (Clarke & Warwick 2001).

The inter-relationships between the samples were analysed using the non-metric multidimensional scaling (MDS) ordination technique (Clarke & Warwick 2001). The MDS routine in PRIMER follows the non-metric MDS procedure described by Kruskal (1964). The non-metric MDS displays the data in a 'map' format, which attempts to satisfy all the conditions imposed by the Bray-Curtis similarity matrix. When displaying the data in this format some distortion or stress is being placed on the similarity rankings (Clarke & Warwick 2001). This stress is measured, and a value given for each ordination, the stress values for 2-dimensional ordinations can be interpreted as stated in Table 2.2.

The hierarchical clustering and non-metric MDS ordination analyses are complemented by the 'similarity percentage' (SIMPER) analysis. The SIMPER routine analyses the species (taxa in this thesis) data and determines the percentage contribution of all species towards the average within group similarity and to the average between group dissimilarity (Clarke & Warwick 2001). In simple terms, results from these analyses show co-presence of species across stations (thus contributing to stations similarity) and also co-absence of species across stations (thus contributing to stations dissimilarity). The dendrogram and MDS ordination plot show how the samples are clustered and displayed as a 2-dimensional 'map', the SIMPER results give an indication of which individual species either contribute to the within group similarity or between group dissimilarity. The SIMPER routine is performed on the fourth root transformed data and requires replicates. It is therefore not possible to perform SIMPER test on the data when the replicates are summed.

Table 2.2. MDS stress values with interpretation of the values (Clarke & Gorley 2006).

	MDS stress value	Interpretation
Stress < 0.05		Excellent representation with no prospect of misinterpretation
	Stress < 0.1	Good representation, no real prospect of misleading interpretation
	Stress < 0.2	Gives a potentially useful 2-dimensional picture
	Stress > 0.3	The points are close to being arbitrarily placed

The DIVERSE routine in PRIMER was used to calculate the Shannon-Wiener Diversity Index (Clarke &Warwick 2001). The Shannon-Wiener Diversity Index (here referred to as 'Shannon') is the most commonly used diversity measure (Clarke & Warwick 2001; Labrune et al. 2006). The Shannon Diversity Index accounts for both the richness, i.e. number of taxa present, and evenness, i.e. number of individuals of each taxon present in the sample, of the taxa present in the sample. It is calculated using the following formula:

 $H' = \sum_i p_i \log_2(p_i)$

Where p_i = proportion of the total count arising from the *i*th species.

The Shannon Diversity Index (H') increases as the number of species increases, but H' will also increase as the proportion of individuals per species becomes more constant (Gray & Elliott 2009).

2.2.5.2 R software

R suite of software facilities for interactive data analysis (R: Core Team 2018) was used for statistical analysis of the data. Macroinvertebrate data were analysed using analysis of variance (ANOVA) on 4th root transformed data judging significance according to permutation tests using R library lmPerm (Wheeler & Torchiano 2016). ANOVA was used for testing for difference in abundance of taxa between year groups of samples.

2.2.5.3 Granulometry data analysis

Sediment particle size data for both Historical and Current time period were analysed using GRADISTAT v8.0 programme (Blott & Pye 2001). In order to characterise the sediment properties collected at each site GRADISTAT was used to calculates the mean, median, mode, sorting, skewness and kurtosis arithmetically and geometrically (in metric units) and logarithmically (in phi units) using moment and Folk and Ward methods (Blott & Pye 2001). The GRADISTAT programme provides results in both tabulated and graphic form. The grain size descriptions used in the GRADISTAT programme are presented in Table 2.3. The results were based on the median grain size of the overall granulometric profile for each sample.

Grain size		Descriptive terminology						
phi $mm/\mu m$		Udden (1914) and Wentworth (1922)	Friedman and Sanders (1978)	GRADISTAT program				
-11	2048 mm		Very large boulders					
			Large boulders	Very large				
-10	1024		Medium boulders	Large				
-9	512	Cobbles	Small boulders	Medium	Boulders			
-8	256		Large cobbles	Small				
-7	128		Small cobbles	Very small				
-6	64			Very coarse				
-5	32		Very coarse pebbles					
-4	16	Pebbles	Coarse pebbles	Coarse				
-3	8		Medium pebbles	Medium	Gravel			
-2	4		Fine pebbles	Fine				
-1	2	Granules	Very fine pebbles	Very fine				
		Very coarse sand	Very coarse sand	Very coarse				
Ω	1	Coarse sand	Coarse sand	Coarse				
1	500 um	Medium sand	Medium sand	Medium	Sand			
2	250	Fine sand	Fine sand	Fine				
3	125	Very fine sand	Very fine sand	Very fine				
4	63							
5	31		Very coarse silt	Very coarse				
6	16	Silt	Coarse silt	Coarse				
7	8		Medium silt	Medium	Silt			
8	4		Fine silt	Fine				
9	\overline{c}	Clay	Very fine silt	Very fine				
			Clay	Clay				

Table 2.3. Sediment grain size adapted for the GRADISTAT programme. From Blott & Pye (2001).

2.3 Beach morphometric information

Beach morphometric information details the physical characteristics of a sandy beach, which can be used to calculate beach indices for categorising beach types (McLachlan & Defeo 2018).

The following physical measurements were included in the beach morphometric calculations: mean sediment grain size (μm) , seawater temperature $({}^{\circ}C)$, salinity, sand fall velocity (cm/s) and tidal range (m) (Appendix D). These values were used to calculate: wave height (cm), wave frequency $(s-1)$, wave period (s) and slope $(°)$ (Appendix D). These were used when calculating the Dean's parameter (Ω) , Relative Tidal Range (RTR) and Beach Index (BI) (Appendix D).

Dean's parameter (Ω) is a dimensionless fall velocity and is an index of the ability of waves to move sand on the beach (McLachlan & Defeo 2018). RTR is a measure which combines the influence of waves and tides on the beach (McLachlan & Defeo 2018). BI is used by ecologists to compare sandy beaches with different tidal ranges, Slope (°) is
the reciprocal of beach face slope and is used to compare sandy beaches with a similar tidal range (McLachlan & Defeo 2018). BI includes values of slope, sand and tide, Slope (°) includes the measurements of sand, tides and waves (McLachlan & Defeo 2018).

The beach morphometric calculations are presented in Appendix D. The results for each site are presented in Chapter 3.

2.4 Personnel during the monitoring programme

During the historical monitoring period (1974-1990) two members of the personnel remained constant, namely the Director and the Scientific Officer. A Technician was part of the team for eleven years from 1976 until 1987. Several field assistants were employed during the historical part of the monitoring programme. Some of the field assistants were part of the team for one season, others returned for several years. During the first year of the monitoring programme the sample collection was carried out by the Director, Scientific Officer and the seasonal field assistants. After the monitoring programme was established the sampling was carried out by the Scientific Officer, Technician and seasonal field assistants with Director joining them occasionally. The samples were hand sorted immediately after the sample collection by everyone involved in the sampling. After receiving training in the identification of macroinvertebrates the identification was carried out by the Scientific Officer and the Technician. Intermittently some samples were sent to Dundee University for verification by the Director and to be included in a Dundee University voucher specimen collection. Consistency in the programme was maintained by the continued presence of the same Director and the same Scientific Officer.

During the Current time period (2002-2016) there have been four posts within the Marine Environmental Unit; Scientific Officer, Biologist, Technician and Summer Student (Table 2.4).

The sandy beach sample collection has been carried out by the Scientific Officer, Biologist and Technician until 2011. In 2011 and from then on, the sampling has been carried out by the Biologist and the Technician. The hand sorting of the samples has mainly been carried out by the summer students, the identification of the samples has been carried out by the Biologist and the Technician, occasionally the hand sorting has been carried out by the Biologist and Technician and occasionally the identification has received assistance from the summer students. Consistency to this period has come from the presence of the same person as a Scientific Officer. The effects of changes in

personnel and their differences in levels of taxonomic expertise are considered in Chapter 4.

rapic 2.4. Matric Environmental Unit personnel, $2002-2010$.									
Year	Scientific Officer	Biologist	Technician	Summer student					
2002	SO ₁	B1	T1	SS ₁					
2003	SO ₁	B1	T1	SS ₂					
2004	SO ₁	B1	T1	-					
2005	SO ₁	B ₂	T1	SS ₃					
2006	SO ₁	B ₂	T1	SS ₄					
2007	SO ₁	B ₃	T1	SS ₅					
2008	SO ₁	B ₃	T1	-					
2009	SO ₁	B ₃	T1 (until Feb'09)	SS ₃					
2010	SO ₁	B ₃	$T2$ (from May'10)	-					
2011	SO1 (until Feb'11)	B ₃	T ₂						
2012	٠	B ₃	T ₂	SS ₆					
2013	$\overline{}$	B ₃	T ₃ (mat. cover)	SS ₇					
2014	$\overline{}$	B ₃	T ₂	SS8 (same as T3)					
2015		B ₃	T ₂	SS ₉					
2016	$\qquad \qquad -$	B ₃	T2	SS ₉					

 Table 2.4. Marine Environmental Unit personnel, 2002-2016.

Chapter 3 Description of study sites

3.1. Introduction

The sandy beach monitoring sites are located within Scapa Flow, a large sheltered water body in the southern part of Orkney Islands, Figure 3.1. The sites have been separated into three groups, (1) Mainland and South Ronaldsay sites, (2) Bay of Ireland sites and (3) Hoy sites according to their geographical location and due to the years when the sampling was carried out (Figure 3.1, Table 3.1).

Figure 3.1. Sandy beach monitoring sites. Bay of Ireland sites are detailed in Figure 3.10. Source: OIC.

Table 3.1. The monitoring sites and years surveyed in Historical and Current time periods. For more detailed information refer to Chapter 2 Table 2.1.

3.2. Methods

All maps were produced using ArcGIS Desktop version 10.3.1.

Shore profile survey details are described in Chapter 2. Methods, Section 2.1.2.

Beach morphometrics were calculated as detailed in Appendix D. Note that Beach Type was classified at station level, recognising differences in average grain size between stations. Although it might seem paradoxical to consider differences in the type of a beach within sites, we have used 'Beach Type' as a synoptic measure of physical conditions at a particular location, intended to capture temporal and spatial variation at both small (within-site) and large (between-site) scales.

3.3. Mainland and South Ronaldsay sites

Scapa Bay, Swanbister Bay and Waulkmill Bay are located on the coast of Mainland Orkney, Widewall Bay is on the coast of South Ronaldsay (Figure 3.1). At Current time period, at these sandy beach sites samples were collected from two sampling stations (Table 3.2).

Site	Top of the transect and OS Grid reference	Sampling stations (ST) surveyed in 2002 – 2016 and OS Grid reference	
Scapa Bay	ST1 - HY 44305 08520	ST6 - HY 44290 08510	ST12 - HY 44271 08464
Swanbister Bay	ST0 - HY 35108 04709	ST7 - HY 35150 04708	ST12 - HY 35495 04678
Waulkmill Bay	ST0 - HY 37820 06577	ST ₁₀ - HY 37867 06498	ST12 - HY 37989 06213
Widewall Bay	ST ₀ - ND 43524 91629	ST8 - ND 43335 91766	ST12 - ND 43261 91848

Table 3.2. Mainland and South Ronaldsay sites, stations included in the monitoring programme and British Ordnance Survey (OS) grid references for the stations. Station 0 = Highest Astronomical Tide

3.3.1. Scapa Bay

Scapa Bay opens up to a south-westerly direction, the sampling stations were located on the eastern side of the bay (Figure 3.2). Scapa Bay has a small working pier, which mostly accommodates three tugs, a pilot boat and a couple of fishing vessels on a regular basis. Approximately once a month a coastal tanker delivers oil products, for example petrol, aviation fuel and low sulphur marine gas oil, to the pier. These products are used in Orkney and demand for the products dictates the frequency of the deliveries. The bay has a mooring for visiting yachts during summer months and often accommodates additional yachts that anchor within the bay. Two whisky distilleries are located nearby: Highland Park and Scapa. Historically Highland Park used to discharge organic effluent into the bay (Atkins & Jones 1990). Between 1974 and 1988 the effluent releases from Highland Park varied from approximately $5,000,000 - 25,000,000$ litres a year (Atkins & Jones 1990). During current monitoring period (2002-2016) Highland Park has not released any effluent to the Crantit Canal (SEPA, pers. comm.). Two small burns discharge into the bay, the Lingro Burn next to Scapa distillery and Crantit Canal middle of the bay. Crantit Dairy has been discharging in to Crantit Canal since 1993, both Scapa Distillery cooling waters and septic tank have been discharging into Lingro Burn since 2004 (SEPA, pers. comm) (Table 3.3). Within the bay, there are sub-tidal seagrass (*Zostera* sp.) and maerl beds both of which are mostly on the eastern area of the bay, south from the Scapa Pier (Orkney Harbour Authority pers. comm). The sandy beach at Scapa Bay is a popular location with dog walkers and day visitors.

Table 3.3. Details of effluent discharges into Crantit Canal and Lingro Burn (SEPA, pers. comm)

Company	SEPA Licence	Details of licence	Lingro Burn	Crantit Canal
Crantit Dairy	CAR/L/1001994	Licence to discharge	N/A	From 07/10/1993
		granted 07/10/1993		onwards
Scapa Distillery	CAR/L/1003120	Licence to discharge	From 20/10/2004	N/A
Cooling Waters		granted 20/10/2004	onwards	
Scapa Distillery	CAR/L/1003118	Licence to discharge	From $20/10/2004$	N/A
Septic tank		granted 20/10/2004	onwards	

Beach morphometric information for Scapa Bay is presented in Table 3.4. The Beach Type, as defined by Dean's parameter and RTR, remained Dissipative: non-barred at both stations since 1974 (Table 3.4), demonstrating that the grain size and beach physical characteristics have remained the same since 1974.

The Scapa Bay survey site has a steep shore profile with a slope of 2.24 (Table 3.4) and a relatively short distance of 65.3 metres from sampling ST1 (bottom of the seawall) to sampling ST12 (Figure 3.3).

Figure 3.2. Scapa Bay sampling stations. Source: Open Street Map, ArcGIS.

Figure 3.3. Scapa Bay shore profile, surveyed March 2016.

3.3.2. Swanbister Bay

Swanbister Bay opens up to an easterly direction (Figure 3.4). The bay is surrounded by the Swanbister farm which keeps cattle and sheep on the fields. There are also three burns, Burn of Fidge, Burn of Swanbister and Burn of Clummar, all of which discharge into the bay. In the south-eastern area of Swanbister Bay there is a ruined pier that was used historically by Swanbister farm, but the pier has not been in use during the monitoring period.

Figure 3.4. Swanbister Bay sampling stations. Source: Open Street Map, ArcGIS.

Beach morphometric information for Swanbister Bay is presented in Table 3.5. At ST7 the Beach Type, as defined by Dean's and RTR, changed from Ultra-dissipative to Dissipative: non-barred to Intermediate and back to Dissipative: non-barred. At ST12 the Beach Type, as defined by Dean's and RTR, varied over the years (Table 3.5) but most noticeably it was Dissipative: non-barred in 1979 and after several changes it returned to Dissipative: non-barred in 2015 and remained the same in 2016.

Swanbister Bay survey site has a steep upper shore with a long gently sloping lower shore with a slope of 0.66 (Table 3.5), the length of the shore from ST0 to ST12 was 390.3 metres (Figure 3.5).

Table 3.5. Swanbister Bay beach morphometric information.

SWANBISTER	

average at +10 Celsius

		Mean												
		grain	Water			Wave Sandfall Wave Wave					Tide Wave			
		size	temp.	Salinity height		velocity	freq	period Deans range height				Beach Type		
	Year	(μm)	$(^{\circ}C)$	(PPT)	(cm)	(cm/sec) (sec-1) (sec)			(Ω)	(m)		(m) RTR (as defined by Deans and RTR)	Slope BI BSI	
Swanbister 7	1986	228.74	8		34.6 109.25	2.38	0.26		3.88 11.83	4.2		1.09 3.84 Ultra-dissipative	0.662.85.6	
Swanbister 7	1987	235.14	8		34.6 109.25	2.72	0.26		3.88 10.37	4.2		1.09 3.84 Ultra-dissipative	0.662.8	5.3
Swanbister 7	1988	203.91	8		34.6 109.25	2.06	0.26		3.88 13.71	4.2		1.09 3.84 Ultra-dissipative	0.662.7	6.0
Swanbister 7	1989	239.48	8		34.6 109.25	2.72	0.26		3.88 10.37	4.2		1.09 3.84 Ultra-dissipative	0.662.8	5.3
Swanbister 7	1990	225.33	8		34.6 109.25	2.38	0.26		3.88 11.83	4.2		1.09 3.84 Ultra-dissipative	0.662.8	- 5.6
Swanbister 7	2006	244.70	8		34.6 109.25	2.72	0.26		3.88 10.37	4.2		1.09 3.84 Ultra-dissipative	0.662.85.3	
Swanbister 7	2014	344.70	8		34.6 109.25	4.44	0.26		3.88 6.35	4.2		1.09 3.84 Dissipative: non-barred	0.663.0	4.2
Swanbister 7	2015	442.40	8		34.6 109.25	6.17	0.26	3.88	4.57	4.2		1.09 3.84 Intermediate	0.663.1	3.5
Swanbister 7	2016	408.20	8		34.6 109.25	5.48	0.26		3.88 5.14	4.2		1.09 3.84 Dissipative: non-barred	0.66 3.1 3.7	
Swanbister 12 1979		343.03	8		34.6 109.25	4.44	0.26	3.88	6.35	4.2		1.09 3.84 Dissipative: non-barred	0.663.04.2	
Swanbister 12 1986		272.36	8		34.6 109.25	3.40	0.26	3.88	8.29	4.2		1.09 3.84 Ultra-dissipative	0.66 2.9	4.8
Swanbister 12 1987		508.34	8		34.6 109.25	7.19	0.26	3.88	3.92	4.2		1.09 3.84 Intermediate	0.663.1	3.1
Swanbister 12 1988		264.06	8		34.6 109.25	3.06	0.26	3.88	9.22	4.2		1.09 3.84 Ultra-dissipative	0.662.9	5.1
Swanbister 12 1989		274.15	8		34.6 109.25	3.40	0.26	3.88	8.29	4.2		1.09 3.84 Ultra-dissipative	0.66 2.9	-4.8
Swanbister 12 2006		257.20	8		34.6 109.25	3.06	0.26	3.88	9.22	4.2		1.09 3.84 Ultra-dissipative	0.66 2.9	5.1
Swanbister 12 2014		5566.54	8		34.6 109.25	51.98	0.26	3.88	0.54	4.2		1.09 3.84 Reflective: low tide terrace w/rip	0.66 4.2 -1.4	
Swanbister 12 2015		363.20	8		34.6 109.25	4.79	0.26	3.88	5.89	4.2		1.09 3.84 Dissipative: non-barred	0.663.0	4.0
Swanbister 12 2016		343.90	8		34.6 109.25	4.44	0.26	3.88	6.35	4.2		1.09 3.84 Dissipative: non-barred	0.663.04.2	
$-50 + 4.47m$ $-57, +1.34m$ $MHWS + 2.7m$														
								Beach profile					$-S12. -0.19m$ MLWS 0m	
틺 390.3m														
	BEACH PROFILE - SWANBISTER, HORIZONTAL SCALE 1:2000, VERTICAL SCALE 1:200 (VERTICAL EXAGGERATION X10) MHWS and MLWS refer to St Mary's, the nearest Scapa Flow port with tabulated tidal height data													

Figure 3.5. Swanbister Bay shore profile, surveyed March 2016.

3.3.3. Waulkmill Bay

Waulkmill Bay opens up to a south-easterly direction (Figure 3.6). The Waulkmill Site of Special Scientific Interest (SNH site code 1598) surrounds the sandy beach monitoring site. The designation is for an area of 66.51 hectares and includes a saltmarsh area at the top of the bay, maritime cliffs in the bay, and is for the presence of Golden-rod casebearer moth (*Coleophora obscenella*) in the area.

Figure 3.6. Waulkmill Bay sampling stations. Source: Open Street Map, ArcGIS.

Figure 3.7. Waulkmill Bay shore profile, surveyed March 2016.

At low tide, undulating sand waves create small pools of water across the shore. Mill Burn links Waulkmill Bay, the saltmarsh area and the Loch of Kirbister. The Loch of Kirbister is a popular area for trout fishing and the Orkney Trout Fishing Association carries out annual trout surveys on the Mill Burn. One of their trout hatcheries is located by the loch and is a Controlled Activities Regulations (CAR) licensed seawater finfish farm (SEPA 2016). The bay is popular with dog walkers and day visitors. It is unknown why the Waulkmill transect is diagonal across the beach.

Beach morphometric information for Waulkmill Bay are presented in Table 3.6. The Beach Type, as defined by Dean's and RTR, remained constantly as Dissipative: nonbarred at ST10 since 1974 and at ST12 since 1986 (Table 3.6), demonstrating the stability of the physical characteristic of the site over time.

Waulkmill Bay survey site has a long gentle profile with a slope of 0.45 (Table 3.6) and a shore length of 403.3 metres from ST0 to ST12 (Figure 3.7).

Table 3.6. Waulkmill Bay beach morphometric information.

3.3.4. Widewall Bay

WAULKMILL BAY average at +10 Celsius

Widewall Bay is a large L-shaped sheltered bay on the island of South Ronaldsay (Figure 3.1), the bay opens up in a south-westerly direction to the Sound of Hoxa. The sandy beach transect is in a north-westerly direction (Figure 3.8).

Figure 3.8. Widewall Bay sampling stations. Source: Open Street Map, ArcGIS.

The inner bay has a large seagrass *Zostera* sp. bed (Thomson et al. 2014) which begins below the lower (ST12) sandy beach sampling station. The bay also has several harbour seal (*Phoca vitulina*) haul-out and pupping sites (Thompson & Harwood 1990), one of them being on a rocky outcrop next to the sandy beach sampling stations. Agricultural land and sparse housing surrounds the bay. The Oback Burn and Oyce of Quindry both discharge into the eastern section of the bay.

Beach morphometric information for Widewall Bay is presented in Table 3.7. The Beach Type, as defined by Dean's and RTR, has remained Ultra-dissipative at both stations throughout the monitoring programme, with the exception on ST8 in 2015 when it was classified as Intermediate: bar and rip channels present (Table 3.7). The ultra-dissipative beach type in most years demonstrates the very sheltered nature of the beach and the stability of the physical parameters at the beach.

The Widewall Bay monitoring site has a steep upper shore and a gently sloping lower shore with a slope of 1.11 (Table 3.7) and a shore length of 347.7 metres from ST0 to ST12 (Figure 3.9).

		Mean													
		grain	Water		Wave	Sandfall	Wave	Wave			Tide Wave				
		size	temp.	Salinity	height	velocity	freq	period	Deans range height			Beach Type			
	Year	(μm)	$(^{\circ}C)$	(PPT)	(c _m)	(cm/sec)	$(sec-1)$	(sec)	(Ω)	(m)	(m)	RTR (as defined by Deans and RTR)	Slope BI BSI		
Widewall 8	1974	187.3	8	34.6	51.23	1.74	0.41	2.46	11.98	4.2	0.51	8.20 Ultra-dissipative	1.11 2.9 5.7		
Widewall 8	1980	280.2	8	34.6	51.23	3.40	0.41	2.46	6.12	4.2	0.51	8.20 Ultra-dissipative	1.11 3.1 4.1		
Widewall 8	1986	228.9	8	34.6	51.23	2.38	0.41	2.46	8.74	4.2	0.51	8.20 Ultra-dissipative	1.11 3.0 4.9		
Widewall 8	1987	205.3	8	34.6	51.23	2.06	0.41	2.46	10.13	4.2	0.51	8.20 Ultra-dissipative	1.11 3.0 5.3		
Widewall 8	1988	194.3	8	34.6	51.23	2.06	0.41	2.46	10.13	4.2	0.51	8.20 Ultra-dissipative	1.11	3.0 5.3	
Widewall 8	1989	196.7	8	34.6	51.23	2.06	0.41	2.46	10.13	4.2	0.51	8.20 Ultra-dissipative	1.11	$3.0 \t5.3$	
Widewall 8		1990 190.7	8	34.6	51.23	2.06	0.41	2.46	10.13	4.2	0.51	8.20 Ultra-dissipative	1.11 2.9 5.3		
Widewall 8	2006	201.9	8	34.6	51.23	2.06	0.41	2.46	10.13	4.2	0.51	8.20 Ultra-dissipative	1.11	$3.0 \t5.3$	
Widewall 8		2014 196.6	8	34.6	51.23	2.06	0.41	2.46	10.13	4.2	0.51	8.20 Ultra-dissipative	1.11 3.0 5.3		
Widewall 8		2015 272.9	8	34.6	51.23	5.13	0.41	2.46	4.06	4.2	0.51	8.20 Intermediate: bar & rip channels	1.11	$3.1 \quad 3.2$	
Widewall 8		2016 261.2	8	34.6	51.23	3.06	0.41	2.46	6.81	4.2	0.51	8.20 Ultra-dissipative	1.11 3.1 4.4		
Widewall 12		1986 209.4	8	34.6	51.23	2.06	0.41	2.46	10.13	4.2	0.51	8.20 Ultra-dissipative	1.11 3.0 5.3		
Widewall 12	1989	183.6	8	34.6	51.23	1.74	0.41	2.46	11.98	4.2	0.51	8.20 Ultra-dissipative	1.11 2.9 5.7		
Widewall 12		2006 212.2	8	34.6	51.23	2.38	0.41	2.46	8.74	4.2	0.51	8.20 Ultra-dissipative	1.11 3.0 4.9		
Widewall 12	2014	184.8	8	34.6	51.23	1.74	0.41	2.46	11.98	4.2	0.51	8.20 Ultra-dissipative	1.11	2.9 5.7	
Widewall 12	2015	250.7	8	34.6	51.23	3.06	0.41	2.46	6.81	4.2	0.51	8.20 Ultra-dissipative	1.11	3.1 4.4	
Widewall 12		2016 259.9	8	34.6	51.23	3.06	0.41	2.46	6.81	4.2	0.51	8.20 Ultra-dissipative	1.11 3.1 4.4		

Table 3.7. Widewall Bay beach morphometric information.

WIDEWALL BAY average at +10 Celsius

Figure 3.9. Widewall Bay shore profile, surveyed March 2016.

3.4. Bay of Ireland sites

The Bay of Ireland monitoring sites: Congesquoy, Dead Sand and Cumminess, are located within the Bay of Ireland, in the north west of Scapa Flow (Figure 3.1). All three sites are north of the Bu Point waste water treatment facility (Figure 3.10). At Current time period, at The Bay of Ireland monitoring sites samples were collected from two sampling stations (Table 3.8).

Figure 3.10. Bay of Ireland monitoring sites in relation to the, Brig O Waithe, Loch of Stenness and Bu Point waste water treatment facility. Source: Open Street Map, ArcGIS.

These intertidal monitoring sites were set up in 1984 prior to the new sewage outfall pipe being built at Bu Point, Bay of Ireland (Jones et al. 1990). The outfall system started discharging raw sewage into the Bay of Ireland in 1986 (ICIT 2004a) and continued to do so until 2006 when the Bu Point sewage treatment facility became operational (Scottish Water, pers. comm.). The Bu Point waste water treatment facility has a secondary treatment in place and discharges approximately 750 m^3 per day into the Bay of Ireland (Scottish Water, pers comm.).

3.4.1. Congesquoy

Congesquoy site is south-east facing (Figure 3.11). The bay is surrounded by agricultural land and one burn, the Burn of Congesquoy, which runs into the bay. A watercourse from Brig O Waithe and Loch of Stenness is located to the north-east from the transect. A carpark with an access to the beach is available. The sandy beach is frequently visited by members of the public during spring low tides for collecting razorfish (*Ensis* spp.).

Figure 3.11. Congesquoy sampling stations. Source: Open Street Map, ArcGIS.

Beach morphometric information for Congesquoy is presented in Table 3.9. The Beach Type, as defined by Dean's and RTR, has remained Ultra-dissipative at both stations since 1983 (Table 3.9), demonstrating the physical stability of this sheltered beach.

The Congesquoy monitoring site has a long gentle profile with a slope of 0.14 (Table 3.9) and a shore length of 173.8 metres from ST0 to ST2 (Figure 3.12).

Table 3.9. Congesquoy beach morphometric information.

CONGESQUOY

Figure 3.12. Congesquoy shore profile, surveyed March 2016.

3.4.2. Dead Sand

Dead Sand is an enclosed embayment with a narrow north-west facing entrance (Figure 3.13). An unnamed burn runs into the embayment, which in turns opens into The Bush and leads to north to Bridge of Waithe (also called the Brig O Waithe) (Figure 3.13). The Bridge of Waithe is a watercourse, which connects Loch of Stenness saline lagoon into Scapa Flow. Agricultural fields and marshy ground surround the bay. No road access is available to this site.

Beach morphometric information for Dead Sand are presented in Table 3.10. At ST1 the Beach Type, as defined by Dean's and RTR, has been Ultra-dissipative in 1986 and for the subsequent four years of surveys, and then changed to Intermediate: bar and rip channels present in 2015 onwards (Table 3.10). At ST2 the Beach Type has changed several times over the years (Table 3.10), but mainly in 1986 it was Intermediate: bar and rip channels present, and it returned to this same beach type in 2016. The change of Beach Type to Intermediate: bar and rip channels present indicates increase in the mean grain size at the beach. The ST1 is in very sheltered location in the middle of the bay, ST2 is in or next to a channel of water running away from the bay. The change at the ST2 is potentially due to the change in the location of the channel which would carry the finer sediment away making the sediment at this sampling station coarser. The cause for the change in the grain size at ST1 is less clear and would need further investigation.

Figure 3.13. Dead Sand sampling stations. Source: Open Street Map, ArcGIS.

Figure 3.14. Dead Sand shore profile, surveyed March 2016.

Table 3.10. Dead Sand beach morphometric information.

average at +10 Celsius Mean

Dead Sand monitoring site is a shallow intertidal embayment with a slope of 0.66 (Table 3.10) and a shore length of 189.7 metres from ST0 to ST2 (Figure 3.14).

3.4.3. Cumminess Bay

Cumminess Bay opens up in southerly direction to Bay of Ireland (Figure 3.15). The bay is surrounded by agricultural land but no burns run into it, some surface run off from the fields around is expected during heavy rains. No road access is available to the beach.

Figure 3.15. Cumminess sampling stations. Source: Open Street Map, ArcGIS.

Beach morphometric information for Cumminess Bay are presented in Table 3.11. The

throughout the monitoring period (Table 3.11), demonstrating the sheltered nature of the site and the stability of the physical parameters at the site over time.

Cumminess Bay monitoring site has a steep upper shore and a relatively steep lower shore with a slope of 0.57 (Table 3.11) and a shore length of 284.2 metres from ST0 to ST4 (Figure 3.16).

Figure 3.16. Cumminess Bay shore profile, surveyed March 2016.

3.5. Hoy sites

CUMMINESS BAY average at +10 Celsius

Bay of Creekland, Bay of Quoys, Lyrawa Bay, Mill Bay, Longhope Bay and Kirk Hope sandy beach monitoring sites are located on the island of Hoy (Figure 3.1). At the Hoy sites sampling has been carried out over varying frequency from 1974 until 1990, Current monitoring programme was started in 2006 (Table 3.1). In the Current time period, up to three stations were selected for monitoring purposes (Table 3.12).

Site	Top of the	Stations surveyed in 2006 - 2014							
	transect								
Bay of Creekland	S _{TO}	ST ₇	ST ₉	ST11					
	HY 23852 04061	HY 23875 04100	HY 23932 04142	HY 24035 04214					
Bay of Quoys	ST ₀	ST ₇	ST10	ST ₁₂					
	HY 24176 03091	HY 24189 03105	HY 24340 03151	HY 24523 03218					
Lyrawa Bay	ST ₀	ST ₈	ST10	N/A					
	ND 29271 98660	ND 29275 98664	ND 29475 98727						
Mill Bay	ST ₀	ST ₈	ST10	ST ₁₂					
	ND 30130 95082	ND 30151 95100	ND 30187 95126	ND 30310 95200					
Longhope Bay	ST ₀	ST ₈	ST10	ST ₁₂					
	ND 27378 89390	ND 27420 89420	ND 27449 89430	ND 27478 89485					
Kirk Hope	ST ₀	N/A	N/A	MLWS					
	ND 33390 89373			ND 33460 89400					

Table 3.12. Hoy sites with details of the stations included in the monitoring programme. Station $0 =$ Highest Astronomical Tide

3.5.1. Bay of Creekland

Bay of Creekland is located on the north-western part of Scapa Flow and is east facing (Figure 3.17). Several unnamed burns run into the bay from the surrounding agricultural land. Within the bay, there is an unused slipway and a cemetery. A passenger ferry terminal, Moaness Pier, is south of the bay. The Bay of Creekland is sheltered from longrange fetch by the island of Graemsay, which is located due northeast from the bay. Between the Bay of Creekland and island of Graemsay is a very narrow strip of water called Burra Sound, which experiences strong tidal currents. A road runs alongside the bay servicing couple of houses and the cemetery.

Figure 3.17. Bay of Creekland sampling stations. Source: Open Street Map, ArcGIS.

Beach morphometric information for Bay of Creekland are presented in Table 3.13. The Beach Type, as defined by Dean's and RTR, has been Intermediate: bar and rip channels at all three stations apart from 2006 in ST7 and ST11 and 1974 at ST9 when the Beach Type was Ultra-dissipative (Table 3.13). The beach is by the fast running Burra Sound, which is likely to contribute to the coarser sand recorded at this beach.

The bay has a steeply sloping upper shore and gently sloping lower shore with a slope of 0.93 (Table 3.13) with a shore length of 240.1 metres from ST0 to ST11 (Figure 3.18).

Table 3.13. Bay of Creekland beach morphometric information.

BAY OF CREEKLAND

average at +10 Celsius

Figure 3.18. Bay of Creekland shore profile, surveyed April 2016.

3.5.2. Bay of Quoys

Bay of Quoys is located on the north-western part of Scapa Flow (Figure 3.19). Whaness Burn, South Burn of Quoys and several unnamed burns run into the bay. A disused quarry on the South Burn of Quoys has an inactive freshwater finfish farm (SEPA 2016). The Whaness Burn has been enlarged by locals to enable them to take their small boats up and down the burn and to store the boats in a small 'homemade' inland anchorage. Agricultural land and a few houses, which have access to the beach, surround the bay;

otherwise, the bay is inaccessible. A passenger ferry terminal, at Moaness Pier, is located to the north of the bay.

Figure 3.19. Bay of Quoys sampling stations. Source: Open Street Map, ArcGIS.

BAY OF QUOYS

Beach morphometric information for Quoys Bay is presented in Table 3.14. The Beach Type, as defined by Dean's and RTR, were Intermediate at each station at the start of the monitoring period in 1986 and changed to Dissipative: non-barred at each station for several years only to change back to Intermediate in 2014 at ST7, 2015 at ST10 and in 2016 at ST12 (Table 3.14). The coarseness of the sediment at this sheltered site was attributed to the local geology by Atkins et al. (1985) and requires further investigation.

The Bay of Quoys has a steep upper shore but a more gently undulating lower shore with a slope of 0.48 (Table 3.14) with a shore length of 369.2 metres from ST0 to ST12 (Figure 3.20).

Figure 3.20. Bay of Quoys shore profile, surveyed April 2016.

3.5.3. Lyrawa Bay

Lyrawa Bay is on the western part of Scapa Flow with east facing bay (Figure 3.21). Two islands off the coast from Lyrawa Bay: Rysa Little and Cava, provide this sandy beach site some degree of shelter from westerly weather. The Lyrawa Burn runs into the Lyrawa Bay, at the top of the shore there is a large area of marshland, which is covered by seawater during spring tides. Within Lyrawa Bay, there is a CAR licensed salmon aquaculture site (SEPA 2016). There is a roadside parking place for cars and a footpath to the beach.

Beach morphometric information for Lyrawa Bay are presented in Table 3.15. The Beach Type, as defined by Dean's and RTR, was Ultra-dissipative when first granulometry samples were collected from ST7 at 1974 and ST10 at 1986 (Table 3.15) and changed to Intermediate: bar and rip channels at ST7 in 2006, 2015 and 2016 and at ST10 in 2015 and 2016. This change indicates the increase in the mean grain size and could be attributed to the change in the season when samples were collected; summer time in 1974- 1989 and winter time from 2006 onwards.

Lyrawa Bay is a gently sloping shore with a slope of 0.15 (Table 3.15) and a shore length of 213.9 meters from ST0 to ST10 (Figure 3.22).

Figure 3.21. Lyrawa Bay sampling stations. Source: Open Street Map, ArcGIS.

LYRAWA BAY

Figure 3.22. Lyrawa Bay shore profile, surveyed April 2016.

3.5.4. Mill Bay

Mill Bay is in the western area of Scapa Flow (Figure 3.23). The bay is sheltered by the island of Fara, which is due east from the site. The Mill Burn (in the north-west of the bay) and several unnamed burns run into the bay.

Figure 3.23. Mill Bay sampling stations. Source: Open Street Map, ArcGIS.

The Mill Burn has an active CAR licensed salmon hatchery, The Milburn Salmon Hatchery (SEPA 2016). The bay itself has an inactive mussel aquaculture site (SEPA 2016). There are several houses with shore access and the shoreline is accessible by an unpaved road. The bay is surrounded by moorland and agricultural land.

Beach morphometric information for Mill Bay are presented in Table 3.16. The Beach Type, as defined by Dean's and RTR, has varied at each station over the years: at ST8 the beach has been defined as Intermediate: bar and rip channels (1974, 1986-1989 and 2006) and Reflective: low tide terrace without rip channels (2014-2016), at ST10 the beach has been defined as Ultra-dissipative (1974, 1979, 2006 and 2014) and Intermediate: bar and rip channels (1986-1989, 2015 and 2016), at ST12 the beach has been defined as Ultradissipative (1974 until 2006) and Intermediate: bar and rip channels (2016) (Table 3.16). The gradation of the sediment grain sizes is clear at Mill Bay with the coarsest mean grain size at the top of the shores (ST8) and the finest grain sizes at the lower shore station (ST12).

The Mill Bay sandy shore transect has a steep upper shore and a steadily declining lower section with a slope of 0.79 (Table 3.16), the length of the shore is 215.2 metres from ST0 to ST12 (Figure 3.24).

Table 3.16. Mill Bay beach morphometric information.

MILL BAY
average at $+10$ Celsius

Figure 3.24. Mill Bay shore profile, surveyed April 2016.

3.5.5. Longhope Bay

Longhope Bay is in the southwestern area of Scapa Flow (Figure 3.25). It is in the western area of a large enclosed and sheltered embayment, the North Bay. Numerous unnamed burns run into the bay. The bay is mainly surrounded by agricultural land and some moorland. There are a several patches of *Zostera* sp. within the bay (Thomson et al. 2014) all of which are below the bottom station (ST12). A number of houses are along the coastline with access to the beach. General access to the shoreline is difficult especially where the monitoring site is.

Figure 3.25. Longhope sampling stations. Source: Open Street Map, ArcGIS.

Beach morphometric information for Longhope Bay are presented in Table 3.17. The Beach Type, as defined by Dean's and RTR, has varied at ST8 and ST10 but remained constant at ST12 (Table 3.17). At ST8 the beach has been defined as Intermediate (1974, 1986, 1988 and 1989), as Reflective: low tide terrace with rip (1987) and as Dissipative: non-barred (2006, 2014 – 2016), at ST10 the beach has been defined as Dissipative: nonbarred (1974, 1989, 2006, 2014 – 2016) and Intermediate (1986-1988), at ST12 the beach type has remained Dissipative: non-barred (1986-1989, 2006). Longhope beach shows the same progression of mean grain sizes as Mill Bay, the coarsest sediment is at the top of the shore station (ST8) with finer mean grain sizes at the low shore station (ST12).

The Longhope Bay sandy shore site has a steep upper shore with a gently sloping lower shore with a slope of 1.10 (Table 3.17), the length of the shore from ST0 to ST12 is 138.2 metres (Figure 3.26).

Table 3.17. Longhope Bay beach morphometric information.

LONGHOPE BAY

3.5.6. Kirk Hope Bay

Kirk Hope Bay is in the southern area of Scapa Flow (Figure 3.27). At Kirk Hope only one sampling station has been included in the monitoring programme, MLWS, the station was named after its location on the beach which was at the Mean Low Water Spring level (MLWS). The bay opens up to the northeast and it receives a small amount of shelter from the island of Switha, which lies due northeast from the site. One unnamed burn runs into the bay from the surrounding agricultural land. Three houses are located on the coastline and have easy access to the beach from the road, which runs close to the western end of the bay. The Kirk Hope cemetery and a RNLI memorial statue is on the northwest side of the bay.

Figure 3.27. Kirk Hope sampling station. Source: Open Street Map, ArcGIS.

Beach morphometric information for Kirk Hope Bay are presented in Table 3.18. The Beach Type, as defined by Deans and RTR, has varied between Dissipative: non-barred (1986, 1987, 1990 and 2014) and Intermediate (1988, 1989, 2006, 2015 and 2016) (Table 3.18). The reason for the changes in the Beach Type are not clear and would need further investigation.

The Kirk Hope site has a very steep shoreline with a slope of 1.10 (Table 3.18) and 74.5 metres from the ST0 to Sampling station MLWS (Figure 3.28).

Table 3.18. Kirk Hope Bay beach morphometric information.

KIRK HOPE BAY
average at $+10$ Celsius

Figure 3.28. Kirk Hope shore profile, surveyed April 2016.

The potential impacts for each sandy beach are brought together in Table 3.19 for easy comparison of sites.

Site	Possible sources of effluent	Features within the site	Physical features	
Scapa Bay	Whisky distillery effluent $1974 - 1988$ approx. $5,000,000 -$ 25,000,000 /year	A working pier Mooring for visiting yachts Popular with dog walkers and day visitors Road alongside the beach Easy access to the	South-westerly facing Two burns run into the bay Sediment: sand at both stations	
		beach Zostera and maerl bed		
Swanbister Bay	Surrounded by a farm, possible source of diffuse pollution	Derelict pier Road alongside the heach Easy access to the beach	East facing Three burns run into the bay Sediment: sand at both stations	
Waulkmill Bay	Loch of Kirbister has a finfish farm and is connected to Waulkmill Bay via Mill Burn	Surrounded by moorland the Waulkmill Site of Special Scientific Interest Access to beach via two footpath, parking provided for dog	South-east facing Sediment: ST10 sand, ST12 slightly gravelly sand	

Table 3.19. Summary of potential impacts at each Scapa Flow sandy beach site.

3.6. Sampling stations at the Orkney sandy beach survey sites

The macroinvertebrate species and abundances are influenced by tidal level (Dexter 1984; Rakocinski et al. 1993), if comparing populations from different tidal levels it is unlikely that like with like are being compared. The sampling in the Current time period included several sampling stations as listed in the Table 3.20.a. The tidal heights of all the Current time sampling stations were recorded during the shore profile surveys carried out in March and April 2016 (Figures 3.3, 3.5, 3.7, 3.9, 3.12, 3.14, 3.15, 3.18, 3.20, 3.22, 3.24, 3.26 and 3.28). To investigate whether the sampling stations at different sites were at same tidal height in 2016 the stations with their tidal heights were tabulated to allow comparisons (Table 3.20.b).

Using the station numbers as a guideline, the data from ST2 of each of the Bay of Ireland sites (Congesquoy, Cumminess and Dead Sand) could be analysed together (Table 3.20.a). In comparison, the Mainland and South Ronaldsay and Hoy sites could be analysed together, if data from ST11 and ST12 were used (Table 3.20.a).

Using the information from the 2016 shore profiles, the grouping of the sites for data analysis purposes would be different (Table 3.20.b). The Congesquoy and Cumminess lower stations, ST2 and ST4 respectively, were at the same tidal height as ST10-ST12 were at other sampling sites. Several sampling stations were over a meter above the MLWS: Dead Sand ST1 and ST2, Quoys ST7, Scapa ST6, Swanbister ST7 and Waulkmill ST10. In 1974 the sampling stations were established using stations at fixed 30 cm vertical intervals (Jones 1980). At three of the sites (Congesquoy, Creekland and Quoys) sampling stations were less than 30 cm vertical height difference from each other. At Congesquoy the two sampling stations (ST1 and ST2) were 20 cm vertical height difference (Table 3.20.b). At Creekland the ST7 and ST9 and at Longhope the ST8 and ST10, should have had 60 cm vertical height differences (as they were two stations apart, each station fixed at 30 cm vertical height difference) but instead they were at 19 cm vertical height difference at Creekland and at 10 cm vertical height difference at Longhope.

The shore profiles and the tidal heights of the stations measured in 2016 should be used as guidelines. Due to the mobile character of sediment at sandy beaches, the profiles of the sites change from season to season and from year to year. This information on the tidal heights of the sampling stations is vital in understanding the site-specific macroinvertebrate data analysis as well as highlighting which sampling stations from different sampling sites were approximately at the same tidal height.

57

Table 3.20. Sampling stations. On the left (A) sampling stations are grouped according to their station numbers, on the right (B) according to their height on the shoreline (surveyed in 2016). Stations which were less than 30cm vertical difference are highlighted in grey.

Chapter 4 Assessing the impact of variability and inconsistency in macroinvertebrate sample identification and enumeration on data analysis

4.1. Introduction

Sandy beach monitoring programmes have several stages during which discrepancies can be unintentionally introduced that affect the data: (1) during sample collection, (2) sample washing and sieving, (3) sample sorting, (4) sample identification and enumeration, (5) data entry and (6) change in personnel and management (Ellis 1988; Ranasinghe et al. 2003; Haase et al. 2006; Jones et al. 2007; Schlacher et al. 2008; Haase et al. 2010; Worsfold & Hall 2010).

The standardisation of sample collection should be considered during the planning of the monitoring programme and implemented by using standard protocols and methods (Holme & McIntyre 1971; McLachlan & Defeo 2018). The main issues with sample collection are the repeatability of relocating sampling locations and the season when samples are collected (Atkins et al. 1989). If the samples in a monitoring programme are not collected from the same location the data used to analyse and draw conclusions from can become meaningless (Ellis 1988) as any changes in the macroinvertebrate community could be due to the change in the sampling location rather than due to changes in the environmental conditions at the shoreline (Brazeiro & Defeo 1996). Macroinvertebrate population abundances fluctuate throughout the year, often reaching a peak in adult populations at the end of the summer with recruitment at the end of the winter or early spring (Leber 1982; Atkins et al. 1989; Baron & Clavier 1994; Brazeiro & Defeo 1996). The recruitment events can vary in scale and timing from year to year, sampling at the same time of the year minimises the effect of seasonal cycles of abundance, so that measured abundance reflects the inter-annual variability (Essink & Beukema 1986).

After sample collection the opportunities for inconsistencies arise after the sample fixing, when the samples are washed and sieved in freshwater to remove any formalin residues (Eleftheriou & Robertson 1988; Kröncke & Reiss 2010; Worsfold & Hall 2010). Depending on how much sediment and how many specimens are present in the sample, the washing time and any loss or damage to the specimens can vary dramatically (Worsfold & Hall 2010).

After the washing and sieving stage further opportunities for inconsistencies arise from the hand sorting of the samples. Hand sorting is carried out by placing the rinsed sample into a sorting tray; in some laboratories, including at the Marine Environmental Unit laboratory, samples are stained using Bengal red stain to aid the hand sorting (Worsfold & Hall 2010). The stain is added to the fixative and stains all living cells bright red or pink which makes the macroinvertebrates stand out during sorting against the white background of the sorting tray. During the sorting all the red/pink specimens are removed and placed into labelled glass bottles for identification at a later stage. Undercounts of specimens can result from incomplete sorting when all the organisms from the tray have not been removed for identification (Ranasinghe et al. 2003; Worsfold & Hall 2010).

Inconsistencies in identification and enumeration can be divided into three types: firstly, misidentification of organisms; secondly as a 'true' enumeration error where the analyst has miscounted the specimens; and thirdly, an enumeration error due to poor laboratory practice. An example of a poor laboratory practice is when during the sorting process polychaetes are damaged and consequently fragmented; counting of both anterior and posterior ends would lead to inflated abundances (Stribling et al. 2003). Generally, it is agreed that in case of fragmented specimens only heads are counted (Stribling et al. 2003; Worsfold & Hall 2010). To evaluate possible sources of errors in biological data Stribling et al. (2003) outlined performance characteristics which enable the quality of taxonomic data to be determined. The percentage difference in enumeration (PDE) and percentage difference in taxonomic disagreement (PTD) are calculated using data from samples which have been analysed (identified and enumerated) by two different analysts (Section 4.3.2. below). The PDE and PTD enable the highlighting of any enumeration or taxonomic issues in samples and therefore provide a tool for biological monitoring programmes to investigate the accuracy of their data.

Barchard $\&$ Pace (2011) demonstrated how data entry by a single person followed by data check by the same person resulted in significant data entry errors and incorrect statistical analysis. Quality control procedures for entering and checking data entry are vital and should be supplemented by double checking of the entered data by a second person or a computer programme (Stribling et al. 2008; Barchard & Pace 2011).

During hand sorting and identification processes there will always be variability between different personnel. Ranasinghe et al. (2003) concluded that even if using specialist analysts for identification and enumeration of difficult taxa, a level of inconsistency will be introduced if all the samples are not identified by the same analyst and this could still introduce an error if they are misidentifying. The inter-operator variability during the identification of specimens mainly depends on the analyst's familiarity with the taxa involved and their experience. In large taxonomic laboratories the inter-operator variability is mitigated, and analysts' skills are standardised using external Quality Assurance (QA) assessments (Jones et al. 2007; Milner & Hall 2016; Worsfold & Hall 2017a, 2017b).

Retrospectively nothing can be done regarding the first three stages described: the sample collection, washing, sieving and sorting of the samples. However, as all macroinvertebrate samples collected as part of the OICHA sandy beach monitoring programme from 2002 onwards have been preserved and stored at the Harbour Authority building, the remaining stages: identification, enumeration and data entry could be retrospectively investigated. As there has been a long-standing and growing need for a critical examination of the quality of taxonomic data recorded from OICHA's sandy beach monitoring programme the decision was made to re-identify, re-enumerate and reenter data from selected sites to investigate any inconsistencies in the data. In the OICHA sandy beach monitoring programme there are 13 sites, each with either two or three stations and each station with five replicate core samples, making it over 2000 core samples with possible discrepancies in identification, enumeration and data entry to contend with. Given the resources available and due to the time constraints of this doctoral research a decision was made to verify the identification and enumeration for three sites, re-enter the data and analyse the data for these three sites using both the Original and the Verified records. "Original" data is here taken to mean the data from the samples which were identified, enumerated and entered to the datasheets during the on-going monitoring programme in 2006-2013 by the personnel at that time. "Verified" data is taken to mean the 2006-2013 samples that were taken from the storage and which were re-identified, re-enumerated and re-entered in 2016/17 by the in-house analysts to enable this analysis.

In this chapter, analyses of the Original and Verified data for Quoys ST7, ST10 and ST12, Congesquoy ST1 and ST2, and Waulkmill ST10 and ST12 are compared to assess if they indicate similar trends in their macroinvertebrate communities over the years. These results will then be used to determine limits of the implications that can be made using the Original data; these decisions will be extrapolated to the other ten sites. The results of the data analyses are compared with each other as the data analyses are examined for the trends and variability characterised by the outcomes of the Original versus Verified data analysis. It will not be possible to ascribe inconsistencies to particular elements of the sampling process, it will be the identification, enumeration and data entry that will be compared between the Original and Verified data.

4.2. Aims

The aims of this chapter are to examine the extent to which inconsistencies and errors in identification and enumeration affect patterns of variation in sandy beach communities between sites and years. This process will inform how the data from these and the remaining sites will be treated and will explain the levels at which patterns of variation can be confidently interpreted.

4.3. Methods

In the Current monitoring programme (2002 onwards) several people have worked on the identification, enumeration and data entry of the samples (Chapter 2 Table 2.4), which has resulted in application of different levels of in-house taxonomic expertise. During the identification process in 2007 – 2010, several macroinvertebrate taxa (including but not exhaustively: Oligochaeta, Capitellidae, Nemertea, Paraonidae) were not confidently identified by the in-house analysts. One of the analysts attended a NE Atlantic Marine Biological Analytical Quality Control Scheme (NMBAQC) Scheme Benthic Invertebrate Taxonomic Workshop (http://www.nmbaqcs.org/) in 2010 during which it became evident that several identification errors and inconsistencies had been made during the identification process prior to 2010. After the NMBAQC workshop all samples from 2002 - 2010 for one site (Scapa) were re-identified by the OICHA in-house analysts to further investigate if there were any data inconsistencies, and if yes, to determine its level.

For quality control of the in-house identification and enumeration and to collate an independently created and verified voucher specimen collection for the macroinvertebrate fauna, all 2014 sandy beach samples from the 13 sites were sent to a taxonomic laboratory, APEM Ltd. The voucher specimen collection was further developed in-house by making an identification guide by photographing each specimen with care taken to highlight any features important for identification and by including relevant identification guides and references for each taxon. This has resulted in a comprehensive identification guide with corresponding voucher specimen collection.

With the existence of the verified voucher specimen collection, identification guide and preliminary understanding of the inconsistencies in the data, in 2017 a decision was made to further clarify the extent of the inconsistencies and errors by verifying all samples from three sites for the years 2006-2013. One site was selected from each group of sites (Chapter 3 Table 3.1): Waulkmill Bay from Mainland and South Ronaldsay; Congesquoy from Bay of Ireland; and Quoys from Hoy. The verification of the samples for the three sites was carried out by the in-house analysts with the use of the voucher specimens and
the in-house identification guide. All specimens were identified to species level where possible. Two analysts carried out the identification simultaneously but on different samples, no formal in-house random checks were conducted but any queries or difficult taxa were discussed between the analysts, as and when required, during the verification process. The identification was not carried out 'blind', photocopies of the original identification sheets were available to the analysts to refer to at all times. This process is still liable to inconsistencies and errors, but these will have been minimised as the reidentification was carried out by three different analysts (two working on the samples simultaneously) with the use of the in-house voucher specimens and identification guide compared with the original process which over the eight years (2006-2013) had eight different people (Chapter 2 Table 2.4) carrying out the identification with no voucher specimens or in-house identification guide.

Verification of the samples was carried out to species level where possible. The identification of the samples from the three sites to species level will give detailed information regarding the macroinvertebrate community of each site. This allows the examination of the effects of different taxonomic aggregation, as well as any other issues with the Original data.

The identification of the OICHA sandy beach macroinvertebrate samples has always been to the lowest taxonomic level possible but due to the different levels of expertise of personnel over the years this has varied from species level identification to class for some taxa. Due to this, and with the additional issue of misidentification of some specimens it was decided to aggregate all data to family level or higher (e.g. order or class when appropriate) for the data analysis. Aggregation to genus or family level has been applied in other studies with similar issues (e.g. Frid et al. 2009; Blanchard et al. 2010) with no loss of information on relevant ecological trends.

4.3.1 Data sets

For the data analysis, two sets of data are used from each sampling station, Original and Verified. Only data from 2006-2013 are used for the analysis due to sandy beach monitoring at Quoys re-starting in 2006 and as 2013 is the last year during which identification of the Original macroinvertebrate data was carried out without the in-house identification guide and voucher specimen collection. In 2014 all samples from all the sites were identified by a taxonomic laboratory which provided a voucher specimen collection for the macroinvertebrate monitoring programme and which was used in the development of the in-house identification guide (Chapter 4 Section 4.3).

The following changes were made to the both Original and Verified datasets: juveniles or any larval phases were removed if they were identified in the datasheets as such, meiofauna (would normally not be retained by 0.5mm sieve) were removed and any taxa which are not normally part of sandy beach communities were removed (Table 4.1).

Taxon	Reason for removal from data sets			
ANNELIDA				
Serpulidae Rafinesque,	Does not include intertidal sandy beach species (Hayward & Ryland 2017). One species within this family, the Ditrupa arietina (O.F. Müller, 1776)			
1815	lives unattached to substrata (mud or sand) but it is only present sub-tidally			
	(Hayward & Ryland 2017).			
Spirorbinae	Does not include intertidal sandy beach species (Hayward & Ryland 2017).			
Chamberlin, 1919				
ARTHROPODA				
Arachnida	Does not include intertidal sandy beach species (Hayward & Ryland 2017)			
Diptera	Does not include intertidal sandy beach species (Hayward & Ryland 2017)			
CHORDATA				
Ascidiacea	Specimens all juvenile, not able to determine species.			
Pleuronectidae	Tidal migrants, temporarily rather than permanently present in the intertidal			
Rafinesque, 1815	area (Hayward & Ryland 2017)			
CNIDARIA				
Actiniidae Rafinesque,	Does not include intertidal sandy beach species (Hayward & Ryland 2017)			
CRUSTACEA				
Acanthonotozomatidae	Does not include intertidal sandy beach species (Hayward & Ryland 2017)			
Stebbing, 1906				
Barnacle Nauplius	Larval phase and not sandy beach species (Young et al. 2002)			
Risso, 1844				
Cirripedia	Larval phase and not sandy beach species (Hayward & Ryland 2017)			
Copepoda	Does not include intertidal sandy beach species (Hayward & Ryland 2017)			
Ostracoda	Does not include intertidal sandy beach species (Hayward & Ryland 2017)			
ECHINODERMATA				
Holothuroidea	Does not include intertidal sandy beach species (Hayward & Ryland 2017)			
Ophiuroidae	Does not include intertidal sandy beach species (Hayward & Ryland 2017)			
MOLLUSCA				
Littorinidae	Does not include intertidal sandy beach species (Hayward & Ryland 2017)			
Children, 1834				
Mytilidae	Does not include intertidal sandy beach species (Hayward & Ryland 2017)			
Rafinesque, 1815				
Skeneopsidae	Does not include intertidal sandy beach species (Hayward & Ryland 2017)			
Iredale, 1915				
NEMATODA	Meiofauna (Gheskiere et al. 2005)			

Table 4.1. Taxa removed from data sets.

Details of the methods used in the statistical data analysis using PRIMER software are detailed in Chapter 2 Section 2.2.5.1.

4.3.2 Performance characteristics

The taxonomic precision or how accurately the analyst has identified all the specimens in a sample can be measured by the percentage taxonomic disagreement (PTD) and the accuracy when counting the specimens is measured by the percentage difference in enumeration (PDE). The PTD and PDE are calculated by comparing the two taxa lists created by the two different analysts when independently identifying the same sample (Stribling et al. 2003, 2008). The number of specimens identified and allocated to the same taxon by both analysts is taken as an agreement. If the specimens are identified to species level but only family level information is required, it is also taken as an agreement as both identifications are correct but at different taxonomic levels.

The PTD and PDE were calculated following Stribling et al. (2003). Both measurements were calculated for all stations, comparing the Original and Verified data and using the following formulae:

$$
PDE = \frac{|n1 - n2|}{n1 + n2} \times 100
$$

Where, *n*1 is the number of specimens counted by the first analyst, and *n*2 the number of specimens counted by the second analysist (Stribling et al. 2003).

$$
PTD = \left[1 - \frac{No. \text{ of agreements}}{N}\right] \times 100
$$

Where, N is the total number of specimens in the larger of the two counts (Stribling et al. 2003).

4.4. Results

4.4.1 Types of inconsistency and error in the data

Once the verification process was completed for the three sites the full scale of the inconsistencies was highlighted (Table 4.2). The verification process highlighted six families which had been identified incorrectly, two taxa at order level were identified to species level, and three taxa (Oligochaeta, Capitellidae and Orbiniidae) when verified were split into two or more taxa. Other inconsistencies found were the recording of juveniles and adults together and double counting of fragmented specimens (Table 4.2).

During the identification process any juvenile or larval phases should ideally be assigned an appropriate qualifier to enable the distinction between juvenile and adults during data analysis. For many species of macroinvertebrates, it is not possible to correctly identify juveniles to species or even family level as the juvenile stages do not exhibit the characteristics of the mature adult phases. For this reason, when identifying macroinvertebrates and entering the data into databases a note should be made if the specimens are juveniles. Presence of juveniles in the samples is an indication of recruitment event success and an important indicator of ecosystem status but is influenced by timing of sample collection and other external environmental factors (Giangrande et al. 1994; Hadfield & Strathman 1996). In this study juveniles are excluded from the analyses, mainly to remove potential large fluctuations in the abundances of taxa present due to changes in the timing of the sampling. The juveniles are recorded during identification but were removed from the data set prior to analysis, it is therefore possible to include them in future data or community analysis. In other studies juveniles have been included to study the timing of macroinvertebrate recruitment events (e.g. Atkins et al. 1989).

In the Original data from Congesquoy, Quoys and Waulkmill, for some of the polychaete specimens both the anterior and posterior ends were identified and counted, which led to inflated abundances. For example, in 2007 Quoys Core 1, in the Original data 62 Capitellidae / Oligochaeta were identified. The verification process analysts highlighted that 'many bits of Oligochaeta in the sample are missing heads' and enumerated a total of 47 Oligochaete and no Capitellidae, a difference of 15 specimens from Original to Verified data. In most samples the discrepancy was not this large but a difference of one or two was often recorded during the verification process. For some of the bivalve and gastropod specimens the empty shells were accounted for in the Original data in addition to the shells with soft tissue inside. In the Verified data only shells with soft tissue were accounted for.

These mistakes could be due to poor standards of practice as well as to the absence of standard protocols and methods. This could particularly be the case when personnel working on the samples have different levels of expertise and when change in personnel has occurred with no training or handover period. Errors in data entry were also noted; in some cases in the data spreadsheets, values had been entered incorrectly. At Quoys ST7 Original data spreadsheet two taxa with superficially similar names were confused with each other at point of data entry, Psammodrilidae and Pyramidellidae. In 2012 nine Psammodrilidae were entered for Quoys ST10 (Table 4.6); the analyst entering the data had made a mistake as no Psammodrilidae were identified in the samples, instead Pyramidellidae were identified in replicates three and four.

Other variations in the identification process were due to consistent misidentification. An example of this is the consistent misidentification of the small gastropod *Ebala nitidissima* (family Murchisonellidae). This gastropod is very small, up to 2.5mm in length, with a spiral conical shell (JMS 1986; Brenzinger et al. 2014) (Figure 4.1.a). In 2006-2013 specimens of *Ebala nitidissima* were only identified to family level and incorrectly as belonging to family Pyramidellidae. Several species within family Pyramidellidae have a spiral conical shell, one of which (*Pyrgiscus fulvocinctus*) is shown in Figure 4.1.b. The shells are not similar in the range of lengths, *E. nitidissima* being up to 2.5 mm and *P. fulvocinctus* (as an example of family Pyramidellidae) up to 8 mm. To an untrained eye this confusion between the two small conical shells could be easily made. In 2006-2013 most identification within the OICHA laboratory were made using Hayward & Ryland (1995) Handbook of the Marine Fauna of North-West Europe which includes identification keys for family Pyramidellidae but not for family Murchisonellidae; this could have compounded the misidentification.

Figure 4.1. a) *Ebala nitidissima,* family Murchisonellidae (length up to 2.5mm), b) *Pyrgiscus fulvocinctus*, family Pyramidellidae (length up to 8 mm).

Credits: a[\) http://species](http://species-identification.org/species.php?species_group=mollusca&menuentry=soorten&id=665&tab=classificatie)[identification.org/species.php?species_group=mollusca&menuentry=soorten&id=665&tab=classificatie](http://species-identification.org/species.php?species_group=mollusca&menuentry=soorten&id=665&tab=classificatie) b) Hayward and Ryland 2017.

There has also been confusion in the identification of specimens belonging to the subclass Oligochaeta and family Capitellidae which is a family of polychaetes. In some samples Oligochaeta were identified as Capitellidae and vice versa. In other samples Capitellidae were marked as unknown or misidentified as belonging to the family Syllidae. After the re-identification process it was concluded that within the subclass Oligochaeta two families are present: Enchytraeidae and Naididae represented by four species *Baltidrilus costatus*, *Paranais litoralis*, *Tubificoides benedii* and *T. pseudogaster*(Table 4.2). Within the family Capitellidae three taxa are present: *Capitella* sp., *Notomastus* sp. and *Mediomastus fragilis* (Table 4.2).

4.4.2 Performance characteristics

The percentage difference in enumeration (PDE) varied from $0.1 - 2.3$ % (Table 4.3). At most sites the enumeration error was less than 0.8% with Quoys ST7 having the highest error of 2.3%. The percentage of taxonomic disagreement (PTD) was more variable between the sites with highest disagreement at Congesquoy ST2 (16.1%) and lowest at Quoys ST12 (1.2%) (Table 4.3). The higher percentage of taxonomic disagreement indicates that the taxonomic disagreements are more likely to influence the data compared to enumeration error.

Site and station	PDE	PTD
Quoys ST7	2.3	13.9
Quoys ST10	0.1	5.1
Quoys ST12	0.1	1.2
Congesquoy ST1	0.8	8.7
Congesquoy ST2	0.4	16.1
Waulkmill ST10	0.6	9.2
Waulkmill ST12	04	15.1

Table 4.3. Percentage difference in enumeration (PDE) and percentage of taxonomic disagreement (PTD) for each station, comparing Verified and Original data.

4.4.3 Quoys Station 7 (ST7)

Most identification errors in Quoys ST7 were with Capitellidae and Oligochaeta (Table 4.4). In the Original data low abundance of Oligochaeta and high abundance of Capitellidae were recorded in 2006 - 2009, but when verified the abundances for both taxa were very similar or Oligochaeta (Enchytraeidae and Naididae) was more abundant. Identification errors of Nemertea and Platyhelminthes were most evident in 2012 and 2013; in the Original data only Nemertea are recorded but when these were verified most of them were Platyhelminthes and only a small number were confirmed as Nemertea (Table 4.4).

Two direct taxonomic name changes were highlighted: Hydrobiidae / Rissoidae when verified were Hydrobiidae, and Pyramidellidae which when correctly identified were Murchisonellidae.

No discrepancy was found in the identification or counts of the majority of the following taxa: Cirratulidae, Nereidae, Opheliidae, Corophiidae, Pontoporeiidae, Cardiidae, Retusidae and Tellinidae. All other taxa have differences between Original and Verified data.

4.4.3.1 Quoys ST7 Original vs Verified data analysis with replicates

When performing a cluster dendrogram with SIMPROF test (both tests explained in Chapter 2 Section 2.2.5) on the Original and Verified data while using replicates, SIMPROF test creates the first significant division of the data at 50% similarity for Original and 56% similarity for Verified data as demonstrated in the Multi-Dimensional Scaling (MDS) ordination by the significant clusters identified by SIMPROF test during cluster analysis (Figure 4.2). For Original data the two main clusters were 2006-2009; and 2010-2013 with one replicate from 2008; for Verified data the two clusters were 2006-2010; and 2011-2013. The two analyses are similar in their general pattern, even if they are not identical. A SIMPER test (Chapter 2 Section 2.2.5) explored these divisions further and showed that the first division in the Original data is driven by a shift in the

composition of the main taxa: in 2006-2009 the three main taxa present in the samples were marine polychaetes of the families Spionidae and Capitellidae and amphipods of the family Pontoporeiidae (Table 4.5), whereas in 2010 the three main taxa present were polychaetes of the families Spionidae and Capitellidae and marine worms of the subclass Oligochaeta (Table 4.5). The change in the taxa indicates a shift from a polychaete and amphipod dominated community to polychaete and oligochaete dominated community. However, the change from polychaete to oligochaete dominated community is due to misidentification of Oligochaeta as Capitellidae in 2006-2009 samples. The division of the two clusters in the Verified data (2006-2010 and 2011-2013) is driven by the change in macrofauna community from one dominated by polychaetes of the families Spionidae and Capitellidae, and oligochaetes of the family Naididae to one dominated by oligochaetes of the family Enchytraeidae, polychaetes of the family Capitellidae and flatworms of the phylum Platyhelminthes (Table 4.5). Given the confusion between the identification of Capitellidae and Oligochaeta this division is consistent with that described for the Original data. The cluster dendrogram data for both Original and Verified data show several further significant divisions; these were not analysed further as the analysis with the replicates summed only presents one significant division (Figure 4.3). Both sets of data indicate a shift in community composition occurring around 2010/2011. The shifts were due to different taxa in the two data sets due to identification errors in the Original data. Major shifts in macroinvertebrate community composition were detectable in Verified data, but the lack of verification means that the changes cannot reliably be characterised in terms of changes in particular taxa.

4.4.3.2 Quoys ST7 Original vs Verified data analysis, replicates summed

When performing the same tests with Quoys ST7 Original and Verified data sets, but summing the replicates for each year, the cluster dendrogram with SIMPROF test has only one significant division of the data at 57% similarity for Original and at 60% similarity for Verified data (Figure 4.3). For both Original and Verified data the SIMPROF test divides the data into two clusters, 2006-2010 and 2011-2013, indicating the same timing of the community shift for both Original and Verified data when replicates are summed. The MDS ordination (Figure 4.3) presents the two clusters for both data with low 2D stress of 0.04 further confirming the shift.

Table 4.4. Quoys ST7 summed abundances recorded for Original and Verified data. Grey highlight indicates taxa which have the most differences for Original and Verified data. Lines denote key changes in personnel: 2007 Biologist changed, 2010 and 2013 Technician changed.

Table 4.5. Quoys ST7 with replicates. Contributions of representative taxa to each year based on PRIMER analysis (cut-off at 60%). Dashed line represents significant SIMPROF separation.

Contrib%: Percentage contribution of taxa to the year group; Cum.%: Cumulative percentage contribution of taxa to the year group; Av. Similarity: The average Bray-Curtis similarity between all pairs of samples within the year group

4.4.4 Quoys Station 10 (ST10)

Two identification errors stand out in Quoys ST10: in 2009, 13 Hydrobiidae / Rissoidae were recorded, but when verified these were identified as Murchisonellidae (Table 4.6); and in 2011, 130 Syllidae were recorded, but when these were verified most were identified as Capitellidae (Table 4.6). In years 2012 and 2013 no Capitellidae were recorded in the Original data, in the Verified data 68 and 32 were recorded respectively. In these two years there were large numbers of 'Unknown' worms which had not been taken into account in the data analysis, only once the samples were re-identified were these 'Unknown' worms accounted for. In other years the verification errors in the Capitellidae and Oligochaeta families involve one or two specimens.

Two direct taxonomic name changes were highlighted: Haustoriidae for which the correct identification is Urothoidae, Cumacea which when identified to family level is Lampropidae and Pyramidellidae for which the correct identification is Murchisonellidae.

After the verification process the following taxa still have the same number of records: Arenicolidae, Phyllodocidae, Ampeliscidae and Ammodytidae. All other taxa have small differences between Original and Verified data.

4.4.4.1 Quoys ST10 Original vs Verified data analysis with replicates

When performing a cluster dendrogram with SIMPROF test on the ST10 Original and Verified data while using replicates, the SIMPROF test creates first significant division of the data at 44% similarity for Original and 51% similarity for Verified data (Figure 4.4). For Original data the two clusters are 2006-2011 and 2012-2013, for Verified data the two clusters are 2006- 2010 and 2011-2013 indicating a shift one year earlier compared to the Original data. The timing of the shift is consistent between the two Quoys stations (ST7 and ST10) in Verified data. A SIMPER test explored this further and showed that this first division of the data in the Original data is driven by a shift in the composition of the taxa (Table 4.7): in 2006-2011 the most abundant three taxa are the amphipods Pontoporeiidae, polychaetes Spionidae, and Syllidae and, in 2012 the dominant three taxa are amphipods Pontoporeiidae, polychaetes Opheliidae, and Paraonidae, the main change being the change in the polychaete taxa present. In 2013 the most abundant taxa change back into polychaete Spionidae and amphipod Pontoporeiidae in Original data. The division of the two clusters in the Verified data are driven by the change in one of the main taxa present, an increase in the abundance of polychaete Capitellidae. This increase in Capitellidae in the Verified data is due to a misidentification of Syllidae in 2011 (Table 4.6). The high abundance of Syllidae in the Original data is driving the significant division in the Original data, and once the samples were correctly identified as Capitellidae; the increase in this taxon drives the significant division in Verified data. This misidentification of one taxon is the factor defining the difference between the shift in the Quoys ST10 Original and Verified data.

4.4.4.2 Quoys ST10 Original vs Verified data analysis, replicates summed

When performing the same tests with Quoys ST10 Original and Verified data but summing the replicates for each year the cluster dendrogram with SIMPROF test has only one significant division of the data at 49% similarity for Original and at 60% similarity for Verified data (Figure 4.5). For Original data the SIMPROF test divides the data into two clusters, 2006-2011 and 2012-2013; for Verified data the SIMPROF test divides the data into following two clusters, 2006-2010 and 2011-2013. The division of both data when the replicates are summed follows the same pattern as is seen when both data sets are analysed with replicates and is consistent with the findings with that for Quoys ST7.

Table 4.6. Quoys Station 10 summed abundances recorded for Original and Verified data. Grey highlight indicates taxa which have the most differences for Original and Verified data. Lines denote key changes in personnel: 2007 Biologist changed, 2010 and 2013 Technician changed.

Figure 4.4. Quoys ST10 Original versus Verified data with replicates. Multidimensional scaling (MDS) and cluster dendrogram analysis. Cluster Figure 4.4. Quoys ST10 Original versus Verified data with replicates. Multidimensional scaling (MDS) and cluster dendrogram analysis. Cluster dendrogram with SIMPROF test, solid black line = significant difference. MDS ordination with significant SIMPROF similarity clusters. dendrogram with SIMPROF test, solid black line = significant difference. MDS ordination with significant SIMPROF similarity clusters.

Table 4.7. Quoys ST10 with replicates. Contributions of representative taxa to each year based on PRIMER analysis (cut-off at 60%). Dashed line represents significant SIMPROF separation.

Contrib%: Percentage contribution of taxa to the year group; Cum.%: Cumulative percentage contribution of taxa to the year group; Av. Similarity: The average Bray-Curtis similarity between all pairs of samples within the year group

4.4.5 Quoys Station 12 (ST12)

The most identification errors in Quoys ST12 are with the family Paraonidae which in Original data were incorrectly identified as belonging to family Orbiniidae (Table 4.8). Inconsistencies in identification were with the order Cumacea, which in Original data was recorded at order level Cumacea but when verified is divided into the two families Bodotriidae and Lampropidae (Table 4.8).

Two direct taxonomic name changes due to consistent misidentification have been highlighted: Haustoriidae for which the correct identification is Urothoidae, and Pyramidellidae for which the correct identification is Murchisonellidae.

After the verification process the following taxa still have the same number of records: Arenicolidae, Ampeliscidae, Caprellidae, Phoxocephalidae and Cardiidae. All other taxa have small differences between Original and Verified data.

4.4.5.1 Quoys ST12 Original vs Verified data analysis with replicates

Note that there are no data for year 2009 for ST12 at Quoys. No samples were collected in 2009; for further information see Chapter 3 Section 3.5.2 Quoys.

When performing a cluster dendrogram with SIMPROF test on the ST12 Original and Verified data while using replicates, the first significant division of the data is at 51% similarity for Original and 54% similarity for Verified data (Figure 4.6). For Original data the two clusters are 2006-2008 with one replicate from 2010 and rest of the replicates from 2010-2013; for Verified data the two clusters are 2006-2008 and 2010-2013. For Verified data the timing of the change is similar to ST7 and ST10 indicating that the SIMPROF test is able to identify a trend of change. A SIMPER test explored this further and showed that this first division in both the Original and Verified data is driven by a shift in the composition of the taxa: the three most abundant taxa in 2006-2008 are amphipods Pontoporeiidae and Oedicerotidae and gastropod Murchisonellidae (Pyramidellidae in Original data), and in 2010 the three most abundant taxa are amphipods Pontoporeiidae, Oedicerotidae, and polychaete Spionidae (Table 4.9). The cluster dendrogram for the Original data shows one further significant division, 2006- 2008 being separated from one replicate from 2010, this is not analysed further as the data with the replicates summed presents one significant division only (Figure 4.7). For Quoys ST12 not many misidentification errors were made (Table 4.8). The most common is a discrepancy in the naming of the taxa and either using old nomenclature, as in case of Urothoidae which was called Haustoriidae or naming taxa consistently by a wrong name, for example Murchisonellidae being incorrectly identified as Pyramidellidae.

4.4.5.2 Quoys ST12 Original vs Verified data analysis, replicates summed

When performing the same tests with Quoys ST12 Original and Verified data but summing the replicates for each year the cluster dendrogram with SIMPROF test has only one significant division of the data at 55% similarity for Original and at 63% similarity for Verified data (Figure 4.7). For both Original and Verified data the SIMPROF test divides the data into two clusters, 2006-2008 and 2010-2013. The division of both data sets when the replicates are summed follows the same pattern as is seen when the data are analysed with replicates.

Table 4.8. Quoys ST12 summed abundances recorded for Original and Verified data. Grey

highlight indicates taxa which have the most differences for Original and Verified data. Lines denote key changes in personnel: 2007 Biologist changed, 2010 and 2013 Technician changed.

	QUOYS ST12 - ORIGINAL DATA			QUOYS ST12 - VERIFIED DATA			
Year	Taxa	Contrib%	$Cum. \%$	Taxa	Contrib%	$Cum. \%$	
2006	Average similarity: 85.15		Average similarity: 84.84				
	Pontoporeiidae	23.31	23.31	Pontoporeiidae	22.39	22.39	
	Oedicerotidae	15.36	38.66	Oedicerotidae	14.74	37.13	
	Spionidae	13.05	51.71	Spionidae	12.4	49.52	
	Pyramidellidae	11.97	63.69	Murchisonellidae	11.57	61.09	
2007	Average similarity: 78.01				Average similarity: 82.29		
	Pontoporeiidae	23.72	23.72	Pontoporeiidae	22.42	22.42	
	Oedicerotidae	13.72	37.44	Murchisonellidae	13.59	36.01	
	Haustoriidae	13.63	51.07	Oedicerotidae	13.08	49.09	
	Corophiidae	12.52	63.59	Urothoidae	12.88	61.97	
2008	Average similarity: 84.80			Average similarity: 81.27			
	Pontoporeiidae	24.89	24.89	Pontoporeiidae	24.01	24.01	
	Oedicerotidae	14.87	39.76	Oedicerotidae	14.37	38.37	
	Pyramidellidae	14.6	54.36	Murchisonellidae	14.1	52.47	
	Corophiidae	13.72	68.07	Corophiidae	13.23	65.7	
2010	Average similarity: 69.51			Average similarity: 71.59			
	Pontoporeiidae	20.44	20.44	Pontoporeiidae	19.47	19.47	
	Spionidae	18.5	38.94	Spionidae	17.28	36.76	
	Oedicerotidae	15.4	54.34	Oedicerotidae	14.54	51.3	
	Syllidae	12.39	66.74	Tellinidae	11.9	63.2	
2011	Average similarity: 74.66			Average similarity: 77.15			
	Pontoporeiidae	18.76	18.76	Pontoporeiidae	17.7	17.7	
	Spionidae	15.66	34.42	Paraonidae	14.84	32.54	
	Oedicerotidae	12.46	46.88	Spionidae	14.77	47.32	
	Tellinidae	11.39	58.26	Oedicerotidae	11.77	59.09	
	Pyramidellidae	11.25	69.51	Tellinidae	10.74	69.83	
2012	Average similarity: 74.81			Average similarity: 76.09			
	Pontoporeiidae	29.69	29.69	Pontoporeiidae	27.55	27.55	
	Oedicerotidae	16.43	46.12	Oedicerotidae	15.23	42.78	
	Corophiidae	12.62	58.74	Corophiidae	11.71	54.49	
	Paraonidae	11.32	70.06	Tellinidae	10.63	65.12	
2013	Average similarity: 68.82			Average similarity: 68.82			
	Pontoporeiidae	31.61	31.61	Pontoporeiidae	31.61	31.61	
	Spionidae	21.66	53.27	Spionidae	21.66	53.27	
	Paraonidae	12.24	65.51	Paraonidae	12.24	65.51	

Table 4.9. Quoys ST12 with replicates. Contributions of representative taxa to each year based on PRIMER analysis (cut-off at 60%). Dashed line represents significant SIMPROF separation.

Contrib%: Percentage contribution of taxa to the year group; Cum.%: Cumulative percentage contribution of taxa to the year group; Av. Similarity: The average Bray-Curtis similarity between all pairs of samples within the year group

4.4.6 Congesquoy Station 1 (ST1)

Most identification errors in Congesquoy ST1 were with Orbiniidae and Paraonidae (Table 4.10). In Original data in 2006-2010 only Orbiniidae were identified and from 2011 onwards both Orbiniidae and Paraonidae were identified. In the Verified data both Orbiniidae and Paraonidae were identified in all years. A small number of inconsistencies in identification were highlighted for Capitellidae and Oligochaeta and for order Cumacea, and for family Lampropidae belonging to order Cumacea.

Three taxonomic name changes were highlighted, one for order Tanaidacea, which when identified to family level were Tanaissuidae, Hydrobiidae / Rissoidae which when verified were Hydrobiidae and for Pyramidellidae which when verified were Murchisonellidae.

After the verification process the following taxa still have the same number of records: Phyllodocidae, Terebellidae, Ampeliscidae, Calliopiidae, Caprellidae, Corophiidae, Crangonidae, Portunidae, Mysidae, and Retusidae. All other taxa have small differences between Original and Verified data.

4.4.6.1 Congesquoy ST1 Original vs Verified data analysis with replicates

When performing a cluster dendrogram with SIMPROF test on the Original and Verified data while using replicates, the first significant division of the data is at 61% similarity for Original and 66% similarity for Verified data (Figure 4.8). For Original data the two clusters are 2006-2007 and 2008-2013, and for Verified data the two clusters are 2006- 2013, and an outlier of one replicate from 2012.

A SIMPER test explored this further and showed that in the Original data the division of data into two clusters was driven by the following changes in the macrofauna: 2006 – 2007 all taxa present had low abundances but were dominated by polychaetes Spionidae, Orbiniidae and Syllidae; 2008-2013 differs from the two earlier years by having higher abundances of polychaetes Spionidae and Opheliidae (Table 4.11). In the Verified data analysis, the clustering was less distinct and replicates from different years group together indicating similar macrofaunal composition in the samples over the years but the SIMPROF outcome suggests that there has been a significant change (Figure 4.8).

4.4.6.2 Congesquoy ST1 Original vs Verified data analysis, replicates summed

When performing a cluster dendrogram with SIMPROF test on the Original and Verified data while replicates are summed, the first significant division of the data is at 68% similarity for Original, and for Verified data there are no significant divisions (Figure 4.9). The Original data are divided into three clusters, 2006-2007, 2008-2010 and 2011-2013, these divisions in the data are comparable with the first three significant divisions when the SIMPROF test is performed on the Original data with replicates.

4.4.6.3 Congesquoy ST1 Original data analysis with Capitellidae, Orbiniidae / Paraonidae and Cumacea corrected

To investigate further the possible causes of the significant divisions in the Original data three inconsistencies in the data were changed. For 2006 the abundance count of 15 for Lampropidae was aggregated to order level Cumacea, for 2007 the abundance count of two for Capitellidae/Oligochaeta was moved to Capitellidae, and for 2006-2013 the abundance of Orbiniidae and Paraonidae were summed and the total abundance relabelled as Orbiniidae / Paraonidae (Table 4.10. hashed pattern). Each one of these inconsistencies were tested first individually, the results were similar to the initial data analysis. When all of these inconsistencies were applied together the results were similar to the Verified data, no significant SIMPROF divisions were created (Figure 4.9. and 4.10.). This indicates that at Congesquoy ST1 several small identification issues and inconsistencies are influencing the Original data analysis and when these inconsistencies are corrected the results are similar with the Verified data.

Table 4.10. Congesquoy ST1 summed abundances recorded for Original and Verified data.

Grey highlight indicates taxa which have the most differences for Original and Verified data. Lines denote key changes in personnel: 2007 Biologist changed, 2010 and 2013 Technician changed. Hashed area denotes inconsistencies which were corrected for analysis described in Section. 4.4.4.3.

analysis. Cluster dendrogram with SIMPROF test, solid black line = significant difference. MDS ordination with significant SIMPROF similarity analysis. Cluster dendrogram with SIMPROF test, solid black line = significant difference. MDS ordination with significant SIMPROF similarity Figure 4.9. Congesquoy ST1 Original versus Verified data with replicates summed. Multidimensional scaling (MDS) and cluster dendrogram **Figure 4.9.** Congesquoy ST1 Original versus Verified data with replicates summed. Multidimensional scaling (MDS) and cluster dendrogram clusters.

Table 4.11. Congesquoy ST1 with replicates. Contributions of representative taxa to each year based on PRIMER analysis (cut-off at 50%). Dashed line represents significant SIMPROF separation.

Contrib%: Percentage contribution of taxa to the year group; Cum.%: Cumulative percentage contribution of taxa to the year group; Av. Similarity: The average Bray-Curtis similarity between all pairs of samples within the year group

4.4.7 Congesquoy Station 2 (ST2)

As in Congesquoy ST1, the most identification errors in Congesquoy ST2 were with Orbiniidae and Paraonidae (Table 4.12). In the Original data during 2006-2010 only Orbiniidae were identified, but from 2011 onwards both Orbiniidae and Paraonidae were identified. In the Verified data both Orbiniidae and Paraonidae were identified in all years. Other errors were with Capitellidae and Oligochaeta; in the Original data Capitellidae were identified every year apart from 2007 when Capitellidae / Oligochaeta were identified, Oligochaeta were identified once in the Original data, in 2011. When the samples were verified two different families within the class Oligochaeta were recorded in the samples: Enchytraeidae in 2009 and 2011, Naididae in 2010 (Table 4.12). Cumacea were recorded in the Original data in 2006-2012, Cumacea (Lampropidae) were recorded in 2006 and 2013. When verified Cumacea (Lampropidae) were present every year 2002- 2013 and Cumacea (Bodotriidae) were present once in 2011. The marine gastropods Murchisonellidae were incorrectly identified as Pyramidellidae in the Original data.

Several nomenclature changes were present: Sabellidae should be Fabriciidae, Haustoriidae should be Urothoidae and order Tanaidacea when identified to family level is Tanaissuidae (Table 4.12).

After the verification process the following taxa still have the same number of records: Magelonidae, Terebellidae, Caprellidae, Crangonidae, Nebaliidae and Mysidae. All other taxa have small differences between Original and Verified data.

4.4.7.1 Congesquoy ST2 Original vs Verified data analysis with replicates

When performing a cluster dendrogram with SIMPROF test on the Original and Verified data while using replicates, the first significant division of the data is at 47% similarity for Original and 67% similarity for Verified data (Figure 4.11). For both Original and Verified data the years are divided into two main clusters in which 2011 is separated from all other years, this division is clearly observed from the MDS ordination (Figure 4.11). In both the Original and Verified data the year 2011 was dominated by three amphipod taxa: Pontoporeiidae, Corophiidae and Haustoriidae (Original data) and Urothoidae (Verified data) therefore separating this year from all other years (Table 4.13).

4.4.7.2 Congesquoy ST2 Original vs Verified data analysis, replicates summed

When the replicates are summed the resulting division in data for both Original and Verified data were same as when the replicates are used, the first division in the data divides the years into two clusters in which 2011 is separated from all other years. The Verified data analysis does not result in significant SIMPROF clusters, whereas the

Original data analysis has eight of which three are shown in the MDS ordination (Figure 4.12).

4.4.7.3 Congesquoy ST2 Original data analysis with Capitellidae, Orbiniidae/Paraonidae, Cumacea and Hydrobiidae/Rissoidae corrected

As in Congesquoy ST1, it was decided to investigate further the possible causes of the significant divisions in the Original data. Four inconsistencies in the Congesquoy ST2 Original data were amended. For 2006 and 2013 the abundance count of 14 and 2, respectively, for Lampropidae were aggregated to order level Cumacea, for 2007 the abundance count of 21 for Capitellidae/Oligochaeta was moved to Capitellidae, for 2006- 2013 the abundance of Orbiniidae and Paraonidae were summed and the abundance relabelled as Orbiniidae / Paraonidae and abundance count of 1 for Hydrobiidae was moved to Hydrobiidae/Rissoidae (Table 4.12. hashed pattern). Each one of these inconsistencies were tested on their own; for Capitellidae and Hydrobiidae/Rissoidae no change in the patterns in the cluster dendrogram or MDS was apparent compared to the initial analysis. For Orbiniidae/Paraonidae, Cumacea/Lampropidae and when all the changes were applied together the results were similar to the Congesquoy ST2 Verified data analysis (Figure 4.13). This is similar to Congesquoy ST1 where several small identification issues and inconsistencies influenced the Original data analysis.

Table 4.12. Congesquoy ST2 summed abundances recorded for Original and Verified

data. Grey highlight indicates taxa which have the most differences for Original and Verified data.

Lines denote key changes in personnel: 2007 Biologist changed, 2010 and 2013 Technician changed. Hashed area denotes inconsistencies which were corrected for analysis described in Section. 4.4.5.3.

Table 4.13. Congesquoy ST2 with replicates. Contributions of representative taxa to each year based on PRIMER analysis (cut-off at 50%). Dashed line represents significant SIMPROF separation.

Contrib%: Percentage contribution of taxa to the year group; Cum.%: Cumulative percentage contribution of taxa to the year group; Av. Similarity: The average Bray-Curtis similarity between all pairs of samples within the year group

4.4.8 Waulkmill Station 10 (ST10)

Several identification errors were made within the taxa Capitellidae and Oligochaeta (Table 4.14). In 2008 in the Original data, specimens belonging to Capitellidae and Oligochaeta were identified as either Capitellidae, Capitellidae / Oligochaeta or Oligochaeta. In 2010 no Oligochaeta were identified, only Capitellidae. When these specimens were verified three taxa were recorded, Capitellidae, Oligochaeta (Enchytraeidae) and Oligochaeta (Naididae). Of these, Capitellidae and Oligochaeta (Enchytraeidae) were the most common and present every year. In the Original data the family Orbiniidae was consistently misidentified in 2006-2010, but from 2011 onwards it was correctly identified as Paraonidae (Table 4.14).

The only taxa, which after the verification process still have the same number of records are Arenicolidae, Corophiidae, Idoteidae, Portunidae and Retusidae. All other taxa have small differences between Original and Verified data.

4.4.8.1 Waulkmill ST10 Original vs Verified data analysis with replicates

When performing a cluster dendrogram with SIMPROF test on the Original and Verified data while using replicates, the first significant division of the data was at 56% similarity for Original and 65% similarity for Verified data (Figure 4.14). The first significant division of data for both Original and Verified data was the same: the first cluster includes two replicates from 2012 (replicates 1 and 2) and the second cluster includes all replicates from years 2006-2011 and three replicates from 2012 (replicates 3, 4 and 5). For Original data further, lower level significant divisions were also created (Figure 4.14). Because the replicates from year 2012 were split within the clusters SIMPER will not be able to explain these divisions. SIMPER test uses the replicates from each year to create the similarity and dissimilarity percentages for a year, therefore pooling the replicates.

4.4.8.2 Waulkmill ST10 Original vs Verified data analysis, replicates summed

When the replicates are summed for both Original and Verified data and the cluster dendrogram with SIMPROF test is performed there were no significant divisions (Figure 4.15, Table 4.15). The MDS ordination for Original data (Figure 4.15.a) indicates a directional shift from 2006-2010 to 2011-2013 by separating these two groups on the x-axis, whereas the MDS ordination for Verified data did not have any discernible trends (Figure 4.15.b).

4.4.8.3 Waulkmill ST10 Original vs Verified data analysis, Paraonidae corrected A possible identification error was highlighted in the Original data relating to Orbiniidae and Paraonidae (Table 4.14). In the Original data in 2006-2010 only Orbiniidae were recorded, from 2011 onwards only Paraonidae were recorded (Table 4.14). To investigate

any possible data analysis errors caused by these misidentifications the Orbiniidae were re-labelled to Paraonidae for 2006-2010 in the Original data, therefore eliminating the artificial change in the macroinvertebrate community from 2010 to 2011. The cluster dendrogram and MDS ordinations for this 'Waulkmill ST10 Original – Paraonidae corrected' data were created (Figure 4.16). The results for these data are comparable with the Verified data set (Figures 4.14 and 4.15) and removed the directional shift on the xaxis. This highlights that the apparent trend first observed using the Original data (Figure 4.15.a) is an artefact of the change in identification and not due to a change in the environmental conditions.

Table 4.14. Waulkmill ST10 summed abundances recorded for Original and Verified data.

Grey highlight indicates taxa which have the most differences for Original and Verified data. Lines denote key changes in personnel: 2007 Biologist changed, 2010 and 2013 Technician changed. Note: Cardiidae: in Original data most specimens were juvenile.

103

Table 4.15. Waulkmill ST10 with replicates. Contributions of representative taxa to each year based on PRIMER analysis (cut-off at 50%). No significant SIMPROF separations present.

Contrib%: Percentage contribution of taxa to the year group; Cum.%: Cumulative percentage contribution of taxa to the year group; Av. Similarity: The average Bray-Curtis similarity between all pairs of samples within the year group

4.4.9 Waulkmill Station 12 (ST12)

The abundances of Capitellidae and Oligochaeta are lower in Waulkmill ST12 compared with those in Waulkmill ST10 and therefore the magnitude of the identification errors in these two taxa were less in Waulkmill ST12 compared to other sites (Table 4.16). The incorrect identification of Paraonidae in 2006-2010 as Orbiniidae, as discussed for Waulkmill ST10, can also be seen in the Waulkmill ST12 Original data as highlighted in Table 4.15. In 2006 Cumacea was identified to the order level only and once verified it was identified as belonging to the family Lampropidae.

The taxa which after the verification process still have the same number of records are Nephtyidae, Phyllodocidae, Psammodrilidae, Terebellidae, Crangonidae, Cardiidae and Veneridae. All other taxa have small differences between Original and Verified data.

4.4.9.1 Waulkmill ST12 Original vs Verified data analysis with replicates

When performing a cluster dendrogram with SIMPROF test on the Original and Verified data while using replicates, the first significant division of the data is at 60% similarity for Original and 65% similarity for Verified data (Figure 4.17). For Original data this division creates two clusters, 2006-2010 and 2011-2013. For Verified data year 2011 is separated from the main cluster of 2006-2010, 2012 and 2013 (Figure 4.17). A SIMPER test explored these divisions further and revealed that for the Original data, in the cluster 2006-2010, polychaete Orbiniidae was present in high abundance whereas in the cluster 2011-2013, only Paraonidae was present (Table 4.17). This change in the Original data from Orbiniidae to Paraonidae is due to the misidentification of the polychaete in the pre-2011 years (highlighted in Table 4.16).

4.4.9.2 Waulkmill ST12 Original vs Verified data analysis. replicates summed

When the replicates are summed for the Original data the resulting divisions are the same as when the replicates are used (Figure 4.18). For the Verified data, when the replicates are summed no significant divisions are created (Figure 4.18) compared with when the replicates are used and one significant division is created (Figure 4.17).

4.4.9.3 Waulkmill ST12 Original vs Verified data analysis, Paraonidae corrected

By correcting the misidentification of Orbiniidae in Original data, the results 'Waulkmill ST12 Original – Paraonidae corrected' are similar to Verified data (Figures 4.17 and 4.19). SIMPER analysis on the 'Paraonidae corrected' data highlighted that the high abundance of Pontoporeiidae, Tellinidae and Orbiniidae / Paraonidae is driving the significant division of 2011 from all other years (Table 4.18). When the misidentification of Orbiniidae is corrected in the Original data (replicates summed) the resulting MDS ordination is similar to the Verified data, and the cluster dendrogram is similar except for

one significant division being present: 2011 is separated from the rest of the years (Figure 4.19).

Table 4.16. Waulkmill ST12 summed abundances recorded for Original and Verified data. Grey highlight indicates taxa which have the most differences for Original and Verified data. Lines denote key changes in personnel: 2007 Biologist changed, 2010 and 2013 Technician changed.

Table 4.17. Waulkmill ST12 with replicates. Contributions of representative taxa to each year based on PRIMER analysis (cut-off at 50%). Dashed line represents significant SIMPROF separation.

Contrib%: Percentage contribution of taxa to the year group; Cum.%: Cumulative percentage contribution of taxa to the year group; Av. Similarity: The average Bray-Curtis similarity between all pairs of samples within the year group

Table 4.18. Waulkmill ST12 Original data - Paraonidae corrected with replicates. Contributions of representative taxa to each year based on PRIMER analysis (cut-off at 50%). Dashed line represents significant SIMPROF separation.

Contrib%: Percentage contribution of taxa to the year group; Cum.%: Cumulative percentage contribution of taxa to the year group; Av. Similarity: The average Bray-Curtis similarity between all pairs of samples within the year group

4.5. Discussion

The main issues highlighted by the verification process and the analysis of the Original and Verified data can be summarised as inconsistencies due to the use of incorrect taxonomic nomenclature and errors due to misidentification and miscounting.

For Murchisonellidae, Urothoidae and Fabriciidae inconsistencies have arisen from the incorrect use of taxonomic names (Table 4.2). The use of incorrect taxonomic names has not affected the outcome of the data analysis and can be corrected by direct label changes. Taxonomic inconsistencies easily infiltrate taxonomic laboratories if on-going quality control, auditing and updating of taxonomic names are not carried out regularly (Ranasinghe et al. 2003; Stribling et al. 2003, 2008; NMBAQC 2018). Updates in taxonomy of marine invertebrates are moving fast with the use of genetic techniques and online publishing of manuscripts (Vandepitte et al. 2018). If reviews of the taxonomic names of the taxa had been carried out the changes in the names would have been corrected many years ago. Stribling et al. (2003) describes how taxonomic bias can exists if continuous misinterpretation of dichotomous keys or outdated keys are used. The using of an old taxonomic guide, the Hayward & Ryland (1995) at OICHA has inadvertently encouraged the use of old taxonomic nomenclature for several of the taxa: Murchisonellidae, Urothoidae and Fabriciidae. Experienced taxonomists are adept at noticing old taxonomic nomenclature or taxonomic synonyms for taxa they are used to working with (Ranasinghe et al. 2003), however when an inexperienced analyst or data user is involved they might not be able to do so. Regular taxonomic nomenclature reviews for taxa recorded in a monitoring programme would enable any changes to the names to be implemented on an on-going basis, direct label changes could also be applied retrospectively to existing records. For the analysis of the remaining ten sandy beach sites in Orkney, the changes of the nomenclature can be corrected by a direct label changes as described in Table 4.2.

The misidentification of polychaete species belonging to the families Orbiniidae and Paraonidae, and polychaetes Capitellidae and class Oligochaeta were highlighted and were shown to have significant influence on the results. The confusion with species belonging to families Orbiniidae and Paraonidae were with the identification errors relating to the species *Paraonis fulgens* from family Paraonidae. Until 2011 this species was misidentified as belonging to the family Orbiniidae, in 2011 (after attending a taxonomic course, sample verification and improved identification skills) the correct identification of this species began. When analysing the Original data, this abrupt change

115

in the species identification is interpreted by SIMPROF analysis as a significant change in the macroinvertebrate community. To mitigate this, the re-labelling of all pre-2011 Orbiniidae as Paraonidae in the Waulkmill Original data corrects the issue. However, not all taxa belonging to the family Orbiniidae are the species *P. fulgens*. At Quoys and Congesquoy *Scoloplos armiger,* a species belonging to the family Orbiniidae, were present. *S. armiger* is a large polychaete (20-50 mm long) (Hayward & Ryland 2017), it is very distinctive from other polychaetes in the Orkney Islands sandy beach samples and it is correctly identified in all samples. The re-labelling of all Orbiniidae as Paraonidae is only a valid mitigation measure if an abrupt change from Orbiniidae to Paraonidae is present in 2011 as it was in Waulkmill. If both Orbiniidae and Paraonidae are recorded in either pre- and post-2011 samples then the summing of the abundances together and relabelling the pooled abundance as Orbiniidae / Paraonidae would be more appropriate as was carried out for Congesquoy ST1 and ST2. The inclusion of both family names in the label signals that both taxa are present but the specific abundances of each are not known.

Other misidentification issues were highlighted with the identification of polychaetes belonging to the family Capitellidae and oligochaetes belonging to the class Oligochaeta. During the verification process the confusion in these two taxa were revealed mainly to be due to the incorrect identification of species *Baltidrilus costatus, Paranais litoralis, Tubificoides benedii* and *T. pseudogaster* which all belong to the family Naididae but were misidentified as Capitellidae (at Quoys ST7). At Waulkmill ST10 the specimens identified as class Oligochaeta when re-identified belonged to the family Enchytraeidae, this was a direct label change as the abundances remained almost unchanged. In both of these sites, Quoys ST7 and Waulkmill ST10, the confusion of Capitellidae and Oligochaeta affected the results of the data analysis.

Laternulidae (Waulkmill ST10), Dorvilleidae (Congesquoy ST1, Waulkmill ST10) and Janiridae (Waulkmill ST10 and ST12) were all misidentified in the Original data. In each case the abundances of the taxa were small, between one and three, and were not found to affect the data analysis.

At Congesquoy it was highlighted that both identification errors and aggregation of taxa to different levels were in combination influencing the results. The misidentification of Orbiniidae and Paraonidae, Capitellidae and Oligochaeta, and the non-aggregation of Lampropidae to order Cumacea, all affected the results. These different errors and inconsistencies in the data contributed to the incorrect trends shown in the results.

In their research on benthic macroinvertebrates Ranasinghe et al. (2003) reported that miscounts were the most common type of error affecting their data; 4.8% of their data records were affected by miscounts compared to 4.5% by misidentifications. These results are opposite to what was found at the Orkney sandy beach sites where misidentifications were between $1.2 - 16.1\%$ of the data records and miscounts $0.1 -$ 2.3%. The percentage of taxonomic disagreement values are higher in the Orkney sandy beach sites compared to Ranasinghe et al. (2003) but others have reported even higher values; 29.6% (Stribling et al. 2008) and 33.8% (Haase et al. 2010). The misidentifications have been attributed to the analysts' differing levels of experience, the condition of the samples and differences of opinion (Ranasinghe et al. 2003), experience of the taxonomists, sample condition and quantity allocated to analyst (Stribling et al. 2008) and poor sample processing and identification (Haase et al. 2010). The experience of the analysts or taxonomists has been highlighted by Ranasinghe et al. (2003), Stribling et al. (2003, 2008) and Haase et al. (2010) and cannot be over emphasised. In the Orkney sandy beach monitoring programme several different analysts, with different levels of expertise have worked on the samples with often outdated identification guides. The effect of the experience and the training which the analyst has received is not only paramount in the identification of the samples but also in the enumeration process. The errors in enumeration were negligible in the Orkney sandy beach samples which gives assurance that even though there are issues in the taxonomic precision of the data the abundances of the taxa have only a small margin of error. Establishing standard operating procedures, updating identification guides, establishing voucher specimen collection, giving appropriate training and by implementing quality control measures the taxonomic skills of the analysts can be improved and the inter-operator variability can be reduced (Ellis 1985, 1988; Ranasinghe et al. 2003; Stribling et al. 2003, 2008, Milner & Hall 2016). During the course of these doctoral studies many of these improvements have been successfully implemented at the OICHA laboratory as described in Section 4.3 of this chapter. However, the improvement in the identification and enumeration process will be an on-going process which will continue as long as sandy beach monitoring is part of the OICHA work programme.

The pooling of the replicates clarifies the inter-annual trends in the macroinvertebrate communities. When replicates for each station for each year were used, the year on year trends were not always clear. The variability between the replicates resulted in outliers, the only stations without significant single replicate outliers were Quoys ST10 and Waulkmill ST12. To perform the SIMPER analysis (Chapter 2 Section 2.2.5) replicates

for each station are required but for the analysis of the temporal trends the pooling of replicates clarifies the trends. Pooling of replicates for each year to investigate temporal trends is applied widely to long term monitoring studies (Whomersley et al. 2007; Blanchard et al. 2010; Chainho et al. 2010; Kröncke et al. 2011; Schulz et al. 2013; Weydmann et al. 2014).

4.5.1 Conclusions

The errors and inconsistencies in the Original data due to misidentifications and identification of specimens to different taxonomic levels (family, order, and class) without consistent aggregation during the monitoring period cumulatively affect the results and the patterns that emerge from the results. The verification process and data analysis of the three sites, Quoys, Congesquoy and Waulkmill, have each highlighted how these issues affect data analysis at different sites. The full extent of the misidentifications and inconsistencies would not be possible without the verification process.

To apply this level of scrutiny and changes to the rest of the ten sandy beaches would be beyond what is achievable in the time available for this thesis. The detailed analysis of the spatial and temporal variability of the macroinvertebrate communities will be focussed on the three sites with verified data; Quoys, Congesquoy and Waulkmill.

Chapter 5 Scapa Flow-wide analysis

5.1 Introduction

The spatial and temporal variability in benthic macroinvertebrate distributions have been widely studied on sandy beaches (Dexter 1984; Ysebaert & Herman 2002; Jarrin et al. 2017; Bae et al. 2018) and in sub-tidal environments (Chainho et al. 2010; Ingels & Vanreusel 2013; Chatzinikolaou et al. 2018). Sandy beaches are inherently very harsh and dynamic environments (Brown & McLachlan 2002; Defeo & McLachlan 2005; Barreiro et al. 2011; McLachlan & Defeo 2018) due to the instability of their substrate and their position as an interface between marine and terrestrial environments. Macroinvertebrate communities on sandy beaches are known to be naturally patchy (Morrisey et al. 1992). The macroinvertebrate communities as well as the physical characteristics of the sandy beaches are driven by three main factors: tidal regime, wave energy and sediment particle size (Short 1996; Defeo & McLachlan 2005). Tidal ranges in the UK vary substantially from a spring tide range of 14 m in Avonmouth (Xia et al. 2010) to 1.9 m in Lowestoft (National Tidal and Sea Level Facility 2018). The Scapa Flow sandy beach monitoring sites are all within the same tidal regime having a spring tidal range maximum of 4.1 m (Appendix F Section 4). On a given beach the sand particle size is determined by the exposure to waves (Short 1996; McLachlan & Defeo 2018); exposed beaches have a coarser sand compared to sheltered beaches with finer sand particles. Oceanic sandy beaches can experience the full force of waves during storm events which can change the profile of the shore and sediment composition in a matter of hours (Morton & Sallenger 2003). Sheltered sandy beaches located in embayments or in the shelter of islands or reefs do not experience the same wave climate as oceanic beaches (Hegge et al. 1996; Jackson et al. 2002), but they can still be affected by storm events which can cause erosion on the shore and change the sand particle composition. The sandy beach monitoring sites in Scapa Flow are all in a sheltered body of water (Chapter 1 Figure 1.1) and do not experience oceanic waves but they are affected by weather-related waves during storms and gale force winds.

Differences between the physical characteristics of the Scapa Flow sandy beach monitoring sites are evident; the beaches range from ultra-dissipative (e.g. Congesquoy) to intermediate (e.g. Kirkhope) beach types, as described in Chapter 3. No two beaches, whether oceanic or sheltered, have identical macroinvertebrate species composition but there will be similarities between sites.

The macroinvertebrate time series data for each sandy beach site are analysed using the MDS ordinations 'map-format' (Chapter 2 Section 2.2.5.1), where the macroinvertebrate time series data are firstly transformed $(\sqrt{\sqrt{2}})$, then the Bray-Curtis similarities are calculated. Bray Curtis similarities are used to 'map' the time series on MDS ordination (e.g. Figure 5.2) (also called the first-stage MDS). Cyclical patterns of the time series data are looked for on the first-stage MDS ordinations. The visual comparison of the first-stage MDS ordinations is not easy and it is always subjective (Clarke & Warwick 2001). To enable an objective comparison of two first-stage MDS time series plots Clarke et al. (2006) proposed a method, the second-stage multi-dimensional scaling (MDS) analysis. The second-stage MDS compares first-stage MDS ordination patterns between two sites, thereby shifting the focus of analysis from the site-specific species composition to the patterns of changes in community structure. Second-stage MDS also calculates a Spearman's rank correlation which gives the measure of how closely the two sample patterns match. First-stage MDS is suitable for analysing time series data for a site in order to determine if there are any year-to-year patterns, second-stage MDS is used to statistically test whether the gradient pattern of two first-stage MDS plots are the same (Clarke & Warwick 2001).

5.2 Aim and hypothesis

The aim of this chapter is to scrutinise the macroinvertebrate time series data from eight stations, each station is from a different sandy beach within Scapa Flow but from the same tidal height, to understand if there are any Scapa Flow-wide regional effects affecting the macroinvertebrate communities, or if patterns are site-specific.

Hypothesis:

- $H_0: p > 0.05$ There are no spatial or temporal differences in the patterns of macroinvertebrate community time series data across the eight sandy beaches monitored, indicating that no Scapa Flow-wide trends are present.
- $H_1: p < 0.05$ There are spatial and/or temporal differences in the patterns of macroinvertebrate community time series data across the eight sandy beaches, indicating that Scapa Flow-wide trends are present.

5.3 Methods

To investigate the long-term patterns in the macroinvertebrate community assemblage structures, the second-stage analysis of MDS ordination was used (Clarke et al. 2006). The analysis requires each station to have the same time points. Data from the Current time period were analysed and only stations with complete data set for the required time period were used.

There were two restricting factors for the selection of the stations for the data analyses, the tidal height of the sampling stations on the shoreline (Chapter 3 Section 3.6) and the availability of the data. Taking these restrictions into account the second-stage MDS analysis was conducted for 2006-2013 & 2016 using data for eight low shore stations: Congesquoy ST2, Creekland ST11, Cumminess ST4, Kirk Hope MLWS, Longhope ST12, Lyrawa ST10, Mill Bay ST12 and Swanbister ST12 (Table 5.1 and Figure 5.1). The low shore stations were selected since they are more likely to be subject to any environmental variability.

Table 5.1. Lowest sampling stations, all within 30cm vertical height interval from each other, with the year's the samples were collected. Years in bold and enclosed within borders were used in the second-stage MDS analysis.

Figure 5.1. Sampling stations included in the second-stage MDS analysis.

The macroinvertebrate abundance data were aggregated to family level, except for: phylum Nemertea, class Oligochaeta, orders Cumacea and Mysida. The abundances for families Orbiniidae and Paraonidae were summed and labelled 'Orbiniidae / Paraonidae' as recommended in Chapter 4. To enable the macroinvertebrate data to be analysed using the first-stage MDS ordination the replicates for each year were summed. For the analyses the data were fourth root transformed prior to creating the Bray-Curtis resemblance matrices. The multivariate statistics were calculated using Primer v6 software package (Clarke & Warwick 2001; Clarke & Gorley 2006).

Spearman's rank correlation calculated by PRIMER was used to define the correlations between the first stage MDS ordinations of the sites. The strength of Spearman's rank correlations are given in Table 5.2. The p-values were calculated using Microsoft Excel.

Table 5.2. Strength of Spearman's rank correlation (Barcelona Field Studies Centre 2019).

Value of coefficient r_s (positive or negative)	Meaning
$0.00 \text{ to } 0.19$	Avery weak correlation
$0.20 \text{ to } 0.39$	A weak correlation
$0.40 \text{ to } 0.69$	A moderate correlation
$0.70 \text{ to } 0.89$	A strong correlation
$0.90 \text{ to } 1.00$	A very strong correlation

5.4 Results

5.4.1 2*006-2013 & 2016 Analysis*

The macroinvertebrate community at seven out of the eight sampling stations did not show any trends (Figure 5.2), the time series points of each station lacking any obvious organisation or direction. Lyrawa ST10 and Cumminess ST4 were the only sites with a consistent directional time trajectory pattern during $2006-2013 \& 2016$. The time series points were generally found to be moving away from the first year of samples (2006) along the x-axis. The other six sampling stations, Congesquoy ST2, Creekland ST11, Kirk Hope MLWS, Longhope ST12, Mill Bay ST12 and Swanbister ST12, had varied time trajectory patterns indicating inter-annual macroinvertebrate population variability in each sampling station, although no overall trend could be detected. Congesquoy ST2 MDS ordination is dominated by the separation of 2011 from all the other years (Figure 5.2).

The second-stage MDS ordination plot does not show distinct grouping of stations (Figure 5.3). Cluster analysis based on the similarity matrices of the sampling stations, groups the sampling stations into three clusters: 1) Swanbister ST12 and Mill Bay ST12; 2) Cumminess ST4, Longhope ST12, Creekland ST11 and Lyrawa ST10; 3) Congesquoy ST2 and Kirkhope MLWS (Figure 5.3 and Figure 5.2). The first-stage MDS time series trajectories were used to interpret these clusters. In the first cluster, the year 2009 was an

outlier from the first-stage MDS time series, at both sampling stations (Swanbister ST12 and Mill Bay ST12) (Figure 5.2). The year 2016 was an outlier from the first-stage MDS time series trajectory at all sampling stations in the second cluster (Figure 5.2). No similarities of the time series patterns are evident for the third cluster (Figure 5.2). The first-stage MDS time trajectory for Congesquoy ST2 had a strong separation of the year 2011 from the rest of the time trajectory (Figure 5.2). In 2011 the macroinvertebrate community at Congesquoy ST2 was dominated by amphipods compared to other years when the community was polychaete dominated (Chapter 6 Table 6.15). There were within-cluster similarities of the sampling stations but no overall trend in the macroinvertebrate communities in Scapa Flow was evident.

The Spearman rank correlation was used to determine the relationship between the sandy beaches (Table 5.3). When data for all years (2006-2013 & 2016) was combined, the strongest correlations (all positive) were between MI ST12 and SW ST12 ($r_s = 0.574$, p < 0.001), CR ST11 and CU ST4 (r_s = 0.353, p < 0.05), CR ST11 and LO ST12 (r_s = 0.537, $p < 0.001$), CR ST11 and LY ST10 ($r_s = 0.555$, $p < 0.001$), CU ST4 and LY ST10 ($r_s =$ 0.521, p< 0.001) and LY ST10 and MI ST12 ($r_s = 0.362$, p < 0.05) (Table 5.3). CO ST2 and KH MLWS had the weakest correlations with the other sandy beaches.

The H_0 can be accepted as there was no evidence of Scapa Flow-wide trends from the analysis of the eight sampling stations for the $2006-2013 \& 2016$ time period as is demonstrated by the low Spearman's rank correlation values, high p-values and the separation of the sampling stations within the second-stage MDS ordination plot. Evidence of similarities with some sites were demonstrated by the clustering of the sites in the second-stage cluster dendogram and by the statistical significance of some sites but the overall low Spearman's rank values imply low confidence in the clusters.

Table 5.3. Spearman rank correlation matrix of every single pair of similarity matrices: Congesquoy ST2, Creekland ST11, Cumminess ST4, Kirk Hope MLWS, Longhope ST12, Lyrawa ST10, Mill Bay ST12 and Swanbister ST12 for time period 2006-2013 & 2016. $r_s = 1$ indicates perfect positive correlation, $r_s = 0$ no association with the patterns, $r_s = -1$ indicates perfect negative correlation. Bold: *p < 0.05, **p < 0.01, ***p < 0.001.

	SW ST ₁₂	CO ST ₂	CR ST11	CU ST4	KH MLWS	LO ST ₁₂	LY ST ₁₀
SW ST ₁₂							
CO _{ST2}	0.090						
CR ST ₁₁	0.067	0.109					
CU ST ₄	-0.228	0.125	$0.353*$				
KH MLWS	0.119	0.191	0.042	0.162			
LO ST ₁₂	-0.214	0.067	$0.537***$	0.313	0.028		
LY ST ₁₀	-0.106	0.024	$0.555***$	$0.521***$	-0.028	$0.494**$	
MI ST ₁₂	$0.574***$	0.076	0.222	0.252	-0.276	-0.044	$0.362*$

Figure 5.2. First-stage MDS time trajectories (2006-2013 & 2016) for Congesquoy ST2, Creekland ST11, Cumminess ST4, Kirk Hope MLWS, Longhope ST12, Lyrawa ST10, Mill Bay ST12 and Swanbister ST12. The closer the distance between two points, the more similar in macroinvertebrate composition they are. Clusters refer to second-stage cluster analysis (Figure 5.3)

Figure 5.3. Second-stage MDS ordination plot and cluster dendrogram showing between year differences for Congesquoy ST2, Creekland ST11, Cumminess ST4, Kirk Hope MLWS, Longhope ST12, Lyrawa ST10, Mill Bay ST12 and Swanbister ST12, in 2006-2013 & 2016. Beach Types indicated on second-stage MDS ordination plot.

5.5 Discussion

Preliminary analysis of the macroinvertebrate community data from all Scapa Flow sandy beach sites demonstrated that the macroinvertebrate communities which were one station apart on the transect (30 cm vertical height difference) were not significantly different (Kakkonen 2016). The sampling stations for the Scapa Flow-wide analysis presented here were selected based on their tidal height on the sandy beaches, all were within 30 cm vertical tidal height. The analysis and comparison of these sampling stations were as close to like with like comparison as was possible within the Scapa Flow sandy beach sampling sites and stations (Chapter 3 Table 3.20).

Second-stage MDS analyses have been used in various situations to determine interannual variability such as in boreal zooplankton in the West Spitsbergen Current (Weydmann et al. 2006); long-term shifts in coral communities in Curaçao and Bonaire (De Bakker et al. 2017); and the habitat use of herbivorous fish in the Red Sea (Afeworki et al. 2013). Clarke et al. (2006) demonstrated the use of the second-stage MDS analysis on several sets of time series data; reef corals in Phuket; macrobenthos in Tees Bay and rocky subtidal macroalga in Livorno, Italy. The study of the soft sediment macrobenthos in Tees Bay is similar to this current study. A time series data of several sites was compared with each other to determine if they show different temporal patterns of community change (Clarke et al. 2006). The five macroinvertebrate sampling sites included by Clarke et al. (2006) in their study were along the Tees Bay coastline, results showed that sites closest together were more similar in their time series patterns compared with sites further away from each other. In the Scapa Flow sites, no such similarities between the sites were observed.

No common overall time series pattern was present at the Scapa Flow sampling stations, the trajectories of change in the community composition over time were different between the sampling stations, thus indicating that the main factors influencing the year to year patterns were specific to each sampling station rather than Scapa Flow-wide trends. The physical characteristics of the eight sampling stations discussed here were: four of the sampling stations (Creekland ST11, Kirk Hope MLWS, Lyrawa ST10 and Mill Bay ST12) had Intermediate Beach Type; three (Cumminess ST4, Longhope ST12 and Swanbister ST12) had Dissipative non-barred Beach type; and one (Congesquoy ST2) had Ultra-dissipative Beach Type (Chapter 3). The different Beach Types signify the presence of different wave climates and sediment particle sizes on the individual beaches. The similarity of the Beach Types at the sampling stations did not predict similarities in the macroinvertebrate time series trajectories, the stations with similar Beach Types were

not grouped together by the second-stage cluster dendrogram or the MDS ordination (Figure 5.3). The second-stage cluster dendrogram grouped the sampling stations into three clusters but the low Spearman rank correlation values indicated that the similarities between the sites were not significant.

The results presented here are consistent with other studies where temporal variability in macroinvertebrate communities was explained by local scale processes (Atkins & Jones 1990; Jarrin et al. 2017; Schooler et al. 2017; Bae et al. 2018). Atkins and Jones (1990) analysed 15-years of data, 1974 – 1988, from four of the Scapa Flow monitoring sites: Scapa Bay, Swanbister Bay, Waulkmill Bay and Mill Bay. By analysing the most common species over the time period and the community fluctuations at each site they concluded that the main regulatory processes were site-specific rather than regional or Orkney-wide. These site-specific processes were identified as high population variability of opportunistic species at Swanbister, Waulkmill and Mill Bay and the effluent discharge from Highland Park Distillery at Scapa Bay (Atkins & Jones 1990). On Californian oceanic beaches Schooler et al. (2017) identified local-scale processes as the main influence on the long-term macroinvertebrate community changes. They identified the decrease in suitable habitat due to the loss of washed up seaweed at the top of the shore at the beaches as one of the main reasons for the decline in species diversity in their monitoring sites (Schooler et al. 2017). The loss of washed up seaweed was not an issue at the Scapa Flow sites, most sites in the monitoring programme have a rocky shore aspect (Atkins et al. 1985) and the seaweed is most often deposited into this area.

5.5.1 Conclusions

The eight Scapa Flow sampling stations included in this analysis all had different temporal patterns in their macroinvertebrate population communities. The results indicate that the main factors influencing the year to year patterns of macroinvertebrate populations were specific to each sampling station, no Scapa Flow-wide trends were apparent.

Chapter 6 Spatio-temporal patterns in intertidal macroinvertebrate communities at three sandy beach sites on Orkney: Quoys, Congesquoy and Waulkmill

6.1 Introduction

In the previous chapter (Chapter 5) the large scale or regional spatio-temporal patterns in macroinvertebrate community composition within Scapa Flow were investigated. It was demonstrated that no Scapa Flow wide patterns were present and that patterns of macroinvertebrate community variability within Scapa Flow are site-specific. The next step is to characterise any patterns or trends at the site scale. Three sites have been selected for a detailed study: Bay of Quoys, Sands of Congesquoy and Waulkmill Bay. These are the only sites for which a data verification process was carried out, as detailed in Chapter 4.

Drivers of spatio-temporal variation in the sandy beach macroinvertebrate communities between different beaches were briefly discussed in Chapter 5 and can be summarised as being physical characteristics of the beaches, including granulometry, exposure to waves and tidal regime (Short 1996; Defeo & McLachlan 2005). Macroinvertebrates have species-specific preferences for a suitable range of particle sizes for their habitat (Brown 1983; McLachlan 1996) and therefore are restricted to certain area of the sandy beach (Brown 1983; McLachlan 1996; McLachlan & Defeo 2018). Change in sediment particle size has been shown to affect the macroinvertebrate diversity and abundance on the shoreline (McLachlan 1996). On sandy beaches the finest sediment particles are near the waterline and the coarsest sediment particles at the top of the shore. This distribution and the transport of the sediment particles are determined by their behaviour within the water column. The action of waves washing on the shore suspends small sediment particles from the seabed, the suspended load, and transports the particles either towards the top of the shore during calm periods or offshore during storm events (McLachlan & Defeo 2018). The motion of the waves on the seabed and shoreline move coarse sediment particles by shear force near the seabed, the bed load, and transports the sediment particles further up the shore (McLachlan & Defeo 2018). The finest sediment particles are suspended in the water column longer than the coarse sediment particles which are deposited at the top of the shore by the waves. A change in the wave climate at a beach can change the sediment particle distribution on the shore (Schlacher et al. 2008). This change can vary at different sites as site-specific factors can mediate the incoming wave energy (McLachlan & Defeo 2018). A storm event can erode the beach by suspending the fine sediment particles, only leaving behind the coarse sediment particles (Scott et al.

2016; Burvingt et al. 2017). Storm events were mentioned in Chapter 5 as possible disturbance events which can change the granulometry of the beach but can also alter the macroinvertebrate population and community by decreasing both diversity and abundance due to storm scouring (Engel et al. 2009). Other one-off events which can affect the macroinvertebrate communities vary from anthropogenic disturbance caused by beach cleaning (Dugan et al. 2003; Gilburn 2012) to extreme weather events including freezing winter temperatures (Beukema 1990). Several aspects of climate change are likely to affect sandy beaches, sea level rise (Schlacher et al. 2008; Le Cozannet et al. 2018; Melet et al. 2018; Orlando et al. 2019), increased seawater temperature (Melet et al. 2018; Orlando et al. 2019), increased extreme weather events (Defeo et al. 2009) and the introduction of non-native species (Brown & McLachlan 2002). Sea level rise and extreme weather events, mainly increased storminess, will lead to habitat loss and change in the sediment transport at sandy beaches (Schlacher et al. 2008; Le Cozannet et al. 2018; Melet et al. 2018; Orlando et al. 2019). Increased sea water temperature will influence the distribution of macroinvertebrates with cold water species potentially being replaced by warm or temperate water species and the establishment of non-native species from warmer areas become more likely (Brown & McLachlan 2002). Changes in macroinvertebrate communities can also be due to long-term chronic pollution caused by surface-run off, waste water treatment facilities, factory effluents or from small scale but persistent hydrocarbon pollution from refuelling of vessels, bilge pump accidents and small spillages at oil terminals (Defeo et al. 2009; McLachlan & Defeo 2018). Chronic pollution from anthropogenic sources can be difficult to determine without on-going monitoring (Jones 1980; McLachlan & Defeo 2018).

Many other factors can contribute to the spatio-temporal variability of macroinvertebrates within a beach. Natural fluctuations in macroinvertebrate populations occur between seasons (Atkins et al. 1989; Bamber 1993) and years (Dörjes et al. 1986; Bamber 1993) due to winter mortality, recruitment success and predation (Essink & Beukema 1986). Macroinvertebrates are naturally patchy within the sandy beach environment because of the effect of the swash on movement and sorting of the sediment, localised food concentrations, aggregations of species and mobility of the species due to tidal movements (McLachlan 1983; Morrisey et al. 1992; Ysebaert & Herman 2002; McLachlan & Defeo 2018). The methodology used for sampling is important in mitigating the patchiness (and therefore spatial fluctuations in the abundance) of the macroinvertebrates by ensuring that an adequate representation of the sandy beach is sampled at any one sampling event (Holme & McIntyre 1971; McLachlan 1983).

Intertidal benthic macroinvertebrates were selected for the sandy beach monitoring programme for potential oil pollution impacts in Orkney based on the best available research and knowledge at the time (Jones & Simpson 1976; Jones 1980). Benthic macroinvertebrates have been used since the 1980s in monitoring studies as indicators of environmental health (Gray & Christie 1983; Hargrave & Thiel 1983; Bilyard 1987; Warwick 1988; Warwick et al. 1990; Dauer 1993; Warwick & Clarke 1993; Kiyko & Pogrebov 1997). Anthropogenic activities on land and at sea affect the health of the aquatic environment and it is often the case that by the time a problem is visible or noticeable it might be too late to act to prevent the impact, but detection can stimulate action to reverse it. The long-term monitoring of intertidal macroinvertebrates in Scapa Flow provides a tool to assess the health of the area as long as the natural population fluctuations are accounted for and the data analyses are carried out promptly after surveys.

6.2 Aims

To assess, understand and explain the extent of spatial and temporal fluctuations in the macroinvertebrate populations against which any future variations or trends can be measured.

6.3 Methods

Sampling at the monitoring sites has been carried out since 1974. The complete data from these sampling events are not available. The Orkney Marine Biology Unit annual reports (Jones 1974; Jones & Simpson 1976, 1977; Jones et al. 1978, 1979; Jones 1980; Jones et al. 1981, 1982; Jones 1983, 1985; Jones et al. 1986-1991) detail the sandy beach monitoring from 1974 to 1990 (see Chapter 2 Table 2.1) but the data for all these sampling events are not held at the Marine Environmental Unit, Orkney Harbour Authority. The data that are available for Quoys, Congesquoy and Waulkmill are summarised in Table 6.1. For each of the sites and stations the following years are included in the Historical and Current time periods:

Quoys: three stations each with two time periods:

- Quoys ST7 Historical time period 1974-1988, Current time period 2006-2016
- Quoys ST10 Historical time period 1976-1988, Current time period 2006-2016
- Quoys ST12 Historical time period 1983-1988, Current time period 2006-2016

Congesquoy: two stations each with two time periods:

- Congesquoy ST1 Historical time period 1983-1989, Current time period 2002-2016
- Congesquoy ST2 Historical time period 1983-1989, Current time period 2002-2016

Waulkmill: two stations each with two time periods:

- Waulkmill ST10 Historical time period 1973-1988, Current time period 2002-2016
- Waulkmill ST12 Historical time period 1978-1988, Current time period 2002-2016

6.3.1 Data analysis and statistics

Data analysis was carried out as described in Chapter 2 Section 2.2.5.

Granulometry statistics were calculated and described as detailed in Chapter 2 Section 2.2.5.3.

Table 6.1. Data available for analysis at Quoys ST7, ST10 and ST12, Congesquoy ST1 and ST2, and Waulkmill ST10 and ST12 in Historical (1974 – 1988) and Current monitoring period (2006 – 2016). Details of year and month of sampling and core size. $X =$ data available.

			Congesquoy			Waulkmill					
	Year Core size Month			Station		Month	Station		Month	Station	
			ST7	ST10 ST12			ST1 ST2				ST10 ST12
1973	0.1								July	X	
1974	0.1	August	X						July	X	
1975	0.1								July	$\mathbf X$	
1976	0.1	July		X					July	X	
1977	0.02								June	X	
1978	0.02								June	X	X
1979	0.02								June	X	X
1980	0.02								June	X	X
1981	0.02	August	X	X					June	X	X
1982	0.02	September	X	X					June	X	X
1983		October			X	June	X	X			
1984	0.02	September	X	X	X	April	X	X	June	X	X
1985	0.02	September	X	X	X	April	X	X	June	X	X
1986	0.02	September	$\mathbf X$	X	X	April	X	X	June	X	$\mathbf X$
1987	0.02	September	X	X	X	April	X	X	June	X	X
1988	0.02	August	X	X	X	April	X	X	June	X	X
1989						April	X	X			
1990						April	X	X			
2002	0.02					March	X	X	March	X	X
2003	0.02					April	X	X	March	X	X
2004	0.02					March	X	X	March	X	X
2005	0.02					March	X	X	March	X	X
2006	0.02	April	X	X	X	March	X	X	March	X	X
2007	0.02	April	X	X	X	March	X	X	March	X	X
2008	0.02	April	X	X	$\mathbf x$	March	X	X	March	X	X
2009	0.02	April	X	X		March	X	X	March	X	$\mathbf X$
2010	0.02	March	X	X	X	March	X	X	February	X	$\mathbf X$
2011	0.02	March	X	X	X	April	X	X	April	X	X
2012	0.02	March	X	X	X	May	X	X	May	X	X
2013	0.02	February	X	X	X	March	X	X	March	X	$\mathbf X$
2014	0.02	March	X	X		March	X	X	March	X	X
2015	0.02	April	X	X		February	X	X	February	$\mathbf X$	X
2016	0.02	April	X	X	X	March	X	\bf{X}	March	X	$\mathbf X$

6.4 Results

6.4.1 Quoys

6.4.1.1 The physical environment

The sediment granulometry has been measured at Quoys for 11 years: 1974, 1979, 1986- 1990, 2006 and 2014-2016 (Figure 6.1).

At ST10 and ST12 the sediment type (Chapter 2 Table 2.3) has been medium sand (0.25- 0.5 mm) (Figure 6.1). In the upper shore station ST7 the sediment type changed in 2014, pre-2014 the sediment type was medium sand (0.25-0.5 mm) which changed to coarse sand (0.50-1.0 mm) from 2014 onwards (Figure 6.1). The change in the sediment grain size to coarse sand at the three stations at Quoys influenced the beach morphometric calculations: the Beach Index (Chapter 1 Section 1.3) changed from Dissipative: nonbarred to Intermediate at each sampling station (2014 at ST7, 2015 at ST10 and after 2006 at ST12, Chapter 3 Table 3.14).

Figure 6.1. Bay of Quoys mean grain size data for each sampling station ST7, ST10 and ST12 using Folk and Ward method.

6.4.1.2 Quoys ST7 macrofauna during Historical and Current time periods

Twenty-four taxa were identified both in the Historical and Current time periods at Quoys ST7 (Table 6.2). Two of the main characterising taxa (with percentage contributions to the similarity) were same in both time periods; polychaete annelids belonging to the family Spionidae (Historical 18.3%, Current 21.4%) and annelids belonging to the class Oligochaeta (Historical 19.3%, Current 37%) (Table 6.3). The third most abundant taxa were different in each time period, amphipods belonging to the family Pontoporeiidae (25.2%) in the Historical time period and polychaetes belonging to the family Capitellidae (17.0%) in the Current time period (Table 6.3). The average similarities for the Historical and Current time period were 62.1% and 65.9%, respectively.

The macroinvertebrate community at Quoys ST7 was dominated by polychaetes and amphipods with few molluscs' present in the samples (Table 6.2). Substantial annual variation for the most abundant taxa present (Spionidae and Pontoporeiidae) was observed (Figure 6.2). The populations of Capitellidae and Oligochaeta were stable, apart from 2014 when a spike in the abundance of Oligochaete $(6,060 \text{ ind. } 0.1 \text{ m}^{-2})$ was recorded (Figure 6.2).

6.4.1.3 Quoys ST10 macrofauna during Historical and Current time periods

Thirty-one taxa were identified in the Historical time period compared with 27 taxa in the Current time period (Table 6.4). The main characterising taxa (with percentage contributions to the similarity) in both time periods were the same; amphipods belonging to the families Pontoporeiidae (Historical 19.4%, Current 30.8%) and Corophiidae (Historical 19.3%, Current 10%) and polychaetes belonging to family Spionidae (Historical 18.7%, Current 21.3%) (Table 6.3). The average similarity for the Historical and Current time periods were 79.2% and 62.5%, respectively.

Although the most abundant taxa remained the same during the two time periods a decrease in the number of crustacean taxa and increase in the number of polychaete taxa from Historical to Current time period were observed (Table 6.4). Molluscs were low in both number of taxa and in abundances during the two time periods. Large year-to-year population fluctuations were observed for the most abundant taxa (Spionidae, Corophiidae and Pontoporeiidae) (Figure 6.2).

6.4.1.4 Quoys ST12 macrofauna during Historical and Current time periods

Thirty-five taxa were identified during the Historical time period compared with 25 taxa during the Current time period (Table 6.5). Two of the main characterising taxa (with percentage contributions to the similarity) were the same for both time periods: amphipods belonging to the family Pontoporeiidae (Historical 19.1%, Current 25.9%) and polychaetes belonging to family Spionidae (Historical 16.1%, Current 15.7%) (Table 6.3). The third most abundant taxa were different in each time periods: amphipods belonging to the family Corophiidae (17.8%) in the Historical time period and amphipods belonging to the family Phoxocephalidae (14.5%) in the Current time period (Table 6.3). The average similarities for the Historical and Current time periods were 74.6% and 64.1%, respectively.

The number of crustacean taxa reduced at Quoys ST12 from Historical time period to Current time period, and the number of annelid and mollusc taxa remained stable (Table 6.5). The annual total abundances were lower in the Current time period compared to the Historical time period. Minimal year-to-year population fluctuation was observed in the most abundant taxa (Spionidae, Corophiidae and Pontoporeiidae) (Figure 6.2).

6.4.1.5 Diversity at Quoys ST7, ST10 and ST12

The diversity (Shannon Diversity $(H'(log_e))$ at Quoys ST7, ST10 and ST12 was mostly \leq 1.5 and occasionally below 1.0, except for some instances (ST7 1982, 1987 and 1988; ST10 1987, 2013 and 2015; ST12 1983, 2010, 2011 and 2013) when values of up to 1.8 were recorded (Tables 6.2, 6.4 and 6.5). The low Shannon Diversity (\leq 1.5) reflects the dominance of a few taxa despite high numbers of taxa present. The results were comparable with the research by Atkins et al. (1985) on 14 sandy beaches (seven of which are not included in this study) on Orkney which all had a low diversity $($ 1.5) or very low diversity (<1.0) .

QUOYS ST7																			
				1974 1981 1982 1984 1985 1986 1987 1988 2006 2007 2008 2009 2010 2011 2012 2013 2014 2015 2016															
ANNELIDA																			
Arenicolidae				$\mathbf{1}$		$\mathbf{1}$	\overline{c}		3		$\overline{2}$	$\mathbf{1}$			$\mathbf{1}$			$\mathbf{1}$	10
Capitellidae	$\overline{2}$	$\overline{7}$	100	$\mathbf{1}$		$\mathbf{1}$	53	21	41	6	87	307	73	16	127	12	5	54	73
Cirratulidae															$\mathbf{1}$		$\mathbf{1}$		
Fabriciidae		$\mathbf{1}$	234	12	$\overline{4}$	17	20	52			$\mathbf{1}$						1	$\mathbf{1}$	
Maldanidae	48																		
Nereididae		$\overline{2}$		$\mathbf{1}$			$\mathbf{1}$	$\mathbf{1}$	1										
Oligochaeta	28	114	250	64	44	80	233	187	45	293	71	87	134	1064	503	424	5722	377	762
Opheliidae	6	112	210	593	41	34	47	40						$\overline{7}$	6	$\mathbf{1}$	36	$\overline{7}$	71
Orbiniidae										12	\overline{c}	$\overline{4}$	$\mathbf{1}$						
Paraonidae										$\overline{2}$			$\mathbf{1}$					$\mathbf{1}$	
Phyllodocidae			$\overline{4}$																
Spaerodoridae									$\overline{4}$										
Spionidae	177	458	557	142	18	35	112	90	1039	89	361	151	332	24	169	33	159	41	188
Syllidae			$\overline{4}$								$\overline{2}$		$\mathbf{1}$	$\mathbf{1}$	$\overline{\mathbf{3}}$			$\overline{4}$	
NEMERTEA	$\overline{2}$	25	20	9	6	37	140	95	1	10		9	$\overline{7}$	$\mathbf{1}$	$\overline{7}$	8		10	
PLATYHELMINTHES														92	79	26	128	28	52
CRUSTACEA																			
Caprellidae								$\mathbf{1}$											
Cirolanidae	$\overline{3}$	8	\mathfrak{S}	263	184	37	19	37	1			$\mathbf{1}$						$\overline{2}$	
Corophiidae		15		$\overline{2}$			$\mathbf{1}$	14						1					
Cumacea								$\overline{7}$	1										
Dexaminidae								$\mathbf{1}$											
Gammaridae			$\overline{4}$		\overline{c}												$\mathbf{1}$		
Hyalidae					$\overline{\mathbf{3}}$	$\mathbf{1}$	9	3			\overline{c}	9		\overline{c}	$\mathbf{1}$	\overline{c}	3		
Idoteidae								$\mathbf{1}$											
Janiridae																	$\overline{4}$	$\mathbf{1}$	
Melitidae																$\mathbf{1}$			
Phoxocephalidae								\overline{c}						$\mathbf{1}$					
Pontoporeiidae	846	589	1148	1341	399	301	517	595	29	78	9	70	$\overline{7}$	6	12	12		95	51
MOLLUSCA																			
Cardiidae								$\mathbf{1}$											
Hydrobiidae									$\mathbf{1}$	5									
Mactridae									1										$\mathbf{1}$
Margaritidae											\overline{c}								
Montacutidae											$\overline{\mathbf{3}}$								
Murchisonellidae						\overline{c}		$\mathbf{1}$		55									
Retusidae							$\mathbf{1}$			$\overline{2}$									τ
Tellinidae									1										
Taxa	8	10	11	11	9	11	13	18	13	10	11	9	8	11	11	9	10	13	9
Abundance (ind. $0.1m-2$)	1112	1331	2536	2429	701	546	1155	1149	1168	552	542	639	556	1215	909	519	6060	622	1215
Diversity $(H'(loge)))$	0.8	1.3	1.6	1.2	1.2	1.5	1.6	1.6	0.5	1.4	1.0	1.4	1.1	0.5	1.3	0.8	0.3	1.3	1.3

Table 6.2. Quoys ST7 summary abundances (ind. 0.1m^{-2}) for Historical and Current periods. The three most abundant taxa for each period are highlighted.
Table 6.3. Quoys ST7, ST10 and ST12 SIMPER results for Historical and Current periods (cut-off at 90%).

QUOYS ST7 - HISTORIC QUOYS ST7 - CURRENT

QUOYS ST10 - HISTORIC QUOYS ST10 - CURRENT

Average similarity: 79.2 Average similarity: 62.5

QUOYS ST12 - HISTORIC QUOYS ST12 - CURRENT

Average similarity: 74.6 Average similarity: 64.1

Table 6.4. Quoys ST10 summary abundances (ind. 0.1m^{-2}) for Historical and Current periods. The three most abundant taxa for each period are highlighted.

Table 6.5. Quoys ST12 summary abundances (ind. 0.1m^{-2}) for Historical and Current periods. The three most abundant taxa for each period are highlighted.

Figure 6.2. Quoys ST7, ST10 and ST12, year-to-year variation in the three most abundant taxa; Spionidae, Corophiidae and Pontoporeiidae during Historical and Current time periods. The abundances of Capitellidae and Oligochaeta and the Total Abundances of all taxa in each station. Abundances in ind. 0.1m^{-2} .

6.4.1.6 Quoys results of data analysis

When testing for differences between the three stations at Quoys, the MDS ordination and cluster dendrogram with SIMPROF test creates two groups: 1) ST7 on its own and 2) samples from ST10 and ST12 together (Figure 6.3). In the Historical time period, both ST10 and ST12 had high abundances of amphipods belonging to the families Corophiidae and Urothoidae, which were the main discriminating taxa between ST7 vs ST10 and ST12. In the Current time period the main discriminating taxa between the two groups of samples were annelids belonging to the class Oligochaeta (high abundance in ST7) and amphipods belonging to the family Pontoporeiidae (high abundance in ST10 and ST12) (Table 6.6).

To fully understand the spatio-temporal patterns at Quoys the samples from the two time periods and from each station were analysed separately.

At Quoys ST7, Historical time period, the MDS ordination and cluster dendrogram with SIMPROF test revealed samples from 1974 to be different from the rest of the years, 1981 1982, 1984 – 1988 (Figure 6.4). To perform the SIMPER test, replicates for each year were required, but for 1974 only one core sample was available and therefore SIMPER was unable to calculate the characterising taxa for this year (Table 6.7).

At Quoys ST7, Current time period, the MDS ordination and cluster dendrogram with SIMPROF test revealed two groups of samples: 1) 2008-2010, and 2) 2011-2016 (Figure 6.4). The taxa contributing to the within-group similarity for years 2008-2016 were polychaetes belonging to the families Spionidae and Capitellidae, and annelids belonging to the class Oligochaeta (Table 6.8). The differences between the two groups of years 1) 2008-2010, and 2) 2011-2016 can be explained by the discriminating taxa, which were polychaetes belonging to the family Opheliidae, annelids belonging to the class Oligochaeta and flatworms belonging to the phylum Platyhelminthes (Appendix E Section 1). No Opheliidae or Platyhelminthes were present in the group 1) 2008 – 2010 samples, the abundance of Oligochaeta were much lower in the group 1) 2008 – 2010 samples compared to the group 2) $2011 - 2016$ samples (Table 6.8).

At Quoys ST10, Historical time period, the MDS ordination and cluster dendrogram with SIMPROF test revealed two groups: 1) 1976, 1985, and 2) 1981-1984, 1986-1988, the separation of the two groups were not significant (Figure 6.5). The SIMPER analysis identified the taxa that characterised the group 1) were amphipods belonging to the families Pontoporeiidae, Corophiidae and Urothoidae (Table 6.9). Group 2) was characterised by amphipods belonging to the families Pontoporeiidae and Corophiidae

and polychaetes belonging to the family Spionidae (Table 6.9). Spionidae and Urothoidae were the taxa that were different characterising taxa between the two groups.

At Quoys ST10, Current time period, the MDS ordination and cluster dendrogram with SIMPROF test revealed two groups: 1) 2006-2010, 2016, and 2) 2011-2015 (Figure 6.5). SIMPER analysis identified the taxa that typified samples from each year (Table 6.10), as mostly amphipods belonging to the family Pontoporeiidae and Urothoidae and polychaetes belonging to family Spionidae. To explore the separation of the two groups, the taxa contributing to the most between-group dissimilarities were identified by SIMPER analysis to be the polychaetes Capitellidae and Opheliidae (Appendix E Section 2). Capitellidae were low abundance in 2006-2010, 2016 samples but at higher abundance $(6-139 \text{ ind. } 0.1 \text{ m}^{-2})$ in 2011-2015 samples. Opheliidae were absent from all samples within group 1) (Table 6.4).

At Quoys ST12, Historical time period, the MDS ordination and cluster dendrogram with SIMPROF test revealed two groups: 1) 1983, 1985-1988, and 2) 1984 (Figure 6.6). SIMPER analysis identified the taxa characterising samples from each year (Table 6.11). In 1984 samples are characterised by polychaetes belonging to the family Spionidae, amphipods belonging to the family Pontoporeiidae and gastropods belonging to the family Murchisonellidae. Group 1) were characterised by amphipods belonging to the families Pontoporeiidae and Corophiidae and polychaetes belonging to the family Spionidae, apart from 1986 when amphipods belonging to the family Urothoidae were more abundant than polychaetes belonging to the family Spionidae (Table 6.11).

At Quoys ST12, Current time period, the MDS ordination and cluster dendrogram with SIMPROF test revealed two groups: 1) 2006-2008, and 2) 2010-2013, 2016 (Figure 6.6). SIMPER analysis identified the taxa characterising samples from each year (Table 6.12). Amphipods belonging to the family Pontoporeiidae were the most abundant taxa in all of the samples. The main taxa contributing to the between group dissimilarities were molluscs belonging to the families Tellinidae and Murchisonellidae, and amphipods belonging to the family Corophiidae (Appendix E Section 3). Tellinidae were absent from group 1) 2006-2008 samples. Murchisonellidae and Corophiidae both had higher abundances in the group 1) samples compared with group 2) samples (Table 6.5).

Table 6.6. Summary of SIMPER results for Quoys ST7, ST10 and ST12 Historical and Current periods: average abundance (%) of discriminating taxa at each time period in each station, the contribution (%) of taxa to dissimilarity of the groups, and cumulative total (%) of contributions (cut-off at 70%).

Figure 6.4. Quoys ST7, Historical and Current data analysis with replicates summed. Multidimensional scaling (MDS) and cluster dendrogram **Figure 6.4**. Quoys ST7, Historical and Current data analysis with replicates summed. Multidimensional scaling (MDS) and cluster dendrogram analysis. Cluster dendrogram with SIMPROF test, solid black line = significant difference. MDS ordination with trajectory in year order. analysis. Cluster dendrogram with SIMPROF test, solid black line = significant difference. MDS ordination with trajectory in year order.

Table 6.7. Summary of SIMPER results for Quoys ST7 Historical period: average abundance (%) of characterising taxa in each year, the contribution (%) of taxa to the within-group similarity, and cumulative total (%) of contributions (cut-off at 90%).

T.

Table 6.8. Summary of SIMPER results for Quoys ST7 Current period: average abundance (%) of characterising taxa in each year, the contribution (%) of taxa to the within-group similarity, and cumulative total (%) of contributions (cut-off at 90%).

П

Table 6.9. Summary of SIMPER results for Quoys ST10 Historical period: average abundance (%) of characterising taxa in each year, the contribution (%) of taxa to the within-group similarity, and cumulative total (%) of contributions (cut-off at 90%).

COL

Table 6.10. Summary of SIMPER results for Quoys ST10 Current period: average abundance (%) of characterising taxa in each year, the contribution (%) of taxa to the within-group similarity, and cumulative total (%) of contributions (cut-off at 90%).

Table 6.11. Summary of SIMPER results for Quoys ST12 Historical period: average abundance (%) of characterising taxa in each year, the contribution (%) of taxa to the within-group similarity, and cumulative total (%) of contributions (cut-off at 90%).

П

Table 6.12. Summary of SIMPER results for Quoys ST12 Current period: average abundance (%) of characterising taxa in each year, the contribution (%) of taxa to the within-group similarity, and cumulative total (%) of contributions (cut-off at 90%).

П

6.4.2 Congesquoy

6.4.2.1 The physical environment

The sediment granulometry was measured at Congesquoy for eight years: 1983, 1986, 1988, 1989, 2006, 2014-2016 (Figure 6.7).

The sediment type (Chapter 2 Table 2.3) has changed from medium sand (0.25-0.50 mm) in 1983 and 1986 to fine sand (0.125-0.25 mm) in 1989, 2006 and 2014 and, back to medium sand (0.25-0.50 mm) from 2015 onwards. This change is consistent at both ST1 and ST2 but has not influenced the Beach Index (Chapter 1 Section 1.3) which remains Ultra-Dissipative throughout the monitoring period (Chapter 3 Table 3.9).

Figure 6.7. Congesquoy mean grain size data (Folk and Ward method).

6.4.2.2 Congesquoy ST1 macrofauna during Historical and Current time periods

Thirty-four taxa were identified in the Historical time period compared with 38 taxa in the Current time period (Table 6.13). The main characterising taxa (with percentage contributions to the similarity) were same in both time periods; polychaetes belonging to the families Syllidae (Historical 14.4%, Current 13.6%) and Spionidae (Historical 14.0%, Current 16.3%), and amphipods belonging to the family Pontoporeiidae (Historical 13.2%, Current 11.1%) (Table 6.14). The average similarities for the Historical and Current time period were 76.1% and 69%, respectively.

The macroinvertebrate community at Congesquoy ST1 is polychaete dominated (Table 6.13). The total abundance has decreased (Table 6.13) during the monitoring programme, the highest abundance was recorded in 1985 $(1,910 \text{ ind. } 0.1 \text{ m}^{-2})$ and lowest in 2007 (330 ind. 0.1m^{-2}). Inter-annual population fluctuations in the three most abundant taxa were observed (Figure 6.8) with largest annual variation observed in the abundances of Spionidae.

6.4.2.3 Congesquoy ST2 macrofauna during Historical and Current time periods

Thirty-five taxa were identified in the Historical time period compared with 46 taxa in the Current time period (Table 6.15). The main characterising taxa (with percentage contributions to the similarity) were same in both time periods; polychaetes belonging to the families Syllidae (Historical 13.7%, Current 13.9%) and Spionidae (Historical 13.3%, Current 15.7%) and amphipods belonging to the family Pontoporeiidae (Historical 12.5%, Current 11.1%) (Table 6.14). The average similarity for the Historical and Current time periods were 76.1% and 67.1%, respectively.

The macroinvertebrate community at Congesquoy ST2, like at ST1, was polychaete dominated (Table 6.15). The total abundance had a decreasing trend over the monitoring programme with the highest abundance recorded in 1988 $(1,944 \text{ ind. } 0.1 \text{ m}^{-2})$ and the lowest in 2013 (688 ind. 0.1m^{-2}) (Table 6.15). Large year-to-year fluctuations in the abundances of the three most abundant taxa were observed (Figure 6.8).

6.4.2.4 Diversity at Congesquoy ST1 and ST2

The diversity (Shannon Diversity (H'(loge)) at Congesquoy ST1 and ST2 varied between 1.3 and 2.3 throughout the monitoring period (Tables 6.13 and 6.15). The diversity at Congesquoy was consistently higher compared to Quoys and Waulkmill and when compared to the 14 sites (seven of which are not included in this study) surveyed by Atkins et al. (1985). The slightly higher diversity at Congesquoy ST1 and ST2 indicate that the stations have more even distribution of taxa and their abundances compared to other sites in Orkney.

Table 6.13. Congesquoy ST1 summary abundances (ind. 0.1m^{-2}) for Historical and Current periods. The three most abundant taxa for each period are highlighted.

CONGESQUOY ST1 - HISTORICAL CONGESQUOY ST1 - CURRENT

CONGESQUOY ST2 - HISTORICAL CONGESQUOY ST2 - CURRENT

Average similarity: 76.1 Average similarity: 67.1

Table 6.15. Congesquoy ST2 summary abundances (ind. 0.1m^{-2}) for Historical (1983 – 1989) and Current (2002 – 2016) periods.The three most abundant taxa for each period are highlighted.

Figure 6.8. Congesquoy ST1 and ST2, year-to-year variation of the three most abundant taxa; Spionidae, Syllidae and Pontoporeiidae, and of Total Abundance. Abundance is in ind. 0.1m^{-2} .

6.4.2.5 Congesquoy results of data analysis

At Congesquoy the macroinvertebrate taxa compositions of the two time periods are different from each other as demonstrated by the clustering of the data into two distinct 'Historical' and 'Current' groups in the MDS ordination and in cluster dendrogram (Figure 6.9). The main discriminating taxa between the two time periods were amphipods belonging to the families Phoxocephalidae, Tanaissuidae and Corophiidae and polychaetes belonging to the family Opheliidae in ST1, and amphipods belonging to the family Phoxocephalidae, and polychaetes belonging to the families Psammodrilidae, Opheliidae and Sphaerodoridae in ST2 (Table 6.16).

There was no grouping of samples according to their location on the beach. The two stations, ST1 and ST2, were similar in their macroinvertebrate composition (Figure 6.9).

To fully understand the spatio-temporal patterns, at Congesquoy, samples from the two time periods and from each station were analysed separately.

The MDS ordination and cluster dendrogram with SIMPROF for Congesquoy ST1, Historical time period, did not reveal any groupings (Figure 6.10). The SIMPER analysis identified the main characterising taxa for the samples as polychaetes belonging to the families Spionidae, Syllidae and Paraonidae and amphipods belonging to the family Pontoporeiidae (Table 6.18).

An MDS ordination and cluster dendrogram with SIMPROF test revealed two groups for the Congesquoy ST1 Current time period: one group of samples from 2002 and 2003 and the second group of samples from 2004-2016 (Figure 6.10). The main characterising taxa for the 2002 and 2003 samples were polychaetes belonging to the families Psammodrilidae, Spionidae and Syllidae and amphipods belonging to the family Pontoporeiidae (Table 6.19). Polychaetes belonging to the family Psammodrilidae was the main taxon contributing to the dissimilarities between the two groups of years with Psammodrilidae being absent from the 2004-2016 sub-group (Appendix E Section 4).

The MDS ordination and cluster dendrogram with SIMPROF test did not reveal any groupings for the Congesquoy ST2 Historical time period (Figure 6.11). A SIMPER test identified the main characterising taxa (Table 6.20) as polychaetes belonging to the families Spionidae and Syllidae and amphipods belonging to the families Pontoporeiidae and Tanaissuidae. These four taxa cumulatively contribute approximately 50% of the total abundance to the within-group similarity of each year (Table 6.20).

The MDS ordination and cluster dendrogram with SIMPROF test revealed three significantly different clusters in the Congesquoy ST2 Current time period: 1) 2002-2004, 2) 2005-2010 & 2012-2016, and 3) 2011 (Figure 6.11). The main characterising taxa for group one (2002-2004) were amphipods belonging to the families Pontoporeiidae and polychaetes belonging to the families Syllidae and Spionidae (Table 6.21). The second group (2005-2010 & 2012-2016) was characterised solely by polychaetes belonging to families Paraonidae, Spionidae and Syllidae (Table 6.21). The third group (2011) was characterised solely by amphipods belonging to the families Pontoporeiidae, Corophiidae and Urothoidae (Table 6.21). The main taxa contributing to the dissimilarities between group one $(2002-2004)$ and group two $(2005-2010 \& 2012-2016)$ were polychaetes belonging to the families Paraonidae, Opheliidae, Psammodrilidae, gastropods belonging to the family Murchisonellidae and amphipods belonging to the family Pontoporeiidae (Appendix E Section 5). The main taxa contributing to the dissimilarities between year 2011 and all other years was the high abundance of amphipods belonging to the families Urothoidae and Pontoporeiidae and the low abundance of polychaetes of the family Paraonidae in 2011 (Appendix E Section 5).

Table 6.16. Summary of SIMPER results for Congesquoy ST1 and ST2 Historical and Current periods: average abundance (%) of discriminating taxa in both stations at each time period, the contribution (%) of taxa to dissimilarity of the groups, and cumulative total (%) of contributions (cut-off at 60%).

Table 6.17. Summary of SIMPER results for Congesquoy ST1 and ST2 Historical and Current periods: average abundance (%) of discriminating taxa at each time period in each station, the contribution (%) of taxa to dissimilarity of the groups, and cumulative total (%) of contributions (cut-off at 60%).

Figure 6.10. Congesquoy ST1 Historical and Current data analysis with replicates summed. Multidimensional scaling (MDS) and cluster dendrogram analysis. Cluster dendrogram with SIMPROF test, solid black line = significant difference. MDS ordination with trajectory in year Figure 6.10. Congesquoy ST1 Historical and Current data analysis with replicates summed. Multidimensional scaling (MDS) and cluster dendrogram analysis. Cluster dendrogram with SIMPROF test, solid black line = significant difference. MDS ordination with trajectory in year order.

Table 6.18. Summary of SIMPER results for Congesquoy ST1 Historical period: average abundance (%) of characterising taxa in each year, the contribution (%) of taxa to the withingroup similarity, and cumulative total (%) of contributions (cut-off at 90%).

Table 6.19. Summary of SIMPER results for Congesquoy ST1 Current period: average abundance (%) of characterising taxa in each year, the contribution (%) of taxa to the withingroup similarity, and cumulative total (%) of contributions (cut-off at 50%).

Figure 6.11. Congesquoy ST2 Historical and Current data analysis with replicates summed. Multidimensional scaling (MDS) and cluster **Figure 6.11.** Congesquoy ST2 Historical and Current data analysis with replicates summed. Multidimensional scaling (MDS) and cluster dendrogram analysis. Cluster dendrogram with SIMPROF test, solid black line = significant difference. MDS ordination with trajectory dendrogram analysis. Cluster dendrogram with SIMPROF test, solid black line = significant difference. MDS ordination with trajectory in year order. in year order.

Table 6.20. Summary of SIMPER results for Congesquoy ST2 Historical period: average abundance (%) of characterising taxa in each year, the contribution (%) of taxa to the withingroup similarity, and cumulative total (%) of contributions (cut-off at 90%). Dashed lines indicate significant groupings.

Table 6.21. Summary of SIMPER results for Congesquoy ST2 Current period: average abundance (%) of characterising taxa in each year, the contribution (%) of taxa to the withingroup similarity, and cumulative total (%) of contributions (cut-off at 50%). Dashed lines indicate significant groupings.

6.4.3 Waulkmill

6.4.3.1 The physical environment

The sediment granulometry has been measured at Waulkmill for ten years, 1974, 1986- 1990, 2006, 2014-2016 (Figure 6.12).

The sediment type (Chapter 2 Table 2.3) mainly fell into the fine sand category up to 2014, but mean grain size has increased in the most recent years (particularly at ST12), for which the sediment would be classified as medium sand. The change in the sediment type has not influenced the Beach Type (Chapter 1 Section 1.3) which has remained as Dissipative: non-barred throughout the monitoring period (Chapter 3 Table 3.6).

Figure 6.12. Waulkmill mean grain size data (Folk and Ward method).

6.4.3.2 Waulkmill ST10 macrofauna during Historical and Current time periods

Twenty-three taxa were identified in the Historical time period compared with 30 taxa in the Current time period (Table 6.22). One of the main characterising taxa (with percentage contributions to the similarity) was same in both time periods, the polychaetes belonging to the family Opheliidae (Historical 28.1%, Current 17%) (Table 6.23). The two other most abundant taxa were different, in Historical time period they were amphipods belonging to the family Pontoporeiidae (37.8%) and polychaetes belonging to the family Capitellidae (8.7%) and in Current time period they were polychaetes belonging to the families Enchytraeidae (19.2%) and Spionidae (17.9%) (Table 6.23). The average similarities for the Historical and Current time period were 41.7% and 71.6%, respectively.

Waulkmill ST10 macroinvertebrate community was polychaete dominated with few amphipod and mollusc taxa present (Table 6.22). For the most abundant taxa at Waulkmill ST10, large inter-annual population fluctuations were observed for Opheliidae and Spionidae in both time periods and for Enchytraeidae in Current time period (Figure 6.13).

6.4.3.3 Waulkmill ST12 macrofauna during Historical and Current time periods

Seventeen taxa were identified in the Historical time period compared with 23 taxa in the Current time period (Table 6.24). Two of the main characterising taxa (with percentage contributions to the similarity) were same in both time periods; amphipods belonging to the family Pontoporeiidae (Historical 42.5%, Current 18.2%) and polychaetes belonging to the family Paraonidae (Historical 11.3%, Current 21.9%) (Table 6.23). The third most abundant taxa in each time period were different; polychaetes belonging to the family Opheliidae (22.2%) in the Historical time period and polychaetes belonging to the family Spionidae (16.3%) in the Current time period (Table 6.23). The average similarities for the Historical and Current time periods were 53.7% and 71.0%, respectively.

The Waulkmill ST12 macroinvertebrate community was dominated by polychaetes with few crustacean and mollusc taxa (Table 6.24). The number of taxa and the total abundance have varied greatly over the monitoring period (Table 6.24, Figure 6.13). The variability during Historical time period can be assigned to the data deficiencies, in the Current time period the overall trend is of decreasing numbers of taxa but constant total abundance; 198 ind. 0.1m^{-2} (2009) to 467 ind. 0.1m^{-2} (2004). Year-to-year fluctuations in the abundances of Paraonidae and Opheliidae were observed with discernible variation in Pontoporeiidae and Spionidae and small fluctuations in the total abundance (Figure 6.13).

6.4.3.4 Diversity at Waulkmill ST10 and ST12

The diversity (Shannon Diversity (H'(loge)) at Waulkmill ST10 and ST12 during Historical time period was much lower (0.2-1.5) compared to the Current time period (1.1-2.1) (Tables 6.22 and 6.24). The average diversity value for Current time period was 1.6 which was slightly higher compared to Quoys ST7, ST10 and ST12. The low diversity at Waulkmill is comparable with the research by Atkins et al. (1985) on 14 sandy beaches on Orkney (seven of which are not included in this study) which all had a low diversity (≤ 1.5) or very low diversity (≤ 1.0) .

Table 6.22. Waulkmill ST10 summary abundances (ind. 0.1m⁻²) for Historical (1973 - 1988) and Current (2002 - 2016) periods. The three most abundant taxa **Table 6.22.** Waulkmill ST10 summary abundances (ind. 0.1m-2) for Historical (1973 – 1988) and Current (2002 – 2016) periods. The three most abundant taxa for each period are highlighted. for each period are highlighted.

Table 6.23. Summary of SIMPER results for Waulkmill ST10 and ST12 Historical and Current periods: average abundance (%) of characterising taxa in both stations at each time period, the contribution (%) of taxa to the within-group similarity, and cumulative total (%) of contributions (cut-off at 90%).

Taxa Contrib% Cum.% Taxa Contrib% Cum.% Pontoporeiidae 39.2 39.2 Opheliidae 29.6 68.8 Capitellidae 7.4 76.2 Spionidae 6.8 83.0 Tellinidae 4.0 87.0 Pontoporeiidae 8.7 76.2 Cirolanidae 2.8 89.8 Average similarity: 41.5

WAULKMILL ST10 - HISTORICAL WAULKMILL ST10 - CURRENT

WAULKMILL ST12 - HISTORICAL WAULKMILL ST12 - CURRENT

Phyllodocidae 2.2 91.9

Average similarity: 53.7 Average similarity: 71.0

Table 6.24. Waulkmill ST12 summary abundances (ind. 0.1m⁻²) for Historical (1973 - 1988) and Current (2002 - 2016) periods. The three most abundant **Table 6.24.** Waulkmill ST12 summary abundances (ind. 0.1m-2) for Historical (1973 – 1988) and Current (2002 – 2016) periods. The three most abundant taxa for each period are highlighted. taxa for each period are highlighted.

Figure 6.13. Waulkmill ST10 and ST12, year-to-year variation in the three most abundant taxa at each station; Capitellidae, Enchytraeidae, Paraonidae, Spionidae, Opheliidae and Pontoporeiidae, and year-to-year variation in Total Abundance. Abundances ind. 0.1m^{-2} .

6.4.3.5 Waulkmill results of data analysis

The MDS ordination and cluster dendrogram with SIMPROF test revealed groupings of the samples according to the time periods, Historical and Current, and according to the level of sampling station on the beach, ST10 and ST12 (Figure 6.14). The main discriminating taxa between the two time periods at ST10 were oligochaetes belonging to the family Enchytraeidae and polychaetes belonging to the families Spionidae and Paraonidae all of which were in low abundance in Historical time period (Table 6.25). The discriminating taxa at ST12 were polychaetes belonging to the families Paraonidae, Spionidae which were in low abundance at Historical time period and amphipods belong to the family Pontoporeiidae which were in high abundance in Historical time period (Table 6.25).

The two stations, ST10 and ST12, have different macroinvertebrate community compositions (Figure 6.14). Dissimilarities between the stations were analysed by a SIMPER test which highlighted amphipods belonging to the family Pontoporeiidae and polychaetes belonging to the families Spionidae, Paraonidae and Opheliidae as the main discriminating taxa for the Historical ST10 and ST12 samples (Table 6.26). The main discriminating taxa between the two stations in the Current time period were oligochaetes belonging to the family Enchytraeidae and polychaetes belonging to the families Opheliidae, Capitellidae and Spionidae (Table 6.26).

To fully understand the spatio-temporal patterns at Waulkmill the samples from the two time periods and from each station are analysed separately.

The MDS ordination and cluster dendrogram with SIMPROF test revealed six significant groups of samples for Waulkmill ST10 Historical period (Figure 6.15):

- 1. 1974
- 2. 1973, 1975, 1976, 1977
- 3. 1978 1981
- 4. 1982, 1984, 1985
- 5. 1986, 1987
- 6. 1988

The SIMPER analysis identified the main characterising taxa for the samples (Table 6.27). For group 1) these were amphipods belonging to the family Pontoporeiidae and polychaetes belonging to the family Opheliidae. Group 2) was characterised by the presence of amphipods belonging to the family Pontoporeiidae and polychaetes belonging to the families Opheliidae and Paraonidae, presence of Paraonidae being the difference between the other groups. Group 3) was characterised by the presence of amphipods

belonging to the families Pontoporeiidae and Corophiidae and polychaetes belonging to the family Opheliidae, the presence of high abundance of Corophiidae being the difference between the other groups. Group 4) and year 1982 signifies the first group with polychaetes as characterising taxa; the polychaetes belonging to the families Spionidae, Capitellidae, and Phyllodocidae. Group 5) was characterised by the presence of polychaetes belonging to the families Spionidae, Opheliidae and Capitellidae, and amphipods belonging to the family Pontoporeiidae. Group 6) was characterised by polychaetes belonging to the family Capitellidae and amphipods belonging to the families Crangonidae and Pontoporeiidae.

The MDS ordination and cluster dendrogram with SIMPROF test revealed two groups of samples and one outlier year for Waulkmill ST10 Current period (Figure 6.15). These groupings are: 1) 2002-2004, 2) 2006-2016 and year 2015 as an outlier. The SIMPER analysis identified the main characterising taxa for the samples (Table 6.28). For years 2002-2004 the main characterising taxa were oligochaetes belonging to the family Enchytraeidae and polychaetes belonging to the families Capitellidae, Paraonidae, Spionidae and Opheliidae. Enchytraeidae were absent or in very low numbers in Historical period and is therefore one of the discriminating taxa between the two time periods. The year 2005 was characterised by the presence of polychaetes belonging to the families Opheliidae and Spionidae and oligochaetes belonging to the family Enchytraeidae, both Capitellidae and Paraonidae were in very low abundances in 2005 (1 and 24 ind. 0.1m⁻², respectively). The discriminating taxa for 2005 samples compared to all other year's samples were the absence of molluscs belonging to the family Tellinidae and the low abundance of oligochaetes belonging to the family Enchytraeidae (Appendix E Section 6). The main characterising taxa for the samples from years 2006-2016 were oligochaetes belonging to the family Enchytraeidae and polychaetes belonging to the families Opheliidae and Spionidae, both Capitellidae and Paraonidae, which were characterising taxa in 2002-2004, were in low abundances.

The MDS ordination and cluster dendrogram with SIMPROF test revealed two groups of samples for Waulkmill ST12 Historical period (Figure 6.16). The samples from years 1978-1981 form one group and the samples from years 1982, 1984-1988 form a second group. The SIMPER analysis identified the main characterising taxa for the samples (Table 6.29). The years 1978-1981 are characterised by the presence of amphipods belonging to the family Pontoporeiidae, polychaetes belonging to the family Opheliidae and molluscs belonging to the family Tellinidae. The second group, years 1982, 1984-

1988, were characterised by the presence of amphipods belonging to the family Pontoporeiidae and polychaetes belonging to the families Spionidae and Paraonidae, showing a change in the dominating polychaete assemblage from Opheliidae to Spionidae and Paraonidae, and low abundances of Tellinidae bivalves.

No significant groups were present in the Waulkmill ST12 Current period data (Figure 6.16). The SIMPER analysis identified the main characterising taxa for Waulkmill ST12 Current period (Table 6.30), as polychaetes belonging to the families Spionidae, Opheliidae and Paraonidae, amphipods belonging to the family Pontoporeiidae and ribbon worms belonging to the phylum Nemertea. Tellinidae were present in higher abundances compared to Historical time period but low in comparison to other taxa in Current time period.

Table 6.25. Summary of SIMPER results for Waulkmill ST10 and ST12 Historical and Current periods: average abundance (%) of discriminating taxa in both stations at each time period, the contribution (%) of taxa to dissimilarity of the groups, and cumulative total (%) of contributions (cut-off at 70%).

Table 6.26. Summary of SIMPER results for Waulkmill ST10 and ST12 Historical and Current periods: average abundance (%) of discriminating taxa at each time period between the two stations, the contribution (%) of taxa to dissimilarity of the groups, and cumulative total (%) of contributions (cut-off at 70%).

Table 6.27. Summary of SIMPER results for Waulkmill ST10 Historical period: average abundance (%) of characterising taxa in each year, the contribution (%) of taxa to the withingroup similarity, and cumulative total (%) of contributions (cut-off at 90%).

Table 6.28. Summary of SIMPER results for Waulkmill ST10 Current period: average abundance (%) of characterising taxa in each year, the contribution (%) of taxa to the withingroup similarity, and cumulative total (%) of contributions (cut-off at 80%).

Table 6.29. Summary of SIMPER results for Waulkmill ST12 Historical period: average abundance (%) of characterising taxa in each year, the contribution (%) of taxa to the withingroup similarity, and cumulative total (%) of contributions (cut-off at 90%).

Table 6.30. Summary of SIMPER results for Waulkmill ST12 Current period: average abundance (%) of characterising taxa in each year, the contribution (%) of taxa to the withingroup similarity, and cumulative total (%) of contributions (cut-off at 80%).

6.5 Discussion

Macroinvertebrate populations are naturally patchy (McLachlan 1983; Morrisey et al. 1992; Ysebaert & Herman 2002; McLachlan & Defeo 2018) and their populations have been shown to fluctuate both seasonally and annually (Warwick & Clarke 1993; Atkins et al. 1989; Ysebaert & Herman 2002). In this study the statistically significant (SIMPROF tests) variability in the macroinvertebrate communities can be characterised by either large fluctuations in the macroinvertebrate population abundances or by a change in the taxa present in the macroinvertebrate populations.

Differences in the abundances of the taxa resulted in statistically significant separation of the time periods and stations at each site. The two time periods were different from each other in all sampling stations. The population fluctuations between the two time periods could be attributed to natural fluctuation related to population dynamics, patchiness of the populations within the intertidal zone, or be due to sampling methods used. The sampling in the Historical time period was carried out at the end of the summer compared with the Current time period when sampling was carried out during winter months. This change in the season of sampling would influence the macroinvertebrate communities present (Atkins et al. 1989). Even within the separate monitoring periods sampling was carried out during several months: August-October in Historical time period and February-April in the Current time period, further increasing the likelihood of sampling different phase of the macroinvertebrate population. At Quoys ST10 the main characterising taxa did not change from Historical to Current time period, the four most abundant taxa in both periods were the same: Pontoporeiidae, Spionidae, Corophiidae and Urothoidae only the abundances changed significantly. Similar circumstances were shown at Congesquoy ST1 and ST2, the most abundant taxa present remained the same; Syllidae, Spionidae and Pontoporeiidae for both ST1 and ST2 but their abundances fluctuated resulting in statistically significant separation. At Waulkmill ST12 the four most abundant taxa remained the same (Pontoporeiidae, Opheliidae, Paraonidae and Spionidae) with the fluctuating abundances of these taxa resulting significant separation of the two time periods. At Quoys ST12 a change in one of the most abundant taxa from Corophiidae in Historical time period to Phoxocephalidae in Current time period was due to the decreased abundance of Corophiidae in the Current time period, both taxa were present in Historical and Current time periods and change was due to population fluctuations. At Waulkmill ST10 only two of the most abundant taxa (Opheliidae and Pontoporeiidae) were recorded in every year from 1973-1988 and 2002-2016 whereas Capitellidae, Spionidae and Enchytraeidae were recorded from 1982 onwards only. The

abrupt start of the recording of Capitellidae, Spionidae and Enchytraeidae alongside with two other polychaetes (Phyllodocidae and Nemertea) imply that the lack of recordings of these taxa in pre-1982 samples was due to data deficiencies rather than the taxa not being present at this sampling station, data deficiencies at Waulkmill will be discussed further later.

The population abundances fluctuated greatly at the different time periods and stations. These groups or clusters of samples identified were statistically significantly different from each other, but the differences were principally due to large fluctuations in the abundances of one or more taxa in the macroinvertebrate assemblages rather than wholesale changes in the taxonomic composition. In their study on intertidal and subtidal benthic communities at Tagua estuary in Portugal Chainho et al. (2010) demonstrated how the fluctuations in the dominant taxa resulted in significant separations rather than the differences in the taxonomic composition. The abundances of a single taxon, Spionidae, at Congesquoy ST1 varied from 46 ind. 0.1m⁻² in 2003 to 449 ind. 0.1m⁻ 2 in 2004, but the baseline community remained the same. Atkins et al. (1989) described the seasonal and annual fluctuations of macroinvertebrate populations at Waulkmill and Scapa, the populations of amphipod *Bathyporeia sarsi* (family Pontoporeiidae), polychaetes *Spio martinensis* and *Malacoceros fuliginosus* (family Spionidae) and *Capitella capitata* (family Capitellidae) experienced great fluctuations in their population's densities both seasonally and annually. Atkins et al. (1989) illustrated the seasonal fluctuation patterns of the above-mentioned species and were able to show how the densities of the species at the two sites varied from year to year, further demonstrating how unpredictable and variable the population densities can be. Ysebaert & Herman (2002) reported similar variability in populations of *B. sarsi* and *P. elegans* in Schelde estuary in Netherlands. The annual variability of Pontoporeiidae and Spionidae were observed in all three sites in this study. The largest population fluctuations for Spionidae and Pontoporeiidae were observed at Congesquoy ST1 and ST2.

At Quoys ST7 a change in one of the abundant taxa from Pontoporeiidae in Historical time period to Capitellidae in Current time period indicates potentially a significant change in the macroinvertebrate community. Amphipods Pontoporeiidae are common sandy beach taxa (McLachlan & Defeo 2018) and are classed as species sensitive to organic pollution (Borja et al. 2000), whereas Capitellidae are an organic pollution indicator species (Read 1987; Pocklington & Wells 1992; Borja et al. 2000; Ferrando & Méndez 2011). This change in taxa could be an indication of change in the environmental conditions on the shore line and will be discussed in detail in Chapter 7. At Quoys ST10 Historical time period a statistically significant division created two groups of years: 1) 1972, 1985 and 2) 1981-1984, 1986-1988. The characterising taxa for both groups were the same (Pontoporeiidae, Spionidae and Corophiidae), but the absence of three taxa (Fabriciidae, Nemertea and Ampeliscidae) in 1972 and 1985 separates these two years from all the others. The absence of the three taxa, Fabriciidae, Nemertea and Ampeliscidae, in 1972 and 1985 could be due to poor recruitment in these two years. The abundances of Fabriciidae (1-20 ind. 0.1m^{-2}), Nemertea (1-22 ind. 0.1m^{-2}) and Ampeliscidae $(3-12 \text{ ind. } 0.1 \text{ m}^{-2})$ were low in the other years (Table 6.4), poor postsampling sample processing could have also been a contributing factor to the absence of the three taxa in 1972 and 1985. At Quoys ST12 the macroinvertebrate communities in year 1984 were significantly different from macroinvertebrate communities in all other years (1983, 1985-1988). This separation was attributable to a high abundance of marine snail, Murchisonellidae, in 1984. The high abundance of Murchisonellidae in 1984 could be a chance event of a random settlement of the taxon at that station. The other characterising taxa (Pontoporeiidae, Spionidae, Corophiidae and Urothoidae) remained the same during Historical time period. During the Historical time period for both ST1 and ST2 at Congesquoy there were no statistically significant changes in the macroinvertebrate communities. The community composition remained stable over time with natural variability of different taxa from year to year.

At Waulkmill ST10 and ST12 in Historical time period several significant separations of the years were observed. These clusters were due to data deficiencies leading to statistically different groups. The full extent of the Waulkmill Historical time period data deficiencies were not known when the site was selected for the analysis. Waulkmill was one of the seven sites for which samples were collected annually from 1974 onwards (Chapter 2 Table 2.1), however once the Historical data were located from the Orkney Islands Council Harbour Authority (OICHA) archives and digitised it became clear that not all of the macroinvertebrate data was held at OICHA. During the Historical time period the sample sorting was carried out at OICHA after which all polychaetes were sent to Dundee University for identification and enumeration. Amphipods and molluscs were identified and enumerated locally, and it was these data that were in the archives, no polychaete data for Waulkmill were held at OICHA. Once the data deficiencies were understood the decision was made to include Waulkmill Historical data in the data analysis. The Current time period at Waulkmill ST10 were separated into two groups: 1) 2002-2004 and 2) 2006-2016 and an outlier year, 2005. All have the same macroinvertebrate communities characterised by Enchytraeidae, Spionidae, Opheliidae and Capitellidae with the year to year fluctuations in the abundances of the taxa separating them into groups. The Current period at Waulkmill ST12 does not have any significant groupings indicating that there have been no changes to the macroinvertebrate community during the 15 years of current monitoring, substantiated by the high average similarity value (71%).

At Congesquoy ST1 Current time period there were statistically significant changes which divided the monitoring years into two groups: 1) 2002, 2003 and 2) 2004-2016. The separation of these two groups is driven by the high abundance of Psammodrilidae in 2002 and 2003. When examining the main characterising taxa (Spionidae, Syllidae and Pontoporeiidae) for the Current time period there were no change in these taxa and the significant groupings could be due to combination of factors: natural fluctuation in the populations, sampling issues at the start of the monitoring programme or to inconsistencies in laboratory processes. At ST1 there was a change from the Historical to the Current time period. The macroinvertebrate community has changed in that three of the amphipod taxa that had high abundances in the Historical time period (Corophiidae, Phoxocephalidae and Tanaissuidae) have either low abundances or were absent in the Current time period, and two polychaete taxa which were rare in the Historical time period (Opheliidae and Psammodrilidae) had higher densities in the Current time period. These changes represent population fluctuations in the abundances of the taxa contributing to the ST1 macroinvertebrate community. Over all at Congesquoy the macroinvertebrate communities have remained the same during the Historical and the Current time periods at both stations.

After year 2006, the sediment type changed at all three sites, Quoys, Congesquoy and Waulkmill: from medium sand to coarse sand at Quoys, and from fine sand to medium sand at Congesquoy and Waulkmill (Figures 6.1, 6.7 and 6.12). Change in the sediment type between Historical and Current time periods was likely to be partly associated with the time of year the samples were collected. Samples from 1973-1990 were collected during the summer or late summer compared to the samples from 2014 onwards which were all collected in the winter or early spring. Sandy beaches are dynamic environments and the sedimentation patterns on the shores are driven by strong winds and storm events in the winter (Schlacher et al. 2008), with associated increased wave climate which back washes the sediment to offshore, and calm summer months when the fine sediment particles, which are suspended in the water column, are deposited back to the shoreline

(Fox & Davis 1978; Masselink & Pattiaratchi 2001). The change in sediment type altered the Beach Type for Quoys, from Dissipative: non-barred to Intermediate Beach Type (Chapter 3 Table 3.9). No changes in the Beach Type were observed at Congesquoy or Waulkmill (Chapter 3 Tables 3.6 and 3.9). The change in the sediment type from 2014 onwards at Congesquoy and Waulkmill was not linked with significant changes in the macroinvertebrate communities as explored by multivariate analyses, the changes in the multivariate analyses were shown to be at years different to the changes in the sediment grain size. At Congesquoy and Waulkmill the changes in the sediment type has not significantly affected the macroinvertebrate community which are adapted to the dynamic environment of sandy beaches. Quoys, Congesquoy and Waulkmill, are within the sheltered waterbody of Scapa Flow (Chapter 3 Figure 3.1) but have different site-specific conditions; Congesquoy is very sheltered within Bay of Ireland, Waulkmill is on the northern shore of Scapa Flow and open to the south/south-easterly direction, Quoys is on the north-western shores of Scapa Flow and has the fast-flowing waters of Burra Sound running past. Quoys is the only site out of the three that has seen its sediment composition change significantly between 2006 and 2014. Prevailing wind direction during 2006- 2014 was south-east for five (2008, 2009, 2010, 2011 and 2014) of the eight years (Appendix A) with a majority of storm force winds from west (Appendix B). In 2009 and 2010 the direction of storm events was from south-west which could have resulted in change in the sediment composition at Quoys ST7. Lack of granulometry data from Quoys for 2007-2013 makes pinpointing the exact time of change impossible. The change in the macroinvertebrate community occurred in 2011, it is therefore possible that the change in sediment grain size happened prior to the sample collection in March 2011. Sediment grain size is a determining factor for macroinvertebrate communities, sheltered beaches with fine sediments being higher in macroinvertebrate biomass compared to exposed beaches with coarse and mobile sediment (Ricciardi & Bourget 1999). Atkins et al. (1985) describe the Quoys site as being unusual due to the combination of relatively coarse sand and extreme shelter, the coarse nature of the sediment at Quoys ST7 might not be of uncharacteristic of the site.

Three stations (Quoys ST7 and ST10; Congesquoy ST2) out of the seven stations analysed experienced a significant change in their macroinvertebrate communities in 2011, with one station (Quoys ST12) experiencing a significant change a year earlier in 2010. At Quoys ST12 the years 2006-2008 were different from the later years (2010- 2016), several taxa either decreased / increased their abundance from one group to the other or were completely absent in a group: Opheliidae and Tellinidae were not present in the group 1); abundances of Urothoidae decreased in group 2); and Capitellidae and Paraonidae increased in group 2). The changes from group 1) to group 2) remained and could therefore be interpreted as a shift in the macroinvertebrate community composition. The presence of the bivalve mollusc Tellinidae in the Current period from 2011 onwards could represent changes in the intertidal environment or alternatively be due to the change in sampling team. Tellinidae are a common sandy beach fauna and are cosmopolitan in their distribution (McLachlan & Defeo 2018), they were common in other Scapa Flow sandy beach sites during the Current time period: Scapa Bay ST12 $(9-63 \text{ ind. } 0.1 \text{ m}^2)$, Swanbister (15-252 ind. 0.1m^{-2}), Creekland (8-22 ind. 0.1m^{-2}), Longhope (33-97 ind. 0.1m⁻²), Lyrawa (1-86 ind. 0.1m⁻²), Mill Bay ST12 (6.5-42 ind. 0.1m⁻²) and Kirkhope MLWS $(12-56 \text{ ind. } 0.1 \text{m}^{-2})$ (J. Kakkonen pers. obs.). Favourable conditions on the shore at Quoys ST12 and the high number of Tellinidae in other areas of Scapa Flow might have enabled the bivalves to populate the lower shore area at Quoys successfully. At Quoys ST7 the change in the macroinvertebrate community from 2011 onwards was driven by the introduction of a new taxa, Platyhelminthes, which had not been recorded in any year before 2011 but was recorded every year from then on in abundances between 26 ind. 0.1m^{-2} (2013) to 128 ind. 0.1m^{-2} (2014) (Table 6.2). Platyhelminthes are marine flat worms and they are an important part of the interstitial fauna on sandy beaches (McLachlan & Defeo 2018). Platyhelminthes were recorded in other OICHA sandy beach sites (Creekland, Dead Sands, Longhope, Lyrawa, Mill Bay) in both Historical and Current time periods and it is possible that the taxa were overlooked at Quoys ST7 in previous years. Another taxon at Quoys ST7 with a marked difference from 2011 onwards was the class Oligochaeta, their abundances increased from mean abundance of 126 ind. 0.1m⁻² in 2002-2010 to mean abundance of 1475 ind. 0.1m⁻² in 2011-2016 indicating a substantial change in the abundance. Oligochaetes are a classic pollution and disturbance indicator (Read 1987; Pocklington & Wells 1992; Ferrando & Méndez 2011) and they are common in transitional waters (McLusky & Elliott 2007). High abundance of Oligochaeta has been known to be a response to a pollution or disturbance event (Ferrando & Méndez 2011). The increased abundance of Oligochaeta at Quoys ST7 could be an indication of increased disturbance at that level of the shoreline. At Quoys ST10 there were two statistically different groups: 1) 2006-2010, 2016 and 2) 2011-2015. The main differences between these two groups can be characterised by the high abundance of Pontoporeiidae and Urothoidae, and absence of Opheliidae in the group 1) and the high abundance of Capitellidae and presence of Opheliidae in the group 2). There was a shift in the macroinvertebrate community at ST10 in 2011 which continued until 2015. From 2011 onwards Capitellidae and Opheliidae were recorded in high numbers and the abundance of other species present were greatly reduced (Table 6.4). The shift in the macroinvertebrate community was reversed in 2016 when the community returned to the same composition as before. Capitellidae is an organic pollution indicator (Read 1987; Pocklington & Wells 1992; Ferrando & Méndez 2011) and its high abundance from 2011 coincides with the high abundance of another pollution and disturbance indicator taxa, Oligochaeta, at both Quoys ST7 and ST10. At Congesquoy ST2 two statistically different groups and an outlier year were revealed: group 1) 2002-2004 were separated from group 2) 2005-2010, 2012-2016 with year 2011 as an outlier (Figure 6.11). Although groups 1) and 2) were significantly separated from each other, community composition was the same and significant changes were due to the abundances of the taxa present not changes in the taxa. The outlier year, 2011, stands out as it has a high diversity of crustaceans (11 taxa) with several of them having higher abundance than in the years before or after (Table 6.15).

The significant changes in the macroinvertebrate communities from 2011 onwards could be due to a change in the sample collection, sorting and sample identification process. 2011 was first year when the sampling and sample processing was carried out by the Biologist and Technician without the Scientific Officer (Chapter 2 Table 2.4), which could have influenced the process. Due to change in personnel the sampling at Quoys ST7, Quoys ST10 and Congesquoy ST2 from 2011 could have been carried out at a slightly different location compared to previous years and therefore caused an erroneous change in the macroinvertebrate population. During all OICHA sandy beach surveys photographs were taken at each sampling station every year. By comparing site photos from before and after 2011 it was possible to ascertain that the sampling locations at Quoys ST7 and ST10, and Congesquoy ST2 remained within the same area (Appendix G Sections 1, 2 and 3), eliminating change in the sampling location as influencing the change in the macroinvertebrate community. However, at Congesquoy it is noticeable that the photo of the sampling in 2011 at ST2 was taken while the sea was still covering the sand. It is not possible to say for certain if the sampling was carried out while water was over the sand, no field notes were taken that year, but it is likely. The sample processing and identification could still be a possible source of variability as discussed in Chapter 4, but as the personnel processing the samples were the same before and after (Chapter 2 Table 2.4) it is unlikely cause for the change in the community composition.

Quoys ST7 is located directly below a small unnamed burn and in close proximity of a larger burn, Whaness Burn (Chapter 3 Figure 3.19), increased freshwater input and nutrients from surrounding fields could be a contributing factor in the increased Oligochaeta and Capitellidae at Quoys ST7 and ST10 from 2011 onwards. Both taxa are known organic pollution indicators and could have had increased population abundances after a heavy rainfall. The rainfall prior to sandy beach sampling in March 2011 (Appendix F Section 1) was within the 30-year average of <100 mm (MET Office 2019), apart from a peak in September 2010 when high rainfall (>300 mm) was recorded. A caution in interpretation of cause and effect should be taken as to fully understand the drivers of this change, further measurements of environmental parameters are required.

Reiss et al. (2006) demonstrated how extreme cold weather of 1995/1996 changed the near shore benthic invertebrate communities at Dogger Bank, southern North Sea significantly compared to offshore benthic communities in the same area. Cold temperature effects on macroinvertebrates have been studied in Wadden Sea tidal flats (Beukema 1990) and in southern North Sea (Neumann et al. 2009; Kröncke et al. 2013) all reporting changes in the macroinvertebrate communities directly after extreme cold weather. The changes in the macroinvertebrate community occurred at Quoys ST12 in 2010, at Quoys ST7 and ST10 in 2011; at Congesquoy ST2 2011 was an outlier. The winters of 2009/2010 and 2010/2011 were both exceptionally cold in Scotland (Prior & Kendon 2011a, 2011b). The cold spells during 22-23 December 2009 and 6-8 January 2010 (Prior & Kendon 2011b) were during neap tides with the lowest tidal height of 0.5m in Scapa Flow (Appendix F Section 4), these low tides were low enough to expose Quoys ST7 which is at a height of +1.7m, but not ST10 or ST12 which were at a height of 0.2m and -0.4m respectively (Chapter 3 Figure 3.20). The period of cold weather in late 2010s (24/11-09/12/2010 and 16-26/12/2010), was named 'The Big Freeze', during which temperatures of -23.3°C were recorded in the Scottish Highlands (Prior & Kendon 2011a). During 24/11-09/12/2010 low tides of 0.5m were experienced in Scapa Flow (Appendix F Section 4), low enough to expose Quoys ST7 but not ST10 or ST12. During the second spell of cold weather the tide was lower at 0.2m, which would have exposed Quoys ST7 and potentially Quoys ST10 depending on atmospheric pressure and wind conditions, but not ST12. Congesquoy ST2 is at a height of -0.15m (Chapter 3 Figure 3.12) and lower than the tides on both of the cold periods in 2011. It is therefore possible that the macroinvertebrate changes seen at Quoys ST7 and ST10 are due to being exposed to cold atmospheric temperatures and lying snow cover. The cold air temperatures in December 2010 (Appendix F Section 2) did not have an immediate impact on the seawater temperatures at Scapa Flow (Appendix F Section 3). The seawater temperatures at Scapa Pier in December to January 2009-2011 were low (<9°C), but no change from the normal range was recorded in 2010 after 'The Big Freeze' (Appendix F Figure 6). The increased number of amphipods and reduced number of polychaetes at Congesquoy ST2 in 2011 could potentially be due to the amphipods' ability to withstand freezing temperatures. Davenport (1979) demonstrated that Gammaridea amphipods could withstand temperatures of -10^oC in intertidal pools in Norway. The amphipods at Congesquoy might have had a better chance of survival in the cold weather during winter 2010/2011 compared to the polychaetes. Polychaetes, Nephtyidae and Cirratulidae, have been reported to have a poor tolerance of low temperatures (George 1968; Beukema et al. 2000) and a study in Wadden Sea tidal flats found ten out of a total of twenty-eight macroinvertebrate species to be sensitive to cold winters (Beukema 1990), they also reported lower macroinvertebrate abundances and diversity after a severe winter. This however was not the case at Quoys where the main difference in 2011 was increased abundance of Oligochaetes and Platyhelminthes. Oligochaetes are opportunistic taxa and could have responded to the cold weather as environmental change. Oligochaetes were in high abundances at Quoys ST7 also in 2014 when abundance of 5722 ind. 0.1m^{-2} were recorded. From 2014 onwards photographs of $1m²$ quadrats were added to the OICHA survey methods and these can assist in understanding the annual variability of the shoreline (Appendix G Section 4). At Quoys ST7 in 2014 the shoreline was covered in algal debris compared to 2015 when only clean sand was present, these changes in the shoreline will affect the macroinvertebrate communities and could have contributed to the high Oligochaete abundance in 2014. Similar conditions could have been present in 2011 but no photos of quadrats were taken.

6.5.1 Conclusions

A significant change in the macroinvertebrate communities has occurred at one out of the three sites highlighting the need for multiple monitoring sites to enable the successful ongoing monitoring of large waterbodies, like Scapa Flow. The analysis highlighted how the macroinvertebrate communities have remained stable at the sites during the Historical time period and how a long gap in the monitoring programme caused issues in the longtime series analyses. Extreme cold weather and change in granulometry were associated with changes in the macroinvertebrate communities, no anthropogenic influences were shown to have influenced the macroinvertebrate communities at the three sites studied.

Chapter 7 Establishing the macroinvertebrate baseline community and ecological quality status for 13 Orkney sandy beaches

7.1 Introduction

A baseline is a minimum or a starting point which is set, and against which any future changes are compared (Humphries & Winemiller 2009; Callaway 2016). In biological monitoring, baseline surveys have been conducted to characterise natural population fluctuations, over short- and long-term timescales so that the scale of response to any future changes in the environment can be measured against this background (Humphries & Winemiller 2009; Pande & Gardner 2009; Villnäs & Norkko 2011; Callaway 2016). A baseline is not necessarily an ideal condition, rather it is the condition (or state of population or assemblage of fauna or flora) which was found at a point in time. Baseline surveys are used in many aspects of marine monitoring: e.g. benthic macroinvertebrates (Borg et al. 1997; Simboura et al. 1998; Puente et al. 2002; Callaway 2016), assessing marine communities in proposed marine protected areas (Durell et al. 2005; Pande & Gardner 2009; Louzao et al. 2010), to assess marine bioinvasions (Campbell et al. 2007; Lehtiniemi et al. 2015) and in marine planning (Day 2008).

Collection of samples along a transect line is agreed to be a sound approach for measuring and describing complete macroinvertebrate community structure on a sandy beach (McLachlan & Defeo 2018). Samples collected at intervals, starting from the top of the shore all the way down to the low tide mark (or vice versa), enable the capture of macroinvertebrates from each zone of the beach. In Chapter 6 the samples from up to three sampling stations sampled along a shore transect were analysed separately for three sites (Quoys, Congesquoy and Waulkmill) to assess spatio-temporal variability. To define and describe the macroinvertebrate community structure at each of the 13 study sites data from the sampling stations (up to three stations) which were sampled in Current time period were be used. The sampling in the Historical time period was carried out at most of the established sampling stations excluding the bedrock or shingle stations at the top of the shore line (Atkins et al. 1985). The Historical baseline was defined in the terms of the stations sampled during the Current period, hence restricted to two or three stations per site, and further details will be given in the methods section below (Section 7.3).

The trends and variability of macroinvertebrate communities are characterised most meaningfully in terms of the common taxa (Frid et al. 2009). Rare taxa are known to contribute up to 70% of total number of species in benthic macroinvertebrate

communities (Gray & Elliott 2009) and are an important part of the macroinvertebrate community contributing to the diversity of macroinvertebrate communities (Davidson et al. 2004). In setting the baseline macroinvertebrate community for a sandy beach the common and rare taxa should be determined. Bamber (1993) described rare taxa as any macroinvertebrate species for which mean abundance was $\langle 1.5 \rangle$ individuals per 0.1m⁻². In comparison Frid et al. (2009) included all taxa representing >0.1% of individuals in their data analysis. Jarrin et al. (2017) removed all taxa which were present only in one sample. Atkins et al. (1985) described 14 sandy beach macroinvertebrate communities in Orkney Islands by listing the dominant species of each station at each site, the authors defined the dominant species as any fauna which was >1% of total abundance of the fauna present. The study by Atkins et al. (1985) was on many of the same sandy beaches considered in this thesis (Mill Bay, Bay of Quoys, Bay of Creeklands, Swanbister Bay, Waulkmill Bay, Scapa Bay and Widewall Bay); the same rule for defining the dominant taxa will be adopted here in establishing the macroinvertebrate baseline communities.

The aggregation of macroinvertebrate data to family level or higher (e.g. order, class or phylum where appropriate) (Chapter 2 Section 2.2.4) has been used throughout this thesis. Table 7.1 summarises the pros and cons of family level versus species level data. By using family level data, and when setting baseline macroinvertebrate communities, any changes in species level are lost. Taxonomic sufficiency as defined by Ellis (1985) is 'the concept that in any project organisms must be identified to a level (species, genus, family, etc.) which balances the need to indicate the biology of organisms present with accuracy in making the identifications', which is a well-studied concept (Warwick et al. 1990; Somerfield & Clarke 1995; Roach et al. 2001; Dauvin et al. 2003; De Biasi et al. 2003; Ruso et al. 2007; De-la-Ossa-Carretero et al. 2012; Chatzinikolaou et al. 2018). De Biasi et al. (2003) analysed macrobenthic data in order to distinguish if there were differences in the results when using species, genus, family, order, class and phylum levels. De Biasi et al. (2003) concluded that when using species and genus level the results were very similar and at family level the results did not show much difference to species and genus level, but all levels higher than family showed changed patterns in the results. Similar results were obtained by Warwick (1988) in his study in which he used multivariate methods to analyse five sets of data (two meiofauna and three macrofauna) aggregated to different taxonomic levels. Warwick (1988) concluded that when using multivariate methods and higher taxonomic groupings (genus, family or order), the results were same as using species level data. Olsgard et al. (1998) concluded that for routine environmental monitoring it is effective to identify macrobenthic samples to family level only. Using

family level data to set the baseline macroinvertebrate data and environmental condition will undoubtedly miss species level dynamics but it has been shown to be sufficient to highlight any changes in the environmental conditions (Warwick et al. 1990; Somerfield & Clarke 1995; Roach et al. 2001; Dauvin et al. 2003; De Biasi et al. 2003; Ruso et al. 2007; De-la-Ossa-Carretero et al. 2012; Chatzinikolaou et al. 2018).

Chapter 6 explored the spatial and temporal variability of macroinvertebrate communities at three sandy beach sites in Scapa Flow (Quoys, Congesquoy and Waulkmill). The abundances of the taxa present were shown to fluctuate between years at each site (Quoys, Congesquoy and Waulkmill). The same distinct taxon groups remained present from year to year and from Historical to Current time period at all but one station, Quoys ST7. The presence of new taxon (Platyhelminthes) at Quoys ST7 indicated a change in the distinct taxa groups present at that station. Chapter 6 concentrated on the characterising taxa and their presence in a time series dataset, in this chapter the two time periods of Historical and Current will be compared with each other across a wider selection of sites.

Shifting baselines and the understanding of what baseline data are has been debated by Pauly (1995) who described the 'shifting baselines' in fisheries biology where each generation of fisheries scientists take the status of stock sizes at the beginning of their career to be the baseline. This 'shifts' the baseline to a more depleted stage for every new generation of fisheries scientists. Shifting baselines have, for example, been discussed in relation to fisheries (Pinnegar & Engelhard 2008), shark populations in Gulf of Mexico (Baum & Myers 2004), Antarctic bivalve molluscs (Reed et al. 2012), Californian kelp forests (Dayton et al. 1998) and benthic macroinvertebrates in Baltic Sea (Villnäs & Norkko 2011).

The colonisation sequence of macroinvertebrates after a disturbance or pollution event on sandy beaches follows a pattern; the first-order opportunistic species colonise the area first, these are small opportunistic macroinvertebrates (for example *C. capitata*) (Borja et al. 2000) which have the ability to find new areas quickly, are able to rapidly increase in numbers, have large population sizes, early maturation and high mortality (Gray 1979; Gray 1981). When a site is heavily polluted the macroinvertebrate, communities have a low species diversity and are dominated by few species and small individuals (Elliott 1993). The second-order opportunistic species (for example *Chaetozone* sp. (Borja et al. 2000)) colonise a polluted or disturbed area after the first-order opportunistic species (pollution indicator species), and both are superseded by the natural or equilibrium state species which vary depending on the sandy beach (Gray 1979; Gray 1981; Elliott 1993). In the equilibrium state the diversity of species is high, with low abundance, the species which are dominant are generally large in size and in weight (Gray 1979; Gray 1981; Elliott 1993). The macroinvertebrate community diversity, the taxa present, and the abundance of the taxa present are all important components when analysing and interpreting macroinvertebrate data for benthic quality.

During the 1980s and 1990s many macrobenthic studies evaluated the use of macroinvertebrates in pollution monitoring (Gray & Christie 1983; Hargrave & Thiel 1983; Bilyard 1987; Warwick 1988; Warwick et al. 1990; Dauer 1993; Warwick & Clarke 1993; Kiyko & Pogrebov 1997; Dean 2008). In 2000 the European Water Framework Directive (WFD 2000) came into effect. The aim of the directive was to establish 'good ecological status' in all waters: inland surface waters, transitional (estuarine) waters, coastal waters and groundwater (Borja et al. 2004) by 2015. Since the directive came into effect scientists in member states of European Union worked towards finding ways to assess the ecological status of water bodies. Within the coastal and transitional water bodies much of research concentrated on using benthic macroinvertebrates as indicators of water quality (Borja et al. 2004; Prior et al. 2004; Borja et al. 2007; Dauvin et al. 2007; Muxica et al. 2007; Borja et al. 2009; Josefson et al. 2009; Borja et al. 2011, 2012a). In the UK the methods were developed by Prior et al. (2004). They considered transitional waterbody typology (mixing characteristics, salinity, mean tidal range, exposure, depth and substratum), reference conditions, boundary areas, historical data and several classification tools, in their research into finding a suitable method for UK waters. In 2014 the Infaunal Quality Index (IQI) was agreed as the classification method for UK by the United Kingdom Technical Advisory Group (UKTAG) (WFD-UKTAG 2014). The IQI is a multimetric index and uses three components, namely AZTI's Marine Biotic Index (AMBI), Simpson's Evenness and the number of taxa (WFD-UKTAG 2014). It would have been possible to calculate IQI for the Scapa Flow sites but as the Scapa Flow macroinvertebrate data were aggregated to family level and not species level and due to time constraints of this study, the decision was made just to apply AMBI software to establish the ecological quality of the sandy beaches. The AMBI software was developed by researchers from AZTI Tecnalia Marine Research Division, Spain in response to the EU Water Framework Directive and the requirements of the ecological status assessment of coastal and estuarine waters (Borja et al. 2000; 2004; 2007; 2009; 2011; 2012a). The AMBI index (see Methods section 7.3.1) has been widely used in assigning ecological quality and environmental conditions for benthic communities (Muxica et al. 2005; Carvalho et al. 2006; Dauvin et al. 2007; Josefson et al. 2009; Gillett et al. 2015; Albayrak et al. 2019).

In the present context, 'baseline data' refers to macroinvertebrate community structure at a point in time (Historical or Current), and 'ecological quality' on a sandy beach refers to the environmental status of a beach of which the macroinvertebrate community structure is an indicator.

7.2 Aims

To develop and test an approach towards the definition of the baseline macroinvertebrate community and the ecological quality for each of the 13 study sites (Scapa Bay, Swanbister Bay, Waulkmill Bay, Widewall Bay, Congesquoy Bay, Cumminess Bay, Dead Sand, Bay of Creekland, Kirk Hope Bay, Longhope Bay, Lyrawa Bay, Mill Bay and Bay of Quoys).

7.3 Methods

During the Current time period samples were collected from up to three stations per study site (Chapter 3). The comparisons between Historical and Current time periods were always based on the same sampling stations. Table 7.2 lists the study sites and their stations which were used in describing the baseline macroinvertebrate communities and their ecological quality (further details of the sites and the stations characteristics were provided in Chapter 3).

Table 7.2. The Orkney sandy beach sites with their sampling stations from both Historical and Current time periods used in describing the baseline macroinvertebrate communities.

Sampling at the 13 sandy beach sites has been carried out since 1974, however, not all the data from the monitoring period were available for analysis, as previously described. Data used in this analysis are shown in Table 7.3.

Table 7.3. Data available for analysis from sandy beach surveys carried out during A. Historical and B. Current time periods. Macroinvertebrate data from years which are crossed out and in light grey were not available for this analysis.

SITE		A. HISTORICAL TIME PERIOD														No. yrs		
Congesquoy													1983 1984 1985 1986 1987 1988 1989					7
Cumminess													1984 1985 1986 1987 1988 1989					6
Dead Sand													1984 1985 1986 1987 1988 1989					6
Scapa		1974 1975 1976 1977 1978 1979 1980 1981 1982 1983 1984 1985 1986 1987 1988 1989																12
Swanbister		1974 1975 1976 1977 1978 1979 1980 1981 1982 1983 1984 1985 1986 1987 1988 1989																15
Waulkmill		1974 1975 1976 1977 1978 1979 1980 1981 1982 1983 1984 1985 1986 1987 1988																14
Widewall													1982 1983 1984 1985 1986 1987 1988 1989					8
Creekland		1974 1975 1976 1977 1978 1979							1981 1982									8
Kirk Hope													1983 1984 1985 1986 1987 1988					6
Longhope		1974 1975 1976 1977											1983 1984 1985 1986 1987 1988				1990	10
Lyrawa		1974 1975 1976 1977										1983 1984 1985 1986				1989 1990		7
Mill Bay		1974 1975 1976 1977 1978 1979 1980 1981 1982 1983 1984 1985 1986 1987 1988																15
Quoys		1974 1975 1976 1977 1978 1979 1980 1981 1982 1983 1984 1985 1986 1987 1988																10
SITE	B. CURRENT TIME PERIOD																	
Congesquoy 2002 2003 2004 2005 2006 2007 2008 2009 2010 2011 2012 2013 2014 2015 2016																		15
Cumminess		2002 2003 2004 2005 2006 2007 2008 2009 2010 2011 2012 2013 2014 2015 2016																15
Dead Sand		2002 2003 2004 2005 2006 2007 2008 2009 2010 2011 2012 2013 2014 2015 2016																15
Scapa		2002 2003 2004 2005 2006 2007 2008 2009 2010 2011 2012 2013 2014 2015 2016																15
Swanbister		2002 2003 2004 2005 2006 2007 2008 2009 2010 2011 2012 2013 2014 2015 2016																15
Waulkmill		2002 2003 2004 2005 2006 2007 2008 2009 2010 2011 2012 2013 2014 2015 2016																15
Widewall		2002 2003 2004 2005 2006 2007 2008 2009 2010 2011 2012 2013 2014 2015 2016																15
Creekland													2006 2007 2008 2009 2010 2011 2012 2013 2014 2015 2016					11
Kirk Hope													2006 2007 2008 2009 2010 2011 2012 2013 2014 2015 2016					11
Longhope													2006 2007 2008 2009 2010 2011 2012 2013 2014 2015 2016					11
Lyrawa													2006 2007 2008 2009 2010 2011 2012 2013 2014 2015 2016					11
Mill Bay													2006 2007 2008 2009 2010 2011 2012 2013 2014 2015 2016					11
Quoys													2006 2007 2008 2009 2010 2011 2012 2013 2014 2015 2016					11

All taxa were aggregated to family level or higher (e.g. order, class or phylum where appropriate) (Chapter 2 Section 2.2.4) and abundances were standardised for ind. 0.1m^{-2} .

Current time period macroinvertebrate data were re-identified and enumerated for three sites, Quoys, Congesquoy and Waulkmill as detailed in Chapter 4. This was not done for the macroinvertebrate samples of the remaining ten sites (Creekland, Cumminess, Dead Sand, Kirk Hope, Longhope, Lyrawa, Mill Bay, Scapa, Swanbister and Widewall) but the macroinvertebrate data for these sites were standardised using information detailed in Table 4.2 (Chapter 4).

For each Historical and Current time periods a single figure for all samples for all stations (Table 7.2) was calculated to show the mean abundance (ind. 0.1m^{-2}) of each taxon at that time period.

The >1% contribution was calculated at an aggregated level for each site within each time period; taxa which contributed >1% to the total abundance of the macroinvertebrate community at either Historical or Current time period were assigned as dominant taxa. In this thesis dominant taxa will be used as a representation of the baseline macroinvertebrate community for each site. The total abundance for each time period was the sum of all the taxa abundances in a site in a time period. The dominant taxa at each time period are considered against both the dominant and rare taxa at the other time period.

The comparison of macroinvertebrate community composition (annual macroinvertebrate data standardised to ind. $0.1m^{-2}$) between Historical and Current time periods were analysed using analysis of variance on $4th$ root transformed data judging the significance of variability according to a permutation test using R library lmperm (Chapter 2 Section 2.2.5.2).

7.3.1 AMBI (AZTI's Marine Biotic Index) software

The version 5.0 of the AMBI software has more than 8400 species (AMBI update June 2017) included from the entire world (Borja et al. 2012b). The AMBI analysis is based on allocating species to five pre-defined ecological Groups (GI-GV), where species in GI are very sensitive to organic enrichment, GII are species indifferent to enrichment, GIII are species tolerant to excess organic matter enrichment, GIV are second order opportunistic species and GV are first-order opportunistic species and pollution indicator species (Table 7.4) (Borja et al. 2000).

The AMBI score is calculated using the percentage abundance of each ecological group in a sample using the following formulae:

$$
AMBI = \frac{(0 x \%GI) + (1.5 x \%GII) + (3 x \%GIII) + (4.5 x \%GIV) + (6 x \%GV)}{100}
$$

Where:

GI – GV represent the ecological groups as described by Borja et al. (2000) and as described in Table 7.4.

The AMBI calculation was run for each site and for each time period using the macroinvertebrate abundances shown in Tables 7.6-18. Several of the taxa in the Scapa Flow macroinvertebrate dataset were not listed in the AMBI species list and for these either a species or genus name was assigned or designated as 'not assigned' or 'ignored' (Appendix H). As an example, Arenicolidae was assigned as *Arenicola marina*; Skeneidae was assigned *Skenea* sp.; Chordata was 'ignored' and Brachyura was 'not assigned' (Appendix H). Assigning species or genus names for the families in the Scapa Flow dataset was possible due to one or more species being known for each macroinvertebrate family, for example four species of Pontoporeiidae are known from the Scapa Flow sites: *Bathyporeia elegans, B. guilliamsoniana, B. pilosa* and *B. sarsi* (Kakkonen pers. obs.). Most species within a family are in the same ecological group for the AMBI calculation (Borja et al. 2000) meaning that where one species was assigned for family with several species recorded from Scapa Flow it would not have affected the AMBI calculation.

The summary of AMBI boundaries is listed in Table 7.5 and illustrated in Figure 7.1.

Table 7.4. Description of the ecological Groups (GI-V) with example taxa. From Borja et al. $(2000).$

Figure 7.1. The AMBI biotic coefficient relating with the Ecological Groups I-V. WFD: Water Framework Directive. From WFD-UKTAG 2014.

7.3.2 Multi-dimensional Scaling (MDS) ordination

The macroinvertebrate communities of the 13 sites were compared using MDS ordination (Chapter 2 Section 2.2.5.1). All taxa were aggregated to family level or higher (e.g. order, class or phylum where appropriate) (Chapter 2 Section 2.2.4) and abundances were standardised for ind. 0.1m^{-2} . . The MDS ordination was performed using macroinvertebrate data from years 2006-2016. For each site data from all sampling stations (up to three) were aggregated to one value, for example the Dead Sand macroinvertebrate abundances from station 1 and 2 were pooled for each year to provide a single value for each taxon for each year.

7.4 Results

7.4.1 Taxonomic composition

At all study sites the macroinvertebrate community has been dominated by three phyla: Annelida, Arthropoda and Mollusca (Figure 7.2). The number of taxa recorded varied from 20 taxa at Scapa in Historical time period to 60 at Longhope in the Current time period (Figure 7.2). At Congesquoy, Cumminess and Lyrawa the number of taxa increased from Historical to Current time period. This increase in number of taxa at these three sites could potentially be due to sampling effort, as the data analysed for Congesquoy, Cumminess and Lyrawa include up to seven years of sampling in Historical time compared to up to 15 in Current time (Table 7.3); increased sampling effort is known to increase the number of taxa recorded (Schooler et al. 2017). At Dead Sand, Kirk Hope and Widewall the number of taxa has remained more or less the same regardless of

increased sampling effort, the macroinvertebrate communities at these sites must consist of restricted number of taxa and the sampling effort at both periods has been sufficient enough to capture the full community present. At Longhope the sampling effort has remained almost the same at both Historical (10yrs) and Current (11yrs) time periods and the number of taxa recorded are 59 in Historical and 60 in Current time (Figure 7.2). The number of taxa has varied between the two time periods, but most differences are likely due to sampling effort.

Figure 7.2. Changes in the taxonomic composition of the macroinvertebrates between Historical and Current time period at each study site. The three sites to the left of hashed line (Congesquoy, Quoys and Waulkmill) were discussed and analysed in detail in Chapter 6. Abbreviations: Co: Congesquoy, Qu: Quoys, Wa: Waulkmill, Cr: Creekland, Cu: Cumminess, De: Dead Sand, Ki: Kirkhope, Lo: Longhope, Ly: Lyrawa, Mi: Mill Bay, Sc: Scapa, Sw: Swanbister, Wi: Widewall Bay. Others: Chordata, Hemichordata, Sipuncula, Phoronida and Echinodermata.

7.4.2 Species accumulation curves

The number of taxa at each sampling station has continued to increase over the monitoring period (Figure 7.3). Either the full complement of taxa at each station has not yet been sampled, the increase in the number of taxa is due to taxa coming in from climate change or other natural event or the analysts are identifying more taxa due to improved sample processing and identification skills. Further work is required to understand this fully.

Figure 7.3. Species Accumulation Curve for each sampling station. Sample Order: Original.

Figure 7.3 (continued) Species Accumulation Curve for each sampling station. Sample Order: Original.
7.4.3 Baseline macroinvertebrate communities

The baseline macroinvertebrate communities are described for each of the 13 sandy beach sites by the presence of dominant taxa, macroinvertebrate data for each site are presented in the Tables 7.6-18.

7.4.3.1 Congesquoy

The number of taxa recorded at Congesquoy increased by ten from 38 taxa in Historical time period to 48 taxa in Current time period (Figure 7.2). For the Historical time period taxa with mean abundance of ≥ 13.92 ind. 0.1m⁻² were classed as dominant (Table 7.6), for Current time period taxa with mean abundance of ≥ 8.90 ind. 0.1m^{-2} were classed as dominant (Table 7.6). The baseline macroinvertebrate community includes 15 taxa; nine of which belong to phylum Annelida, five to phylum Crustacea and one to phylum Mollusca. Polychaetes Capitellidae, Maldanidae, Orbiniidae, Paraonidae, Spionidae and Syllidae, and amphipods Corophiidae, Lampropidae, Pontoporeiidae and Tanaissuidae were dominant in both Historical and Current time periods. The mean abundances of most of the dominant taxa were similar in both time periods apart from Syllidae, Pontoporeiidae and Tanaissuidae which all decreased in their mean abundance from Historical to Current time period. Polychaeta Opheliidae, Psammodrilidae and bivalve Tellinidae were dominant in Current time but rare in the Historical time period, whereas polychaeta Sphaerodoridae and amphipod Phoxocephalidae were dominant in Historical and rare in Current time periods. The abundances of eight taxa (Table 7.6) were statistically different between the time periods, indicating different patterns of annual variability in their abundances.

All taxa that were dominant in either Historical or Current time periods, were at least rare in both time periods. If either Historical or Current macroinvertebrate data were solely used for setting up the baseline macroinvertebrate data, the baseline would be slightly different in each time period.

The AMBI scores for Congesquoy at Historical and Current time period were 1.9 and 1.8, respectively (Figure 7.4), indicating slightly disturbed condition with unbalanced benthic community health (Table 7.5) in both time periods. The organic pollution indicator Capitellidae belongs to the AMBI ecological Group GV, the presence of this indicator taxon increased the AMBI score for the site. Congesquoy is north of the Stromness waste water treatment facility and south of Loch of Stenness, both of which are known organic discharge point sources to the Bay of Ireland area (Scottish Environment Protection Agency 2019).

7.4.3.2 Creekland

At Creekland the number of taxa increased by eight from Historical (36 taxa) to Current (44 taxa) time period (Figure 7.2). At the Historical time period taxa with mean abundance of \geq 12.88 ind. 0.1m⁻² and at Current time period taxa with mean abundance of \geq 13.87 ind. 0.1m⁻² were classed as dominant (Table 7.7). At Creekland the baseline macroinvertebrate community consisted of 12 taxa of which eight belong to phylum Annelida and four to the phylum Crustacea (Table 7.7). Five taxa were dominant at both time periods: Capitellidae/Oligochaeta, Cirratulidae, Spionidae, Syllidae and Pontoporeiidae. Six taxa (Maldanidae, Opheliidae, Orbiniidae, Paraonidae, Corophiidae, and Oedicerotidae) were rare at Historical time period but dominant in Current time period due to their increased mean abundances in Current time period. One taxon (Cirolanidae) was dominant at Historical time period and rare at Current time period due to the decrease in the mean abundance. All but two taxa had statistically different abundances between the time periods (Table 7.7).

All taxa that were dominant in either Historical or Current time periods, were at least rare in both time periods. If either Historical or Current macroinvertebrate data were solely used for setting up the baseline macroinvertebrate data, the baseline would be slightly different in each time period.

The AMBI scores for Creekland at Historical and Current time periods were 2.3 and 2.7, respectively (Figure 7.4) indicating slightly disturbed condition with unbalanced benthic community health (Table 7.5) in both time periods. The mean abundances of Capitellidae / Oligochaeta were pooled for Creekland macroinvertebrate data due to the confusion in their identification and as recommended in Chapter 4. The presence of the organic pollution indicator taxa Capitellidae and Oligochaeta in the samples impacted on the AMBI score; an increase in their mean abundance from Historical to Current time period contributed to the increased AMBI score. The presence of these two taxa could be due to diffuse pollution from the agricultural run-off from the adjacent cultivated land (Chapter 3 Table 3.6). The increased AMBI score in the Current time period remained within the same AMBI Index boundaries (Table 7.5) as the Historical score and therefore the environmental health of the site is judged to be the same (Figure 7.4).

7.4.3.3 Cumminess

At Cumminess the number of taxa increased the most from Historical to Current time period compared to all other sites, an increase of 21 taxa (Figure 7.2). This increase was likely due to sampling effort as discussed earlier (Section 7.4.1). At the Historical time

period taxa with mean abundance of ≥ 8.74 ind. 0.1m⁻² and at Current time period taxa with mean abundance of \geq 9.49 ind. 0.1m⁻² were classed as dominant (Table 7.8). At Cumminess the baseline macroinvertebrate community consisted of 13 taxa of which six belong to phylum Annelida, one to phylum Mollusca, five to phylum Crustacea and one to phylum Nemertea (Table 7.8). Ten taxa were dominant in both time periods: Capitellidae, Maldanidae, Orbiniidae, Spionidae, Syllidae, Corophiidae, Oedicerotidae, Pontoporeiidae, Urothoidae and Nemertea. Two taxa (Psammodrilidae and Tellinidae) were rare at Historical time period and one taxon (Cirolanidae) was rare at the Current time period, all being dominant in the other time period. The mean abundances of the dominant taxa have remained similar in both time periods apart from Spionidae which mean abundance increased by 95% from Historical to Current time period. Taxa which were rare in Historical and dominant in the Current time, and vice versa, all had changes in the mean abundances of 50% or more. These increases in abundances are high and could be due to an increase in organic enrichment to the site or the timing of the sampling in Current time period coinciding with higher abundances compared to the Historical time period. Tellinidae was the only taxon with a statistically significant difference in abundance between the two time periods.

All taxa that were dominant in either Historical or Current time periods, were at least rare in both time periods. If either Historical or Current macroinvertebrate data were solely used for setting up the baseline macroinvertebrate data, the baseline would be slightly different in each time period.

The AMBI scores for Cumminess at Historical and Current time were 1.3 and 1.8 respectively (Figure 7.4) indicating slightly disturbed condition with impoverished benthic community (Table 7.5) in both time periods. The AMBI scores for both time periods were very close to the boundary with classification to unpolluted (Table 7.5). High abundance of Spionidae, taxa belonging to the ecological Group GIII (Borja et al. 2000) increased the AMBI score in Current time period. Cumminess is located in the Bay of Ireland, east from the Stromness waste water treatment facility (Chapter 3 Figure 3.1). Spionid polychaetes are interface feeders with palps which in different species of Spionidae can be adapted to deposit or suspension feeding (Fauchald & Jumars 1979). The presence of high abundance of Spionid polychaetes at Cumminess could be due to the proximity of the site to the Stromness waste water treatment facility.

7.4.3.4 Dead Sand

At Dead Sand the number of taxa from Historical (23) to Current (24) time periods increased by only one (Figure 7.2). At Historical time period taxa with mean abundance of \geq 140.27 ind. 0.1m⁻² and at Current time period taxa with mean abundance of \geq 63.14 ind. $0.1\,\mathrm{m}^{-2}$ were classed as dominant (Table 7.9). The macroinvertebrate baseline community at Dead Sand consisted of seven taxa of which five belong to phylum Annelida and two to phylum Crustacea (Table 7.9). Five taxa, Capitellidae, Fabriciidae, Oligochaeta, Spionidae and Corophiidae were dominant in both Historical and Current time periods. Two taxa, Nereididae and Pontoporeiidae, were rare at Historical time period. One dominant taxon Capitellidae and the two rare taxa, Nereididae and Pontoporeiidae, increased in their abundance at Current time period, Corophiidae mean abundance remained constant and Fabriciidae, Oligochaeta and Spionidae all decreased in their abundances (Table 7.9). All but two taxa had statistically different abundances between the time periods (Table 7.9).

All taxa that were dominant in either Historical or Current time periods, were at least rare in both time periods. If either Historical or Current macroinvertebrate data were solely used for setting up the baseline macroinvertebrate data, the baseline would be slightly different in each time period.

The AMBI score for Dead Sand at both Historical and Current time was 3.5 (Figure 7.4), this indicates moderately disturbed condition with benthic community in transition to polluted condition (Table 7.5). The high AMBI scores for both time periods were due to the presence of two organic pollution indicator taxa Capitellidae and Oligochaeta (ecological Group GV), both of which were present in high abundances. In addition, there were three other dominant taxa belonging to ecological Group GIII: Spionidae, Corophiidae and Nereididae (Borja et al. 2000). Dead Sand is a shallow embayment south of Brig O'Waithe which connects to the Loch of Stenness saline lagoon. The Loch of Stenness catchment area suffers from high levels of nutrient input from the surrounding farmland and from sewage discharges (ICIT 2004b), and this nutrient load has been transported to Dead Sand. The Dead Sand site itself is surrounded by agricultural land and has several unnamed burns running into it from the adjacent land, which is another likely source of nutrients to the site.

7.4.3.5 Kirk Hope

At Kirk Hope the number of taxa increased by two from Historical (34 taxa) to Current periods (36 taxa) (Figure 7.2). At Historical time period taxa with mean abundance of

 \geq 18.63 ind. 0.1m⁻² and at Current time period taxa with mean abundance of \geq 7.32 ind. $0.1\,\text{m}^{-2}$ were classed as dominant (Table 7.10). At Kirk Hope the macroinvertebrate baseline data consisted of eleven taxa of which eight belong to phylum Annelida, two to phylum Crustacea and one to phylum Mollusca (Table 7.10). Five taxa, namely Capitellidae, Oligochaeta, Spionidae, Syllidae and Pontoporeiidae, were dominant in both Historical and Current time periods. Five taxa, namely Maldanidae, Opheliidae, Orbiniidae, Phyllodocidae, Corophiidae and Tellinidae were rare at Historical time period but were dominant at Current time period due to their increased abundances. Six taxa had statistically different abundances (Table 7.10).

All taxa that were dominant in either Historical or Current time periods, were at least rare in both time periods. If either Historical or Current macroinvertebrate data were solely used for setting up the baseline macroinvertebrate data, the baseline would be slightly different in each time period.

The AMBI scores for Kirk Hope at Historical and Current times were 3.8 and 1.6, respectively (Figure 7.4). The Historical score indicates a moderately disturbed condition with benthic community in transition to polluted condition (Table 7.5), which improved at Current time period to slightly disturbed condition with unbalanced benthic community (Table 7.5). The improved AMBI score is due to the decreased abundance of Capitellidae and Spionidae from Historical to Current time period, and the increased abundance of Pontoporeiidae. Capitellidae and Spionidae belong to the ecological Groups GV and GIII, respectively, whereas Pontoporeiidae belongs to the ecological Group GI (Borja et al. 2000). Jones et al. (1991) described Kirk Hope sandy beach site as having, 'high diversity of species and little evidence of pollution'. Kirk Hope is located in the southern area of Scapa Flow (Chapter 3 Figure 3.1) away from any obvious pollution sources. During the Historical time period it is possible that sewage from the houses by the shoreline would have discharged directly into the bay increasing organic pollution at the site and facilitated the population growth of Capitellidae and Spionidae. The AMBI index, based on macroinvertebrate abundance data, highlights improvement in the environmental condition of Kirk Hope sandy beach site.

7.4.3.6 Longhope

Longhope has the highest number of taxa present of the thirteen sites, 76 taxa, of which 59 were recorded during Historical time period and 60 during Current time period (Figure 7.2). At Historical time period taxa with mean abundance of \geq 17.69 ind. 0.1m⁻² and at Current time period taxa with mean abundance of >10.84 ind. 0.1m^{-2} were classed

as dominant (Table 7.11). The macroinvertebrate baseline community at Longhope consisted of 17 taxa, of which nine belong to phylum Annelida and four to both Crustacea and Mollusca (Table 7.11). Nine taxa, namely Opheliidae, Orbiniidae, Spionidae, Syllidae, Corophiidae, Pontoporeiidae, Cardiidae, Hydrobiidae and Montacutidae were dominant in both Historical and Current time periods. Capitellidae, Fabriciidae, Oligochaeta, Ampeliscidae and Oedicerotidae were dominant in Historical time period but were classed as rare at Current time period. Maldanidae was dominant at Current time but was not recorded during Historical time period. Arenicolidae and Tellinidae were rare at Historical time period and dominant taxa at Current time. Large decreases in the abundances from Historical to Current time period were observed for two taxa, namely Fabriciidae and Spionidae, and a large increase was observed for Hydrobiidae.

One new taxon was recorded as dominant in Current time period: Maldanidae. Maldanidae has been recorded at eight other sandy beach sites in Scapa Flow (Congesquoy, Creekland, Cumminess, Dead Sand, Kirkhope, Mill Bay, Quoys, and Swanbister). The arrival of Maldanidae at Longhope constitutes a distinct taxonomic change in the macroinvertebrate baseline community from Historical to Current time period. The remaining 16 taxa which represent the macroinvertebrate baseline community were recorded in both time periods as described above. The baseline macroinvertebrate communities would have been different if dominant macroinvertebrates from only Historical or Current time period were used.

The AMBI score for Longhope at both Historical and Current time was 2.3 (Figure 7.4); this indicates slightly disturbed condition with impoverished benthic community (Table 7.5). Jones et al. (1991) described the Longhope mid-shore stations 7 and 8 (ST8 included in this thesis) as showing evidence of organic pollution. The high abundances of ecological Group GIII taxa (Fabriciidae, Oligochaeta, and Spionidae) in the Historical time period supports this statement. Maldanidae which was only recorded during Current time period belongs to the ecological Group GI, which consist of species sensitive to organic pollution and which are present in unpolluted conditions (Borja et al. 2000). The decreased abundance of Fabriciidae, Oligochaeta and Spionidae in the Current time period, the presence of Maldanidae and increased abundance of Tellinidae (ecological Group GI) would indicate an improvement in the benthic community health. However, the AMBI score at Current time period remained the same as in Historical time which could be due to the presence Hydrobiidae (ecological Group GIII) which was recorded in

increased abundance at Current time period, increasing the AMBI score for Current time period.

7.4.3.7 Lyrawa

At Lyrawa 46 taxa have been recorded during the monitoring programme, of which 29 were recorded at Historical time period and 41 in the Current time period (Figure 7.2). At Historical time taxa with mean abundance of ≥ 3.27 ind. 0.1m⁻² and at Current time taxa with mean abundance of \geq 7.26 ind. 0.1m⁻² were classed as dominant (Table 7.12). The macroinvertebrate baseline community at Lyrawa consists of ten taxa, of which six belong to phylum Annelida, two to phylum Crustacea and one to both Mollusca and Nemertea (Table 7.12). Three taxa, namely Capitellidae / Oligochaeta, Spionidae and Pontoporeiidae, were dominant at both time periods. Remaining seven taxa, Opheliidae, Orbiniidae, Paraonidae, Syllidae, Corophiidae, Tellinidae and Nemertea, were all rare at Historical time period and dominant at Current time period. All but one taxon had statistically different abundances (Table 7.12) indicating different patterns of annual variability in their abundances.

All taxa that were dominant in either Historical or Current time periods, were at least rare in both time periods. If either Historical or Current macroinvertebrate data were solely used for setting up the baseline macroinvertebrate data, the baseline would be slightly different in each time period.

The AMBI scores for Lyrawa at Historical time and Current time period were 4.0 and 3.1, respectively (Figure 7.4). The Historical score indicates a moderately disturbed condition with benthic community in transition to polluted condition (Table 7.5), which has improved at Current time period to slightly disturbed condition with unbalanced benthic community (Table 7.5). The abundances of taxa in the historical time period were disproportionally skewed to three taxa, Capitellidae / Oligochaeta, Spionidae and Pontoporeiidae, whilst all other taxa were rare. The AMBI calculation for the Lyrawa Historical time period would have been dominated by the three taxa of which one belongs to the ecological Group GV (Capitellidae / Oligochaeta) and another to ecological Group GIII (Spionidae). During the Current time period the dominant taxa and their abundances are more evenly distributed with several ecological Group GI taxa (Opheliidae, Orbiniidae, Pontoporeiidae, Tellinidae) present resulting in decreased AMBI score and improved benthic community health.

7.4.3.8 Mill Bay

At Mill Bay 67 taxa were recorded, of which 53 were recorded at Historical time period and 48 at Current time period (Figure 7.2). At the Historical time period taxa with mean abundance of \geq 54.52 ind. 0.1m⁻² and at Current time period taxa with mean abundance of \geq 19.22 ind. 0.1m⁻² were classed as dominant (Table 7.13). The macroinvertebrate baseline community at Mill Bay consists of eleven taxa, of which eight belong to phylum Annelida and three to phylum Crustacea (Table 7.13). Seven taxa, namely Capitellidae, Cirratulidae, Fabriciidae, Oligochaeta, Spionidae, Corophiidae and Pontoporeiidae were dominant at both Historical and Current time periods. Orbiniidae and Phoxocephalidae were dominant at Historical time and rare at Current time period due to their decreased abundances. Opheliidae and Syllidae were rare at Historical time periods and dominant at Current time period. Seven taxa had statistically different abundances (Table 7.13).

All taxa that were dominant in either Historical or Current time periods, were at least rare in both time periods. If either Historical or Current macroinvertebrate data were solely used for setting up the baseline macroinvertebrate data, the baseline would be slightly different in each time period.

The AMBI scores for Mill Bay at Historical and Current time period were 2.5 and 3.1, respectively (Figure 7.4) both indicating slightly disturbed condition with unbalanced benthic community (Table 7.5). No change in the benthic community health was observed even though the abundances of several of the taxa has changed greatly. Mean abundance of Fabriciidae decreased from 2384.56 ind. 0.1m^{-2} in Historical time to 335.08 ind. 0.1m^{-2} at Current time, which is an 85.6% decrease, and an even larger decrease of 92.9% was observed for the abundance of Corophiidae (Table 7.13). During Historical time stations 9 and 10 (ST10 included in this thesis) experienced organic enrichment but the source of this was not identified (Jones et al. 1991). The presence of organic enrichment during Historical time would endorse the presence of both Fabriciidae and Corophiidae in such high abundances as both are filter and deposit feeders (Fauchald & Jumars 1979).

7.4.3.9 Quoys

At Quoys 55 taxa were recorded, of which 44 were recorded at Historical time period and 35 at Current time period (Figure 7.2). At the Historical time period taxa with mean abundance of ≥ 48.44 ind. 0.1m⁻² and at Current time period taxa with mean abundance of \geq 10.75 ind. 0.1m⁻² were classed as dominant (Table 7.14). The macroinvertebrate baseline community at Quoys consisted of 11 taxa of which five belong to the phylum Annelida, four to phylum Crustacea and one to each Mollusca and Platyhelminthes (Table 7.14). Five taxa, namely Spionidae, Corophiidae, Phoxocephalidae, Pontoporeiidae and Urothoidae, were dominant in both Historical and Current time periods. Four taxa, namely Capitellidae, Oligochaeta, Opheliidae and Murchisonellidae, were rare at Historical and dominant at Current time periods. Paraonidae and Platyhelminthes were dominant at Current time and not recorded at Historical time period.

Two new taxa were recorded at Current time period: Paraonidae and Platyhelminthes (Table 7.14). Both are widely recorded at other sandy beach sites in Scapa Flow (Creekland, Dead Sand, Longhope, Lyrawa and Mill Bay). Paraonidae are small polychaete worms found from intertidal to depth of 69m (Hartley 1981). Platyhelminthes (flatworms) are an important part of the interstitial fauna of sandy beaches (McLachlan & Defeo 2018) and are found in many habitats including mud and sand (Fish & Fish 1996). The presence of both Paraonidae and Platyhelminthes as dominant taxa suggests a small but distinct change in the macroinvertebrate community at Quoys. The baseline macroinvertebrate communities would have been different if dominant macroinvertebrates from only Historical or Current time period were used.

The AMBI scores for Quoys Historical and Current time period were 1.8 and 2.8, respectively (Figure 7.4) indicating slightly disturbed condition with unbalanced benthic community health (Table 7.5) in both time periods. The AMBI score increased in Current time period, this could be due to higher abundance of Oligochaeta (ecological Group GV) and lower abundance of Pontoporeiidae (ecological Group GI) compared to the Historical time period (Table 7.14).

7.4.3.10 Scapa

At Scapa 36 taxa were recorded, of which 20 were recorded at Historical time and 30 at Current time period (Figure 7.2). At the Historical time period taxa with mean abundance of \geq 8.04 ind. 0.1m⁻² and at Current time period taxa with mean abundance of \geq 2.66 ind. $0.1\,\mathrm{m}^{-2}$ were classed as dominant (Table 7.15). The macroinvertebrate baseline community at Scapa consists of six taxa of which three belong to the phylum Annelida, and one each to Crustacea, Mollusca and Nemertea (Table 7.15). Four taxa, Capitellidae, Oligochaeta, Spionidae and Pontoporeiidae, were dominant in both Historical and Current time periods, and Tellinidae and Nemertea were rare at Historical and dominant at Current time period. Four taxa had statistically different abundances (Table 7.15).

All taxa that were dominant in either Historical or Current time periods, were at least rare in both time periods. If either Historical or Current macroinvertebrate data were solely

used for setting up the baseline macroinvertebrate data, the baseline would be slightly different in each time period.

The AMBI scores for Scapa at Historical and Current time periods were 4.8 and 2.4, respectively (Figure 7.4). The Historical score indicates a moderately disturbed condition with benthic community in polluted condition (Table 7.5), which improved at Current time period to slightly disturbed condition with unbalanced benthic community (Table 7.5). In the 1970s and 1980s Scapa Bay received organic effluent from two distilleries, Scapa and Highland Park (Atkins & Jones 1990), which explains the high abundance of Capitellidae at the site. No organic effluent from the distilleries has been discharged during the Current time period. The absence of organic effluent discharges to Scapa Bay has improved the benthic community health.

7.4.3.11 Swanbister

At Swanbister 58 taxa were recorded, 42 at Historical time period and 47 at Current time period (Figure 7.2). At the Historical time period taxa with mean abundance of ≥ 58.60 ind. 0.1m⁻² and at Current time period taxa with mean abundance of \geq 27.98 ind. 0.1m⁻² were classed as dominant (Table 7.16). The macroinvertebrate baseline community at Swanbister consisted of seven taxa of which five belong to the phylum Annelida and one to each of Crustacea and Mollusca (Table 7.16). Six taxa, Capitellidae, Fabriciidae, Oligochaeta, Opheliidae, Spionidae and Pontoporeiidae, were dominant in both time periods. Tellinidae was rare in Historical and dominant in the Current time period. Three taxa had statistically different abundances (Table 7.16).

All taxa that were dominant in either Historical or Current time periods, were at least rare in both time periods. If either Historical or Current macroinvertebrate data were solely used for setting up the baseline macroinvertebrate data, the baseline would be slightly different in each time period.

The AMBI scores for Swanbister at Historical and Current time period were 2.5 and 4.3, respectively (Figure 7.4). The Historical score indicates a slightly disturbed condition with unbalanced benthic community (Table 7.5) which declined at Current time period to moderately disturbed condition with benthic community in polluted condition (Table 7.5). The decline in benthic community health from Historical to Current time period could be attributed to the increase in abundance of Capitellidae (ecological Group GV) and decrease of Pontoporeiidae (ecological Group GI). Jones et al. (1991) demonstrated the increase in the organic effluent at stations 5, 6 and 7 (ST7 included in this thesis) by the

increased abundance of *Capitella capitata*, Oligochaeta and nematode worms. The organic enrichment at the Swanbister must have continued since the Historical time period as demonstrated by the decrease in benthic community health in the Current time period (Figure 7.4).

7.4.3.12 Waulkmill

At Waulkmill 39 taxa were recorded, 26 in the Historical time period and 34 in the Current time period (Figure 7.2). At the Historical time period taxa with mean abundance of \geq 3.07 ind. 0.1m⁻² and at Current time period taxa with mean abundance of \geq 5.49 ind. $0.1\,\mathrm{m}^{-2}$ were classed as dominant (Table 7.17). The macroinvertebrate baseline community at Waulkmill consisted of nine taxa of which five belong to the phylum Annelida, two to phylum Crustacea and one to each Mollusca and Nemertea (Table 7.17). Six taxa, Capitellidae, Opheliidae, Paraonidae, Spionidae, Pontoporeiidae and Nemertea, were dominant in both time periods. Oligochaeta and Tellinidae were rare at Historical time period and dominant in Current time period. Corophiidae was dominant in Historical and rare at Current time period. Seven taxa had statistically different abundances (Table 7.17).

All taxa that were dominant in either Historical or Current time periods, were at least rare in both time periods. If either Historical or Current macroinvertebrate data were solely used for setting up the baseline macroinvertebrate data, the baseline would be slightly different in each time period.

The AMBI scores for Waulkmill at Historical and Current time period were 1.3 and 2.6, respectively (Figure 7.4) indicating slightly disturbed condition with unbalanced benthic community health (Table 7.5) in both time periods. The AMBI score increased from Historical to Current time period but remained within the boundaries of slightly disturbed condition. Increased abundance of Oligochaeta, an organic pollution indicator taxon and from ecological Group GV, contributed to the increased AMBI score.

7.4.3.13 Widewall

At Widewall Bay 35 taxa were recorded, 29 at Historical time period and 28 at Current time period (Figure 7.2). At the Historical time period taxa with mean abundance of \geq 2.71 ind. 0.1m⁻² and at Current time period taxa with mean abundance of \geq 4.51 ind. $0.1\,\text{m}^{-2}$ were classed as dominant (Table 7.18). The macroinvertebrate baseline community at Widewall consisted of nine taxa of which four belong to phylum Annelida, three to phylum Crustacea and two to phylum Mollusca (Table 7.18). Three taxa, namely Capitellidae, Spionidae and Pontoporeiidae, were dominant in both Historical and Current time periods. Oligochaeta, Phyllodocidae, Ampeliscidae and Oedicerotidae were dominant in Historical time period and rare at Current time period. Cardiidae and Hydrobiidae were rare at Historical and dominant at Current time period. Four taxa had statistically different patterns of abundances (Table 7.18) indicating different patterns of annual variabilities in their abundances.

All taxa that were dominant in either Historical or Current time periods, were at least rare in both time periods. If either Historical or Current macroinvertebrate data were solely used for setting up the baseline macroinvertebrate data, the baseline would be slightly different in each time period.

The AMBI scores for Widewall at Historical and Current time periods were 1.5 and 3.0, respectively (Figure 7.4) indicating slightly disturbed condition with unbalanced benthic community health (Table 7.5) in both time periods. The AMBI score increased from Historical to Current time period but remained within the boundaries (1.2<AMBI≤3.3) of slightly disturbed condition. The dramatic increase of Hydrobiidae (ecological Group GIII) from 1.13 to 321.73 ind. 0.1m^{-2} contributed to the increased AMBI score.

Table 7.6. Congesquoy, abundance (ind. 0.1m^2) of all taxa at Congesquoy ST1 and ST2 in Historical and Current time periods. Abundance of dominant taxa in bold, rare taxa in italics or in the lower section of the table. Statistical significance: $* \le 0.05$, $** \le 0.01$, $*** \le$ 0.001. Statistical significance refers to the outcome of permutation tests comparing abundances between Historical and Current periods.

Figure 7.4. AMBI values based on the average abundance (ind. 0.1m⁻²) of individuals of each taxa at Historical and Current time period at each study site. Abbreviations: CO-H: Congesquoy Historical, CO-C: Congesquoy Current, CR-H: Creekland Historical, CR-C: Creekland Current, CU-H: Cumminess Historical, CU-C: Cumminess Current, DE-H: Dead Sand Historical, DS-C: Dead Sand Current, KI-H: Kirkhope Historical, KI-C: Kirkhope Current, LO-H: Longhope Historical, LO-C: Longhope Current, LY-H: Lyrawa Historical, LY-C: Lyrawa Current, MI-H: Mill Bay Historical, MI-C: Mill Bay Current, QU-H: Quoys Historical, QU-C: Quoys Current, SC-H: Scapa Historical, SC-C: Scapa Current, SW-H: Swanbister Historical, SW-C: Swanbister Current, WA-H: Waulkmill Historical, WA-C: Waulkmill Current, WI-H: CU-C: Cumminess Current, DE-H: Dead Sand Historical, DS-C: Dead Sand Current, KI-H: Kirkhope Historical, KI-C: Kirkhope Current, LO-H: Longhope Historical, LO-C: Longhope Current, LY-H: Lyrawa Historical, LY-C: Lyrawa Current, MI-H: Mill Bay Historical, MI-C: Mill Bay Current, QU-H: Quoys Historical, QU-C: Quoys Current, SC-H: Scapa Historical, SC-C: Scapa Current, SW-H: Swanbister Historical, SW-C: Swanbister Current, WA-H: Waulkmill Historical, WA-C: Waulkmill Current, WI-H: study site. Abbreviations: CO-H: Congesquoy Historical, CO-C: Congesquoy Current, CR-H: Creekland Historical, CR-C: Creekland Current, CU-H: Cumminess Historical, Widewall Historical, WI-C: Widewall Current Pairs of sites are grouped together. Stations = Study site. Widewall Historical, WI-C: Widewall Current Pairs of sites are grouped together. Stations = Study site.

Table 7.7. Creekland, abundance (ind. 0.1m^{-2}) of all taxa at Creekland ST9, ST10 and ST12 in Historical and Current time periods. Abundance of dominant taxa in bold, rare taxa in italics or in the lower section of the table. Statistical significance: $* \le 0.05$, $** \le 0.01$, $*** \le$ 0.001. Statistical significance refers to the outcome of permutation tests comparing abundances between Historical and Current periods.

Table 7.8. Cumminess, abundance (ind. 0.1m^2) of all taxa at Cumminess ST2 and ST4 in Historical and Current time periods. Abundance of dominant taxa in bold, rare taxa in italics or in the lower section of the table. Statistical significance: $* \le 0.05$, $** \le 0.01$, $*** \le 0.001$. Statistical significance refers to the outcome of permutation tests comparing abundances between Historical and Current periods.

Table 7.9. Dead Sand, abundance (ind. 0.1m^2) of all taxa at Dead Sand ST1 and ST2 in Historical and Current time periods. Abundance of dominant taxa in bold, rare taxa in italics or in the lower section of the table. Statistical significance: $* \le 0.05$, $** \le 0.01$, $*** \le$ 0.001. Statistical significance refers to the outcome of permutation tests comparing abundances between Historical and Current periods.

Table 7.10. Kirk Hope, abundance (ind. 0.1m^{-2}) of all taxa at Kirk Hope sampling station MLWS in Historical and Current time periods. Abundance of dominant taxa in bold, rare taxa in italics or in the lower section of the table. Statistical significance: $* \le 0.05$, $** \le 0.01$, *** ≤ 0.001. Statistical significance refers to the outcome of permutation tests comparing abundances between Historical and Current periods.

KIDK HOPE Dominant taxa (Historical > 18.63 ind. 0.1m⁻² Current > 7.32 ind. 0.1m⁻²)

Skeneidae 0.09

Veneridae 0.06

Table 7.11. Longhope, abundance (ind. 0.1m^{-2}) of all taxa at Longhope ST8, ST10 and ST12 in Historical and Current time periods. Abundance of dominant taxa in bold, rare taxa in italics or in the lower section of the table. Statistical significance: $* \le 0.05$, $** \le 0.01$, *** \leq 0.001. Statistical significance refers to the outcome of permutation tests comparing abundances between Historical and Current periods.

LONGHOPE - Dominant taxa (Historical ≥ 17.69 ind. 0.1m ² , Current ≥ 10.84 ind. 0.1m ²)											
	Mean Abundance $(ind. 0.1m-2)$ Historical Current		Statistical significance		Mean Abundance $(ind. 0.1 m-2)$ Historical Current		Statistical significance				
ANNELIDA				CRUSTACEA							
Arenicolidae	1.20	24.98		Ampeliscidae	28.46	7.74					
Capitellidae	38.17	6.44		Corophiidae	19.21	39.45					
Fabriciidae	503.92	0.20	$**$	Oedicerotidae	26.33	2.24					
Maldanidae		20.52	***	Pontoporeiidae	113.21	58.36					
Oligochaeta	137.68	10.58									
Opheliidae	33.37	15.77		MOLLUSCA							
Orbiniidae	19.34	23.11		Cardiidae	32.92	15.50					
Spionidae	475.84	52.83		Hydrobiidae	59.75	415.64	***				
Syllidae	180.65	231.70		Montacutidae	23.70	68.41					
				Tellinidae	1.64	31.89	***				
LONGHOPE - Rare taxa contributing less than 1% to the total abundance											
ANNELIDA				CRUSTACEA							
Aphroditidae		0.09		Aoridae	0.47						
Cirratulidae	0.73	0.88		Caprellidae	2.17	0.11					
Dorvilleidae	0.07			Cheirocratidae		0.05					
Glyceridae	0.20	0.29		Cirolanidae		0.08					
Hesionidae	0.27	0.06		Crangonidae	0.62	0.03					
Magelonidae		0.06		Cumacea	1.64	1.30					
Nephtyidae	6.04	9.94		Dexaminidae	0.13						
Nereididae	0.03			Gammaridae	2.14	0.45					
Onuphidae	0.03			Holognathidae	0.05						
Paraonidae	0.23	2.30		Hyalidae	0.27						
Pholoidae	0.06	0.05		Idoteidae	0.13						
Phyllodocidae	17.32	6.67		Ischyroceridae		0.05					
Polynoidae	0.03	0.30		Janiridae	0.26						
Psammodrilidae		2.76		Leucothoidae		0.05					
Scalibregmidae	0.23	5.15		Lysianassidae	0.50						
Sigalionidae		0.08		Melitidae		0.18					
Sphaerodoridae	0.93	0.05		Microprotopidae	7.98						
Terebellidae	0.03	0.23		Mysida	0.07	0.03					
Trichobranchidae		0.64		Phoxocephalidae	1.83	0.05					
				Pseudocumatidae		0.05					
HEXAPODA				Tanaissuidae		0.03					
Neanuridae	0.07			Urothoidae	0.29						
NEMERTEA	15.14	7.00		MOLLUSCA							
Heteronemertea		0.03		Akeridae	2.93	1.50					
				Mactridae	0.03	3.59					
PHORONIDA	0.03			Margaritidae	0.03	0.05					
				Murchisonellidae	0.50	4.08					
PLATYHELMINTHES		0.06		Myidae	0.40	0.77					
				Opisthobranchia	1.77						
SIPUNCULA				Pharidae	0.03	0.03					
Sipunculidea	0.03			Philinidae		0.24					
				Retusidae	7.43	8.80					
				Rissoidae	0.23	0.24					
				Skeneidae		0.05					

Veneridae 0.39 0.55

Table 7.12. Lyrawa, abundance (ind. 0.1m^{-2}) of all taxa at Lyrawa ST8 and ST10 in Historical and Current time periods. Abundance of dominant taxa in bold, rare taxa in italics or in the lower section of the table. Statistical significance: $* \le 0.05$, $** \le 0.01$, $*** \le 0.001$. Statistical significance refers to the outcome of permutation tests comparing abundances between Historical and Current periods.

Table 7.13. Mill Bay, abundance (ind. 0.1m^{-2}) of all taxa at Mill Bay ST8, ST10 and ST12 in Historical and Current time periods. Abundance of dominant taxa in bold, rare taxa in italics or in the lower section of the table. Statistical significance: $* \le 0.05$, $** \le 0.01$, *** \leq 0.001. Statistical significance refers to the outcome of permutation tests comparing abundances between Historical and Current periods.

	Mean Abundance $(ind. 0.1m-2)$		Statistical significance		Mean Abundance $(ind. 0.1 m-2)$		Statistical significance
	Historical Current				Historical Current		
ANNELIDA				CRUSTACEA			$**$
Capitellidae	279.47	203.67		Corophiidae	906.01	64.24	**
Cirratulidae	157.74	225.70	***	Phoxocephalidae	107.94	1.52	\mathbf{k}
Fabriciidae	2384.56	335.08	$\frac{d\mathbf{r}}{dt}$	Pontoporeiidae	247.90	81.94	
Oligochaeta Opheliidae	345.28	197.44					
Orbiniidae	33.40 67.33	104.41 18.45	$**$				
Spionidae	795.11	529.11					
Syllidae	30.34	106.56	***				
MILL BAY - Rare taxa contributing less than 1% to the total abundance							
ANNELIDA				CRUSTACEA			
Aphroditidae	0.12			Ampeliscidae	6.47	0.92	
Arenicolidae	3.37	0.97		Brachyura		0.03	
Cephalothrichidae		0.03		Calliopiidae	0.04	0.03	
Dorvillidae		0.03		Cirolanidae	1.38	1.77	
Eunicidae	4.06			Crangonidae	0.28	0.03	
Glyceridae		0.06		Cumacea	21.18	3.05	
Hesionidae	0.04			Dexaminidae	0.02		
Lumbrineridae	0.02			Gammaridae	0.09	0.27	
Magelonidae		0.11		Hyalidae	0.02		
Maldanidae	3.71			Microprotopidae	0.11		
Nephtyidae	0.24	0.53		Mysida	0.04		
Nereididae	0.16	0.26		Oedicerotidae	0.07	0.06	
Paraonidae	0.08			Portunidae	0.07	2.18	
Pholoidae	0.02			Talitridae	0.02		
Phyllodocidae	4.23	2.20		Tanaissuidae	0.04		
Psammodrilidae		8.48		Urothoidae	5.43	1.39	
Scalibregmidae		1.27					
Sphaerodoridae		0.06		MOLLUSCA			
Terebellidae	0.48			Akeridae		0.03	
				Cardiidae	0.87	1.42	
				Hydrobiidae		0.39	
NEMERTEA	29.38	5.11		Lepidochitonidae	0.02		
				Limapontiidae	0.11		
PLATYHELMINTHES				Mactridae Montacutidae		0.15 0.79	
		0.97			1.77		
				Murchisonellidae	0.20	3.83	
CHORDATA Ammodytidae	0.20	0.03		Myidae Opisthobranchia	0.10 0.16	0.06	
				Philinidae		0.21	
				Retusidae	11.21	9.70	
				Rissoidae	0.02		
				Semelidae			
				Skeneidae	0.02		
					0.02 1.27	2.73	
				Tellinidae		4.48	

Veneridae 0.01 0.14

Table 7.14. Quoys, abundance (ind. 0.1m^{-2}) of all taxa at Quoys ST7, ST10 and ST12 in Historical and Current time periods. Abundance of dominant taxa in bold, rare taxa in italics or in the lower section of the table. Statistical significance: $* \le 0.05$, $** \le 0.01$, $***$ ≤ 0.001. Statistical significance refers to the outcome of permutation tests comparing abundances between Historical and Current periods.

Table 7.15. Scapa Bay, abundance (ind. 0.1m^2) of all taxa at Scapa Bay ST6 and ST12 in Historical and Current time periods. Abundance of dominant taxa in bold, rare taxa in italics or in the lower section of the table. Statistical significance: $* \le 0.05$, $** \le 0.01$, $*** \le$ 0.001. Statistical significance refers to the outcome of permutation tests comparing abundances between Historical and Current periods.

Table 7.16. Swanbister, abundance (ind. 0.1m^2) of all taxa at Swanbister ST7 and ST12 in Historical and Current time periods. Abundance of dominant taxa in bold, rare taxa in italics or in the lower section of the table. Statistical significance: $* \le 0.05$, $** \le 0.01$, $*** \le 0.001$. Statistical significance refers to the outcome of permutation tests comparing abundances between Historical and Current periods.

Table 7.17. Waulkmill Bay, abundance (ind. 0.1m^{-2}) of all taxa at Waulkmill Bay ST10 and ST12 in Historical and Current time periods. Abundance of dominant taxa in bold, rare taxa in italics or in the lower section of the table. Statistical significance: $* \leq 0.05$, ** \leq 0.01, *** \leq 0.001. Statistical significance refers to the outcome of permutation tests comparing abundances between Historical and Current periods.

WAULKMILL BAY - Dominant taxa (Historical ≥ 3.07 **ind.** 0.1m^2 **, Current** ≥ 5.49 **ind.** 0.1m^2 **)**

Table 7.18. Widewall Bay, abundance (ind. 0.1m^{-2}) of all taxa at Widewall Bay ST8 and ST12 in Historical and Current time periods. Abundance of dominant taxa in bold, rare taxa in italics or in the lower section of the table. Statistical significance: $* \le 0.05$, $** \le 0.01$, $*** \le$ 0.001. Statistical significance refers to the outcome of permutation tests comparing abundances between Historical and Current periods.

7.4.4 Between site comparison of macroinvertebrate communities (2006-2016)

The MDS ordination shows separation between sites with some overlap (Figure 7.5) demonstrating that between-site differences were greater than within-site differences. The macroinvertebrate communities in the Scapa Flow sites were most similar for sites close to one another compared to more distant sites. Cumminess and Congesquoy, both within Bay of Ireland were similar to each other in their macroinvertebrate communities; Creekland and Quoys; Mill Bay and Lyrawa; Kirk Hope and Longhope; and Scapa, Waulkmill and Swanbister were plotted in the same order as they are from west to east (Figure 3.1). Dead Sand and Widewall are distinctly separate from the other sites. Quoys had the largest within-site variability shown by the big ellipse encircling the samples, compared with Dead Sand which has a very small, highly grouped samples over the 2006- 2016 period (Figure 7.5).

The first (PC1) and second (PC2) principal components of the Principal Components Analysis (PCA) on physical characteristics of the sandy beaches explained 61.3% and 17.3% of the total variance, respectively (Figure 7.6). Wave frequency, wave height and wave frequency are the main influencing factors on PC1, grading the sandy beaches depending the wave climate (Figure 7.6). Sandy beaches which are south-facing (Scapa, Waulkmill, Swanbister and Cumminess) are all grouped together to the left of PC1, in very similar fashion to the order they are in the MDs ordination (Figure 7.4). Dead Sand is separated from the other sites to the left of PC1, again mirroring the MDS ordination. Further analysis is required to understand the full relationship of the physical and biological factors.

Figure 7.5. Multidimensional scaling (MDS) ordination of Scapa Flow sandy beach sites, 2006-2016. Each dot represents a year.

Figure 7.6. Principal Components Analysis (PCA) ordination of Scapa Flow sandy beach sites, 2006, 2014-2016. Each label represents a year.

7.5 Discussion

Oceanic high energy, reflective sandy beaches have a lower species diversity and abundance of macroinvertebrates compared to sheltered low energy, dissipative beaches (Brown & McLachlan 2002). Scapa Flow is an enclosed water body with no ocean swell and only wind generated waves (Howie et al. 1975; Barne et al. 1997). Within Scapa Flow there are many islands and embayments making some areas even more sheltered (Barne et al. 1997). The mean number of taxa in the Scapa Flow sites was 48, with the lowest taxa recorded at Scapa (25) and Dead Sand (32). The lowest numbers of taxa were not observed in the most exposed locations but in the sites with a history of disturbance. Scapa Bay is characterised as Dissipative: non-barred Beach Type (Chapter 3 Table 3.4) and was in Historical time period moderately disturbed (Figure 7.2). The Scapa sandy beach site has improved in its ecological quality but the number of taxa at the site has remained low. Dead Sand sandy beach site has been alternating between the classification types Ultra-dissipative and Intermediate: bar with rip channels (Chapter 3 Table 3.10) due to the change in the granulometry of the site. Dead Sand is an embayment within Bay of Ireland with shallow shore profile and has been classed as moderately disturbed since 1984. Degradation and high levels of organic enrichment decrease the species diversity but increases the abundance of the species present (Gray 1979; Gray & Elliott 2009). The two sampling stations included in this analysis for both Scapa and Dead Sand were located high on the shoreline $(>0.77m$ from MLWS level) compared to other sites (Chapter 3 Table 3.20) which could also explain the low taxa diversity as the diversity of macroinvertebrate communities increase lower down the shoreline (Dexter 1984), therefore explaining the low taxa diversity at the two sites. The highest number of taxa was observed at Longhope (76 taxa) which is classified as Dissipative: non-barred Beach Type and is located within sheltered embayment of North Bay; the lowest sampling station at Longhope is at -0.14m from MLWS (Chapter 3 Table 3.20) which is considerably lower than the stations at Scapa or Dead Sand. On average 18 more taxa were recorded at the sites with three sampling stations (average number of taxa 61) compared to sites with two sampling stations (average number of taxa 43), highlighting that increased number of sampling stations along the intertidal area increases the number of taxa recorded.

The numbers of taxa at the Scapa Flow sites were higher than in sheltered sandy beaches elsewhere in Scotland (Eleftheriou & McIntyre 1976; Eleftheriou & Robertson 1988). Forty-four species were recorded at a sheltered Traigh Mhor beach on island of Barra. The most abundant species at Traigh Mhor were Spionidae (*Pygospio elegans*), Oligochaeta, Capitellidae and Cardiidae (*Cerastoderma edule*) (Eleftheriou & McIntyre 1976). Forty-six species were recorded at Gullane, a sheltered beach in Firth Forth (Eleftheriou & Robertson 1988). The Scapa Flow macroinvertebrate data were aggregated to family level or higher, therefore the number of species at the Scapa Flow sites would be higher than the aggregated value. The high diversity at the Scapa Flow sites reflects the very sheltered nature of the sites. In comparison an exposed sandy beach on the north coast of mainland Scotland, Dunnet Beach, only had ten species of macroinvertebrates recorded of which Spionidae (*Scolelepis squamata*), Paraonidae (*Paraonis fulgens*) and Oedicerotidae (*Pontocrates norvegicus*) were the most abundant (Eleftheriou & McIntyre 1976). The mean number of macroinvertebrate species (24 species) on the sandy beaches of East Coast of Scotland were comparable with the temperate North Sea ecoregion mean number of 15-20 species estimated by Barboza & Defeo (2015). In their review of 256 sandy beaches around the world, when calculating the number of species for the North Sea ecoregion Barboza & Defeo (2015) only included sandy beach macroinvertebrate data from north coast of Spain and Belgium. No data from Scotland or any other country in the northern part of North Sea was included. The number of species stated by Barboza & Defeo (2015) for the North Sea region is lower than the East Coast of Scotland research has shown (mean number of 24 species) and it is considerably lower than the mean number of taxa (48 taxa) in the current study.

Three taxa were observed as dominant taxa at every sandy beach site in Scapa Flow: Spionidae, Pontoporeiidae and Capitellidae. Atkins et al. (1985) recorded four taxa, Spionidae (*Pygospio elegans*), Pontoporeiidae (*Bathyporeia* sp.), Capitellidae (*Capitella capitata*) and Oligochaeta, as ubiquitous and amongst the dominant taxa at the 14 sandy beach sites they surveyed in Orkney. Oligochaeta was not one of the ubiquitous dominant taxa in the current study as two new sites were included in the monitoring programme, Congesquoy and Cumminess, from 1983 and 1984, respectively. Oligochaeta were recorded at Congesquoy and Cumminess but as rare in both Historical and Current time periods (Tables 7.6 and 7.8). These two sites (Congesquoy and Cumminess) were not part of the macroinvertebrate fauna reviewed by Atkins et al. (1985) as their study discussed surveys carried out in 1981 and 1982. The presence of the three dominant taxa (Spionidae, Pontoporeiidae and Capitellidae) at the Scapa Flow sites in Historical and Current time periods is an important factor to consider, loss of any of these taxa from a site would signify a change in the baseline macroinvertebrate community and would require further investigation to determine a cause for the change. The results from Scapa Flow sites showing that three dominant taxa were present at all sites from Historical to

Current time period is reflected at Swansea Bay and at East Frisian island of Norderney, Germany (Dörjes et al. 1986; Callaway 2016). From benthic surveys carried out in 1984 and in 2014 in Swansea Bay, five common species were found to be persistent over time *Nucula nitidosa* (Nuculidae), *Spisula elliptica* (Mactridae), *Spiophanes bombyx* (Spionidae), *Nephtys hombergii* (Nephtyidae) and *Diastylis rathkei* (Cumacea) (Callaway 2016). The constancy of the same eight dominant taxa over time (1976-1985) were described for the intertidal macroinvertebrate communities at East Frisian island of Norderney, Germany (Dörjes et al. 1986).

Macroinvertebrate taxa can be rare either spatially or in terms of overall abundance (Resh et al. 2005). In this thesis rare taxa were assigned as such according to their abundance (<1% of the total faunal abundance (Jones et al. 1985)). The rare taxa were separated from the dominant taxa when describing the baseline macroinvertebrate communities, but they were retained in the table as rare taxa could have dropped in and out of the dominant category between Historical and Current time periods. In a study on rare species in macroinvertebrate community analysis Checon & Amaral (2017) concluded that use of dominant species was sufficient in describing the changes in a macroinvertebrate community. They recommended the use of family- and genus-level identification but highlighted that genus-level identification should be preferred. In routine monitoring surveys there is much debate about the use of different taxonomic groups and the level of taxonomic sufficiency required for monitoring. Identifying polychaetes to family level is seen as sufficient by Olsgard & Somerfield (2000) whilst Bevilacqua et al. (2009) consider family level identification being sufficient for molluscs. Family level identification for macroinvertebrates in benthic monitoring is considered suitable for pollution and disturbance monitoring (Ferraro & Cole 1990; Warwick et al. 1990; Somerfield & Clarke 1995; Roach et al. 2001; Dauvin et al. 2003; Ruso et al. 2007; Dela-Ossa-Carretero et al. 2012; Chatzinikolaou et al. 2018).

The taxa which were dominant either in Historical or Current time period were combined to describe the baseline macroinvertebrate community against which any future changes can be compared against.

Shifting baselines can be an issue where no historical baseline is present and when a baseline is set using the current knowledge only with no understanding of past activities which might have affected the community in question (Pauly 1995: Humphries $\&$ Winemiller 2009; Villnäs & Norkko 2011). The understanding of spatial and temporal variability of macroinvertebrate communities and environmental variables is a key factor for setting baselines in any environment (Villnäs & Norkko 2011). At the Scapa Flow sites long-term Historical and Current macroinvertebrate data were available and baselines were set and described for both time periods. At each of the 13 sites the baseline macroinvertebrate community would have been different had the baseline been set using the dominant taxa at either Historical or Current time period only. A shift in the baseline community from Historical to Current time period due to the variability in the abundances of the dominant taxa was observed at 11 sites: Congesquoy, Creekland, Cumminess, Dead Sand, Kirk Hope, Lyrawa, Mill Bay, Scapa Bay, Swanbister, Waulkmill Bay and Widewall Bay. At Longhope and Quoys, there were a shift in the baseline community due to both variability in abundances of the dominant taxa and to the presence of new dominant taxa in the Current time period. One new taxon (Maldanidae) at Longhope and two new taxa (Paraonidae and Platyhelminthes) at Quoys were recorded at Current time period only. The changes in the abundances of the dominant taxa from one time period to another could be due to natural population fluctuations; macroinvertebrate community populations are known to fluctuate both annually and seasonally (Warwick & Clarke 1993; Atkins et al. 1989; Atkins & Jones 1990). Apart from natural population fluctuations causing the change in the macroinvertebrate community abundances, several other sources of variability should be considered as possible cause for the variability: samples were collected at two different time periods (Historical and Current) each period with different sampling methods (Chapter 2 Section 2.1) including time of the year when samples were collected, sampling personnel and sample processing; the data available for each sandy beach site at Historical time period ranged from 6-15 years and for Current time period from 11-15 years. (Table 7.2). The Historical macroinvertebrate samples were collected in 1970s and 1980s during the summer months, the Current macroinvertebrate samples were collected in 2000s and 2010s in the winter and early spring. When comparing the mean abundances of macroinvertebrates between these two time periods differences in the taxa abundances could have been due to the timing of the sample collection, different people collecting the samples, inconsistencies in the sample processing as discussed in Chapter 4, or due to the populations actually having different abundances in the two time periods. Therefore, a change in the abundance of a taxon will not be considered as a significant change in the baseline macroinvertebrate community.

None of the Scapa Flow sites were classed as undisturbed by the AMBI analysis (Figure 7.4). Organic pollution events or persistent run-off from agricultural land, oil pollution, wrack subsidies and storm events all could disturb the natural balance of macroinvertebrate communities (Morton & Sallenger 2003; Defeo et al. 2009; Engel et al. 2009). The slightly disturbed status of the sandy beaches in Scapa Flow show that each one of the sites were under some level of disturbance. Scapa Bay, Dead Sand and Swanbister Bay were sites with clear pollution events causing the lowered ecological quality, at other sites the reason for the slightly disturbed status is unclear. The abundances of the taxa present were an integral part of the calculations when the macroinvertebrate populations were used for evaluating the ecological quality and benthic community health (Borja et al. 2000). A change in the abundance of a taxon or several taxa can influence the AMBI calculation and therefore change the perceived ecological quality of a sandy beach. Kirk Hope, Lyrawa and Scapa all improved their ecological quality from Historical to Current time period (Figure 7.4). Swanbister is the only site where the ecological quality worsened from Historical to Current time period. In these cases, change in the abundances of the taxa present changed the ecological quality and benthic community health and was therefore seen as a significant change in the ecological quality.

The macroinvertebrate communities of the sites closest to each other had similar macroinvertebrate communities compared to sites further apart. Transition of macroinvertebrate communities from site to site was demonstrated by the organisation of three of the sites in the MDS ordination and PCA plots, namely Scapa, Waulkmill and Swanbister, which were in the order as they are on the shores of Scapa Flow from east to west. The baseline macroinvertebrate communities at the Scapa Flow sites are driven by localised effects which in some cases can be narrowed down to two or three beaches being more similar compared to other beaches. Two sites, Dead Sand and Widewall were separated from the other sites, indicating that their macroinvertebrate baseline communities were at distinct extremes of a continuum of variation across sites. Local and small-scale recruitment has been stated as a factor driving the similarities of adjacent sandy beach macroinvertebrate communities (Checon et al. 2018). Dead Sand and Widewall, which on the MDS ordination were separated from the other sites, are both based in enclosed bays with high spatial separation from the other sites. Recruitment from the nearby sandy beaches to these sites is less likely, explaining the dissimilarity of their macroinvertebrate communities.

The descriptions of the macroinvertebrate baseline communities, ecological quality and benthic community health information for the 13 Scapa Flow sites will now be used as the baseline for the Orkney Islands Council Harbour Authority's (OICHA) on-going sandy beach monitoring programme, against which any future changes can be compared.

Further detailed analysis is required to fully understand the cause for the taxonomic group changes at Longhope and Quoys and for the ecological quality changes in Kirk Hope, Lyrawa and Scapa and Swanbister.

7.5.1 Conclusions

The Scapa Flow sites macroinvertebrate community diversities were in accordance with sheltered, low energy sandy beaches with high number of taxa, each taxon with relatively low abundance. The baseline macroinvertebrate communities were described on the basis of dominant taxa (>1% total faunal abundance), three taxa (Spionidae, Pontoporeiidae and Capitellidae) were ubiquitous to all sites. The abundances of the baseline macroinvertebrate communities were variable but only two sites (Longhope and Quoys) had changes in the dominant taxonomical groups. Four sites had changes in their ecological and benthic community health: three (Kirk Hope, Lyrawa and Scapa) improved and one (Swanbister) worsened. The baseline macroinvertebrate community descriptions and status of the ecological quality of each beach will enable annual comparison of the results from the monitoring programme to advise OICHA, and other interested stakeholders, if, and to which extend, any changes have occurred from year to year. This timely analysis of the data and ecosystem status has not been possible prior to this work.

Chapter 8 Discussion

This research work undertaken during this project is the first time the long-term macroinvertebrate time series data from the 13 Scapa Flow sandy beaches have been brought together and analysed. The benthic macroinvertebrate data were collected during two different time periods under two different monitoring regimes with a gap of 13 years between them. The Orkney Islands Council Harbour Authority (OICHA) have been the custodians of the Historical and Current macroinvertebrate monitoring data from the 13 Scapa Flow sandy beaches since 1974. Since 1990 when a scientific paper on four of the sites, Scapa, Waulkmill, Swanbister and Mill Bay for the time period 1974-1988, was published (Atkins & Jones 1990) the macroinvertebrate data have remained untouched. The monitoring programme was re-started in 2002 (as detailed in Chapters 1 and 2) and continued annually but with no data analysis. Previous to the research reported in this thesis, the Marine Environmental Unit (MEU) at OICHA had not been able to provide an on-going assessment of the state of the Scapa Flow sandy beaches.

8.1 Summary of the thesis research process

Prior to analysis a considerable time was spent in finding, examining and preparing the Historical and Current data. In the process of this research, it has become evident that two wider aspects of the monitoring programme needed improvement: 1) sample collection and field surveys, and 2) sample processing, identification and macroinvertebrate data quality. Each of these topics will be addressed now.

1) Sample collection and field surveys.

In the Historical time period granulometry samples were collected from each sampling station at the same time as the macroinvertebrate samples. Granulometry samples had only been collected once, in 2006, at Current time period. The granulometry sample collection has now been included in the monitoring programme since 2014. The collection and analysis of the granulometry samples enabled the beach morphometric calculations (Appendix D), which in turn enabled the classification of the beaches to Beach Types (Chapter 3) and the comparison of Beach Types between Historical and Current time periods. The sediment of a sandy beach is an important physical parameter as different macroinvertebrates have a preference for different sediment grain sizes (Eleftheriou & McIntyre 2005) and a change in the granulometry at a site can be due to change in the wave climate or due to increased storminess. In 2016 shore profiles of the Scapa Flow sandy beaches were surveyed (by K. Cooper on contract to OICHA);

previously the shores had been profiled in 1981 and 1982 (Atkins et al. 1985). The shore profile surveys mapped the sampling stations at each site and recorded the heights of each sampling station in relation to MLWS. The knowledge of the heights of the sampling stations informed the site and station selection for the Scapa Flow-wide data analysis (Chapter 5). Organic carbon samples were collected from the sampling stations in the Historical time period. Samples for organic carbon were collected in 2019, for the first time in the Current time period.

2) Sample processing, identification and macroinvertebrate data quality.

The absence of detailed sample processing and identification guidelines and Quality Control procedures in the Current time period were highlighted in Chapter 4. During this thesis research the following positive outcomes have assisted in the understanding of the sample processing, identification and macroinvertebrate data quality: i) attendance at a taxonomic workshop, ii) the identification of all samples from 2014 by a taxonomic laboratory, iii) the creation of voucher specimen collection, iv) development of Scapa Flow sandy beach-specific identification guide, and v) the re-identification and reenumeration of all macroinvertebrate samples from 2002-2013 for three sites (Quoys, Congesquoy and Waulkmill).

Once the inconsistencies and variability in the macroinvertebrate data were fully explored in Chapter 4, spatio-temporal variability was investigated. Firstly, Scapa Flow-wide (Chapter 5) to understand if any regional, large scale patterns were present, and secondly at three sites (Quoys, Congesquoy and Waulkmill) to understand sampling stationspecific spatio-temporal variability and to investigate changes in the macroinvertebrate communities between the two time periods (Chapter 6). To establish the baseline macroinvertebrate communities and to determine the environmental condition of each site, data from both Historical and Current time period were used (Chapter 7). Now, for the first time since 1974, baseline macroinvertebrate communities and environmental conditions for the 13 Scapa Flow sandy beaches have been established, against which any future changes can be compared.

8.2 Beach morphometrics

The widely used Beach Index (Short & Wright 1983; Masselink & Short 1984; McLachlan & Defeo 2018), was used to categorise the Orkney sandy beaches. All sites were classified as Dissipative or Ultra-dissipative, with five (Dead Sand, Creekland, Quoys, Lyrawa Bay, Mill Bay, and Kirkhope Bay) also classified as Intermediate in some
years during the monitoring period. For nine of the beaches the Beach Type did not change from Historical to Current time period, indicating that at these beaches there have not been any changes in the physical characteristics of the beaches since 1974. At the four sites (Dead Sand, Lyrawa, Mill Bay and Kirk Hope) which changed in their Beach Type, their mean grain size increased in the Current time period, i.e. the sediment became coarser. This could be due to the samples been collected in the winter time in the Current time period compared to in the summer time during the Historical time period. It would be an interesting and informative exercise to collect the sediment samples from these four beaches at the same time of the year as the Historical sampling was carried out to inform if the change in the beach type is due to seasonal effect or due to changes over time.

The sandy beaches are all located within Scapa Flow, a naturally sheltered body of water and the classification of the sites reflects the sheltered nature of the area. The sites are in low wave energy areas, the longest fetch being approximately 13 km, for the southerly facing Scapa Bay (Barne et al. 1997). Sheltered sandy beaches have a higher species number compared to exposed beaches (Brown & McLachlan 2002), the high mean number of taxa (48) at the Scapa Flow was in agreement with this statement. The Beach Type was not a predictor of macroinvertebrate community structure (Chapter 7), beaches which were near to each other had more similar macroinvertebrate communities compared to beaches further apart with same Beach Type.

8.3 Data inconsistencies

Data quality and accuracy are of paramount importance in benthic macroinvertebrate long-term monitoring programmes (Ellis 1988; Ranasinghe et al. 2003; Stribling et al. 2003; Stribling et al. 2008) where samples are collected, and results compared against each other to detect any responses to possible changes in the environment. The most common type of inconsistency in the three Scapa Flow sites examined in detail were the use of old taxonomic names or consistently misidentifying a taxon and using an incorrect family name. These inconsistencies did not affect the detection of overall patterns and trends within the set of sites. However, without the verification process these inconsistencies would have remained unnoticed in the data with the potential for errors and misleading interpretations to influence ongoing monitoring. The post-sampling procedures of sample sorting, identification and enumeration all are liable for operator variability and for errors (Ranasinghe et al. 2003; Stribling et al. 2003; Stribling et al. 2008). All the samples in this study had already been sorted so only the re-identification and re-enumeration were possible. Enumeration was found not to be a significant issue,

the error at all but one sampling station being <1%. Ranasinghe et al. (2003) reported enumeration error values of 1.0-3.1% whereas Stribling et al. (2008) reported enumeration error rates of 0.5-1.9% with one sample as high as 29.6%. The percentage of taxonomic disagreement (PTD) (which measures how accurately two analysts have identified a set of samples and gives a percentage for the number of disagreements) at the three sites examined varied from 1.2-16.1% which is lower compared to levels of 8.1- 29.6% recorded by Stribling et al. (2008) for their freshwater macroinvertebrate samples. Three taxa were highlighted in the three Orkney sandy beach sites as problem taxa for identification, namely polychaetes belonging to families Capitellidae and Paraonidae, and annelids belonging to the class Oligochaeta. Oligochaeta are known to be difficult to identify (Worsfold 2003); the analysts at the MEU were not familiar with the two different families (Enchytraeidae and Naididae) and four different genera present (*Grania*, *Baltidrilus*, *Paranais* and *Tubificoides*) and how to identify them. The Oligochaeta recorded from the Orkney beaches vary morphologically from the very small, opaque Enchytraeidae to large and conspicuous Naididae. Three different genera (*Capitella*, *Mediomastus* and *Notomastus*) belonging to the family Capitellidae were recorded from the Orkney sites, all of which have the same morphological appearance of narrowing gradually towards the posterior end. The presence of several different taxa from Oligochaeta and Capitellidae, the variability of the morphological characteristics within both Oligochaeta and Capitellidae and the inexperience of the analysts carrying out the work led to taxonomical errors. Stribling et al. (2003) highlight the importance of QC in all laboratory processes; no QC protocols were in place at the MEU during the Current time period (2002-2016). The implementation of QC procedures for sample sorting, identification, enumeration and data entry and regular updating of these procedures was implemented in 2017 as a result of the research undertaken for this thesis. These procedures will enable direction of training and guidelines and to highlight any areas which need improving. Other improvements to the laboratory processes were the ongoing updating of the OICHA-based voucher specimen collection; during the identification process if analysts come upon any difficult specimens to identify these are removed and sent for expert identification. The MEU is a stand-alone small laboratory serving the Orkney Harbour Authority and the Orkney Islands Council. As such there is no requirement for the MEU to join the UK-wide NE Atlantic Marine Biological Analytical Quality Control Scheme (NMBAQC) of which commercial and government laboratories working in regulatory roles participate. The improved procedures, including the analyst's regular attendance at the NMBAQC taxonomic workshops, will ensure that the macroinvertebrate data from the programme is of high quality. High quality data are defined as data that are accurate and fit for data users (Strong et al. 1997).

8.4 Challenges and limitations

There were challenges in using the Orkney macroinvertebrate data. The Historical monitoring programme was implemented and carried out by a team of scientists from Dundee University. Extensive macroinvertebrate samples and environmental data were gathered at each monitoring site and several publications on the monitoring programme were published (e.g. Jones 1980; Atkins et al. 1985; Atkins et al. 1989; Atkins & Jones 1990). The published manuscripts and the annual Orkney Marine Biological Unit (OMBU) reports give summaries of all the monitoring carried out in the Historical time period (Jones 1974; Jones & Simpson 1976, 1977; Jones et al. 1978, 1979; Jones 1980; Jones et al. 1981, 1982; Jones 1983, 1985; Jones et al. 1986-1991). In each annual OMBU report it is mentioned that all the specific details of each survey including all raw data from each survey are held at OICHA for future reference. However, at the start of this thesis research and once all the data files and folders were located from the OICHA archives, and entered into spreadsheets, it was clear that not all of the macroinvertebrate data were there. Months of going through all areas of the archives followed but no further data files were located. All the personnel at Dundee University who were involved in the Orkney monitoring programme left the university, through retirement or otherwise, and the unit carrying out the work (the Environmental Advisory Unit) had ceased business, therefore decision was made to continue with the data available. In the 1970s and early 1980s computerised data storage was not readily available; all data were entered longhand into notebooks or typed using typewriters. Long-term storage of such data required good organisation and enough space for storage. It is most likely that all the data were once held at MEU, but unfortunately the full complement is not currently at OICHA. The data deficiencies due to the data not being available and any influence on data analysis were demonstrated by the Waulkmill ST10 and ST12 data analysis (Chapter 6). Several apparently significant clusters were identified, differing in their macroinvertebrate community composition, and all these were shown to be artefacts of data deficiencies. From the point of view of integrity of monitoring and thus the usefulness of long-term time-series, it is unfortunate that the programme was terminated in 1990.

8.4.1 Changes in the monitoring programme

When the monitoring was re-started in 2002 several key elements of the programme were changed. This included changes in, the time of the sample collection to winter months compared to summer months in the original monitoring programme. Atkins et al. (1989) sampled two Orkney beaches (Scapa Bay and Waulkmill Bay) every 4-8 weeks for four years and they were able to demonstrate that the most suitable period for annual sampling at these sites in Scapa Flow would be June. The mid-summer sampling would indicate this to be the best time for annual sampling, with reduced influence of interannual variations in over winter mortality and the timing of recruitment (Atkins et al. 1989). The change from the summer sampling to winter sampling confounds seasonal variation with real differences between the Historical and Current time periods.

Another aspect of the monitoring programme that was changed when the sampling was re-started in 2002 was the sampling of selected stations, maximum three, along the transect line, compared with all stations in the soft sediment sections of the shoreline in the Historical time period. Atkins et al. (1985) detail the bedrock or shingle sections of the shoreline which were not sampled. The three Hoy sites described (Creekland, Quoys and Mill Bay) were all bedrock or shingle down to ST6, ST6 and ST7, respectively. The first sampling station along each transect line in the Current time period was the first sediment station, ST7, ST7 and ST8, for Creekland, Quoys and Mill Bay respectively. The pattern of sampling at Hoy sites, where highest stations sampled were immediately below the rock or shingle (e.g. Quoys ST7, Figure 3.19), was not followed at all Mainland sites. In most cases, the highest stations sampled for Mainland sites, were some distance below the top of the sandy part of the shores (e.g. Waulkmill ST10, Figure 3.6). At the Bay of Ireland (Congesquoy, Cumminess and Dead Sand) the stations monitored were labelled ST1-ST4 but at both Congesquoy and Cumminess the sampling stations were at lower shore levels compared to other sites and stations labelled similarly (Chapter 3 Table 3.20). The inconsistencies in the selection of the stations for monitoring at the different sandy beach sites makes the macroinvertebrate data comparison between sites more difficult. Macroinvertebrate species and their abundance are influenced by tidal level (Dexter 1984) and comparing populations from different tidal levels is not comparing like with like. The Scapa Flow sandy beach sites were fixed at 30 cm vertical height intervals (Jones 1985), in preliminary data analysis stations which were maximum 30 cm height difference apart were not significantly different (Kakkonen 2016). Data from different sandy beach site sampling stations were selected for Scapa Flow-wide analysis (Chapter 5) on the basis that their measured height (surveyed in 2016) was within 30 cm vertical height interval. This process of selecting sampling stations was considered the best available method for the comparison of between sites.

8.5 Rationale for the Scapa Flow sandy beach monitoring programme

No Scapa Flow wide trends were seen but site-specific trends were observed at Quoys, where the macroinvertebrate community changed from 2011 onwards. This change was linked with extreme cold weather events in two years running, cumulatively affecting the macroinvertebrate communities at the site. It was not possible to ascertain why other sites were not affected by the cold weather. It might be considered that characteristics of a good monitoring site include the requirement for the site to be in an undisturbed state and for the taxa present to have minimal natural variability both in numbers of taxa and in abundances; these characteristics would indicate a relatively stable overall condition against which any disturbance can be measured. At the three of the Scapa Flow sandy beach sites (Quoys, Congesquoy and Waulkmill) large interannual population and community fluctuations were present at all sampling stations and all of the sites were classed as having at least slightly disturbed ecological status. Regardless of this, the macroinvertebrate baseline community and the dominant taxa remained the same from Historical to Current time period in all, but two sites and the ecological quality status of the sites remained unchanged at all but four sites. Although the sites have large interannual fluctuations in the macroinvertebrate population abundances in the long-term, they are remaining constant. Three taxa were ubiquitous to all of the Scapa Flow sandy beaches, namely Pontoporeiidae, Spionidae and Capitellidae. The three taxa might be worth investigating if they had the required properties to monitor disturbances and become target taxa for monitoring in Scapa Flow. Target taxa are commonly used in nonnative species monitoring programmes (Bishop & Hutchings 2011; Collin et al. 2015) and the use of target taxa could potentially be considered for monitoring sandy beaches.

8.5.1 Monitoring for oil pollution

The monitoring of the sandy beaches in Scapa Flow can be described as operational monitoring (Gray & Elliott 2009). Operational monitoring, as defined by Gray and Elliott (2009) (Table 1.1), is "monitoring which is related to a specific human activity and is carried out to establish the status of the sea at risk and to assess changes in that status resulting from programmes of measures". Nine of the Scapa Flow sandy beach monitoring sites (Creekland, Longhope, Lyrawa, Mill Bay, Quoys, Scapa, Swanbister, Waulkmill and Widewall) were established to monitor to provide baseline data and reference condition in case of an oil pollution. There have been no recorded oil spills attributable to the operation of the Flotta Oil terminal or due to ship-to-ship oil transfers in Scapa Flow (OICHA unpublished data). The Flotta Oil Terminal is still operational, and the oil cargo ship-to-ship transfers have been on the increase since 2015 (Chapter 1

Section 1.2.1). If there was an oil spill in Scapa Flow, the macroinvertebrate data from the on-going monitoring of sandy beaches would become an important part in monitoring the recovery of the area. The macroinvertebrate data would be used to detect changes in community composition and abundances and in ecological quality status, as was demonstrated in Chapter 7 by using AMBI software. De la Huz et al. (2005) were able to describe the changes in the macroinvertebrate communities by comparing data collected prior to the 'Prestige' oil spill in 2002 and immediately after the event, observing a loss of up to 66.7% of the total species richness in the most affected beaches. Benthic macroinvertebrates have been used to evaluate the changes in the benthos, reduction in macroinvertebrate diversity and abundances, and its recovery in soft sediment environments after many oil spill events: 'Braer' in the Shetland Islands (Kingston et al. 1995), 'Exxon Valdez' in Prince William Sound, Alaska (Feder & Blanchard 1998), 'Hebei Spirit' in west Korea (Yu et al. 2013) and 'Deepwater Horizon' in the northern Gulf of Mexico (Beyer et al. 2016). On-going sampling is an essential part of describing the potentially shifting baseline against which impacts would be detected. Given the ongoing activities of the oil industry in and around Scapa Flow there is a continuing need to be prepared for oil spill events.

The long-term intertidal macroinvertebrate monitoring programme is unique to Orkney. SOTEAG in Shetland began intertidal macroinvertebrate monitoring of Sullom Voe in 1977 but discontinued it after eight years (Jones 1995). Many 'then' and 'now' surveys have been carried out for intertidal macroinvertebrates (Borja et al. 2010; Chainho et al. 2010; Schooler et al. 2017), but none have continued their monitoring over such a long time period as the OICHA programme.

8.5.2 Monitoring for the effects of organic enrichment

Three of the Orkney sandy beach monitoring sites, namely Congesquoy, Cumminess and Dead Sand, were set up to monitor the effluent discharges from the Stromness waste water treatment facility (WWTF). The three sites had not shown any effects due to organic pollution from the sewage effluent during monitoring carried out in 1984-1989 (Atkins et al. 1991), and two out of the three sites, Congesquoy and Cumminess, have not shown any deterioration during the Current monitoring programme (Chapter 7 Section 7.4.2.1 Congesquoy, Section 7.4.2.3 Cumminess). Dead Sand is a shallow, enclosed embayment in the north-eastern area of Scapa Flow and it was highly enriched with organic matter in 1984-1989, the source of which was considered to be agricultural run-off into the Loch of Stenness (Atkins et al. 1991). Nutrient enrichment from diffuse source pollution is still a cause of concern in the Loch of Stenness and Bay of Ireland area (SEPA 2016). In both Historical and Current time periods the fauna at Dead Sand was dominated by organic pollution indicator species, including Oligochaeta and Capitellidae, and suspension feeders, Spionidae and Fabriciidae. The environmental condition of the site has remained moderately disturbed since 1984 and its continuing worth as a monitoring site is questionable, however it could act as a reference point for the moderately disturbed state. Also, as Dead Sand has been part of an on-going monitoring programme for over 30 years, stopping the macroinvertebrate monitoring at this site would mean breaking a long data series. Instead of discontinuing monitoring at this site, reductions in monitoring frequency allowing redirection of resources to less disturbed sites might be justifiable. Effective approaches to reducing sampling frequency, whilst retaining the ability to detect recovery, should be investigated. This would allow continued monitoring at this site and would be more cost-effective. Currently the sample identification and enumeration take considerable time due to high abundance of several of the taxa at both ST1 and ST2 samples. The point source pollution from the Stromness WWTF was highlighted as one of the pressures on Scapa Flow coastal areas including Loch of Stenness (SEPA 2016). In addition, in 2017 the waste water network in Stromness was upgraded and all Scottish Water sewers in Stromness were linked to the Stromness WWTF (Scottish Water 2019), which will have increased the throughput of waste water at the WWTF. The continued monitoring at these two sites will be able to highlight if the upgrading of the sewage network in Stromness has an effect on the intertidal macroinvertebrate communities.

8.5.3 Potential for non-native species monitoring

All 13 sandy beach monitoring sites could complement the on-going marine non-native species (NNS) monitoring programme in Scapa Flow (Kakkonen et al. 2019). The NNS monitoring programme currently includes only sub-tidal soft sediment sites (Kakkonen et al. 2019), and the addition of intertidal soft sediment sites to the NNS monitoring programme would enhance the programme without adding any extra fieldwork to the MEU schedule. For this to be possible, all the samples would need to be identified to species level, as the identification of taxa to genus or family level would not provide the information required for NNS monitoring (Ojaveer et al. 2014). Species level identification has been implemented for all Scapa Flow sandy beach samples collected from 2017 onwards. The identification of 2017 samples were started in 2019, after a year-long delay while new procedures were being put in place for each step of the sample processing. A trial period is now being run where the time taken in each stage of the sample processing, including identifying all samples to species level is being recorded. Once macroinvertebrate samples from 2017 are identified to species level, and after any

specimens which have been put aside for sending away to taxonomic laboratory for identification, have been completed, a full review of the cost effectiveness of the species identification will be carried out.

8.5.4 Potential for Climate Change monitoring

Climate change is associated with sea temperature increase, sea level rise and increased storminess (Brown & McLachlan 2002; Doney et al. 2012). Sea level rise and increased storminess will lead to increased erosion and habitat loss at sandy beaches, which combined with increased sea temperatures will lead to species and population changes due to altered environmental conditions. Long-term data sets are vital in understanding changes in marine ecosystems (Brown & McLachlan 2002; Doney et al. 2012; Schooler et al. 2017) and the Orkney sandy beach macroinvertebrate data are unique in that respect. Orkney Islands are in an interesting geographical position for many species' distribution. Rocky shores in Orkney have been part of the Marine Biodiversity and Climate Change (MarClim) monitoring since 2001 and repeated surveys in 2014 and 2015 have highlighted that the northern limit for the barnacle *Chthamalus montagui* is in Orkney (Burrows et al. 2017). As with the NNS monitoring, all samples would need to be identified to species level increasing time used in identification, training and laboratory processing. As a research project, the potential for including the Orkney sandy beach macroinvertebrate data in climate change monitoring is clear but as a monitoring programme for a commercially operating Harbour Authority the applicability is questionable. If the decision was made to continue to identify all sandy beach samples to species level for the data to be included in the NNS monitoring, then there would be no additional cost to the identification only in relation to the data analysis and report writing.

8.6 Ecological health of the sites

Using AMBI analysis (Chapter 7), the environmental status in the Historical monitoring period were classed as either moderately disturbed (Dead Sand, Kirkhope, Lyrawa Bay and Scapa Bay) or slightly disturbed (Congesquoy, Creekland, Cumminess, Longhope, Mill Bay, Quoys, Swanbister, Waulkmill and Widewall) (Chapter 7 Figure 7.4). During the Current time period the environmental status for three of the sites (Kirkhope, Lyrawa and Scapa Bay) improved and for one of the sites (Swanbister) deteriorated. In both time periods none of the sites achieved undisturbed environmental status and therefore could not be used as a reference ('pristine') condition. This does not mean, of course, that they cannot be treated as a baseline against which any future condition may be compared. In comparison The Scottish Environment Protection Agency's (2019) overall surface water classification and benthic invertebrate Infaunal Quality Index (IQI) (Scottish Environment Protection Agency 2015) for Scapa Flow were both 'Good'. The different environmental conditions achieved by AMBI, surface water classification and IQI could be due to sampling of different habitats; intertidal macroinvertebrates in this thesis compared to sub-tidal macroinvertebrates for the IQI, while the overall surface water classification took into account all environmental monitoring carried out by SEPA in Scapa Flow. The calculation of IQI for the Scapa Flow sandy beach sites would enable thorough comparison with the SEPA sub-tidal IQI and should be implemented for the 2014 and 2017 macroinvertebrate data which will all have been identified to species level (once identification is completed). The AMBI analysis for all of the sites using family level data from 2002-2016 should be carried out to understand the year-to-year fluctuations. This would enable the understanding of the baseline fluctuations on the benthic community indicating the health of the sites.

8.7 Scientific record keeping

The analysis and interpretation of the data from the two time periods highlights the importance of good scientific practice and record keeping. Much of the information gaps for both time periods were due to either to incomplete reports (Jones 1974; Jones $\&$ Simpson 1976, 1977; Jones et al. 1978, 1979; Jones 1980; Jones et al. 1981, 1982; Jones 1983, 1985; Jones et al. 1986-1991) or absence of reports. Changes and decisions regarding the monitoring programme, e.g. re-starting the sampling in 2002, station selections at the sandy beach sites, change of the season for the sampling, were made in the Current time period with no records of these being made in any format, paper or electronic. Questions which remain unanswered include, how the sampling stations were changed from tape measure distances to OS grid references and why the sampling was changed into winter sampling with reduced sampling stations. All these aspects, which were part of the project management, influenced the data analysis when comparing Current data with the Historical time period. Good scientific research record keeping includes details of the planning of the research, any decisions made regarding the research, details of what, where and when was carried out and why, data management and analysis and any reports produced as part of the research (Schreier et al. 2006; Goodman et al. 2014). At Waulkmill the changes in the faunal composition during Historical time period were attributable to the poor data management and highlight the need for good scientific record keeping. This thesis brings together information on the long-term monitoring programme and has been paramount in establishing good scientific record keeping and data management at the MEU as part of the sandy beach monitoring programme. New forms to track and record laboratory processes are now in place for; rinsing, sorting and for identification of the macroinvertebrate samples and for data entry.

8.8 Critique of the PhD research

The research reported in this thesis was driven by the necessity to understand what data were available (both historically and in present time) and what analysis was possible to carry out with the data. At the start of this doctoral research it was, wrongly, assumed that all macroinvertebrate data were available for both time periods, and therefore data inventory was not carried out until year two of the thesis process. In hind sight, this should have been one of the first tasks to be completed. Only by understanding what data were available, the sites for Verification should have been chosen. By choosing the sites prior to this process one of the sites, Waulkmill, was chosen but later was proven to have large data gaps which affected the data analysis. Carrying out doctoral research part-time and over seven years (2012-2019) is a long time during which some aspects did not become apparent until years into the research process, e.g. data availability (year two) and the importance of calculating Beach Type (year seven).

The changes in the macroinvertebrate communities at the sandy beaches (Chapters 6 and 7) have not been statistically linked to any physical or environmental variables. Changes in the macroinvertebrate communities are seen at two of the sites (Quoys and Longhope) but the reasons for these are not yet fully understood. Collection of granulometry data was re-started for Current time period in 2014; if the importance of these physical samples had been understood earlier the collection of these samples could have been re-started in 2012 at the start of the thesis research. The same could be applied for the organic carbon samples, which have not been collected during the Current time period but have since 2019 been included into the monitoring programme. Once data for these parameters have been collected the biological data (macroinvertebrates) can be analysed with the physical data using the BEST procedure in PRIMER. As described by Clarke & Gorley (2006), 'BEST procedure is used to find the 'best' match between the multivariate among-sample patterns of an assemblage and that from environmental variable associated with those samples'.

8.9 Further work

Bringing data together from a historical long-term monitoring programme is not straight forward. This research is the starting point and further analysis and work can be now undertaken to further our understanding of the ecology of the 13 sandy beaches.

Baseline data for each of the 13 sandy beaches were described but the level of change in the macroinvertebrate communities that would be acceptable for OICHA is still to be established. To enable the setting of the level of acceptable change a power analysis of how many samples are required to detect a change is required. Power analysis allows the determination of sample size required to detect an effect of a given size with a given degree of confidence (Quick-R 2019).

To explain the patterns in the biota several variables could be used: time of year, seasonality, particle size and changes in sediment (sorting, porosity, permeability, anoxic layer, redox potential). For the Current time period four years of granulometry data were available; in the Historical time period a maximum of seven years of granulometry data was available compared with 15 years and 17 years of biological macroinvertebrate data. Availability of physical data is a limiting factor in the data analysis; once further data are collected, as mentioned in Section 8.8, the BEST procedure in PRIMER can be applied. Other data analysis options are to look for changes in the faunal guilds, for example feeding mechanisms, mobility and reproduction strategies.

Another addition to the monitoring programme which would improve the data analysis process is to measure the biomass of the taxa present and to collect organic carbon data for each sampling station. Measuring the wet weight of the samples could be added as a step after the identification process and would enable calculation of biomass/abundance ratio.

Biological changes, e.g. non-native species (NNS), recruitment, predator / prey interactions including presence of wading birds that could affect and change the macroinvertebrate community were not considered in this study but are recommended for future data analysis.

The volume of data available at OICHA and the multitude of possibilities for data analysis are such that many more research projects could utilise the data, especially now that the data are in correct format and the inconsistencies in the data are known.

8.10 Recommendations for the monitoring programme

The sandy beach monitoring is an important part of the Harbour Authority's marine environment monitoring programme (OICHA 2018) along with NNS and rocky shore monitoring. All monitoring carried out by MEU has to be cost effective and proven to provide information and data that can be used in supporting the operation of the Harbour Authority.

253

Continuation of the sandy beach monitoring programme in its current format, sampling in the winter, reduced sampling stations, and family level identification, has now been proven to be sufficient to demonstrate changes in the macroinvertebrate communities and environmental health of the sites. The following recommendations for the monitoring programme are proposed:

- 1) Continue the sandy beach monitoring to indicate any changes in the system potentially arising from oil terminal activities and from ship-to-ship transfers, including oil spills; and arising from increased organic enrichment originating from discharges from the Stromness waste water treatment facility;
- 2) Consider the potential for including the sandy beach macroinvertebrate data as part of the NNS monitoring programme, once point 6 (below) has been completed;
- 3) Retain the same sampling season (winter) and the number of stations as per the Current time period. All Current time sampling stations at each sandy beach site were used in establishing the baseline macroinvertebrate communities, to be able to compare against this baseline the same sample collection methods and regime should be adhered to;
- 4) Collect organic carbon samples from each sampling station;
- 5) Consider reduced sampling frequency at the moderately disturbed site, Dead Sand;
- 6) Consider benefits of species vs. family level identification. The trial period of identifying all macroinvertebrate samples to species level will inform the decision to either continue with the species level identification at all sites, to implement species level identification to selected sites only or to return to family level identification. The samples from 2002-2016, with the exception of data from 2014 which were identified to species level by a taxonomic laboratory, will have to remain as they are, although there is potential for these samples to be identified to species level, and at the same time to be verified, by students or researchers in the future;
- 7) Consider all potential aspects which can affect the faunal composition including but not exclusively: time of year, seasonality, particle size and changes in sediment (sorting, porosity, permeability, anoxic layer, redox potential), NNS, recruitment, predator / prey interactions.
- 8) Consider the use of the three families Pontoporeiidae, Spionidae and Capitellidae as target taxa; these are ubiquitous across all Scapa Flow sandy beach sites and have the potential to be indicators of change if they were found to be significantly changed in abundances in future monitoring;
- 9) Implement the annual analysis of the macroinvertebrate data using AMBI to determine ecological health of the sites;
- 10) Consider the use of IQI calculations for 2014 and 2017 macroinvertebrate data to enable comparison with SEPA's environmental classification of Scapa Flow.

8.11 Conclusions

The long-term data from 13 Scapa Flow sandy beaches demonstrated that majority of the sandy beaches monitored in Scapa Flow have not changed from Historical to Current time period. The Beach Type and Ecological Quality Status remained the same at nine sites and the baseline macroinvertebrate communities remained the same at 11 sites. The sandy beaches selected in the Historical time period for the intertidal macroinvertebrate monitoring have proven to be suitable for the long-term monitoring programme.

This long-term monitoring programme including both historical and recent sampling highlights the true level of variability inherent in the dynamics of macroinvertebrate communities at the Scapa Flow sandy beaches. This provides the context for measuring the significance of any perturbations due to environment or other impacts.

Chapter 9 References

Afeworki, Y., Videler, J.J. & Bruggemann, J.H., 2013, *Seasonally changing habitat use patterns among roving herbivorous fishes in the southern Red Sea: the role of temperature and algal community structure*, Coral Reefs, **32** (2), 475-485.

Albayrak, S., Caglar, S., Mulayim, A., Kurt-Sahin, G., Balkis, H., Cinar, N.F., Atabay, H., Tutak, B., & Bahceci, H., 2019, *A case study: Ecological Quality Status of Susurluk River Basin (Marmara Sea)*, FEB-FRESENIUS ENVIRONMENTAL BULLETIN, p.769.

Armonies, W. & Reise, K., 2000, *Faunal diversity across a sandy shore*, Marine Ecology Progress Series, **196**, 49-57.

Atkins, S.M. & Jones, A.M., 1990, *Studies on natural and anthropogenic influences on macrofauna of sandy shores at four sites in Orkney over a 15-year period*. In Estuaries and Coasts: Spatial and Temporal Comparisons, Elliott M, Ducroroy JP (eds). ECSA Symposium, pp. 139-143, Olsen & Olsen, Fredensborg, Denmark.

Atkins, S.M., Jones, A.M. & Simpson, J.A., 1985, *The fauna of sandy beaches in Orkney: a review*. Proceedings of the Royal Society of Edinburgh, **87B,** 27-45.

Atkins, S.M., Simpson, J.A. & Jones, A.M., 1989, *The importance of the seasonal component in sandy shore monitoring: Examples from low diversity habitats in Orkney*. In J. McManus & M. Elliott (eds): Developments in Estuarine and Coastal Study Techniques, pp. 21-27, Olsen & Olsen, Fredensborg, Denmark.

Bae, H., Lee, J.H., Song, S.J., Ryu, J., Noh, J., Kwon, B.O., Choi, K. & Khim, J.S., 2018, *Spatiotemporal variations in macrofaunal assemblages linked to site-specific environmental factors in two contrasting nearshore habitats*, Environmental Pollution, **241,** 596-606.

Bamber, R.N., 1993, *Changes in the infauna of a sandy beach*, Journal of Experimental Marine Biology and Ecology, **172** (1-2), 93-107.

Barboza, F.R. & Defeo, O., 2015, *Global diversity patterns in sandy beach macrofauna: a biogeographic analysis*, Scientific Reports, **5**, 14515.

Barchard, K.A. & Pace, L.A., 2011, *Preventing human error: The impact of data entry methods on data accuracy and statistical results*, Computers in Human Behavior, **27** (5), 1834-1839.

Barcelona Field Studies Center, 2019, *Spearman's Rank Correlation Coefficient Rs and Probability (p) Value Calculator*, <https://geographyfieldwork.com/SpearmansRankCalculator.html> [Accessed 05 August 2019]

Barne, J.H., Robson, C.F., Kaznowska, S.S., Doody, J.P., Davidson, N.C., & Buck, A.L., (eds), 1997, *Coasts and the Seas of the United Kingdom, Region 2 Orkney*, Joint Nature Conservation Committee, Peterborough, UK.

Barnett, B.E., 1984, *Observations on the intertidal fauna of the South Bank of the Humber Estuary*, Marine Environmental Research, **13** (1), 33-53.

Baron, J. & Clavier, J., 1994, *Structure of and seasonal variations in a sandy beach macrofauna on the southwest coast of New Caledonia (SW Pacific Ocean)*, Asian Marine Biology, **11**, 1-7.

Barreiro, F., Gómez, M., Lastra, M., López, J. & De la Huz, R., 2011, *Annual cycle of wrack supply to sandy beaches: effect of the physical environment*, Marine Ecology Progress Series, **433**, 65-74.

Baum, J.K. & Myers, R.A., 2004, *Shifting baselines and the decline of pelagic sharks in the Gulf of Mexico*, Ecology Letters, **7**, 135-145.

Beukema, J.J., 1979, *Biomass and species richness of the macro-benthic animals living on the tidal flats of the Dutch Wadden Sea*, Netherlands Journal of Sea Research, **10** (2), 236-261.

Beukema, J.J., 1990, *Expected Effects of Changes in Winter Temperatures on Benthic Animals Living in Soft Sediments in Coastal North Sea Areas*. In: Expected Effects of Climatic Change on Marine Coastal Ecosystems, pp. 83-92, Springer, Dordrecht.

Beukema, J.J., Essink, K. & Dekker, R., 2000, *Long*‐*term observations on the dynamics of three species of polychaetes living on tidal flats of the Wadden Sea: the role of weather and predator–prey interactions*, Journal of Animal Ecology, **69** (1), 31-44.

Bevilacqua, S., Fraschetti, S., Musco, L. & Terlizzi, A., 2009, *Taxonomic sufficiency in the detection of natural and human-induced changes in marine assemblages: a comparison of habitats and taxonomic groups*, Marine Pollution Bulletin, **58** (12), 1850- 1859.

Beyer, J., Trannum, H.C., Bakke, T., Hodson, P.V. and Collier, T.K., 2016, *Environmental effects of the Deepwater Horizon oil spill: a review*, Marine Pollution Bulletin, **110** (1), 28-51.

Bilyard, G.R., 1987, *The Value of Benthic Infauna in Marine Pollution Monitoring Studies*, Marine Pollution Bulletin, **18** (11), 581-585.

Bishop, M.J. and Hutchings, P.A., 2011, *How useful are port surveys focused on target pest identification for exotic species management?*, Marine Pollution Bulletin, **62** (1), 36- 42.

Blanchard, A.L., Feder, H.M. & Hoberg, M.K., 2010, *Temporal variability of benthic communities in an Alaskan glacial fjord, 1971 – 2007*, Marine Environmental Research, **69,** 95-107.

Blott, S.J. & Pye, K., 2001, *GRADISTAT: a grain size distribution and statistics package for the analysis of unconsolidated sediments*, Earth surface processes and Landforms, **26** (11), 1237-1248.

Borg, J.A., Micallef, S.A., Pirotta, K., & Schembri, P.J., 1997, *Baseline marine benthic surveys in the Maltese Islands (Central Mediterranean)*, In Proceedings of the third international MEDCOAST conference on the Mediterranean coastal environment, Malta, 1-8.

Borja, A., Barbone, E., Basset, A., Borgersen, G., Brkljacic, M., Elliott, M., Garmendia, J.M., Marques, J.C., Mazik, K., Muxika, I., Magalhaes, Neto, J., Norling, K., Rodriguez, J.G., Rosati, I., Rygg, B., Teixeira, H. & Trayanova, A., 2011, *Response of single benthic metrics and multi-metric methods to anthropogenic pressure gradients, in five distinct European coastal and transitional ecosystems*, Marine Pollution Bulletin, **62**, 499-513.

Borja, A., Franco, J. & Pérez, V., 2000, *A Marine Biotic Index to Establish the Ecological Quality of soft-Bottom Benthos Within European Estuarine and Coastal Environments*, Marine Pollution Bulletin, **40** (12), 1100-1114.

Borja, A., Franco, J., Valencia, V., Bald, J., Muxika, I., Belzunce, M.J. & Solaun, O., 2004, *Implementation of the European water framework directive from the Basque country (northern Spain): a methodological approach*, Marine Pollution Bulletin, **48**, 209-218.

Borja, A., Dauer, D.M., Elliott, M. and Simenstad, C.A., 2010, *Medium-and long-term recovery of estuarine and coastal ecosystems: patterns, rates and restoration effectiveness*, Estuaries and Coasts, **33** (6), 1249-1260.

Boria, A., Dauer, D.M. & Grémare, A., 2012a, *The importance of setting targets and reference conditions in assessing marine ecosystem quality*, Ecological Indicators, **12,** 1- 7.

Borja, A., Mader, J. & Muxica, I., 2012b, *Instructions for the use of the AMBI index software (Version 5.0)*, Revista de Investigación Marina, AZTI-Tecnalia, **19** (3), 71-82.

Borja, A., Miles, A., Occhipinti-Ambrogi, A. & Berg, T., 2009, *Current status of macroinvertebrate methods used for assessing the quality of European marine waters: implementing the Water Framework Directive*, Hydrobiologia, **633**, 181-196.

Borja, A., Josefson, A.B., Miles, A., Muxika, I., Olsgard, F., Phillips, G., Rodríguez, J.G. & Rygg, B., 2007, *An approach to the intercalibration of benthic ecological status assessment in the North Atlantic ecoregion, according to the European Water Framework Directive*, Marine Pollution Bulletin, **55**, 42-52.

Brazeiro, A. & Defeo, O., 1996, *Macroinfauna zonation in microtidal sandy beaches: is it possible to identify patterns in such variable environments?* Estuarine, Coastal and Shelf Science, **42** (4), 523-536.

Brenzinger, B., Wilson, N.G. & Schrödl, M., 2014, *Microanatomy of shelled* Koloonella cf. minutissima *(Laseron, 1951) (Gastropoda: 'lower'Heterobranchia: Murchisonellidae) does not contradict a sister-group relationship with enigmatic Rhodopemorpha slugs*, Journal of Molluscan Studies, **80** (5), 518-540.

Brown, A.C., 1983, *The ecophysiology of sandy beach animals—a partial review*. In Sandy beaches as ecosystems, pp. 575-605, Springer, Dordrecht.

Brown, A.C. & McLachlan, A., 2002, *Sandy shore ecosystems and the threats facing them: some predictions for the year 2025*, Environmental Conservation, **29** (1), 62-77.

Burrows, M.T., Twigg, G., Mieszkowska, N. & Harvey, R., 2017, *Marine Biodiversity and Climate Change (MarClim): Scotland 2014/15*, Scottish Natural Heritage Commissioned Report No. 939, Edinburgh.

Burvingt, O., Masselink, G., Russell, P. & Scott, T., 2017, *Classification of beach response to extreme storms*, Geomorphology, **295**, 722-37.

Callaway, R., 2016, *Historical data reveal 30-year persistence of benthic fauna associations in heavily modified waterbody*, Frontiers in Marine Science, **3** (Article 141), 1-13.

Campbell, M.L., Gould, B. & Hewitt, C.L., 2007, *Survey evaluations to assess marine bioinvasions*, Marine Pollution Bulletin, **55** (7-9), 360-378.

Carvalho, S., Gaspar, M.B., Moura, A., Vale, C., Antunes, P., Gil, O., Da Fonseca, L.C. & Falcao, M., 2006, *The use of the marine biotic index AMBI in the assessment of the ecological status of the Óbidos lagoon (Portugal)*, Marine Pollution Bulletin, **52** (11), 1414-1424.

Chainho, P., Silva, G., Lane, M.F., Costa, J.L., Pereira, T., Azeda, C., Almeida, P.R., Metelo, I. & Costa, M.J., 2010, *Long-term trends in intertidal and subtidal benthic communities in response to water quality improvement measures*, Estuaries and Coasts, **33** (6), 1314-1326.

Chatzinikolaou, E., Mandalakis, M., Damianidis, P., Dailianis, T., Gambineri, S., Rossano, C., Scapini, F., Carucci, A. & Arvanitidis, C., 2018, *Spatio-temporal benthic biodiversity patterns and pollution pressure in three Mediterranean touristic ports*, Science of The Total Environment, **624**, 648-660.

Checon, H.H., & Amaral, A.C., 2017, *Taxonomic sufficiency and the influence of rare species on variation partitioning analysis of a polychaete community*, Marine Ecology, **38** (1), p.e12384.

Checon, H.H., Corte, G.N., Esmaeili, Y.M.S. & Amaral, A.C.Z., 2018, *Nestedness patterns and the role of morphodynamics and spatial distance on sandy beach fauna: ecological hypotheses and conservation strategies*, Scientific Reports, **8** (1), 3759.

Clarke, K.R. & Gorley, R.N., 2006, PRIMER v6: User Manual/Tutorial, PRIMER-E: Plymouth. 190 pp.

Clarke, K.R., Somerfield, P.J., Airoldi, L., Warwick, R.M., 2006, *Exploring interactions by second-stage community analyses*, Journal of Experimental Marine Biology and Ecology, **338** (2), 179-192.

Clarke, K.R., Warwick, R.M., 2001, *Change in marine communities: an approach to statistical analysis and interpretation*, 2nd edition, PRIMER-E: Plymouth.

Collin, S.B., Tweddle, J.F. & Shucksmith, R.J., 2015, *Rapid assessment of marine nonnative species in the Shetland Islands, Scotland*, BioInvasions Records, **4** (3), 147-155.

Le Cozannet, G., Castelle, B., Ranasinghe, R., Wöppelmann, G., Rohmer, J., Bernon, N., Idier, D., Louisor, J. & Salas-y-Mélia, D., 2018, *Uncertainties of sandy shoreline change projections as sea level rises*, Sci Rep in review.

Dahl, E., 1952, *Some aspects of the ecology and zonation of the fauna on sandy beaches*, Oikos, **4** (1), 1-27.

Dauer, D.M., 1993, *Biological Criteria, Environmental Health and Estuarine macrobenthic community structure*, Marine Pollution Bulletin, **26** (5), 249-257.

Dauvin, J.C., Gomez, G.J. & Salvande, F.M., 2003, *Taxonomic sufficiency: an overview of its use in the monitoring of sublittoral benthic communities after oil spills*, Marine Pollution Bulletin, **46** (5), 552.

Dauvin, J-D., Ruellet, T., Desroy, N., & Janson, A-L., 2007, *The ecological quality status of the Bay of Seine and The Seine estuary: Use of biotic indices*, Marine Pollution Bulletin, **55**, 241-257.

Day, J., 2008, *The need and practice of monitoring, evaluating and adapting marine planning and management—lessons from the Great Barrier Reef*, Marine Policy, **32** (5), 823-831.

Dayton, P.K., Tegner, M.J., Edwards, P.B., Riser, K.L., 1998, *Sliding baselines, ghosts, and reduced expectations in kelp forest communities*, Ecological Applications, **8** (2), 309- 322.

Davenport, J., 1979, *Cold resistance in* Gammarus duebeni *Liljeborg*, Astarte, **12**, 21-26.

Davidson, I.C., Crook, A.C. & Barnes, D.K., 2004, *Quantifying spatial patterns of intertidal biodiversity: is movement important?,* Marine Ecology, **25** (1), 15-34.

Dean, H.K., 2008, *The use of polychaetes (Annelida) as indicator species of marine pollution: a review*, Revista de Biologia Tropical, **56** (4), 11-38.

De Bakker, D.M., Meesters, E.H., Bak, R.P., Nieuwland, G., Van Duyl, F.C., 2017, *Longterm Shifts in Coral Communities On Shallow to Deep Reef Slopes of Curaçao and Bonaire: Are There Any Winners?* Frontiers in Marine Science, **3**, 247.

De Biasi, A.M., Bianchi, C.N., & Morri, C., 2003, *Analysis of macrobenthic communities at different taxonomic levels: an example from an estuarine environment in the Ligurian Sea (NW Mediterranean)*, Estuarine, Coastal and Shelf Science, **58**, 99-106.

De la Huz, R., Lastra, M., Junoy, J., Castellanos, C. & Vieitez, J.M., 2005, *Biological impacts of oil pollution and cleaning in the intertidal zone of exposed sandy beaches: preliminary study of the "Prestige" oil spill*, Estuarine, Coastal and Shelf Science, **65** (1- 2), 19-29.

De Jonge, V.N., Elliott, M. & Brauer, V.S., 2006, *Marine monitoring: its shortcomings and mismatch with the EU Water Framework Directive's objectives*, Marine pollution bulletin, **53** (1-4), 5-19.

De-la-Ossa-Carretero, J.A., Simboura, N., Del-Pilar-Ruso, Y., Pancucci-Papadopoulou, M.A., Giménez-Casalduero, F., & Sánchez-Lizaso, J.L., 2012, *A methodology for applying Taxonomic Sufficiency and benthic biotic indices in two Mediterranean areas*, Ecological Indicators, **23**, 232-241.

Defeo, O., & McLachlan, A., 2005, *Patterns, processes and regulatory mechanisms in sandy beach macrofauna: a multi-scale analysis*, Marine Ecology Progress Series, **295**, 1-20.

Defeo, O. & McLachlan, A., 2013, *Global patterns in sandy beach macrofauna: Species richness, abundance, biomass and body size*, Geomorphology, **199**, 106-114.

Defeo, O., McLachlan, A., Schoeman, D.S., Schlacher, T.A., Dugan, J., Jones, A., Lastra, M. & Scapini, F., 2009, *Threats to sandy beach ecosystems: a review*, Estuarine, Coastal and Shelf Science, **81** (1), 1-12.

Dexter, D.M., 1984, *Temporal and Spatial Variability in the Community Structure of the Fauna of Four Sandy Beaches in South-eastern New South Wales*, Australian Journal of Marine and Freshwater Research, **35**, 663-672.

Doney, S.C., Ruckelshaus, M., Duffy, J.E., Barry, J.P., Chan, F., English, C.A., Galindo, H.M., Grebmeier, J.M., Hollowed, A.B., Knowlton, N. & Polovina, J., 2011, *Climate change impacts on marine ecosystems*, Annual Review of Marine Science, **4**, 11-37.

Dugan, J.E., Hubbard, D.M., McCrary, M.D. and Pierson, M.O., 2003, *The response of macrofauna communities and shorebirds to macrophyte wrack subsidies on exposed sandy beaches of southern California*, Estuarine, Coastal and Shelf Science, **58**, 25-40.

dit Durell, S.L.V., McGrorty, S., West, A.D., Clarke, R.T., Goss-Custard, J.D. & Stillman, R.A., 2005, *A strategy for baseline monitoring of estuary special protection areas. Biological conservation*, **121** (2), 289-301.

Dörjes, J., Michaelis, H. & Rhode, B., 1986, *Long-term studies of macrozoobenthos in intertidal and shallow subtidal habitats near the island of Norderney (East Frisian coast, Germany)*, In Long-Term Changes in Coastal Benthic Communities (pp. 217-232). Springer, Dordrecht.

Eleftheriou, A., & McIntyre, A.D., 1976, *The Intertidal Fauna of Sandy Beaches: A Survey of the Scottish Coast*, In Scottish Fisheries Research Report (Vol. 6), Department of Agriculture and Fisheries for Scotland.

Eleftheriou, A., & McIntyre, A.D., (eds.), 2005, *Methods for the Study of Marine Benthos*, 3rd edn., Blackwell Science Ltd.

Eleftheriou, A., & Robertson, M.R., 1988, *The Intertidal Fauna of Sandy Beaches: A Survey of the East Scottish Coast*, In Scottish Fisheries Research Report (Vol. 38). Department of Agriculture and Fisheries for Scotland.

Elliott, M., 1993, *Recent developments in macrobenthic community analysis*, Proceedings of the 4th National Symposium on Oceanography and Fisheries, 144-155.

Elliott, M., 1993. *The quality of macrobiological data*, Marine Pollution Bulletin, **26** (1); 2-3.

Ellis, D., 1985, *Taxonomic sufficiency in pollution assessment*, Marine Pollution Bulletin, **16** (12), 459.

Ellis, D.V., 1988, *Quality control of biological surveys*, Marine Pollution Bulletin, **19** (10), 506-512.

Engel, V.D., Hyland, J.L. & Cooksey, C., 2009, *Effects of Hurricane Katrina on benthic macroinvertebrate communities along the northern Gulf of Mexico coast*, Environmental Monitoring and Assessment, **150** (1-4), 193-209.

Essink, K., & Beukema, J.J., 1986, *Long-term changes in intertidal flat macrozoobenthos as an indicator of stress by organic pollution*, Hydrobiologia, **142** (1), 209-215.

Fauchald, K. & Jumars, P.A., 1979, *The diet of worms: a study of polychaete feeding guilds*, Oceanography and Marine Biology Annual Review, **17**, 193-284.

Feder, H.M. & Blanchard, A., 1998, *The deep benthos of Prince William Sound, Alaska, 16 months after the Exxon Valdez oil spill*, Marine Pollution Bulletin, **36** (2), 118-130.

Ferrando, A., & Méndez, N., 2011, *Effects of organic pollution in the distribution of annelid communities in the Estero de Urías coastal lagoon, Mexico*, Scientia Marina, **75** (2), 351-358.

Ferraro, S.P. & Cole, F.A., 1990, *Taxonomic level and sample size sufficient for assessing pollution impacts on the Southern California Bight macrobenthos*, Marine Ecology Progress Series. Oldendorf, **67** (3), 251-262.

Fish, J.D. & Fish, S., 1996, *A student's guide to the seashore*. Cambridge University Press.

Folk, R.L. & Ward, W.C., 1957, *Brazos River bar [Texas]; a study in the significance of grain size parameters*, Journal of Sedimentary Research, **27** (1), 3-26.

Fox, W.T., & Davis JR., R.A., 1978, *Seasonal variation in beach erosion and sedimentation on the Oregon coast*, Geological Society of America Bulletin, **89** (10), 1541-9.

Frid, C.L.J., Garwood, P.R., & Robinson, L.A., 2009, *Observing change in a North Sea benthic system: A 33 year time series*, Journal of Marine Systems, **77**, 227-236.

George, J.D., 1968, *The effect of the 1962-63 winter on the distribution of the cirratulid polychaetes,* Cirratulus cirratus *(Müller) and* Cirriformia tentaculata *(Montagu) in the British Isles*, The Journal of Animal Ecology, **37** (2), 321-337.

Giangrande, A., Geraci, S., & Belmonte, G., 1994, *Life-cycle and life-history diversity in marine invertebrates and the implications in community dynamics*, Oceanography and marine Biology: an Annual Review, **32**, 305-333.

Gibbs, R.J., Matthews, M.D., & Link, D.A., 1971, *The relationship between sphere size and settling velocity,* Journal of Sedimentary Research, **41** (1), 7-18.

Gilburn, A.S., 2012, *Mechanical grooming and beach award status are associated with low strandline biodiversity in Scotland*, Estuarine, Coastal and Shelf Science, **107**, 81-88.

Gillett, D.J., Weisberg, S.B., Grayson, T., Hamilton, A., Hansen, V., Leppo, E.W., Pelletier, M.C., Borja, A., Cadien, D., Dauer, D. & Diaz, R., 2015, *Effect of ecological group classification schemes on performance of the AMBI benthic index in US coastal waters*, Ecological Indicators, **50**, 99-107.

Gheskiere, T., Vincx, M., Weslawski, J.M., Scapini, F., & Degraer, S., 2005, *Meiofauna as descriptor of tourism-induced changes at sandy beaches*, Marine Environmental Research, **60** (2), 245-265.

Goodman, A., Pepe, A., Blocker, A.W., Borgman, C.L., Cranmer, K., Crosas, M., Di Stefano, R., Gil, Y., Groth, P., Hedstrom, M. and Hogg, D.W., 2014, *Ten simple rules for the care and feeding of scientific data*, PLoS Computational Biology, **10** (4), e1003542.

Gray, J.S., 1979, *Pollution-induced changes in populations*, Philosophical Transactions of the Royal Society of London, B, Biological Sciences, **286** (1015), 545-561.

Gray, J.S., 1981, *The ecology of marine sediments (Vol. 2)*, Cambridge University Press, Cambridge.

Gray, J.S., & Christie, H., 1983, *Predicting long-term changes in marine benthic communities*, Marine Ecology – Progress Series, **13**, 87-94.

Gray, J.S., & Elliott, M., 2009, *Ecology of Marine Sediments from Science to Management*, Oxford University Press, Oxford.

Gray, J.S., & Mirza, F.B., 1979, *A possible method for the detection of pollution-induced disturbance on marine benthic communities*, Marine Pollution Bulletin, **10**, 142-146.

Haase, P., Murray-Bligh, J., Lohse, S., Pauls, S., Sundermann, A., Gunn, R., & Clarke, R., 2006, *Assessing the impact of errors in sorting and identifying macroinvertebrate samples*, Hydrobiologia, **566**, 505-521.

Haase, P., Pauls, S.U., Schindehütte, K., & Sundermann, A., 2010, *First audit of macroinvertebrate samples from an EU Water Framework Directive monitoring program: human error greatly lowers precision of assessment results*, Journal of the North American Benthological Society, **29** (4), 1279-1291.

Hadfield, M.G., & Strathmann, M.F., 1996, V*ariability, flexibility and plasticity in life histories of marine invertebrates*, Oceanologica Acta, **19** (3-4), 323-334.

Hargrave, B.T., & Thiel, H., 1983, *Assessment of Pollution-induced changes in benthic community structure*, Marine Pollution Bulletin, **14** (2), 41-46.

Hartley, J.P., 1981, *The family Paraonidae (Polychaeta) in British waters: a new species and new records with a key to species*, Journal of the Marine Biological Association of the United Kingdom, **61** (1), 133-149.

Hayward, P.J., & Ryland, J.S., (eds.), 1995, *Handbook of the marine fauna of North-West Europe*, 1st edn., Oxford University Press.

Hayward, P.J., & Ryland, J.S., (eds.), 2017, *Handbook of the marine fauna of North-West Europe*, 2nd edn., Oxford University Press.

Hegge, B., Eliot, I., & Hsu J., 1996, *Sheltered sandy beaches of southwestern Australia*, Journal of Coastal Research, **12** (3), 748-760.

Hemery, L.G., Politano, K.K., & Henkel, S.K., 2017, *Assessing differences in macrofaunal assemblages as a factor of sieve mesh size, distance between samples, and time of sampling*, Environmental Monitoring and Assessment, **189** (8), 413.

Holme, N.A., & McIntyre, A.D., 1971, *Methods for the study of marine benthos*, Blackwell Scientific Publication, Oxford.

Howie, F., Read, J., & Turnbull, M., 1975, *Flotta oil handling terminal: marine environment protection*, Petroleum Review, **29** (344), 529-536.

Humphries, P. & Winemiller, K.O., 2009, *Historical impacts on river fauna, shifting baselines, and challenges for restoration,* BioScience, **59** (8), 673-684.

ICIT- International Centre for Island Technology, 2004a, *Review of the uses of the lochs and land in the Stenness catchment, Orkney, and associated environmental impacts,* Scottish Natural Heritage Commissioned Report No.066 (Part 10 of 12) (ROAME No. F01LA02).

ICIT - International Centre for Island Technology, 2004b, *The Lochs of Stenness and Harray: key environmental issues and recommendations for future management*, Scottish Natural Heritage Commissioned Report No.066 (Part 12 of 12) (ROAME No. F01LA02).

ICOE, 2016a, *Case Study 1 – General Report on the Onshore and Offshore Environment Records 1973-2016*. Study by International Centre for Oil and the Environment (ICOE) Research Limited On behalf of Repsol Sinopec Resourses UK Limited.

ICOE, 2016b, *Case Study 2 – The Environmental Determinants of Flotta's Infrastructure*. Study by International Centre for Oil and the Environment (ICOE) Research Limited On behalf of Repsol Sinopec Resourses UK Limited.

Ingels, J., & Vanreusel, A., 2013, *The importance of different spatial scales in determining structural and functional characteristics of deep-sea infauna communities*, Biogeosciences, **10** (7), 4547-4563.

Jackson, N.L., Nordstrom, K.F., Eliot, I., & Masselink, G., 2002, *'Low energy' sandy beaches in marine and estuarine environments: a review*, Geomorphology, **48** (1-3), 147- 162.

Jarrin, J.M., Vanaverbeke, J., Fockedey, N., & Dominguez-Granda L, 2017, *Surf zone fauna of Ecuadorian sandy beaches: Spatial and temporal patterns*, Journal of Sea Research, **120**, 41-49.

JMS, 1986, Ebala nitidissima *(Montagu, 1803):* Turbo nitidissimus *Montagu, 1803* Eulimella nitidissima *(Montagu, 1803)*, Journal of Molluscan Studies, **52,** Issue Supplement 16, 629–632.

Jones, A.M., 1974, *Interim report of the Orkney Marine Biology Unit, July 1974 – December 1974*, Centre for Industrial Research and Consultancy, University of Dundee. Unpublished report.

Jones, A.M., 1980, *Monitoring studies associated with an oil reception terminal*, Rapports et Proces Verbaux des Reunions, **179**, 194-200. Unpublished report.

Jones, A.M., 1983, *Orkney Marine Biology Unit Summary Report for 1982*, Centre for Industrial Research and Consultancy, University of Dundee. Unpublished report.

Jones, A.M., 1985, *Orkney Marine Biology Unit Summary Report for 1983-1984*, Centre for Industrial Research and Consultancy, University of Dundee. Unpublished report.

Jones, A.M., 1995, *Annual changes in selected aspects of soft-shore biology*, Proceedings of the Royal Society of Edinburgh, **103** B, 201-218.

Jones, A.M., Atkins, S.M., Caudwell, C.M., Noble, S.M., & Seatter, I., 1991, *Orkney Marine Biology Unit, Summary Report, June 1990 – May 1991*, Environmental Advisory Unit, University of Dundee. Unpublished report.

Jones, A.M., Atkins, S.M., Noble, S., Seatter, I., & Simpson, J.A., 1990, *Orkney Marine Biology Unit, Summary Report, June 1989 – May 1990*, Environmental Advisory Unit, University of Dundee. Unpublished report.

Jones, A.M., & Jones, Y.M., 1981, *The soft shore environment of Sullom Voe & the north mainland of Shetland*, Proceedings of the Royal Society of Edinburgh, **80**B, 203-218.

Jones, A.M., Jones, Y.M., & Simpson, J.A., 1978, *Third report of the Orkney Marine Biology Unit, December 1976 – December 1977*, Centre for Industrial Research and Consultancy, University of Dundee. Unpublished report.

Jones, A.M., Jones, Y.M., Simpson, J.A., & Baxter, J.M., 1979, *Fourth report of the Orkney Marine Biology Unit, December 1977 – December 1978*, Centre for Industrial Research and Consultancy, University of Dundee. Unpublished report.

Jones, A.M., Jones, Y.M., Simpson, J.A., & Baxter, J.M., 1981, *Fifth report of the Orkney Marine Biology Unit, January 1979 – December 1980*, Centre for Industrial Research and Consultancy, University of Dundee. Unpublished report.

Jones, A.M., & Simpson, J.A., 1976, *First report of the Orkney Marine Biology Unit, July 1974 – December 1975*, Centre for Industrial Research and Consultancy, University of Dundee. Unpublished report.

Jones, A.M., & Simpson, J.A., 1977, *Second report of the Orkney Marine Biology Unit, December 1975 – December 1976*, Centre for Industrial Research and Consultancy, University of Dundee. Unpublished report.

Jones, A.M., Simpson, J.A., Atkins, S.M., Baxter, J.M., & Saum, S., 1982, *Sixth report of the Orkney Marine Biology Unit, January 1981 – December 1981*, Centre for Industrial Research and Consultancy, University of Dundee. Unpublished report.

Jones, A.M., Simpson, J.A., Atkins, S.M., & Noble, S., 1986, *Orkney Marine Biology Unit Summary Report for 1985*, Centre for Industrial Research and Consultancy, University of Dundee. Unpublished report.

Jones, A.M., Simpson, J.A., Atkins, S.M., & Noble S., 1987, *Summary report of the Orkney Marine Biology Unit, January 1986 – March 1987*, Environmental Advisory Unit, University of Dundee. Unpublished report.

Jones, A.M., Simpson, J.A., Atkins, S.M., & Noble, S., 1988, *Summary report of the Orkney Marine Biology Unit, March 1987 – May 1988*, Environmental Advisory Unit, University of Dundee. Unpublished report.

Jones, A.M., Simpson, J.A., Atkins, S.M., & Noble, S., 1989, *Orkney Marine Biology Unit, Summary Report, June 1988 – May 1989*, Environmental Advisory Unit, University of Dundee. Unpublished report.

Jones, J.I., Clarke, R.T., Blackburn, J.H., Gunn, R.J.M., Kneebone, N.T., & Neale, M.W., 2007, *Lake benthic macroinvertebrates II: Quantifying uncertainty in sampling methodology*, CEH Project Number: C02466NEW, Science Report: SC030294/SR2. Bristol, Environment Agency.

Josefson, A.B., Blomqvist, M., Hansen, J.L.S., Rosenberg, R., & Rygg, B., 2009, *Assessment of marine benthic quality change in gradient of disturbance: Comparison of different Scandinavian multi-metric indices*, Marine Pollution Bulletin, **58**, 1263-1277.

Kakkonen, J.E., 2016, *Benthic macro-invertebrate monitoring, Review reports for Mainland Orkney and Hoy*, unpublished report for Orkney Islands Council.

Kakkonen, J.E., Worsfold, T.M., Ashelby, C.W., Taylor, A., & Beaton, K., 2019, *The value of regular monitoring and diverse sampling techniques to assess aquatic non-native species: a case study from Orkney*, Management of Biological Invasions, **10** (1), 46-79.

Kingston, P.F., Dixon, I.M.T., Hamilton, S. and Moore, D.C., 1995, *The impact of the Braer oil spill on the macrobenthic infauna of the sediments off the Shetland Islands*, Marine Pollution Bulletin, **30** (7), 445-459.

Kiyko, O.A., & Pogrebov, V.B., 1997, *Long-term benthic population changes (1920 – 1930s-Present) in the Barents and Kara Seas*, Marine Pollution Bulletin, **35** (7-12), 322- 332.

Kleiss, J.M. & Melville, W.K., 2010, *Observations of wave breaking kinematics in fetchlimited seas*, Journal of Physical Oceanography, **40** (12), 2575-2604.

Kruskal, J.B., 1964, *Multidimensional scaling by optimizing goodness of fit to a nonmetric hypothesis*, Psychometrika, **29** (1), 1-27.

Kröncke, I., & Reiss, H., 2010, *Influence of macrofauna long-term natural variability on benthic indices used in ecological quality assessment*, Marine Pollution Bulletin, **60**, 58- 68.

Kröncke, I., Reiss, H., & Dippner, J.W., 2013, *Effects of cold winters and regime shifts on macrofauna communities in shallow coastal regions*, Estuarine, Coastal and Shelf Science, **119**, 79-90.

Kröncke, I., Reiss, H., Eggleton, J.D., Aldridge, J., Bergman, M.J., Cochrane, S., Craeymeersch, J.A., Degraer, S., Desroy, N., Dewarumez, J.M. and Duineveld, G.C., Essink, K., Hillewaert, H., Lavaleye, M.S.S., Moll, A., Nehring, S., Newell, R., Oug, E., Pohlmann, T., Rachor, E., Robertson, M., Rumohr, H., Schratzberger, M., Smith, R., Berghe, E.V., van Dalfsen, J., van Hoey, G., Vincx, M., Willems, W., Rees, H.L., 2011, *Changes in North Sea macrofauna communities and species distribution between 1986 and 2000*, Estuarine, Coastal and Shelf Science, **94**, 1-15.

Labrune, C., Amouroux, J.M., Sarda, R., Dutrieux, E., Thorin, S., Rosenberg, R., Grémare, A., 2006, *Characterization of the ecological quality of the coastal Gulf of Lions (NW Mediterranean). A comparative approach based on three biotic indices*, Marine Pollution Bulletin, **52** (1), 34-47

Leber, K.M., 1982, *Bivalves (Tellinacea: Donacidae) on a North Carolina Beach: Contrasting Population Size Structures and Tidal Migrations*, Marine Ecology Progress Series. Oldendorf, **7** (3), 297-301.

Lehtiniemi, M., Ojaveer, H., David, M., Galil, B., Gollasch, S., McKenzie, C., Minchin, D., Occhipinti-Ambrogi, A., Olenin, S. & Pederson, J., 2015, *Dose of truth—monitoring marine non-indigenous species to serve legislative requirements*, Marine Policy, **54**, 26- 35.

Louzao, M., Anadón, N., Arrontes, J., Álvarez-Claudio, C., Fuente, D.M., Ocharan, F., Anadón, A. & Acuña, J.L., 2010, *Historical macrobenthic community assemblages in the Avilés Canyon, N Iberian Shelf: Baseline biodiversity information for a marine protected area*, Journal of Marine Systems, **80** (1-2), 47-56.

Masselink, G., & Pattiaratchi, C.B., 2001, *Seasonal changes in beach morphology along the sheltered coastline of Perth, Western Australia*, Marine Geology, **172** (3-4), 243-263.

Masselink, G., & Short, A.D., 1993, *The effect of tide range on beach morphodynamics and morphology: a conceptual beach model*, Journal of Coastal Research, **1**, 785-800.

Maurer, D., Watling, L., Leathem, W., & Kinner, P., 1979, *Seasonal changes in feeding types of estuarine benthic invertebrates from Delaware Bay*, Journal of Experimental Marine Biology and Ecology, **36**, 125-155.

McLachlan, A., 1983, *Sandy beach ecology—a review*, In Sandy beaches as ecosystems (pp. 321-380). Springer, Dordrecht.

McLachlan, A., 1996, *Physical factors in benthic ecology: effects of changing sand particle size on beach fauna*, Marine Ecology Progress Series, **131**, 205-217.

McLachlan, A., & Defeo, O., 2018, *The ecology of sandy shores*, 3rd edition, Academic Press, London.

McLusky, D.S., & Elliott, M., 2007, *Transitional waters: a new approach, semantics or just muddying the waters*?, Estuarine, Coastal and Shelf Science, **71** (3-4), 359-363.

Melet, A., Meyssignac, B., Almar, R. & Le Cozannet, G., 2018, *Under-estimated wave contribution to coastal sea-level rise*, Nature Climate Change, **8** (3), 234.

MET Office, 2019, *Orkney: Loch of Hundaland climate. Averages Table for Climate period: 1981-2010*, <https://www.metoffice.gov.uk/public/weather/climate/gft8q91es> [Accessed 25/01/2019]

Milner, C., & Hall, D., 2016, *Benthic Invertebrate Component. Own Sample Exercise Protocol*, APEM Ltd.

Morrisey, D.J., Howitt, L., Underwood, A.J., & Stark, J.S., 1992, *Spatial variation in softsediment benthos*, Marine Ecology Progress Series, **81**, 197-204.

Morton, R.A., & Sallenger, Jr. A.H., 2003, *Morphological impacts of extreme storms on sandy beaches and barriers*, Journal of Coastal Research, **19** (3), 560-573.

Murray, E., Dalkin, M.J., Fortune, F., & Begg, K., 1999, *Marine Nature Conservation Review Sector 2. Orkney: area summaries. Coasts and seas of the United Kingdom, MNCR series*, Joint Nature Conservation Committee, Peterborough, U.K.

Muxica, I., Borja, Á., & Bald, J., 2007, *Using historical data, expert judgement and multivariate analysis in assessing reference conditions and benthic ecological status, according to the European Water Framework Directive*, Marine Pollution Bulletin, **55**, 16-29.

Muxika, I., Borja, A. & Bonne, W., 2005, *The suitability of the marine biotic index (AMBI) to new impact sources along European coasts*, Ecological Indicators, **5** (1), 19- 31.

National Tidal and Sea Level Facility, 2018, <https://www.ntslf.org/about-tides/tides> Accessed [06/09/2018]

Neumann, H., Reiss, H., Rakers, S., Ehrich, S. and Kröncke, I., 2009, *Temporal variability in southern North Sea epifauna communities after the cold winter of 1995/1996*, ICES Journal of Marine Science, **66** (10), 2233-2243.

NMBACQ, 2018, *NE Atlantic Marine Biological Analytical Quality Control Scheme web site*, <http://www.nmbaqcs.org/> [Accessed 14/02/2018]

Ojaveer, H., Galil, B.S., Minchin, D., Olenin, S., Amorim, A., Canning-Clode, J., Chainho, P., Copp, G.H., Gollasch, S., Jelmert, A. & Lehtiniemi, M., 2014, *Ten recommendations for advancing the assessment and management of non-indigenous species in marine ecosystems*, Marine Policy, **44**, 160-165.

Olsgard, F., & Somerfield, P.J., 2000, *Surrogates in marine benthic investigations*‐*which taxonomic unit to target*? ,Journal of Aquatic Ecosystem Stress and Recovery, **7** (1), 25- 42.

Olsgard, F., Somerfield, P.J. & Carr, M.R., 1998, *Relationships between taxonomic resolution, macrobenthic community patterns and disturbance*, Marine Ecology Progress Series, **172**, 25-36.

Orlando, L., Ortega, L., & Defeo, O., 2019, *Multi-decadal variability in sandy beach area and the role of climate forcing*, Estuarine, Coastal and Shelf Science, **218**, 197-203.

OICHA, 2018, *Orkney Islands Council Harbour Authority Annual Report 2017-2018*, <https://www.orkneyharbours.com/port-authority/info/brochures>

Orkney Islands Council Harbour Authority, 2017, *Ballast Water Management Policy for Scapa Flow (adopted in December 2013)*, Orkney, Scotland.

Pauly, D., 1995, *Anecdotes and the shifting baseline syndrome of fisheries*, TREE, **10**, 430.

Pande, A. & Gardner, J.P., 2009, *A baseline biological survey of the proposed Taputeranga Marine Reserve (Wellington, New Zealand): spatial and temporal variability along a natural environmental gradient,* Aquatic Conservation: Marine and Freshwater Ecosystems, 19 (2), 237-248.

Paté‐Cornell, M.E., 1993, *Learning from the Piper Alpha Accident: A Postmortem Analysis of Technical and Organizational Factors*, Risk Analysis, **13** (2), 215-32.

Pearson, T.H., 1970, *The benthic ecology of Loch Linnhe and Loch Eil, a sea-loch system on the west coast of Scotland. I. The physical environment and distribution of the macrobenthic fauna*, Journal of Experimental Marine Biology and Ecology, **5** (1), 1–34.

Pearson, T.H., 1971, *The benthic ecology of Loch Linnhe and Loch Eil, a sea-loch system on the west coast of Scotland. III. The effect on the benthic fauna on the introduction of pulp mill effluent*, Journal of Experimental Marine Biology and Ecology, **6** (3), 211-233.

Pearson, T.H., 1976, *The benthic ecology of Loch Linnhe and Loch Eil, a sea-loch system on the west coast of Scotland. IV. Changes in the benthic fauna attributable to organic enrichment*, Journal of Experimental Marine Biology and Ecology, **20** (1), 1-40.

Pinnegar, J.K., & Engelhard, G.H., 2008, *The 'shifting baseline' phenomenon: a global perspective*, Reviews in Fish Biology and Fisheries, **18**, 1-16.

Pocklington, P., & Wells, P.G., 1992, *Polychaetes key taxa for marine environmental quality monitoring*, Marine Pollution Bulletin, **24** (12), 593-598.

Prior, A., Miles, A.C., Sparrow, A.J., & Price, N., 2004, *Development of a classification scheme for the marine benthic invertebrate component, water framework directive. Phase I and II-Transitional and Coastal Waters,* R & D Interim Technical Report E1- 116, E2-132, Environment Agency.

Prior, J., & Kendon, M., 2011a, *The UK winter of 2009/2010 compared with severe winters of the last 100 years*. Weather, **66**: 4-10.

Prior, J., & Kendon, M., 2011b, *The disruptive snowfalls and very low temperatures of late 2010*, Weather, **66**, 315-321.

Puente, A., Juanes, J.A., García-Castrillo, G., Álvarez, C., Revilla, J.A. & Gil, J.L., 2002, *Baseline study of soft bottom benthic assemblages in the Bay of Santander (Gulf of Biscay)*, In Nutrients and Eutrophication in Estuaries and Coastal Waters, Springer, Dordrecht.

Quick-R, 2019, Power analysis overview <https://www.statmethods.net/stats/power.html> [Accessed 26/07/2019]

R: Core Team, 2018, *R: A language and environment for statistical computing*, R Foundation for Statistical Computing, Vienna, Austria, URL [https://www.R-project.org/](https://www.r-project.org/)

Ranasinghe, J.A., Montagne, D.E., Weisberg, S.B., Bergen, M., & Velarde, R.G., 2003, *Variability in the identification and enumeration of marine benthic invertebrate samples and its effect on benthic assessment measures*, Environmental Monitoring and Assessment, **81** (1-3), 199-206.

Rakocinski, C.F., Heard, R.W., LeCroy, S.E., McLelland, J.A., & Simons, T., 1993, *Seaward change and zonation of the sandy-shore macrofauna at Perdido Key, Florida, USA*, Estuarine, Coastal and Shelf Science, **36** (1), 81-104.

Read, P., 1987, *The intertidal benthos and sediments of particulate shores in the Firth of Forth, Scotland, with particular reference to waste water discharges*, Proceedings of the Royal Society of Edinburgh, Section B: Biological Sciences, **93** (3-4), 401-413.

Reed, A.J., Thatje, S., & Linse, K., 2012, *Shifting baselines in Antarctic Ecosystems; Ecophysiological response to warming in* Lissarca miliaris *at Signy Island, Antarctic*, PLoS ONE, **7** (12), e53477.

Reiss, H., Meybohm, K. & Kröncke, I., 2006, *Cold winter effects on benthic macrofauna communities in near-and offshore regions of the North Sea*, Helgoland Marine Research, **60** (3), 224.

Resh, V.H., Bêche, L.A. & McElravy, E.P., 2005, *How common are rare taxa in longterm benthic macroinvertebrate surveys*?, Journal of the North American Benthological Society, **24** (4), 976-989.

Ricciardi, A., & Bourget, E., 1999, *Global patterns of macroinvertebrate biomass in marine intertidal communities*, Marine Ecology Progress Series, **185**, 21-35.

Roach, A.C., Jones, A.R. & Murray, A., 2001, *Using benthic recruitment to assess the significance of contaminated sediments: the influence of taxonomic resolution*, Environmental Pollution, **112** (2), 131-143.

Rosenberg, R., & Möller, P., 1979, *Salinity stratified benthic macrofaunal communities and long-term monitoring along the west coast of Sweden*, Journal of Experimental Marine Biology Ecology, **37**, 175-203.

Ruso, Y.D.P., De la Ossa Carretero, J.A., Casalduero, F.G. and Lizaso, J.S., 2007, *Spatial and temporal changes in infaunal communities inhabiting soft-bottoms affected by brine discharge*, Marine Environmental Research, **64** (4), 492-503.

Schlacher, T.A., Schoeman, D.S., Dugan, J., Lastra, M., Jones, A., Scapini, F., McLachlan, A., 2008, *Sandy beach ecosystems: key features, sampling issues, management challenges and climate change impacts*, Marine Ecology, **29** (s1), 70-90.

Schooler, N.K., Dugan, J.E., Hubbard, D.M., & Straughan, D., 2017, *Local scale processes drive long*‐*term change in biodiversity of sandy beach ecosystems*, Ecology and Evolution, **7** (13), 4822-4834.

Schulz, K.G., Bellerby, R.G.J., Brussaard, C.P., Büdenbender, J., Czerny, J., Engel, A., Fischer, M., Koch-Klavsen, S., Krug, S., Lischka, S., & Ludwig, A., 2013, *Temporal biomass dynamics of an Arctic plankton bloom in response to increasing levels of atmospheric carbon dioxide*, Biogeosciences, **10**, 161-180.

Schückel, U., & Kröncke, I., 2013, *Temporal changes in intertidal macrofauna communities over eight decades: A results of eutrophication and climate change*, Estuarine, Coastal and Shelf Science, **117**, 210-218.

Schreier, A.A., Wilson, K. & Resnik, D., 2006, *Academic research record-keeping: Best practices for individuals, group leaders, and institutions*, Academic medicine: Journal of the Association of American Medical Colleges, **81** (1), 42-47.

Scott, T., Masselink, G., O'Hare, T., Saulter, A., Poate, T., Russell, P., Davidson, M., Conley, D., 2016, *The extreme 2013/2014 winter storms: Beach recovery along the southwest coast of* England, Marine Geology, **382**, 224-41.

Scottish Environment Protection Agency, 2015, *Improving the quality of Scotland's water environment Orkney and Shetland area management plan 2010–2015, Supplementary to the river basin management plan for the Scotland river basin district*, SEPA.

Scottish Environment Protection Agency, 2019, *Overall status of surface waters for the latest available year*, [https://www.sepa.org.uk/data-visualisation/water-classification](https://www.sepa.org.uk/data-visualisation/water-classification-hub/)[hub/](https://www.sepa.org.uk/data-visualisation/water-classification-hub/) [Accessed 01/02/2019]

SEPA - Scottish Environment Protection Agency, 2016, <http://aquaculture.scotland.gov.uk/map/map.aspx> [Accessed 23/10/2016]

SEPA – Scottish Environment Protection Agency, 2019, <https://www.sepa.org.uk/environment/water/monitoring/> [Accessed 23/07/2019]

Scottish Water, 2019, News bulleting on Scottish Water Website [https://www.scottishwater.co.uk/investment-and-communities/your](https://www.scottishwater.co.uk/investment-and-communities/your-community/stromness)[community/stromness](https://www.scottishwater.co.uk/investment-and-communities/your-community/stromness) [Accessed 31/01/2019]

Short, A.D., 1996, *The role of wave height, period, slope, tide range and embaymentisation in beach classifications: a review*, Revista Chilena de Historia Natural, **69** (4), 589-604.

Short, A.D., & Wright, L.D., 1983, *Physical variability of sandy beaches*, In Sandy Beaches as Ecosystems (pp. 133-144), Springer, Dordrecht.

Simboura, N., Zenetos, A., Pancucci‐Papadopoulou, M.A., Thessalou‐Legaki, M. & Papaspyrou, S., 1998, *A baseline study on benthic species distribution in two neighbouring gulfs, with and without access to bottom trawling*, Marine Ecology, **19** (4), 293-309.

Somerfield, P.J. & Clarke, K.R., 1995, *Taxonomic levels, in marine community studies, revisited*, Marine Ecology Progress Series, **127**, 113-119.

SOTEAG, 2014, *Chemical & macrobenthic monitoring in Sullom Voe sediments 2014*, A report to the Shetland Oil Terminal Environmental Advisory Group by SGS M-Scan Ltd.

SOTEAG, 2016, *Chemical and macrobenthic monitoring in Sullom Voe sediments 2016*, A report to the Shetland Oil Terminal Environmental Advisory Group by SGS United Kingdom Limited.

SOTEAG, 2018, *Chemical and macrobenthic monitoring in Sullom Voe sediments 2018*, A report to the Shetland Oil Terminal Environmental Advisory Group by SGS United Kingdom Limited and Eco Marine Consultants Limited.

Start, R.D., Layton, C.M., Cross, S.S., & Smith, J.H., 1992, *Reassessment of the rate of fixative diffusion*, Journal of Clinical Pathology, **45** (12), 1120-1121.

Stribling, J.B., Moulton, S.R., & Lester, G.T., 2003, *Determining the quality of taxonomic data*, Journal of the North American Benthological Society, **22** (4), 621-631.

Stribling, J.B., Pavlik, K.L., Holdsworth, S.M., & Leppo, E.W., 2008, *Data quality, performance, and uncertainty in taxonomic identification for biological assessments*. Journal of the North American Benthological Society, **27** (4), 906-919.

Strong, D.M., Lee, Y.W. & Wang, R.Y., 1997, *Data quality in context*, Communications of the ACM, **40** (5), 103-110.

Thomson Ecology Ltd, 2015, *Laboratory Systems Handbook, TEM10 Particle Size Analysis*, Issue 007 13/05/2015, unpublished.

Thomson, M., Jackson, E., with Kakkonen, J., 2014, *Seagrass* (Zostera) *beds in Orkney*. Scottish Natural Heritage Commissioned Report No. 765.

Thompson, P.M., & Harwood, J., 1990, *Methods for estimating the population size of common seals,* Phoca vitulina, Journal of Applied Ecology, **27**, 924-938.

Tucker, M.J., & Pitt, E.G., 2001, *Waves in ocean engineering*. Volume 5. Elsevier Science, Oxford, UK.

Vandepitte, L., Vanhoorne, B., Decock, W., Vranken, S., Lanssens, T., Dekeyzer, S., Verfaille, K., Horton, T., Kroh, A., Hernandez, F., & Mees, J., 2018, *A decade of the World Register of Marine Species – General insights and experiences from the Data Management Team: Where are we, what have we learned and how can we continue?*, PLoS ONE, **13** (4), e0194599.

Villnäs, A., & Norkko, A., 2011, *Benthic diversity gradients and shifting baselines: implications for assessing environmental status,* Ecological Applications, **21** (6), 2172- 2186.

Warwick, R.M., 1988, *The level of taxonomic discrimination required to detect pollution effects on marine benthic communities*, Marine Pollution Bulletin, **19** (6): 259-268.

Warwick, R.M., & Clarke, K.R., 1993, *Increased variability as a symptom of stress in marine communities*, Journal of Experimental Marine Biology and Ecology, **172**, 215- 226.

Warwick, R.M., Platt, H.M., Clarke, K.R., Agard, J., & Gobin, J., 1990, *Analysis of macrobenthic and meiobenthic community structure in relation to pollution and disturbance in Hamilton Harbour, Bermuda*, Journal of Experimental Marine Biology and Ecology, **138**, 119-142.

Weydmann, A., Carstensen, J., Goszczko, I., Dmoch, K., Olszewska, A., Kwasniewski, S., 2014, *Shift towards the dominance of boreal species in the Arctic: inter-annual and* *spatial zooplankton variability in the West Spitsbergen Current*, Marine Ecology Progress Series, **501**, 41-52.

WFD, 2000, *Water Framework Directive 2000/60/EC*, OJ L 327, 22.12.2000, 1–73.

WFD-UKTAG, 2014, *United Kingdom Technical Advisory Group Transitional and Coastal Water Assessment Method, Benthic Invertebrate Fauna, Infaunal Quality Index*, c/o Scottish Environment Protection Agency, Scotland.

Wheeler & Torchiano, 2016, *lmPerm: Permutation Tests for Linear Models*, R package version 2.1.0. [https://CRAN.R-project.org/package=lmPerm](https://cran.r-project.org/package=lmPerm)

Whomersley, P., Schratzberger, M., Huxham, M., Bates, H., & Rees H., 2007, *The use of time-series data in the assessment of macrobenthic community change after the cessation of sewage-sludge disposal in Liverpool Bay (UK)*, Marine Pollution Bulletin, **54** (1), 32- 41.

Worsfold, T., 2003, *Introduction to Oligochaetes*, NMBAQC Workshop 2003 Report. <http://www.nmbaqcs.org/> [Accessed 27/06/2018]

Worsfold, T., & Hall, D., 2010, *Guidelines for processing marine macrobenthic invertebrate samples: a Processing Requirements Protocol Version 1.0*., Unicomarine Report to the National Marine Biological Analytical Quality Control Scheme.

Worsfold, T., & Hall, D., 2017a, *Benthic Invertebrate Component. Own Sample Exercise Protocol*. APEM Ltd.

Worsfold, T., & Hall D., 2017b, *NE Atlantic Marine Biological Analytical Quality Control Scheme, Annual Report 2016/2017*. APEM Ltd.

Wright, L.D., & Short, A.D., 1984, *Morphodynamic variability of surf zones and beaches: a synthesis*, Marine Geology, **56** (1-4), 93-118.

Xia, J., Falconer, R.A., & Lin, B., 2010, *Hydrodynamic impact of a tidal barrage in the Severn Estuary, UK*, Renewable Energy, **35** (7), 1455-1468.

Young, C.M., Sewell, M.A., & Rice, M.E., (eds.), 2002, *Atlas of marine invertebrate larvae (Vol. 6)*, San Diego: Academic press.

Ysebaert, T., & Herman, P.M., 2002, *Spatial and temporal variation in benthic macrofauna and relationships with environmental variables in an estuarine, intertidal soft-sediment environment*, Marine Ecology Progress Series, **244**, 105-124.

Yu, O.H., Lee, H.G., Shim, W.J., Kim, M. and Park, H.S., 2013, *Initial impacts of the Hebei Spirit oil spill on the sandy beach macrobenthic community west coast of Korea*, Marine Pollution Bulletin, **70** (1-2), 189-196.

Appendix A Sandy Hill Annual Windroses 2002 – 2016 (mean wind speed)
Appendix A (continued)

Appendix A (continued)

Appendix B Sandy Hill Annual Windroses 2002 – 2016 (Force 10+)

Appendix B (continued)

Appendix B (continued)

Appendix C Taxa recorded and their authorities

Appendix C (continued)

Appendix D Beach morphometric information

This appendix details the values and calculations used for attaining the beach morphometric information for each sandy shore monitoring site.

1.0 Beach morphometric calculations

Values used for grain size, water temperature, salinity and tidal range are summarised in table below.

1.1 Wave height, wave period and wave frequency

The monitoring sites are all enclosed within Scapa Flow and are therefore fetch limited. Wind generated waves are classed as fetch limited when the distance from shore to shore, or the area from where the waves start to build up to when they reach the opposing shore is from 0-500km (**Kleiss & Melville 2010**). For calculating wave height, wave period and wave frequency the fetch of each monitoring site was measured using FreeMapTools [\(www.freemaptools.com/measure-distance.htm\)](http://www.freemaptools.com/measure-distance.htm). An example of the measuring tool is given for one of the sites, Lyrawa Bay on Hoy, for which the fetch was measured as 3.18 km.

Appendix D (continued)

Once the fetch was known for the sites the following formula was used for calculating the wave height (cm):

 $H_{\text{mO}} = 0.0163 \text{ X}^{1/2} U_{10}$ (Tucker and Pitt 2001)

where

 $X =$ Fetch in km

 U_{10} = wind speed at 10m above mean sea level. 20 m/s for the Scapa Flow sites (Woolf, D. pers. comm.).

For fetch-limited seas, the wave period (sec) is calculated by using the following formula:

 $T_m = 0.566 \, X^{0.3} \, U_{10}^{0.4}$ (Tucker and Pitt 2001)

Wave frequency (\sec^{-1}) is calculated by using the following formula:

Wave frequency = $1/T_m$ (Tucker and Pitt 2001)

Appendix D (continued)

1.2 Sand fall velocity

Sand fall velocity was calculated using tables from Gibbs et al. (1971) who have determined the settling velocities (cm/sec.) of particles based on their sizes (mean grain size (μm)). The Scapa Flow sites were all calculated using the settling velocities measured in water at 10° C, this is average sea water temperature in Scapa Flow (Appendix F. Section 3).

1.3 Dean's parameter (Ω) and Relative Tide Range (RTR)

Dean's parameter is dimensionless, RTR units are in meters, for further information on both see Chapter 1 Section 1.3.

Dean's parameter (DFV) and Relative Tide Range (RTR) are calculated using the following formulae:

DFV $(\Omega) = H_b/WT$ (McLachlan and Defeo 2018) $RTR = Tide/H_b$ (McLachlan and Defeo 2018)

where

 H_b = significant breaker wave height (m)

 $W =$ sand fall velocity (cm/sec.)

 $T =$ wave period (sec.)

Tide = maximum spring tide range (m)

Appendix D (continued)

1.4 Beach type as defined by Dean's parameter (Ω) and RTR

Beaches can be characterised into different types depending on the Dean's parameter and the RTR (Table D.1).

Table D.1. Beach types as defined by Dean's parameter (Ω) and Relative Tide Range (m). (Short and Wright 1983; Wright and Short 1984, Masselink and Short 1993; McLachlan and Defeo 2017)

	Dean's parameter (Ω)	Relative Tide Range (m)
Reflective	<2	
Reflective: low tide terrace with rip	<2	$3 - 7$
Reflective: low tide terrace without rip	<2	
Intermediate	$2 - 5$	
Intermediate: bar and rip channels	$2 - 5$	
Dissipative: barred	$>$ 5	
Dissipative: non-barred	$> \! 5$	
Ultra-dissipative	>5	

1.5 Slope*

Slope* is the reciprocal of the beach face slope and calculated using the following formulae:

Beach Face Slope $=(y1-y2)/d$

 $y1$ = height of the beach at bottom of the shore

 $y2$ = height of the beach at top of the shore

 $d =$ horizontal distance from y1 to y2

 $Slope* = 1/Beach face slope$ (McLachlan and Defeo 2018)

1.6 Beach Index

Beach Index (BI) is calculated using the following formula:

 $BI = log10$ (sand ⋅ tide / slope) (McLachlan and Defeo 2018)

where

 $sand = sand fall velocity (cm/sec.)$

tide = tide range (m)

slope $=$ beach face slope

Appendix E Summary of SIMPER Dissimilarity Results

Summary of SIMPER dissimilarity results for Current time period for the following sites and stations:

The tables summarise following information: average abundance (%) of discriminating taxa between each pair of years, average dissimilarity between the years, dissimilarity standard deviation, the contribution (%) of taxa to dissimilarity of the groups, and cumulative total (%) of contributions (cut-off at 90%).

Section 1. Quoys ST7 Current

Appendix e. Section 2. Quoys ST10 Current

Appendix E. Section 3. Quoys ST12 Current

Appendix E. Section 4. Congesquoy ST1 Current

Section 5. Congesquoy ST2 Current

Section 6. Waulkmill ST10 Current

Appendix F Environmental Data

Section 1. RAINFALL - Orkney Daily Rainfall Data

Met Office (2006): MIDAS: UK Daily Rainfall Data. NCAS British Atmospheric Data Centre, *23/01/2019*. <http://catalogue.ceda.ac.uk/uuid/c732716511d3442f05cdeccbe99b8f90>

Abstract of the data from CEDA Archive:

The UK daily rainfall data describe the rainfall accumulation and precipitation amount over a 24-hour period. The data are collected by observation stations across the UK and transmitted within the following message types: WADRAIN, NCM, AWSDLY, DLY3208, SSER and WAMRAIN. The data spans from 1853 to present.

Figure 1. 2010 Orkney rainfall data for Loch of Hundaland.

Appendix F (continued)

Figure 2. 2011 Orkney rainfall data for Loch of Hundaland.

Section 2. ATMOSPHERIC TEMPERATURE - Wick Weather Station

<https://www.metoffice.gov.uk/public/weather/climate-historic/#?tab=climateHistoric> [Accessed 23/01/2019]

Wick Weather Station location: Lat 58.457908 Long -3.0952692

Mean daily maximum temperature (tmax)

Mean daily minimum temperature (tmin)

The monthly mean temperature is calculated from the average of the mean daily maximum and mean daily minimum temperature i.e. (tmax+tmin)/2.

Appendix F (continued)

Appendix F (continued)

Section 3. SEAWATER TEMPERATURE - Scapa Pier, Scapa Flow

Scapa Pier location: Lat 58.956877 Long -2.9730892

Figure 5. Daily average seawater temperature at Scapa Pier, Scapa Flow January 2002-December 2016. Measured as part of Marine Scotland Science long-term monitoring programme.

Figure 6. Daily average seawater temperature at Scapa Pier, Scapa Flow for the period of 20 November – 31 March in 2009-2012. Measured as part of Marine Scotland Science long-term monitoring programme.

Section 4. TIDE TIMETABLES - Widewall, Scapa Flow, 2009 and 2010

Tidal information for Widewall is from Orkney Harbour Authority, with permission from UK Hydrographic Office. Periods of extreme cold weather are highlighted in yellow.

SCOTLAND - WIDEWALL BAY

 $\overline{\mathbf{3}}$

SCOTLAND - WIDEWALL BAY

LAT 58°49'N LONG 3°01'W

 \mathbf{t}

SCOTLAND - WIDEWALL BAY

LAT 58°49'N LONG 3°01'W

 $\overline{\mathbf{3}}$

Appendix G Selected sampling locations

Section 1. Quoys ST7 sampling in 2009 and 2012

Section 3. Congesquoy ST2 sampling locations

Section 4. Quoys ST7 sampling station in 2014 and 2015

Appendix H Changes for AMBI calculations

Taxa as in dataset Changed in AMBI Taxa as in dataset Changed in AMBI

Ammodytidae changed by Ammodytes tobianus (not assigned) Phyllodocidae changed by Eteone longa (III) Ampharetidae changed by Ampharetides sp. (I) Polynoida changed by POLYNOIDAE (not assigned) Ampithoidae changed by Ampithoe rubricata (I) Pontoporeiidae changed by Bathyporeia sarsi (I) Arenicolidae changed by Arenicola marina (III) Pseudocumatidae changed by Pseudocuma longicorne (II) Atylidae changed by Atylus sp. (I) Scalibregmatidae changed by Scalibregma inflatum (III) Bodotriidae changed by Iphinoe trispinosa (I) Scalibregmidae changed by Scalibregma inflatum (III) Calliopiidae changed by Calliopius laeviusculus (not assigned) Semelidae changed by Abra sp. (III) Caprellidea changed by Caprella sp. (II) Sigalonidae changed by Sigalion sp. (II) Cheirocratidae changed by Cheirocratus sp. (I) Sipunculidea changed by SIPUNCULA (I) Cirolanidae changed by Eurydice pulchra (I) Skeneidae changed by Skenea sp. (I) Corophiidae changed by Corophium crassicorne (III) Sphaerodoridae changed by Sphaerodoridium minutum (II) Dexaminidae changed by Dexamine thea (III) Syllidae changed by Parexogone hebes (II) Dorvillidae changed by Dorvillea sp. (not assigned) Talitridae changed by Talitrus saltator (I) Enteropneusta changed by Enteropneusta sp. (II) Tanaissuidae changed by Tanaissus lilljeborgi (III)
Eunicida changed by EUNICIDAE (II) Terebellidae changed by Lanice conchilega (II) Eunicida changed by EUNICIDAE (II) Terebellidae changed by Lanice conchilega (II)

Fabriciidae changed by Fabricia stellaris (II) Urothoidae changed by Urothoe marina (I) Fabriciidae changed by Fabricia stellaris (II) Urothoidae Urothoidae changed by Syllidia armata (II) CHORDATA Hesionidae changed by Syllidia armata (II) CHORDATA ignored

Hysildae changed by Apobyale prevostii (I) Holographidae ignored Hyalidae changed by Apohyale prevostii (I) Holognathidae ignored

Idoteidae changed by Idotea balthica (II) Brachyura not assi **Changed by Idotea balthica (II)** Brachyura not assigned Ischyroceridae changed by Ericthonius sp. (I) Cephalothrichidae not assigned Janiridae changed by Janira sp. (I) Decapoda not assigned Lampropidae changed by Lamprops fasciatus (I) Eusiridae not assigned Lepidochitonidae changed by Lepidochitona sp. (II) Heteronemertea not assigned Leucothoidae changed by Leucothoe sp. (I) Isaeidae not assigned Limapontiidae changed by Limapontia sp. (I) Leuconidae not assigned Lineidae changed by Cerebratulus sp. (III) Loveniidae not assigned Lumbrineridae changed by Lumbrineris cingulata (II) Neanuridae not assigned Margaritidae changed by Margarites sp. (II) Opisthobranchia not assigned Megaluropidae changed by Megaluropus sp. (I) Portunidae not assigned changed by Megaluropus sp. (I) Portunidae not assigned

changed by Melita palmata (I) Prochidae not assigned Melitidae changed by Melita palmata (I) Trochidae not assigned Microprotopidae changed by Microprotopus sp. (I) Uristidae not assigned Montacutidae changed by Kurtiella bidentata (III) Murchisonellidae changed by Murchisonella occidentalis (I) Myidae changed by Mya arenaria (II) Mysidacea changed by Paramysis helleri (II)
Mysidae changed by Paramysis helleri (II) changed by Paramysis helleri (II) Nebalidae changed by Nebalia bipes (V) Nebaliidae changed by Nebalia bipes (V) Nereidae changed by Hediste diversicolor (III) Nuculidae changed by Nucula sp. (I) Oedicerotidae changed by Perioculodes longimanus (II) Opheliidae changed by Ophelia rathkei (I) Orbiniidae changed by Scoloplos armiger (III) changed by Paraonis fulgens (III) Pectinariidae changed by Lagis koreni (IV)
Periplomatidae changed by Cochlodesma pra changed by Cochlodesma praetenue (not assigned) Philinidae changed by Philine quadripartita (II)
Pholoidae changed by Pholoe baltica (I) changed by Pholoe baltica (I)

Aphroditidae changed by Aphrodita sp. (I) Psammodrilidae changed by Psammodrilus balanoglossoides (I)