# Behind Anemone Lines: Determining the environmental drivers influencing lagoonal benthic communities, with special reference to the anemone *Nematostella vectensis*.

by

**Jessica R. Bone**

Bournemouth University

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# Copyright Statement

# Behind Anemone Lines: Determining the environmental drivers influencing lagoonal benthic communities, with special reference to the anemone *Nematostella vectensis*.

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## Abstract

Climate change induced sea level rise and increase in associated storms is impacting the coastal zone worldwide. Lagoons are a transitional ecosystem on the coast that are threatened with habitat loss due to ingress of seawater, though conversely this also represents an opportunity for lagoon habitat creation. It is important to quantify the spatio-temporal trends of macrozoobenthic communities and abiotic factors to determine the ecological health of lagoon sites. Such information will ensure optimal and adaptive management of these rare and protected ecosystems. This thesis examines the spatial distribution of macrozoobenthic assemblages and the abiotic and biotic factors that may determine their abundance, richness and distribution at tidally restricted urban lagoon at Poole Park on the south coast of England. The macrozoobenthic assemblages were sampled using a suction corer during a spatially comprehensive survey in November 2017, in addition to aquatic and sediment variables such as salinity, temperature, organic matter content and silt content. Species richness and density were significantly lower in areas of high organic matter and silt content, indicative of hostile conditions. There were no correlations between pelagic fauna and macrozoobenthic fauna which suggests that top-down control of macrozoobenthic species is not significant enough to influence their distribution. Salinity and temperature were spatially homogenous but macrozoobenthic assemblages indicate longer term variability; the euryhaline annelid *Hediste diversicolor* dominates at sample sites adjacent to surface water outflow pipes. The non-native protected Starlet Sea Anemone *Nematostella vectensis* was also significantly negatively correlated with organic matter and silt content. It is known to be sensitive to hypoxic-sulfidic conditions associated with high organic matter sediments. The anemone's indiscriminate and efficient method of prey capture in high macrozoobenthic densities may disproportionately affect prey species, limiting their availability to native predators and negatively affect higher trophic levels. The anemone’s effect on native communities should be subject to further study. This thesis will serve as a baseline to compare subsequent surveys to, particularly post dredging and island construction works planned to commence in Poole Park lagoon in the late autumn of 2018.

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I am relieved I have finally finished my project but also sad that work with Poole Harbour lagoons has come to a temporary end – I will always be interested in future studies in the harbour. I have learnt many lessons during my MRes and I feel prepared to embark on a PhD in the future – though I feel a break is in order first!

# Chapter 1 - Macrozoobenthic Communities in Poole Park Lagoon

## 1. Introduction

### 1.1 Threats to Coastal Zones

Climate change induced sea level rise presents an imminent threat to coastal zones. It is deemed very likely that 95% of the ocean area will experience sea level rise, affecting 70% of coastlines worldwide (IPCC 2014). Hence, they are disproportionately threatened by habitat loss from the associated effects of climate change and coastal development, which are predicted to significantly decrease biodiversity within coastal regions (Hawkins et al. 2016; Hawkins 2012). Semi-enclosed coastal systems (SECs), including transitional ecosystems such as lagoons and saltmarshes, are among the most vulnerable coastal ecosystems to these anthropogenic pressures, particularly when associated with river mouth systems (Newton et al. 2014; Newton et al. 2012; Nicholls and Cazenave 2010; Eisenreich 2005).

Along the south coast of the UK, the estimated sea level rise compared to 1990 levels will be between 18.4cm and 25.8cm by 2050. The impact on Scottish coasts is less severe, with sea level rise estimated between 10.5cm and 18cm due to vertical land movement from loss of glaciation leading to local isostatic change (Bradley et al. 2009; IPCC 2007). The most pertinent threat is the flooding of low-lying coastal land with longer term effects involving changes to coastal geomorphology, such as increased erosion and changes to sediment dynamics. Additionally, warming oceans will intensify atmospheric pressure gradients leading to increased frequency and intensity of coastal storm events (Murphy et al. 2010; Woth et al. 2006; Dorland et al. 1999). The hydrological cycle for northern and central Europe is also projected to change, increasing precipitation and freshwater input to lagoon sites (IPCC 2007).

Subsequently, the salinity regime of many SECs may become less predictable and turbidity may increase due to stronger run-off currents.

## 1.2 Lagoons

Lagoons are typically defined as shallow bodies of brackish water partially separated from the adjacent sea by a barrier and where there is a restricted tidal exchange (Brown 1997; Barnes 1989). They are uncommon on the European Atlantic coast, comprising just over 5% of the coastline, with some subtypes rarer still, though these figures are now several decades old and require updating (Barnes 1980). Climate change related threats may lead to an increase in dystrophic events and mass mortality of lagoon fauna. Habitat loss is also a risk due to the inundation of seawater or infilling of sediment if a site is unable to retreat into the hinterland due to land reclamation (Carrasco et al. 2016; Anthony et al. 2009).

The definition of a lagoon has been applied to a broad spectrum of lagoonal sites that can include atypical characteristics, such as depths exceeding 2m (Oban nam Fiadh, Uist, and Arne lagoon, Dorset), but meet the physiographical criteria and host lagoonal specialists that would otherwise be outcompeted by their marine counterparts (Howson et al. 2014; Wheeler 2013). Lagoons are a Priority Habitat in Annex I of the EU Habitats Directive and under the UK’s Biodiversity Action Plan, and several UK sites have been selected as Special Areas of Conservation (SACs) including over 50 individual lagoons across 10 UK SAC sites (Williams 2006). Barnes (1989) identified a total of 41 lagoon sites in Britain in the late 1980s, some of which have been since reclassified, such as Poole Harbour on the southern English coast, which is now considered an estuary due to the salinity regime and volume of tidal exchange (May and Humphreys 2005). Most of Barnes’ original sites were on the English coast, but over 100 lagoon sites have since been identified on the Scottish coasts, predominantly confined to the Northern and Western isles (Howson et al. 2014).

### 1.2.1 Lagoon Ecosystem Services

A wide range of ecosystem services are provided by lagoons as they are highly productive, prevent erosion, are key sources of food for local communities, and provide wildlife refuge and nursery areas. They are also significantly important for the cultural services sector, providing cultural heritage, aesthetics, education and recreation (Velasco et al. 2017; Lopes and Videira 2013; Barbier et al. 2011). Increases in temperature, sea level rise and changes in freshwater input are predicted to negatively affect all ecosystem services provided by lagoons (Newton et al. 2018).

### 1.2.2 Effect of Sea Level Rise on Lagoons

Lagoons are a transitional system so by their very nature are ephemeral, exacerbated further by climate change and coastal squeeze. The predicted sea level rise, based on UKCP09 projections for London, is between 23cm and 53cm by the year 2095. The UK's stance on sea level rise was developed in the late 1990s by the Department of Environment and Rural Affairs (DEFRA) and accepts the likelihood of water ingress in coastal areas. It has several conditions it assigns to portions of coastline depending on the threat level and feasibility of different management strategies; *No active intervention, Hold the line, Managed realignment,* and *Advance the line* (Esteves 2014). In the scenario of *No active intervention* and *Managed realignment* current lagoon habitats may be lost as their defining barriers are overtopped. However, inevitably there will also be low-lying coastal sites that can *become* lagoons, providing refugia for lagoon specialists and the associated reliant food web. As lagoons are hydrologically impounded and the invertebrate fauna generally have a limited dispersal range, connectivity between lagoon sites can be low. It is important to consider a site within the wider context of regional lagoon ecosystems and network (Herbert et al. 2018; Perez-Ruzafa et al. 2018; Ghezzo et al. 2015).

## 1.3 Natural History of Poole Harbour

Having lagoonal characteristics itself, Poole Harbour was formed in the late Holocene and is an estuary of rivers Frome, Sherford, Corfe and Piddle (see

Figure 1). It is a shallow natural harbour with a maximum area of $13.9^2$ miles and an average depth of 48cm with a microtidal range of 0.6m-1.8m and a double high tide for a total duration of 16 of 24 hours (May and Humphreys 2005; May 2005; McClusky and Elliot 2004). The harbour's proportionally narrow entrance permits 22% to 45% of the water volume to leave on the ebb tide. Salinity range increases with distance from the harbour mouth, ranging by as much as 29.4‰ in Wareham Channel where freshwater input is greatest. It possesses extensive intertidal mud flats fringed with saltmarsh and reedbeds and is considered to host most types of British coastal habitat (JNCC 2008b; Gray 1985). The northeast coast of the harbour is significantly urbanised with the town of Poole, its surrounding suburbs and a recently expanded port. Such land reclamation has led to a local sea level rise of 26cm since the late 1800s (Edwards 2001; Pethick 1993). Conversely, the southwest coast possesses a notable absence of development, and most of the coastal land is owned by conservation and heritage charities and are designated nature reserves. For example, Arne is owned by the Royal Society for the Protection of Birds (RSPB) and Studland by the National Trust. Five islands exist in the central harbour area with little to no permanent human residence. This includes most notably Brownsea Island, the largest of the islands with an area of $0.77^2$ miles and owned by the National Trust, and Furzey Island, home to 22 oil wells of Wytch Farm Oil Field.

A site of international avian importance, Poole Harbour is a RAMSAR site, a Special Protected Area (SPA), and several Sites of Specific Scientific Interest (SSSIs) are locally designated under European Union Birds Directive and Habitats Directive legislation. Large numbers of resident, migratory and over-wintering birds utilise Poole Harbour and its various habitats to roost, feed and breed, and peak in the wintertime with a total abundance of approximately 25,000 individuals. Notable species include the protected avocet *Recurvirostra avosetta* (Linnaeus 1758), black-tailed godwit *Limosa limosa* (Linnaeus 1758), and common tern *Sterna hirundo* (Linnaeus 1758) (JNCC 2008b). The cord grass *Spartina anglica* (C.E. Hubb) dominates saltmarsh which occupies $1.24^2$ miles of the coastal area (Corkhill and Edwards 2006). This is decreasing in areas due to erosion, competition from the common reed *Phragmites australis* (Cavanilles) where salinity is low, and trampling

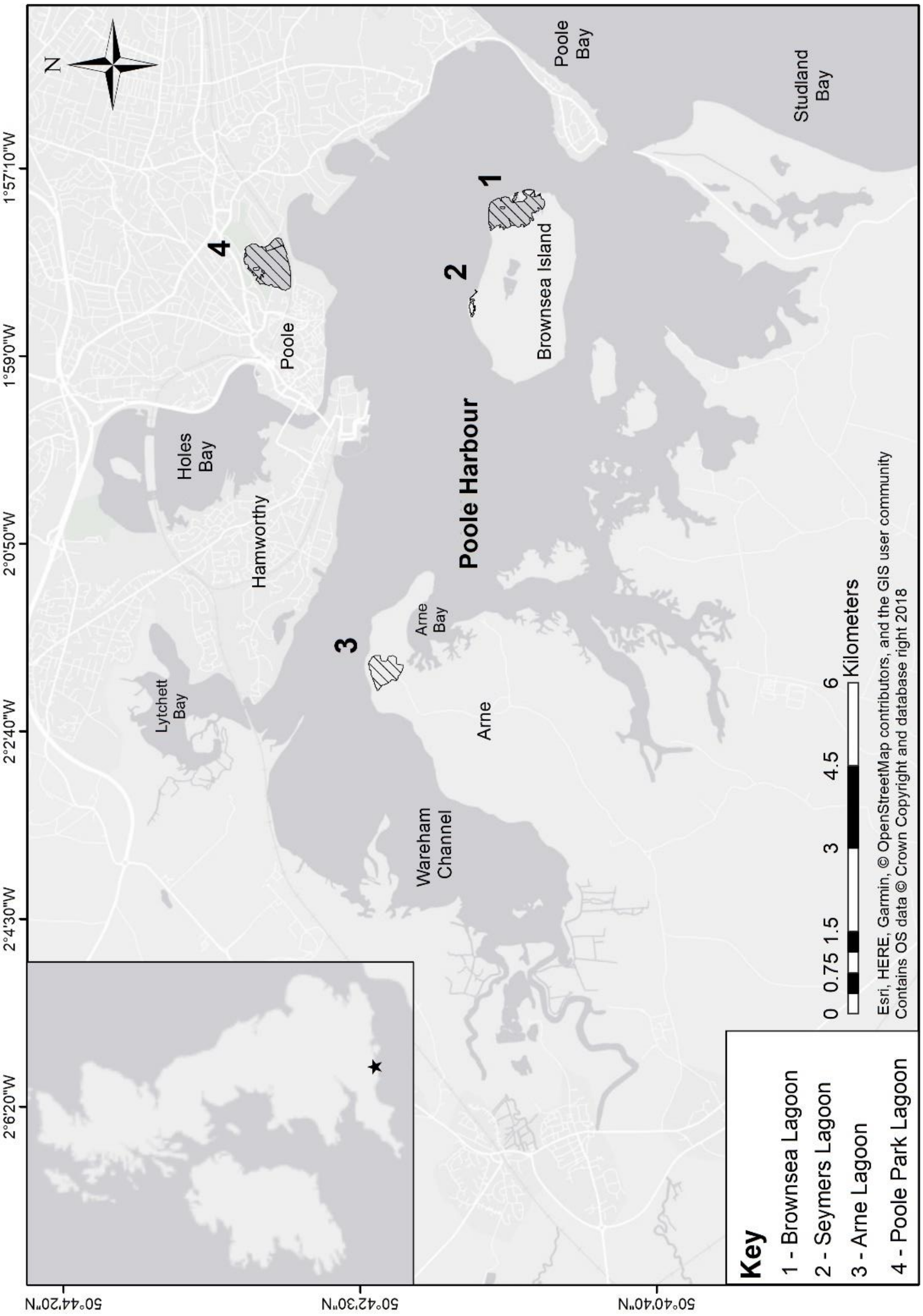

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Figure 1 The Poole Harbour lagoons.

occurs from the non-native Sika deer *Cervus nippon* (Temminck 1838) (House et al. 2005; Edwards 2004; Gray et al. 1991). Several small stony islands in the Wareham Channel host the largest nesting population of Schedule 1 protected Mediterranean gulls *Ichthyaetus melanocephalus* (Temminck 1820). However, these are fully immersed during high spring tides and are vulnerable to strong wave action and storm damage (Hopper 2016).

A maximum of 88 invertebrate species have been recorded in sediment cores from the harbour with a dominance of annelid and mollusc species, with the ragworm *Hediste (Nereis) diversicolor* (O.F Muller 1776) characteristic of assemblages throughout the harbour. Several lagoonal specialists also occur in the harbour such as the bivalve *Cerastoderma (Cardium) glaucum* (Bruguiere 1789) (Thornton 2016; Herbert et al. 2010; Barnes 1980).

The harbour includes four known lagoons; Brownsea, Seymers, Arne and Poole Park lagoon (see Figure 1).

### 1.3.1 Brownsea Lagoon

Brownsea Island lagoon was created in the late 1800s following the flooding of agricultural land on Brownsea Island and the subsequent enclosure by a sea wall (Herbert et al. 2018). The lagoon is fed by a freshwater stream and is connected to the harbour via an electronic pump and sluice. It is the longest established lagoon in terms of management within Poole Harbour and has been protected from development by the National Trust, who own the island, and the Dorset Wildlife Trust, who lease part of the land. It is also the most diverse, with the greatest species richness and abundance (Bone 2017; Herbert et al. 2010). It is incorporated within Poole Harbour SPA due to its importance as a feeding and roosting site for thousands of migrating and resident waterfowl that frequent the harbour each year. The current status of this threatened lagoon site is to *Hold the Line* in the short-term and *Managed Realignment* in the long term (National Trust 2015; Guthrie and Eggiman 2014).

### 1.3.2 Arne Lagoon

Arne lagoon, a repurposed clay quarry, was deliberately filled with sea water in 2012 via a channel that meanders through marshland, connecting it to Poole Harbour (RSPB 2012). Recent observations suggest that the lagoon has since been colonised by characteristic lagoon species such as *Hydrobiid* snails and the estuarine ragworm *H. diversicolor*, and the sand gaper *Mya arenaria* (Linnaeus 1758) was also observed *in situ* (Herbert et al. *in press*). Several species of coastal bird, such as cormorant *Phalacrocorax carbo* (Linnaeus 1758), spoonbill *Platalea leucorodia* (Linnaeus 1758) and little egret *Egretta garzetta* (Linnaeus 1766), were observed at the edge, demonstrating its capacity to provide a crucial habitat for the harbour’s internationally important populations of avifauna (personal observation).

### 1.3.3 Seymers Lagoon

Seymers lagoon was created incidentally in the late 1800s following the ingress of seawater in a clay mining works. Following this, a natural inlet channel formed, allowing the exchange of seawater. Its hydrodynamic regime limits the biodiversity in this small eutrophic lagoon, but characteristic specialist benthic fauna has been recorded, in addition to a number of bird species including teal *Anas crecca* (Linnaeus 1758) and shelduck *Tadorna tadorna* (Linnaeus 1758) (Bone 2017). It has not benefitted from lagoon-specific management like the main Brownsea lagoon but undergoes regular monitoring from the Dorset Wildlife Trust (personal communication). It is a small lagoon and would benefit from expansion and dredging the channel to improve the flushing regime.

### 1.3.4 Poole Park Lagoon

Poole Park lagoon is a recreational boating lake in an urban park in the Poole conurbation. It was created in 1890 when the railway line along the southern edge was constructed, impounding an intertidal bay with the railway embankment. Historically the lagoon supported high abundances of the lagoon cockle *C. glaucum* but this species now occurs only in low numbers (Harrison et al. 2016; Boyden and Russell 1972). It is now a sluiced lagoon that is flushed once a month on the high spring tide and faces significant anthropogenic pressures, such as pollution from surface water runoff that discharges into the lagoon via pipes. The limited tidal exchange combined with highly variable freshwater input can lead to dramatic ranges in salinity, limiting the species richness. Litter from the park users accumulates in the lagoon, such as plastic bottles, to the more unusual such as electric kettles (personal observation).

Anthropogenic disturbance and lack of intertidal mudflats limits the avifauna that visit the lagoon and is dominated by opportunistic species such as mute swans *Cygnus olor* (Gmelin 1789), Canada and greylag geese *Branta canadensis* and *Anser anser* (Linnaeus 1758), mallard ducks *Anas platyrhynchos* (Linneaus 1758), and gulls. The proliferation of algal blooms and nuisance swarms of non-biting midges (Chironomidae) that undergo their larval stages in the lagoon sediment have indicated that Poole Park lagoon is a degraded and hypereutrophic habitat (Harrison et al. 2016).

## 1.4 Rationale

It is evident that there is a relative paucity of literature and data on temperate lagoons, particularly on the coasts of the UK, with much work on European lagoons focused in the Mediterranean. Much of the work done by British lagoon ecologists Martin Sheader, Roger Bamber and Richard Barnes was pivotal in providing a baseline of physico-chemical parameters typical of British lagoons and the lagoonal fauna. However, apart from a handful of more recent studies, surveys and reports (see Howson et al. 2014), little has been done since the 1990s, particularly on English lagoons. Since then, taxonomic scientific techniques and methods have become more accurate and precise, which is crucial to determine positive identification of some of the more ambiguous invertebrates, such as the morphologically similar lagoon cockle *C. glaucum* and common cockle *Cerastoderma edule* (Linnaeus 1758) (Hummel et al. 1994; Barnes 1980). Additionally, it is important to maintain accurate data on the distribution, extent and ecological health of British lagoons in an ever-changing climate to ensure adaptive management.

There is historical evidence of lagoons becoming inundated with seawater in Poole Harbour. Blue Lagoon, once a disused saltern, became intertidal in the late 1980s leading to the loss of a regional lagoon habitat and a local population of the starlet sea anemone *Nematostella vectensis* (Stephenson 1935) (Sheader et al. 1997; Sheader and Sheader 1992; Sheader and Sheader 1985). With Brownsea lagoon facing habitat loss in the future, there is a need to assess the extant lagoons and potential/ unconfirmed lagoon sites in the region to determine their habitat health, their potential as refugia and to provide baseline data that will inform their management for the future. A survey focused on water quality was conducted in Poole Park lagoon in 2015 with some biotic data collected, giving a limited insight into the benthic communities. Macrozoobenthic communities are well-known indicators of habitat health and spatially comprehensive data would provide an assessment of the ecological condition of Poole Park and its capacity to support species from higher trophic levels such as birds (Arbi et al. 2017; Dauvin 2007; Rakocinski and Zapfe 2004; Simboura and Zenetos 2002).

In the late autumn of 2018 the lagoon will undergo works to improve public access and engagement, create islands for use by terns, and to improve overall ecology. Collecting spatially comprehensive abiotic and biotic data now will provide a baseline allowing the status of the lagoon's ecosystem health to be compared before and after the works have been completed.

## 1.5 Aims and Objectives

A study conducted in Poole Park in 2015 gave a snapshot view of macrozoobenthic diversity but focused on water quality. Biotic samples were only taken from two sites and recorded 13 species, with the annelid *H. diversicolor* and amphipod *Monocorophium (Corophium) insidiosum* (Crawford 1937) dominating (Harrison et al. 2016). This study aims to collect spatially comprehensive biotic and abiotic data in Poole Park lagoon to determine the primary environmental variables that dictate the macrozoobenthic communities and species. Measuring the sizes of the most widespread and abundant species can provide useful indicators of spatial ecological health (Petchey and Belgrano 2010).

Decreasing salinity with decreased proximity to the site where the lagoon connects to the sea is seen in Brownsea Lagoon, and also demonstrated on a larger scale in Poole Harbour (Herbert et al. 2010; Barnes 1989). Salinity may be higher closer to the sluice gate with hyposalinity occurring in the north of the lagoon where surface water input from pipes is greatest. It is predicted that proximity to the sluice gate will be negatively correlated with species diversity due to the higher energetic costs associated with osmoregulation (Arndt 1989).

The aims and objectives for this study are:

1) Collect spatially comprehensive baseline abiotic and biotic data by:
    a) Using standardized, replicable field and laboratory methodology
    b) Obtaining abiotic data measurements such as salinity, temperature, particle size, organic matter content, sediment depth and water depth
    c) Obtaining biotic data from benthic cores and fish traps.

2) Identify relationships between abiotic variables and assemblages by:
    a) Using ArcGIS to visually identify spatial trends
    b) Conducting statistical analyses
    c) Analysing the assemblage data within the context of abiotic variables using appropriate biotic metrics
    d) Measuring annelid *H. diversicolor* and amphipod *M. insidiosum* to determine size class frequency distributions to establish resident populations.

3) Identify, if any, relationships between pelagic and benthic fauna by:
    a) Using ArcGIS to visually identify spatial trends
    b) Conducting statistical analyses.

4) Discuss the ecological health of Poole Park lagoon and its potential for improved management by:
    a) Using existing literature, including Harrison et al.'s report from 2016
    b) Analysing results within the context of trophic ecology and comparing it to other lagoons in the region.

# 2. Methods and Materials

## 2.1 Study Site

Poole Park lagoon is a 0.21km$^2$ sluiced lagoon and the gate is opened once monthly on the spring high tide for flushing and is otherwise impounded. It also receives freshwater input from pipes that drain surface water from a catchment of approximately 2km$^2$ from the Poole town conurbation, including a licensed sewage overflow. As a result, freshwater input can be very variable and lead to extreme temporal values in salinity, contribute to contaminant loading and exacerbate eutrophication in the lagoon (Harrison et al. 2016).

Five islands were created in 2007 in the northwest with material dredged from the lagoon and bordered with wooden fencing, chicken wire and reeds to prevent erosion and access by geese and swans (see Figure 2). Since then, some of the islands have subsided and fallen into disrepair and are frequently used by avifauna. In the north and east, reedbeds have been used to create sheltered inlets and are in a similarly poor state. Reedbed One in the north is often populated by geese, swans and gulls which are fed by the users of the car park adjacent to the reedbed. Reedbed Two in the east is less accessible to park users as hedgerows prevent direct access to the lagoon. In the southeast a Concrete Walkway borders the Model Boating Area. The Sluice Gate is in the south of the lagoon and discharges into Parkstone Bay in the north of Poole Harbour. Rockley Watersports operate in the lagoon and are based in The Kitchen Café premises, offering schools and the public access to stand-up paddle-boarding, sailing, kayaking, and peddle boats.

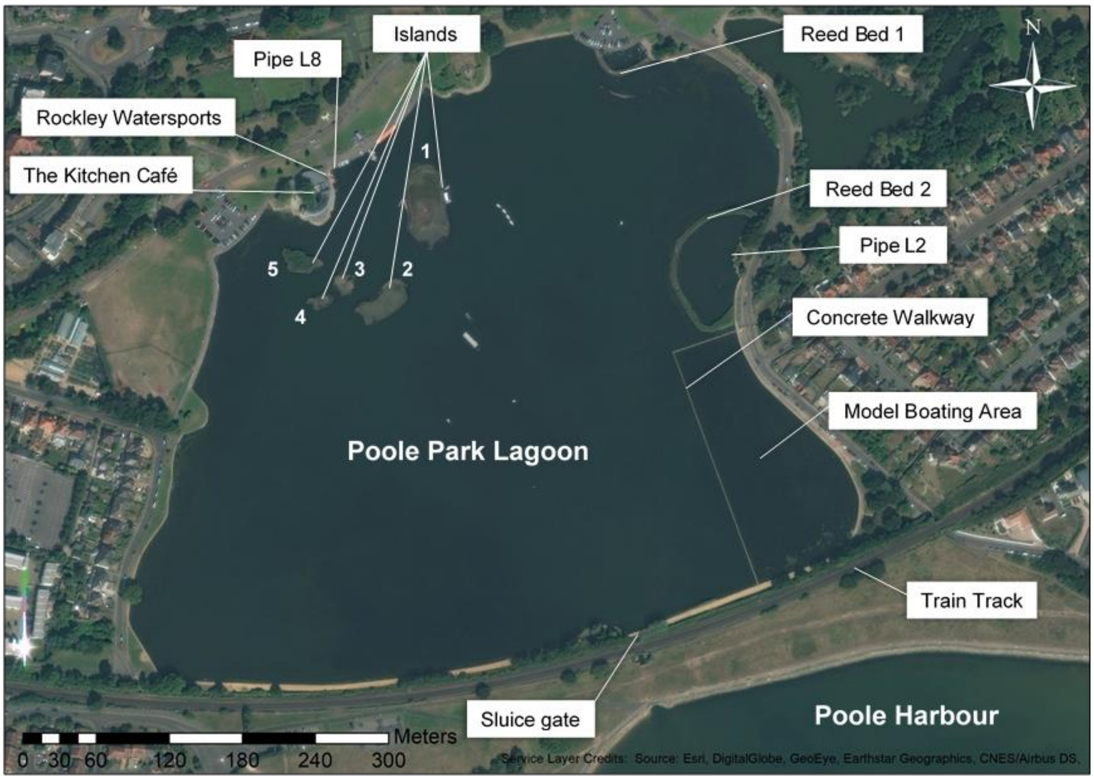

Figure 2 A map of Poole Park lagoon including key features.

Historically the lagoon was flushed on a frequent basis which regulated the salinity regime enough to sustain a relatively stable ecosystem. However, in recent years budgetary constraints have led to an infrequent flushing regime, leading to hypertrophic conditions and stochastic salinities. Unpleasant hydrogen sulphide odours and swarms of midges (Chironomidae), which undergo their larval phase in the mud, have been the subject of park user complaints and can be symptomatic of a degraded aquatic habitat. In 2015, the salinity varied by >25‰ throughout the year, and in early February 2015 varied by approximately 20‰ following a single flushing event. Lagoon depth is predominantly <1m but reaches 1.5m in depth around the islands and in the northeast. Historically, sediment depth is highest in the central to southwest region and overall ranges between 0-200cm (Harrison et al. 2016).

## 2.2 Abiotic Data

### 2.2.1 Fieldwork

Abiotic data at Poole Park lagoon was collected on November 8th and 28th 2017 and February 7th, 2018 with permission from Poole Borough Council and Natural England. The lagoon was accessed with a rigid inflatable boat (RIB). On November 8th and 28th, 49 sediment sample cores were obtained from sample points distributed approximately 65m apart on a grid using a 10cm diameter suction corer (see Figure 3). Samples were stored in plastic resealable bags and labelled with waterproof permanent marker and internally with waterproof paper and pen. Samples were frozen the same day to prevent decomposition of organic content. Aquatic and bathymetric data were obtained on February 7th, 2018. A ranging pole was used to determine water and sediment depth from a RIB and conductivity and temperature data were obtained using a Hach HQ40D multimeter probe.

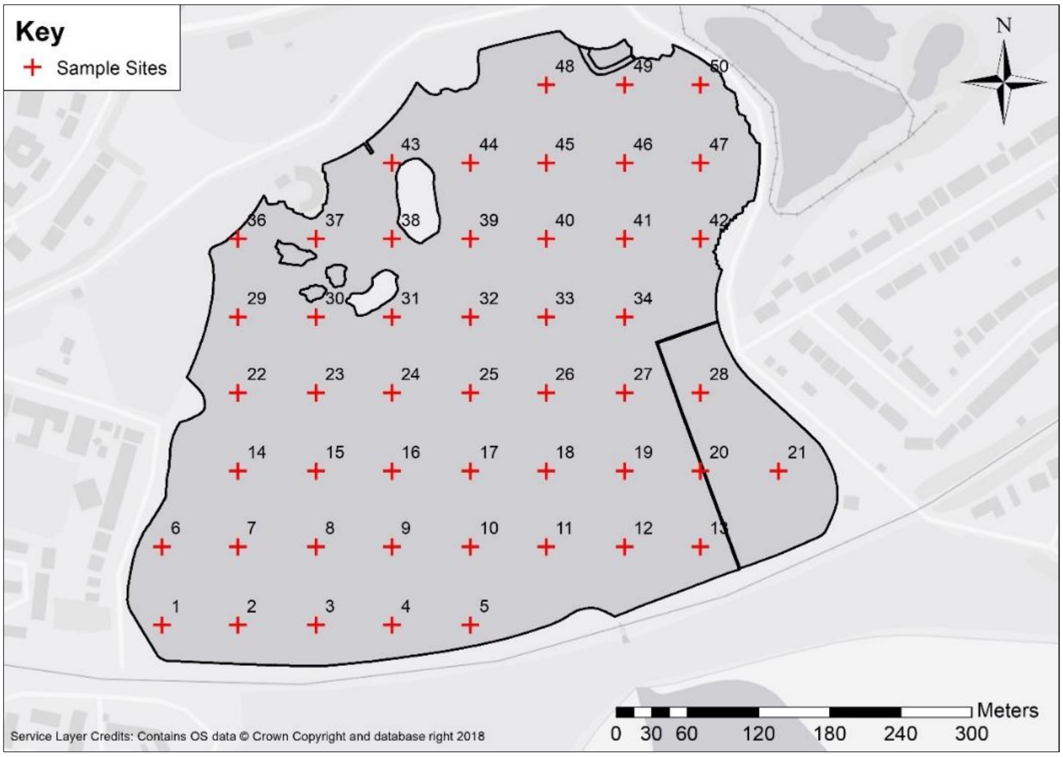

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Figure 3 A map of sampling sites.

### 2.2.2 Laboratory Analysis

The sediment samples were defrosted overnight, homogenised and sub samples were taken for sediment analysis. Organic content was measured by drying a homogenised subsample in a Memmert drying oven at 105˚C for 48h and then placing in a Carbolite chamber furnace at 450˚C for 12h and measuring the loss of mass on ignition (% LOI). Particle size analysis was obtained using a Malvern Mastersizer 3000 laser diffractometer with subsamples of the processed sediment. Samples were put through a 2mm sieve and particles greater than 2mm were weighed separately. Sieved samples were added to distilled water until the obscuration threshold was exceeded and readings commenced. Percentage silt content was determined by grain sizes ≤63µm and percentage sand content was determined by grain sizes >63µm (Wentworth 1922).

## 2.3 Biotic Data

### 2.3.1 Fieldwork

Fieldwork was conducted on November 7th and 8th 2017 and access was obtained with a RIB. Benthic sampling was conducted according to the procedural guidelines for sampling sediment species using cores as detailed in the JNCC Marine Monitoring Handbook and as per other benthic surveys at local lagoons (Bone 2017; Harrison et al. 2016; Herbert et al. 2010; Dalkin and Barnett 2001). Benthic fauna was sampled by taking fifty evenly distributed sediment cores using a 10cm diameter suction corer. Samples were immediately stored in plastic resealable bags, labelled with waterproof permanent marker and internally with waterproof paper and pen, before being sieved in-situ using a 0.5mm sieve to remove fine sediments and clay lumps. These fauna samples were taken back to the laboratory the same afternoon as collection and fixed using 10% formalin and seawater and stored in plastic pots labelled internally and externally.

Following a review of the macrozoobenthic and environmental data, pelagic faunal data were obtained to investigate the relationships between pelagic and benthic faunal communities in areas at the minima and maxima of organic matter content. Pelagic fauna hereafter refers to pelagic and demersal species and invertebrate species missed with core sampling such as crabs. This was to assess how biotic interactions from pelagic fauna may influence macrozoobenthic spatial distribution, particularly in areas where abiotic variables such as organic matter appear to have a strong influence, and accurately reflect the three-dimensional dynamism of the lagoon benthos. Pelagic fauna surveys were conducted in June 2018. Sites were chosen by visually assessing an interpolated GIS map showing the distribution of organic matter content (%) (see Figure 4). Ten sites showing the lowest percentage of organic matter and ten sites showing the highest percentage of organic matter were selected from the same points where the core samples had been taken for accurate spatial comparison. Permission to undertake the work was granted by the Environment Agency.

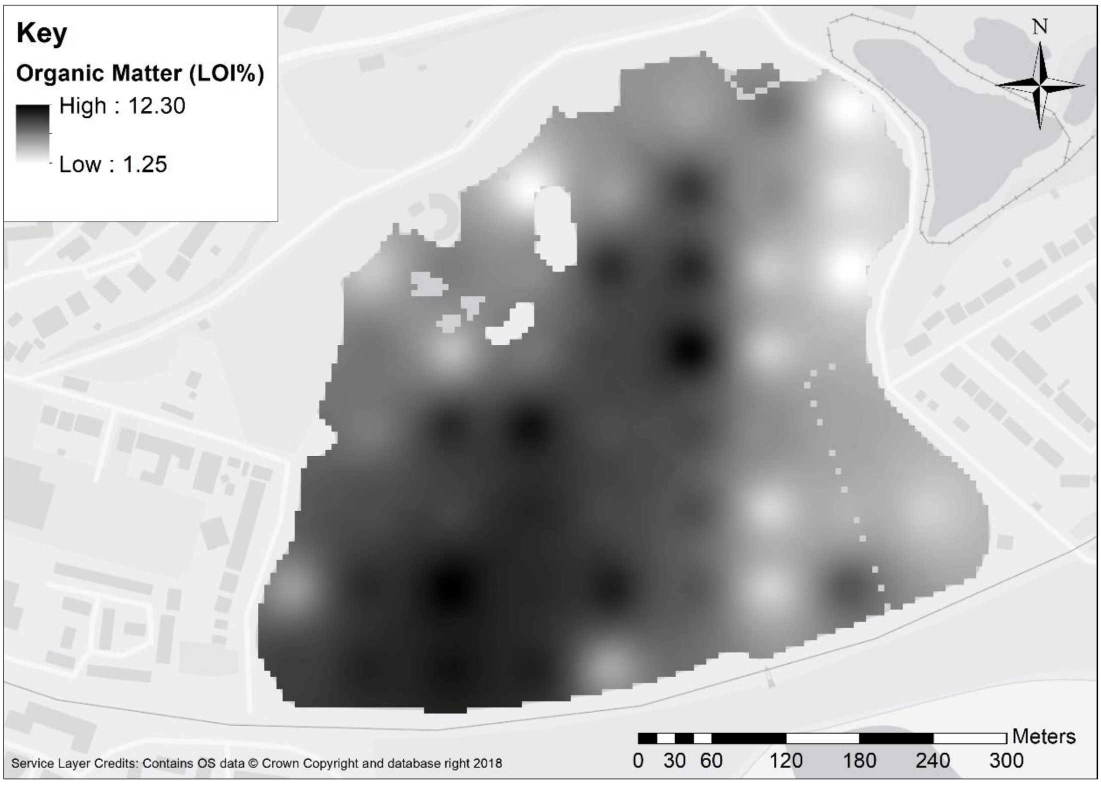

Figure 4 A map showing the distribution of benthic organic matter content in Poole Park lagoon.

Crayfish style traps were used to obtain pelagic data (see Figure 5). A previous study by Harrison et al. (2016) used seine nets to obtain fish biodiversity data but this method was used at the lagoon edges and was therefore unsuitable for the deeper and more central sample sites for this survey. Additionally, because the aim was to identify possible interactions with benthic macroinvertebrates, seine netting may have missed key pelagic and demersal nekton such as the common ditch shrimp *Palaemon (Palaemonetes) varians* (Leach 1813) and European green crabs *Carcinus maenas* (Linnaeus 1758). Crayfish style traps also had the benefit of being left *in situ* during nocturnal and crepuscular periods when many fish species are most active.

A field test was conducted prior to deployment where two traps were left overnight about 10m apart; one was baited with a small piece of tilapia (*Tilapia* sp.) fish and the other was not baited. The baited trap caught approximately 80 common ditch shrimp, eight common gobies *Pomatoschistus microps* (Kroyer 1838), and two European green crabs. The non-baited trap caught seven common gobies and 26 common ditch shrimp. Following this pilot study, it was decided not to use bait due to the length of time the traps would be deployed as the attraction of predators could lead to the mortality of other species caught in the traps. Additionally, the bait plume may attract individuals not local to the trap, providing inaccurate data (Heagney et al. 2007; Cappo et al. 2004).

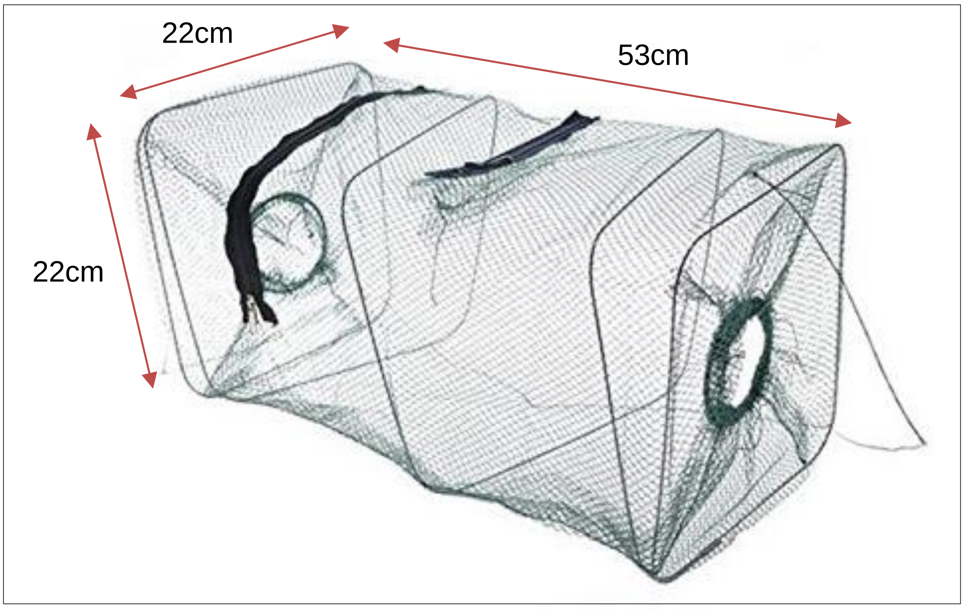

Figure 5 The collapsible crayfish style trap used for this study.

Twenty fish traps (22x22x53cm) with a 3mm mesh size were deployed for no longer than 24 hours at twenty sites (see Figure 6) overnight and collected the following morning to catch nocturnal and crepuscular species. Due to the restricted availability of Rockley Watersports providing boat access, traps were dropped at 11.00 on Thursday 28th June and collected at 10.30 on Friday 29th June. These timings were repeated for Thursday 19th July and Friday 20th July. Traps were removed from the lagoon and submerged in a tub filled with lagoon water on the boat to prevent mortality of sample specimens. Numeric abundance was determined before specimens were immediately returned to the site they were collected from.

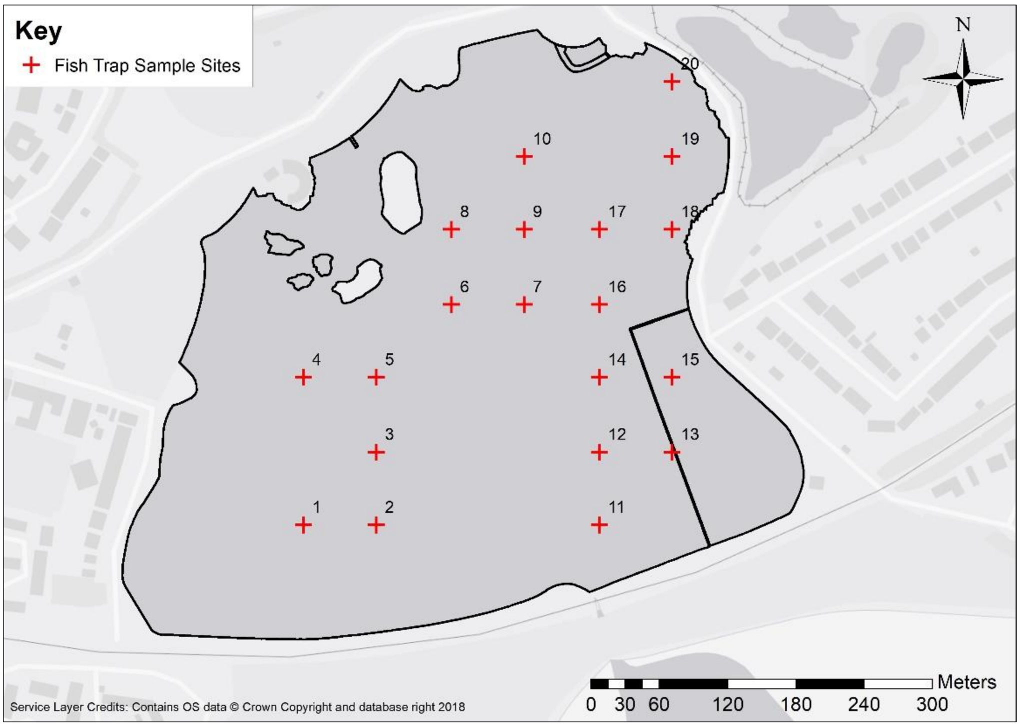

Figure 6 Map of the twenty sample sites where pelagic fauna were sampled. One to ten were at sites with the highest organic matter and eleven to twenty were at sites with the lowest organic matter.

### 2.3.2 Laboratory Analysis

The benthic fauna samples were rinsed of formalin with water in a ventilated fume cupboard for viewing under a microscope. Invertebrates were picked from the samples under a low power stereo microscope and placed in small tubes containing 70% Industrial Methylated Spirit (IMS) for identification and quality control purposes. Identification of individuals was made using appropriate keys to highest taxonomic resolution. Specimens without a head were discounted, as were bivalves which did not contain living tissue. The ragworm *H. diversicolor* and amphipod *M. insidiosum* were measured using a stage micrometer calibrated with an eyepiece graticule. The width of the head (peristomium) and length of the body from the prostomium to pygidium (when entire specimen was present) was taken for *H. diversicolor*. For entire specimens, body size data were collated on a spreadsheet and a line of best fit was found for the linear relationship between head width to body length, to extrapolate body length data for incomplete specimens. The body length of *M. insidiosum* was measured from the tip of the rostrum to the tip of the telson of all complete specimens.

## 2.4 Data Analysis

Data were collated onto Microsoft Office Excel spreadsheets for analysis. Species richness (S) and numeric abundance (N) were counted. Diversity indices, including Margalef's species diversity index, Shannon-Weiner index, and Pielou's evenness were produced with the DIVERSITY function (Clarke and Gorley 2006). Generally, biodiversity indices are most useful when used in comparison with another dataset to identify spatial or temporal change. Therefore, the mean biodiversity index values for Brownsea lagoon, Seymers lagoon and Poole Harbour mudflats are also given. These sites were chosen due to their inclusion within the Poole Harbour regional network and because singular surveys were conducted using the same standardised methodology in late Autumn (Herbert et al. 2010). It should be noted however that surveys at Brownsea and Seymers took samples from the lagoon perimeter and were not as spatially comprehensive or were not conducted at a fine scale. Previous Poole Park lagoon data from Harrison et al. (2016) has been omitted as it is incompatible for the purposes of this comparison because data were only collected from two central points and collated over several months.

Percentage of dominant taxa was also calculated. Density of individuals per $m^2$ was calculated by multiplying abundance by 127.324, the quotient of $1m^2$ divided by the suction core area ($\pi r^2 = 78.54$). IBM SPSS v. 25 was used to perform linear regressions and, where appropriate, polynomial regressions. Linearity was first established by visual inspection of a scatterplot. Normally distributed residuals were assessed by visual inspection of a normal probability plot. Where outliers were identified, the analyses were performed both with and without them.

An independent samples t-test was run to identify significant differences between pelagic fauna of areas of high and low organic matter, and between the surveys of June and July. The correct values as per the outcome of the Levene's Test for Equality of Variances are reported.

A multivariate analysis was conducted using Primer 7th Edition software (Clarke and Gorely 2006). Abundance data was square root transformed to standardise the data and to reduce influence of extreme values. A Bray-Curtis similarity matrix was created for the sample sites and a cluster analysis (CLUSTER) was run to identify statistically significant groupings of sample sites based on the assumption of no known assemblage structure similarities. A similarity profile analysis (SIMPROF) was run to detect structure in the macrozoobenthic assemblages.

The BEST function was used for the BIO-ENV procedure to determine which abiotic factors correlated with overall assemblage patterns using the resemblance matrix of biotic data, following a square root transformation, and normalised abiotic data, following a log transformation (Clarke and Gorley 2006). The following abiotic variables were considered: Silt Content, Organic Matter Content (OM Content), Median Grain Size (MGS), Water Depth (WDepth), Sediment Depth (SDepth), Salinity, Temperature, Distance from Sluice Gate, Distance from Southwest Shore (Distance from SW Shore), Distance from East Shore (Distance from E Shore), and Distance from North Shore (Distance from N Shore). A draftsman plot was created of all the untreated variables and a visual inspection of linear relationships within some plots suggested co-linearity.

Abiotic variables were tested for co-linearity using a Pearson correlation test in SPSS. A subset of co-linear variables was identified exceeding the accepted threshold of 0.7; Silt Content, OM Content and MGS (Dormann et al. 2013). MGS was obtained from the Malvern Mastersizer output during the particle size analysis. MGS was chosen as a proxy to represent the co-linear sediment variables and Silt Content and OM Content were omitted from Primer computations (Clarke and Warwick 2001). To ensure this variable was the best representative, interpolated layers of Silt Content, OM Content, and MGS were visually compared on an ArcMap of Poole Park which showed that MDS was closely associated with Silt and adequately reflected the OM Content distribution. Following this, the MGS proxy variable was renamed 'Sediment' in subsequent analyses to ensure interpretation of results considers the subset represented and not the singular retained variable. Temperature and Distance from N Shore were co-linear, but both remained in subsequent analyses as they were not functionally related in the same way the co-linear sediment variables were. Distance from SW Shore and Distance from N Shore were also co-linear, but both remained in subsequent analyses. Co-linearity was not deemed to be incidental as the sample dataset was large and representative of the full geographic area being surveyed.

Following the removal of co-linear sediment variables, a draftsman plot of untreated remaining abiotic variables was visually inspected, and skewed variables were log transformed (SDepth, Salinity, Temperature). A BIO-ENV procedure was run using the following abiotic variables: SDepth, Salinity, Temperature, Distance from Sluice, Distance from SW Shore, Distance from E Shore, Distance from N Shore, and Sediment. The correlation method was Spearman rank with 999 permutations using Euclidean resemblance.

ArcMap (ArcGIS) was used to provide spatially representative maps of abiotic and biotic data. Abiotic data were subject to the Inverse Distance Weighted (IDW) interpolation function, which extrapolates abiotic variables from existing sample points to provide a contoured surface and show gradients across the lagoon area. IDW interpolation was chosen over other available interpolation functions as the sample points were densely and evenly distributed enough for IDW to capture the extent of local variation necessary for analysis.

Interpolation was not utilised for biotic data which were instead overlaid as points for visual assessment of correlations with interpolated abiotic variables.

Lagoon biotopes were assigned as per the classification in Bamber (1997), however they are not particularly inclusive and require revision and expansion.

# 3. Results

## 3.1 Abiotic Conditions and Patterns

Most environmental variables measured displayed clear spatial patterns or gradients (see Appendix Table 1 for full results).

Sediment factors included Silt Content, MGS, OM Content, and SDepth. Silt Content ranged between 18.7 and 95.4% by 76.6% and had a mean value of 73.3%. Silt Content was highest in the central-southwest region (henceforth referred to as the CSW) and the central-northeast region (henceforth referred to as the CNE) with at least 85% of particles ≤63μm. Silt Content decreases rapidly eastward across a defined line almost perpendicular to the sluice gate and is mostly 65% or less in the east, northeast and northwest edge, decreasing to <35% in three periphery sites in the north (see Figure 7).

MGS ranged between 18.3μm and 230.4μm by 212.1μm and had a mean value of 57.9μm. MGS follows a very similar spatial pattern to Silt Content with the finest grains of ≤73μm occupying the CSW and becoming progressively and rapidly coarser towards the CNE, the northwest of Island One and a periphery site in the southeast (see Figure 7).

OM Content ranged from 1.2% to 12.3% by 11.1% and had a mean value of 7.4%. OM Content was similar in spatial distribution to MGS and Silt Content, with the same east, northeast and northwest periphery sites showing lower OM Content with ≤4.5% and the CSW showing up to 12.3%. Unlike Silt Content and MGS, OM Content is also low in the central east area, not just periphery sites (see Figure 7).

SDepth ranged from 0.1 to 2.0m by 1.9m and had a mean value of 0.6m. It shows a broadly similar pattern to the other sediment factors but is a lot simpler. SDepth is deepest at the central south edge at 2m, just west of the sluice gate. Almost a third of the SDepth is shallow with ≤0.3m in the east and northeast, including a southwest periphery site (see Figure 7). SDepth from the current study is broadly similar to the SDepth measured by Harrison et al. (2016) with the CSW dominated by deeper sediments and the east and northeast very shallow, indicating very little movement of sediments since 2015.

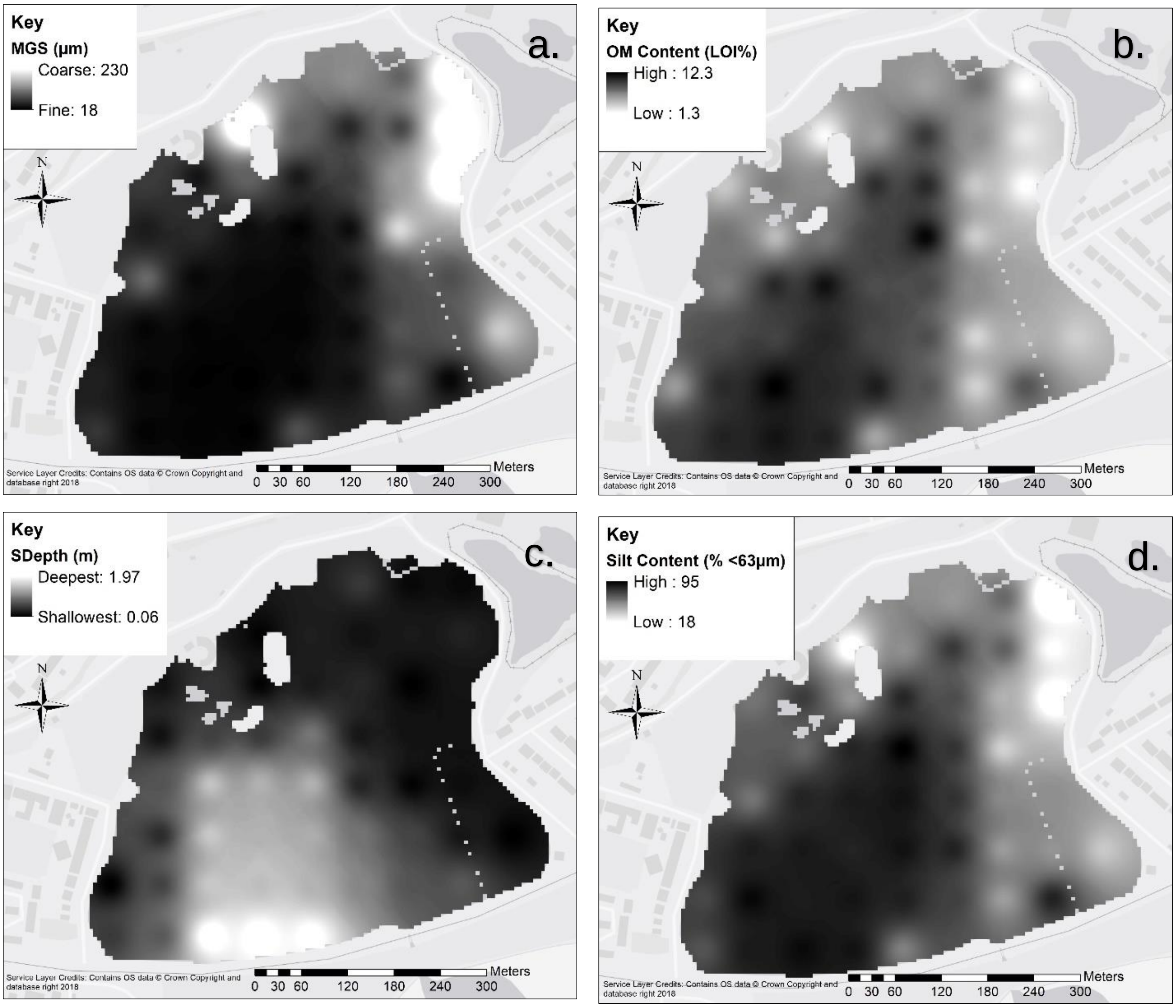

Figure 7 The distribution of sediment factors in Poole Park lagoon including a) MGS, b) OM Content, c) SDepth and d) Silt Content.

Aquatic variables included Temperature, Salinity and Water Depth (WDepth). Temperature ranged from 1.3°C to 3.7°C by 2.4°C, had a mean value of 2.8°C and was overall relatively homogenous. A southwest to northeast gradient can be seen with colder water in the south and becoming warmer in the northeast (see Figure 8). Two periphery sites on the south bank are particularly cool ranging between 1.3°C and 2.0°C but the sites immediately surrounding these anomalous points are between 2.4°C and 2.8°C which is characteristic of the southern temperature regime of that day.

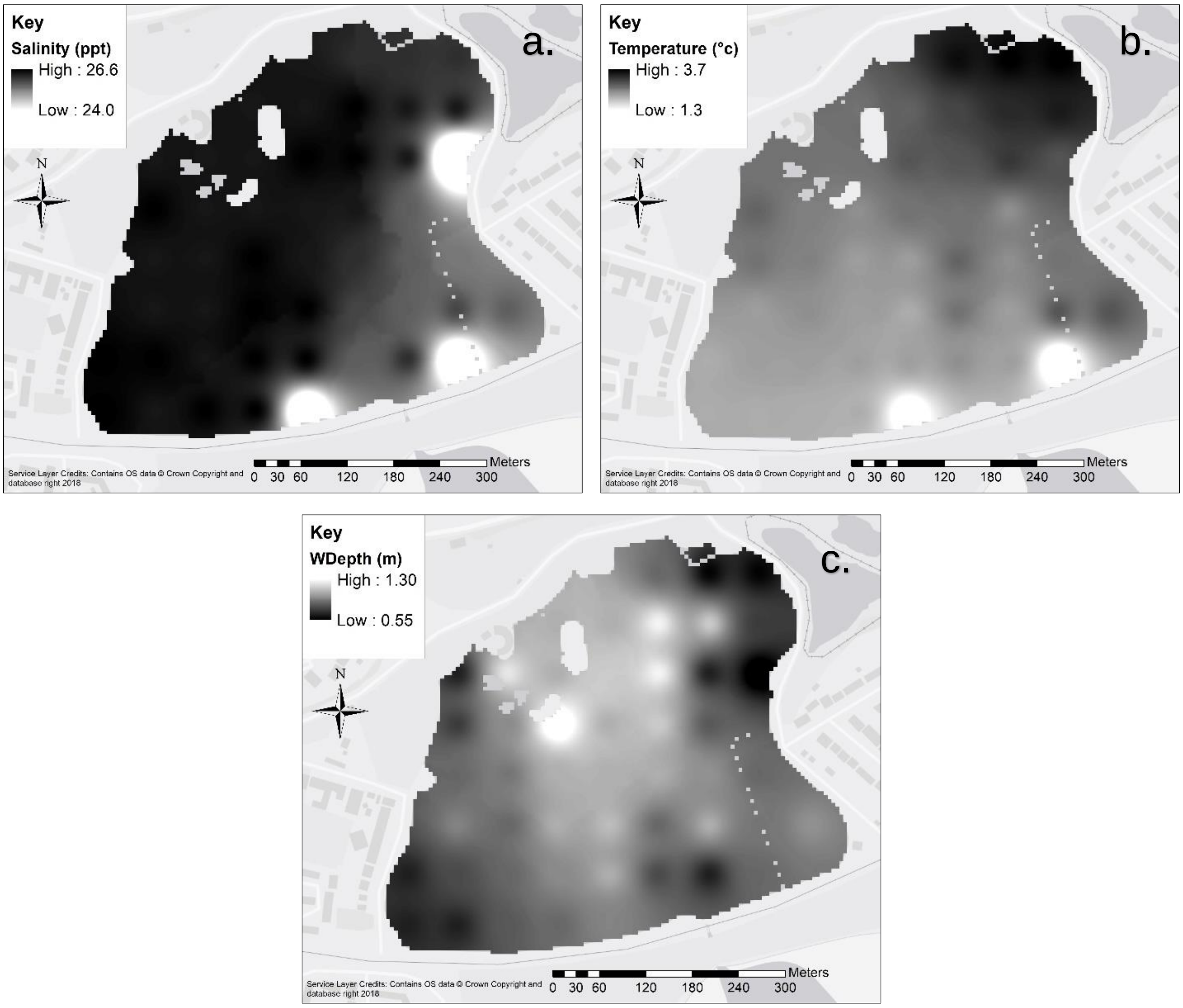

Figure 8 The distribution of aquatic factors in Poole Park lagoon including a) Salinity, b) Temperature and c) WDepth.

Salinity ranged from 24‰ to 26.6‰ by 2.6‰ and has a mean value of 26.4‰. Salinity does not follow the southwest-northeast gradients seen in other environmental variables and appears more northwest-southeast (see Figure 8). Over half of the lagoon ranges between 26.3‰ and 26.6‰ and then in the southeast decreases to 26.0‰ with three periphery sites in this area as low as 24‰. Two of these three points are shared with the anomalous sites for Temperature, suggesting a freshwater source may be locally affecting these areas.

WDepth ranges from 0.6m to 1.3m by 0.8m and has a mean value of 0.9m WDepth is deepest by up to 1.3m in the central north which then decreases in depth to its shallowest points in the southwest and northeast corners (see Figure 8). Bathymetry in this study is very similar to the bathymetry recorded in Harrison et al. (2016).

## 3.2 Overall Macrozoobenthic Biodiversity

The overall numeric benthic abundance across the lagoon totalled 2260 individuals with a total of 15 species and a mean density of 5873m$^2$ which was influenced by sites with disproportionately higher densities; most sample sites were below average density with just 35% above average density. Density ranged from 127 individuals per m$^2$ to 50,930 individuals per m$^2$. The most abundant phylum was Crustacea (see Figure 9) with a mean density of 3448 individuals per m$^2$, dominated by the lagoon amphipod *M. insidiosum* and the amphipod *Microdeutopus gryllotalpa* (Costa 1853). The second most abundant phylum, and the most species rich, was Annelida with a mean density of 940 individuals per m$^2$, dominated by the ragworm *H. diversicolor*. The least abundant phylum was Mollusca with a mean density of 75 individuals per m$^2$, dominated by the lagoon spire snail *Ecrobia (Hydrobia) ventrosa* (Montagu 1803). Phylum Cnidaria and Class Insecta were the least taxonomically rich with only one representative species each, though with relatively high mean densities of 868 and 541 individuals per m$^2$ respectively. Lagoon specialists, indicated with an asterisk * in Table 1, comprised 52% of the fauna with a mean density of 3061m$^2$ (Bamber et al. 2001). Non-native

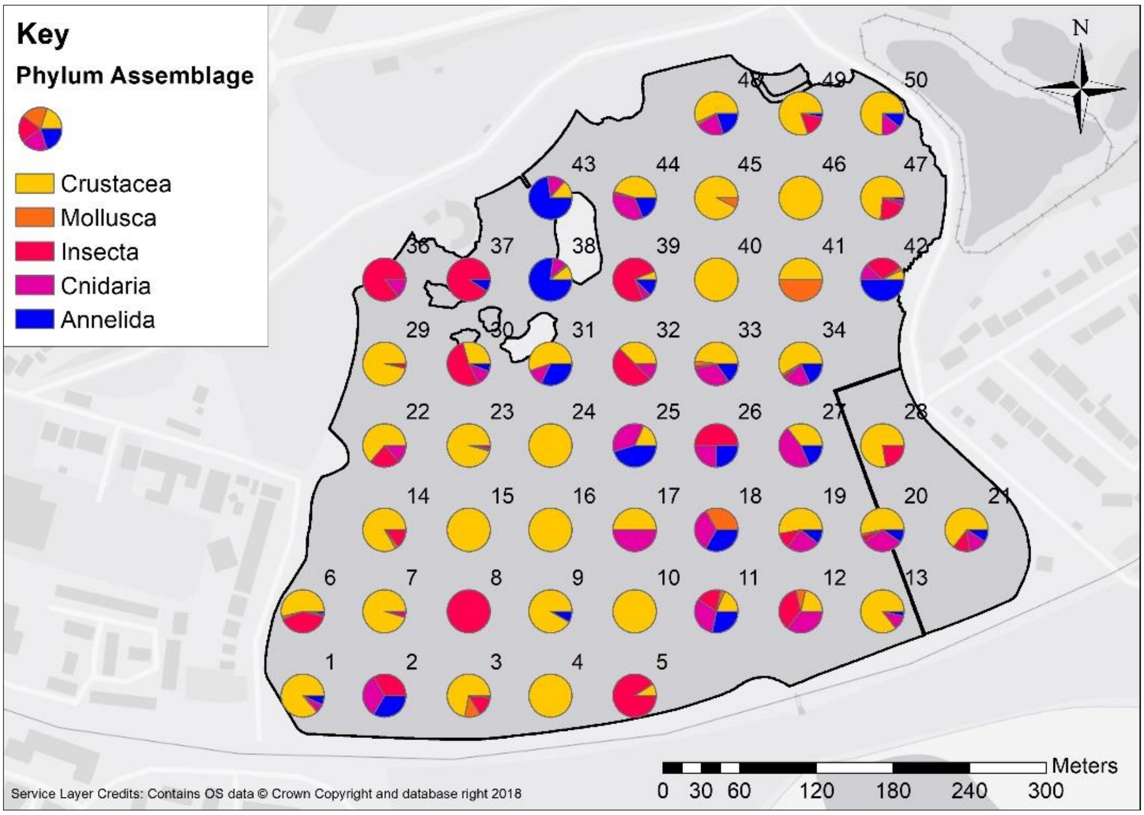

Figure 9 The spatial distribution of phylum assemblages.

and cryptic species (including the anemone *N. vectensis*), indicated with ▲ in Table 1, comprised 15% of the fauna with a mean density of 891m$^2$. *M. arenaria* is included as a cryptic species as, although historically native, it became extinct in the Pleiocene Epoch 1.6 million years ago and has since been introduced from extant populations in the 1500s or 1600s (Petersen et al. 1992; Foster 1946).

Table 1 The species, mean and maximum density of the macrozoobenthic fauna of Poole Park. Note that the minimum density for all species was 0.

| Scientific Name | Common Name | Mean Density ($m^2$) | Maximum Density ($m^2$) |
|---|---|---|---|
| PHYLUM CNIDARIA | | | |
| _Nematostella vectensis_*▲ | Starlet sea anemone | 867.88 | 9803.90 |
| PHYLUM ANNELIDA | | | |
| _Ficopomatus enigmaticus_▲ | Australian tubeworm | 2.60 | 127.32 |
| _Hediste diversicolor_ | Ragworm | 649.61 | 7512.10 |
| _Phyllodoce mucosa_ | Polychaete | 2.60 | 127.32 |
| _Polydora_ sp. | Polychaete | 202.68 | 2164.50 |
| _Tubificoides_ sp. | Oligochaete | 83.15 | 3055.80 |
| PHYLUM CRUSTACEA | | | |
| _Gammarus locusta_ | Amphipod | 15.59 | 381.97 |
| _Melita palmata_ | Amphipod | 44.17 | 763.94 |
| _Microdeutopus gryllotalpa_ | Amphipod | 1239.46 | 14515.00 |
| _Monocorophium insidiosum_* | Amphipod | 2148.92 | 18462.00 |
| PHYLUM MOLLUSCA | | | |
| _Cerastoderma glaucum_* | Lagoon cockle | 12.99 | 254.65 |
| _Ecrobia ventrosa_* | Lagoon spire snail | 31.18 | 509.3 |
| _Mya arenaria_▲ | Sand gaper | 20.79 | 763.94 |
| _Peringia ulvae_ | Spire snail | 10.39 | 381.97 |
| PHYLUM INSECTA | | | |
| Chironomidae sp. | Chironomid larvae | 540.48 | 3819.70 |

Table 2 Regression analyses for overall density and abiotic factors. Significant p values in bold. Site 34 was a consistent outlier for all regression analyses.

| **Density and Abiotic Factors of Poole Park Lagoon** | | | | | | |
|---|---|---|---|---|---|---|
| **Variable** | **Regression** | **$R^2$** | **DF** | **F** | **p** | **Unstandardized Co-efficient B** |
| OM Content | Linear | 10.8 | 1, 47 | 5.709 | **0.021** | -864 |
| Silt Content | Linear | 12.8 | 1, 47 | 6.879 | **0.012** | -158 |
| MGS | Linear | 14.2 | 1, 47 | 7.778 | **0.008** | 60.222 |
| SDepth | Linear | 6.8 | 1, 47 | 3.455 | 0.069 | -3948 |
| Temperature | Quadratic | 4.2 | 2, 46 | 1.006 | 0.373 | 4271 |
| Salinity | Quadratic | 12.6 | 2, 46 | 3.326 | **0.045** | -8588 |
| WDepth | Linear | 0.7 | 1, 47 | 0.342 | 0.562 | -4606 |
| Distance from SW Shore | Linear | 9.2 | 1, 47 | 4.786 | **0.034** | 16 |
| Distance from E Shore | Linear | 4.3 | 1, 47 | 2.095 | 0.154 | -13 |
| Distance from N Shore | Linear | 5.5 | 1, 47 | 2.722 | 0.106 | -13 |
| Distance from Sluice | Linear | 0.4 | 1, 47 | 0.172 | 0.68 | 4.57 |

Overall invertebrate density was significantly negatively correlated with OM Content, Silt Content and Salinity and significantly positively correlated with MGS (see Table 2, Figure 10). Density was lowest in the CSW portion of the lagoon and highest around the north and eastern edge. Density peaked at Site 34, which was consistently flagged as an outlier in regression analyses, with 50,928 individuals per $m^2$. Density was poorest at Site 4 with just 1 individual in the core sample, extrapolated to a density of 127 individuals per $m^2$.

Species richness was significantly negatively correlated with OM Content, SDepth, Silt Content and significantly positively correlated with Distance from SW Shore and MGS (see Table 3, Figure 11). Species richness was lowest in the CSW portion of the lagoon and highest around the north and eastern edge. Species richness peaked at Site 34, which was consistently flagged as an outlier in regression analyses, with 11 species. Richness was poorest at Sites 4, 8, 15, 40 and 46 with 1 species. Mean species richness was 4.

Table 3 Regression analyses for species richness and abiotic factors. Significant p values (<0.05) in bold.

| Species Richness and Abiotic Factors of Poole Park Lagoon | | | | | | |
|---|---|---|---|---|---|---|
| **Variable** | **Regression** | **$R^2$** | **DF** | **F** | **p** | **Unstandardized Co-efficient B** |
| OM Content | Linear | 14.2 | 1, 47 | 7.757 | **0.008** | -0.277 |
| Silt Content | Linear | 17.8 | 1, 47 | 10.151 | **0.003** | -0.053 |
| MGS | Linear | 17.2 | 1, 47 | 9.761 | **0.003** | 0.019 |
| SDepth | Linear | 13.4 | 1, 47 | 7.263 | **0.01** | -1.548 |
| Temperature | Linear | 2.0 | 1, 47 | 0.953 | 0.334 | 0.754 |
| Salinity | Quadratic | 7.2 | 2, 46 | 1.771 | 0.181 | -1.723 |
| WDepth | Linear | 1.9 | 1, 47 | 0.902 | 0.347 | -2.086 |
| Distance from SW Shore | Linear | 9.2 | 1, 47 | 4.747 | **0.034** | 0.004 |
| Distance from E Shore | Linear | 5.4 | 1, 47 | 2.709 | 0.106 | -0.004 |
| Distance from N Shore | Linear | 4.1 | 1, 47 | 1.985 | 0.165 | -0.003 |
| Distance from Sluice | Linear | 0.1 | 1, 47 | 0.033 | 0.858 | 0.001 |

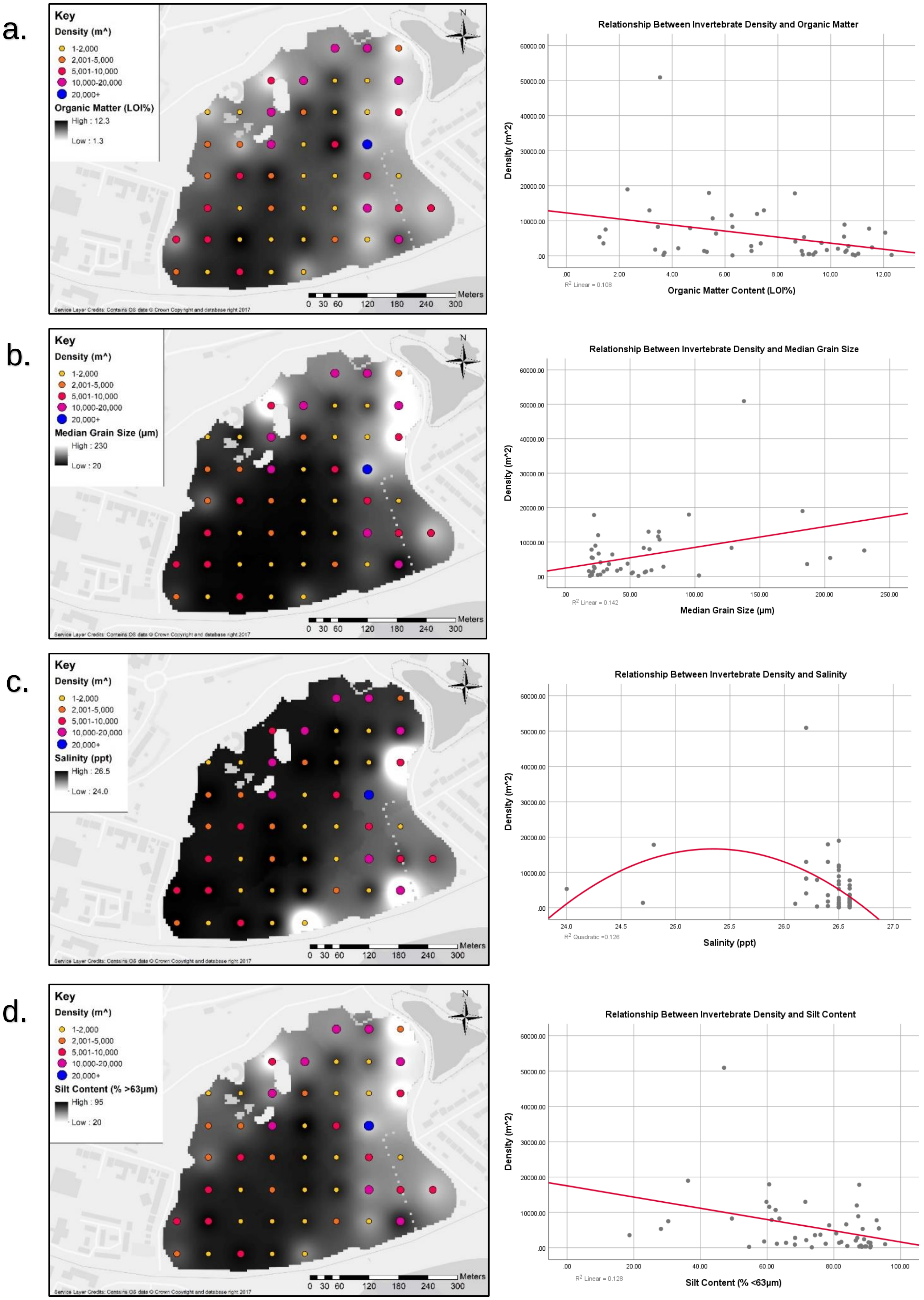

Figure 10 Overall density with significantly associated abiotic variables, including maps of the spatial distribution of abiotic gradients and invertebrate density and scatter graphs demonstrating relationships. a) OM Content, b) MGS, c) Salinity and d) Silt Content.

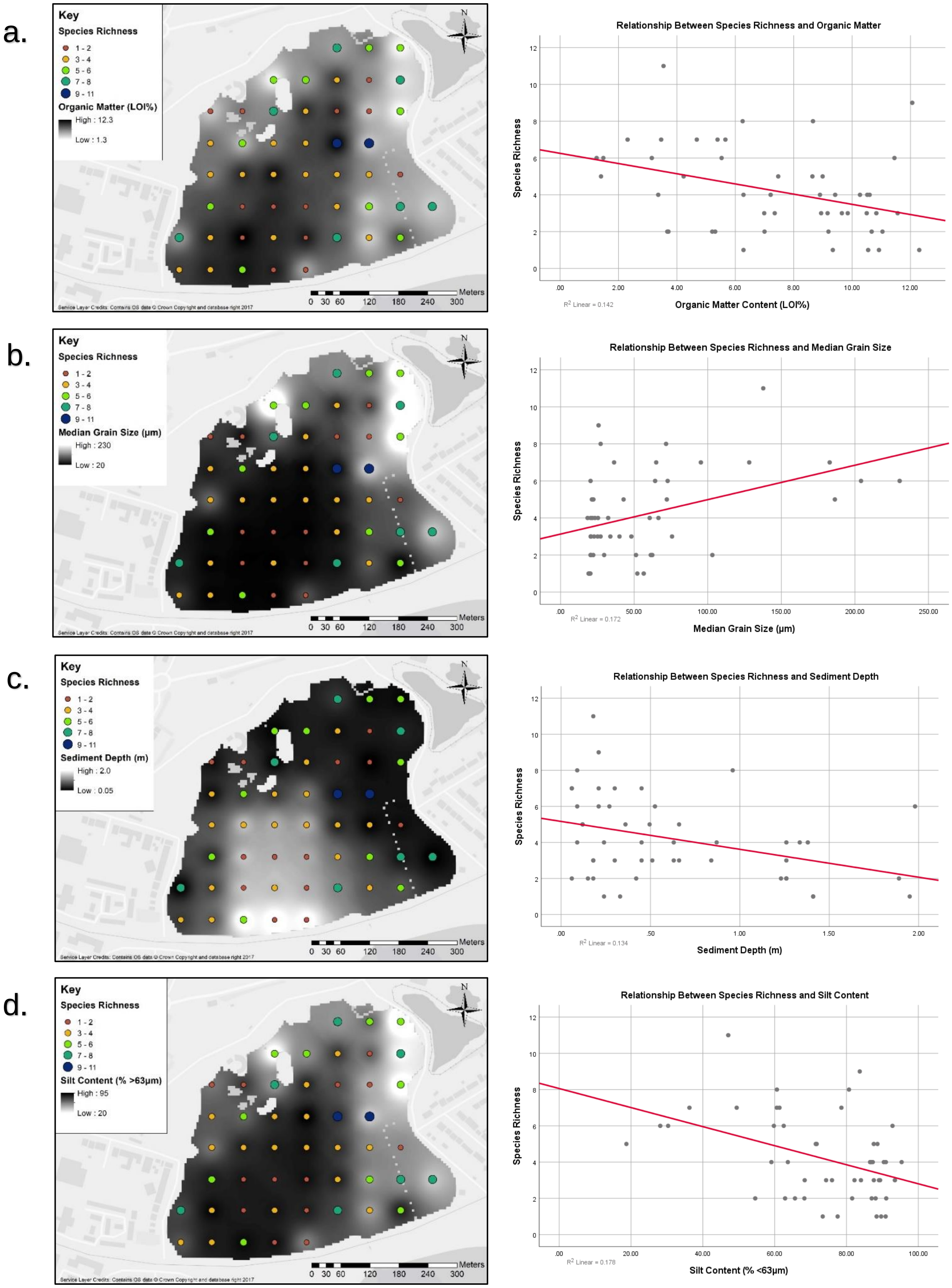

Figure 11 Species richness with significantly associated abiotic variables, including maps of the spatial distribution of abiotic gradients and species richness and scatter graphs demonstrating relationships. a) OM Content, b) MGS, c) SDepth and d) Silt Content.

## 3.3 Matching Abiotic Variables to Assemblages

BIO-ENV indicated that the BEST matching environmental variables to the resemblance matrix of the square root-transformed data was SDepth with a Spearman rank correlation (Rho) of 0.130 and significance value of 0.46. The addition of Temperature or OM Content to Sdepth, which were included in the next best models, reduced the correlation further to 0.105 and 0.099 respectively. These correlations are quite low. Variables Distance from Sluice, Distance from SW Shore, and Distance from E Shore were not among the top ten correlations.

## 3.4 Biodiversity Indices

Shannon-Weiner, Margalef and Pielou’s Evenness biodiversity indices were used to assess biodiversity (see Table 4) though their relative values varied.

Table 4 The Shannon-Weiner, Margalef and Pielou's Evenness biodiversity indices of Poole Park lagoon.

| Biodiversity Index | Range | Mean |
|---|---|---|
| Shannon-Weiner (H’) | 0 – 1.84 | 0.95 |
| Margalef (d) | 0 – 1.82 | 0.96 |
| Pielou’s Evenness (J) | 0 - 1 | 0.68 |

For example, Site 2 only had 3 species and a numeric abundance of 3 but scored above average for all three indices (Shannon-Weiner: 1.10, Margalef: 1.82, Pielou’s Evenness: 1.0), as the distribution of abundance between species was evenly split and there was no overwhelming dominance. Site 6 had 7 species and a numeric abundance of 50 but scored below average for all three indices, as 4 of those species only numbered 1 individual, with the remaining abundance distributed between the remaining 3 species (Shannon-Weiner: 0.31, Margalef: 0.42, Pielou’s Evenness: 0.44).

The biodiversity indices for Brownsea, Seymers, Poole Park lagoon and Poole Harbour mudflats were compared (see Table 5).

Table 5 The Shannon-Weiner, Margalef and Pielou's Evenness biodiversity indices of comparative lagoon sites Brownsea, Seymers and Poole Park and Poole Harbour mudflats (Bone 2017; Herbert et al. 2010).

| Poole Harbour Site | Shannon-Weiner (H') | Margalef (d) | Pielou's Evenness (J) |
|---|---|---|---|
| Brownsea Lagoon | 1.67 | 2.01 | 0.63 |
| Seymers Lagoon | 0.84 | 0.97 | 0.86 |
| Poole Harbour Mudflats | 1.49 | 1.28 | 0.63 |
| Poole Park Lagoon | 0.95 | 0.96 | 0.68 |

As previously noted by Bone et al. (2017), though in reference to historic Poole Park macrozoobenthic data (Harrison et al. 2016), Poole Park and Seymers lagoon are most similar in biodiversity as both are considered eutrophic degraded habitats, though Poole Park ranks lower overall. Poole Harbour mudflats and Brownsea lagoon rank considerably higher with Margalef and Shannon-Weiner and have much higher species richness and abundance than Seymers or Poole Park. However, both rank lower with Pielou's Evenness at 0.63.

## 3.5 SIMPROF Analysis and Groupings

This study has collected comprehensive assemblage data which the Primer SIMPROF function divided into four statistically significant groupings (groups A, B, C and D) (see Figure 12). Statistically significant groups are denoted by

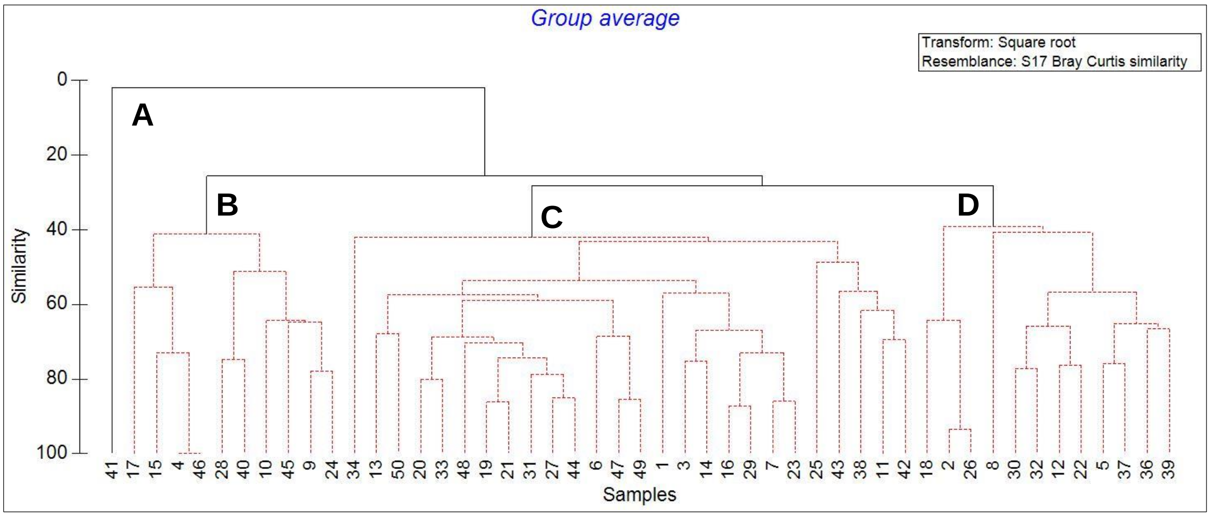

Figure 12 The SIMPROF groupings of Poole Park lagoon.

complete black lines. The assemblages, species richness, and relative abundance of these groupings was analysed with reference to complementary abiotic data, to identify trends and patterns.

### 3.5.1 Group A

Group A notably consisted of Site 41 only and is characterised as a site of extremes; it exhibited the lowest Silt Content (54.58%), OM Content (3.67%) and SDepth (0.06m) and the highest Temperature (3.3°C). Its faunal assemblage consisted of relatively rare species; one snail *E. ventrosa*, and one amphipod *Gammarus locusta* (Linnaeus 1758). Site 41 was in the north east immediately adjacent to Reedbed 2 that enclosed a small section of water, so would experience high energy conditions when windy. It was also relatively shallow, which likely accounted for the higher Temperature. Its proximity to a structure may have led to a local high abundance of fish predators that limited richness and abundance, in addition to coarser sediment just 6cm in depth, limiting its capacity to support infaunal invertebrates.

### 3.5.2 Group B

Group B consisted of 10 samples (Sites 4, 9, 10, 15, 17, 24, 28, 40, 45, 46) and was characterised by very low abundances (n = ≤19, mean = 7.6), low species richness (S = ≤4, mean = 2, total S = 6) and was dominated almost exclusively by amphipods (92%), including *M. insidiosum, M. gryllotalpa,* and *Melita palmata* (Montagu 1804). Group B also included all four monospecific sites dominated by the lagoon amphipod *M. insidiosum*. Other invertebrates within this grouping included the annelid *Phyllodoce mucosa* (Orsted 1843), which was singularly present at Site 9, two Chironomid larvae, two anemones (*N.vectensis*), and *C. glaucum* singularly present at Site 45. Group B had the highest mean Silt Content, OM Content and deepest SDepth with 83.52%, 9.46% and 0.93m respectively. Mean Salinity and Temperature was 26.53‰ and 2.81°C respectively. Spatially these sites were clustered in two main areas; the central-south region and the northeast region, with one inside the Model Boating Area. The low abundance and richness of this group, particularly low abundances of predominantly infaunal species, is indicative of hostile conditions, supported by the unfavourable sediment factors which would suggest hypoxic-sulphidic sediments. Motile amphipods dominate these sites likely due to their capacity to swim to more favourable conditions when necessary.

### 3.5.3 Group C

Group C consisted of 26 samples (Sites 1, 3, 6, 7, 11, 13, 14, 16, 19, 20, 21, 23, 25, 27, 29, 31, 33, 34, 38, 42, 43, 44, 47, 48, 49, 50) and was characterised by higher than average abundances (mean = 79, n = 11 to 400) and higher than average species richness (mean = 5.7, S = 2 to 11, total S = 14). Amphipods *M. insidiosum, M. gryllotalpa, M. palmata,* and *G. locusta* dominated taxa again by 60% but Group C assemblages were characterised by a higher proportion of annelid worms (17%), including *H. diversicolor, Polydora* sp. and *Tubificoides* sp., which dominated five sites and were present in 20 of 26 sites. A higher proportion of infaunal and epifaunal molluscs and *N. vectensis* were also present. The anomalous Site 34 was included in this grouping, with an overall abundance of 400 and a total of 11 species. Other invertebrates within this grouping included one Australian tubeworm *Ficopomatus enigmaticus* (Fauvel 1923) individual singularly present at Site 38, Chironomid larvae, Hydrobiid snails *Peringia (Hydrobia) ulvae* (Pennant 1777) and *E. ventrosa,* bivalves *C. glaucum* and *M. arenaria*, and the anemone *N. vectensis.* The mean SDepth for Group C was 0.48m and had a moderate mean OM Content of 6.63%, and a moderate mean Silt Content of 67.61%. Mean Salinity and Temperature was 26.28‰ and 2.84°C respectively. The moderate sediment factors at these sites have facilitated the colonisation of greater numbers of infaunal species as the sediment is likely to be better oxygenated and little to no interstitial hydrogen sulphide. The predatory non-native anemone *N. vectensis* is present at these sites where there are higher abundances of prey species.

Site 34 is situated between two hard structures; the Concrete Walkway to the southeast and Reedbed 2 to the north. Hard structures within a soft-bottomed habitat represent areas of productivity due to their capacity to be colonised by species that require a hard substrate, and the habitat complexity this creates for other species. The Concrete Walkway is encrusted with Australian tubeworm reefs which anecdotally are sites of high densities of motile species feeding and seeking shelter among the calcareous tubes, including isopods, common ditch shrimp, gobies and unidentified fish fry. Bryozoans and green algae have also been seen growing on the reefs. However, this anomalous site supports the need for taking more than one sample at each site to compare same-site samples to see if there is much biotic variation on a microspatial scale, particularly as it was adjacent to Site 41, with only nine individuals of one species present.

### 3.5.4 Group D

Group D consisted of 12 samples (Sites 2, 5, 8, 12, 18, 22, 26, 30, 32, 36, 37, 39) and was characterised by lower than average abundances ($n = \leq 22$, mean = 9.8) and low species richness ($S = \leq 4$, mean = 3, total S = 6). The overall dominant taxa were Chironomid larvae (55%) and was the dominant taxa in 10 of 12 sites, including the fifth monospecific site. Group D also included the ubiquitous amphipods *M. insidiosum* and *M. gryllotalpa*, which were the second most abundant phylum (17%), and low abundances of annelid worms *H. diversicolor* and *Polydora* sp.. The anemone *N. vectensis* was also present in low abundances and a single Hydrobiid snail *P. ulvae* was present at Site 12. Group D had a mean high Silt and OM Content and relatively deep SDepth with 78.83%, 7.62% and 0.71m respectively. Salinity and Temperature were 26.33‰ and 2.72°C respectively. There was no clear spatial pattern for this grouping, but Group D assemblages did not stretch to the northeast corner or eastern edge of the lagoon.

### 3.5.5 Spatial Distribution of SIMPROF Groupings

When mapped, the distribution of the SIMPROF groupings do not display any clear spatial distribution patterns. As SDepth was identified by BIO-ENV to be weakly associated with macrozoobenthic assemblages, an interpolated SDepth layer was added to the map (see Figure 13).

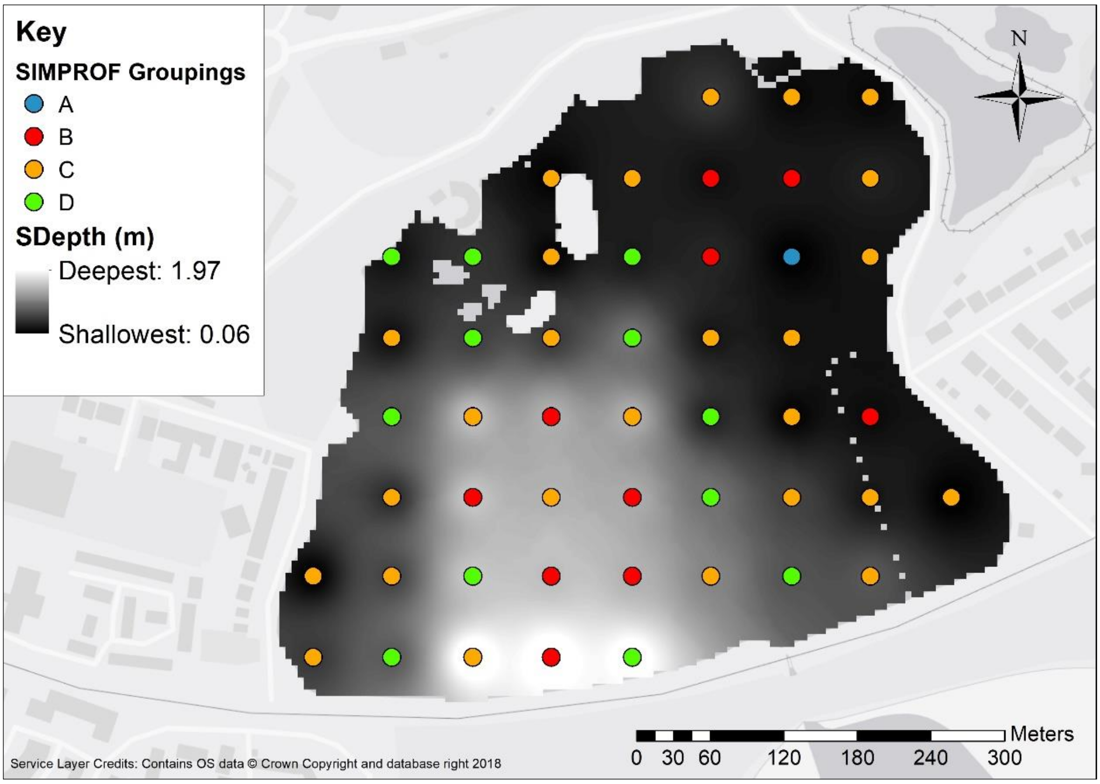

Figure 13 The spatial distribution of statistically significant SIMPROF groupings in Poole Park lagoon with SDepth.

Group B was predominantly clustered within central and southern deeper SDepth areas, but four sites were in the shallower northeast. Group C was predominantly clustered within shallow SDepth areas throughout the lagoon, but four sites were in the deeper central to southern area. Group D can be found on the fringes of the central deeper SDepth area, with some sites in both the deepest and shallowest areas, with no discernible pattern. Group A, the single site, is in the shallow northeast.

## 3.6 Lagoon Biotopes

Considering the English and Welsh Biotopes (Bamber 1997) Group C and D was best suited to ENLag.IMS.Ann, infralittoral muddy sand with Chironomids, Hydrobiids, *H. diversicolor, C. glaucum,* and amphipods. Group B was best suited to ENLag.Veg with its high OM Content and strong dominance of amphipod crustaceans.

## 3.7 Invertebrate Distribution

Every site sampled was populated with invertebrates. Of the 49 sample sites, 10% were monospecific, with four sites dominated by the lagoon amphipod *M. insidiosum* and one site dominated by Chironomid larvae. The abundance at these monospecific sites did not exceed nine individuals. At 59% of sites (29 of 49) the dominant taxa were amphipod species, either *M. insidiosum*, *M. gryllotalpa* or on one occasion *G. locusta* with a dominance ranging from 33% to 100% of the assemblage*.* The second most abundant taxa were Chironomid larvae occurring at 22% of sites (11 of 49) with a dominance ranging from 33% to 100% of the assemblage.

The least numerically abundant species (<20) included the Hydrobiid snails *P. ulvae* and *E. ventrosa*, the bivalves *M. arenaria* and *C. glaucum*, the polychaetes *F. enigmaticus* and *P. mucosa*, and the amphipods *M. palmata* and *G. locusta.* Apart from *P. ulvae* and the polychaetes, these relatively rare species can be found at Site 34 or its neighbouring sites, indicating a hot spot for species richness. The results for the anemone *N. vectensis* can be found in Chapter Two.

### 3.7.1 Annelids

The *F. enigmaticus* individual was found still within its calcareous tube on the southwest corner of Island One, presumably having dislodged from a reef colonising the Island fencing (see Appendix Figure A1). The *P. mucosa* individual, a carnivorous scavenger normally occurring in estuarine and soft-bottomed coastal areas, was found in the southwest centre in the area of high Silt and OM Content (see Appendix Figure A1).

The most abundant annelid *H. diversicolor* was largely absent from the CSW and CNE and most abundant along the west side of the Concrete Walkway, around the northwest edge and Islands and peaked at Site 38 (see Appendix Figure A1).

The annelid *Polydora* sp. was completely absent in the CSW right up to the southwest bank and in the CNE (see Appendix Figure A2). It occurred in low abundances around the Island One, northeast edge, Sites 33 and 34, by the sluice gate and west of the Concrete Walkway. The oligochaete *Tubificoides* sp. occurred at just four sites and was otherwise completely absent throughout the lagoon (see Appendix Figure A2). It was found at Site 34, in the southeast corner and northwest of Island One.

### 3.7.2 Crustaceans

The amphipods *M. gryllotalpa* and *M. insidiosum* were the most abundant species and accounted for several monospecific sites. However, though singularly present, *M. insidiosum* was by no means abundant with a maximum of nine individuals at the most abundant monospecific site, suggesting hostile conditions were not limited to the sediment. *M. gryllotalpa* was present at most sites and scattered throughout the lagoon with no discernible site groupings where they were consistently absent (see Appendix Figure A2). *M. insidiosum* was present at all but eight sites, peaking at Site 34, and was also notably abundant in the northeast corner (see Appendix Figure A3). Three sites *M. insidiosum* was absent from were clustered around the northwest-west edge by The Kitchen Café.

*M. palmata* was distributed fairly randomly, occurring in sites in the centre and perimeter and in the northwest, east and west (see Appendix Figure A3). *G. locusta* was found at Site 34 and an adjacent site, and two perimeter sites; one adjacent to the sluice gate and another in the southwest corner (see Appendix Figure A3).

### 3.7.3 Molluscs

*P. ulvae* was found at two sites, one on the south perimeter and one near the centre, both high Silt and OM Content sites (see Appendix Figure A4). *E. ventrosa* was not only more abundant than *P. ulvae,* its distribution was more widespread; predominantly in the northeast around Reedbed 2 and also found at the southern site with *P. ulvae* (see Appendix Figure A4). Empty shells of both species were more abundant in samples than live specimens.

*M. arenaria* was found at Site 34 and at two perimeter sites adjacent to the Sluice Gate (see Appendix Figure A4). All *M. arenaria* individuals were <1cm in size, suggesting one of two things; that conditions are unfavourable for rapid growth or that they were recently recruited juveniles (<1 year).

*C. glaucum* was found at one site in the southwest corner but the majority were found in three sites in the northern area (see Appendix Figure A5). The high energy of the north is likely favourable for these filter feeding bivalves. Specimens were a mixture of juveniles and young adults (<2cm).

### 3.7.4 Chironomid Larvae

Chironomid larvae were largely absent in the CSW and central northeast, though the site at which they were the sole species occurred in the southwest (see Appendix Figure A5). There were no areas where Chironomid larvae were particularly abundant though they peaked in numbers in the northeast corner.

Although many species demonstrated areas of particularly high abundance or areas where they were notably absent, their spatial distribution was patchy overall. Many species had singular sites of absence or high abundance scattered randomly throughout. High abundance sites were often adjacent to sites where the same species was absent and vice versa, demonstrating the characteristically patchy nature of lagoon macrozoobenthic species and assemblages.

## 3.8 Size Classes

### 3.8.1 Lagoon amphipod *M. insidiosum*

Some specimens were unavailable for measuring after being sent for identification and some were too fragile and broke when being handled.

Nonetheless 688 individuals were measured, comprising 83% of the total abundance for *M. insidiosum.* Approximately 70% were between 1.1-2.0mm and frequency decreases either side of this (see Figure 14). This pattern is reflected at sites with higher abundances. The largest individuals (3.6-4.0mm) were found at Site 23 with all other size classes present (see Figure 15). Six sites that comprised of just one size class were always between 1.6-2.0mm. Size classes ranged the most along the Concrete Walkway and along the west bank. There was no discernible pattern with environmental variables.

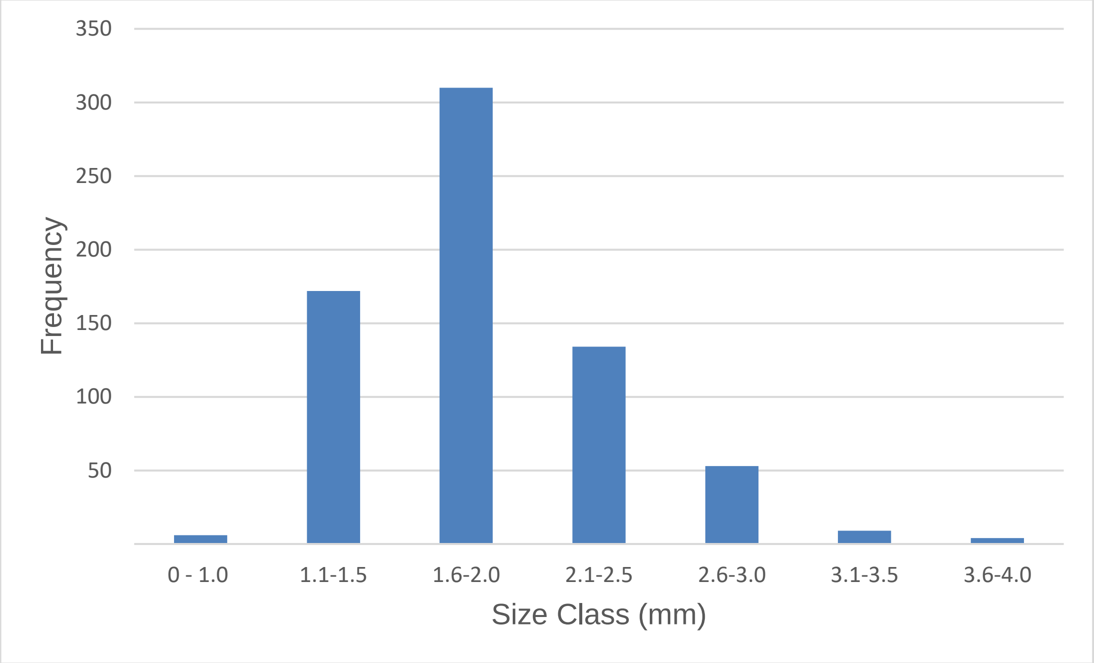

Figure 14 The size class frequency for the amphipod *M. insidiosum*.

Figure 15 The size class distribution of the lagoon amphipod *M. insidiosum.*

### 3.8.2 Ragworm *H. diversicolor*

Seventy percent of *H. diversicolor* specimens were measured, with some being too fragile to handle. The dominant size class was 0-1.0cm and there was a negative correlation between frequency and size class (see Figure 16). No specimen exceeded 7cm though *H. diversicolor* can grow to twice this size. Size class 1.1-2.0cm occurs in all but two sites (see Figure 17). Seven sites were dominated by a single size class which was either 1.1-2cm or 3.1-4cm. Sites in the north exhibited the greatest variety of size classes and greater abundances, particularly around Island One and both Reedbeds. Conversely the largest specimens (6.1-7.0cm) were found in the southeast by the Concrete Walkway.

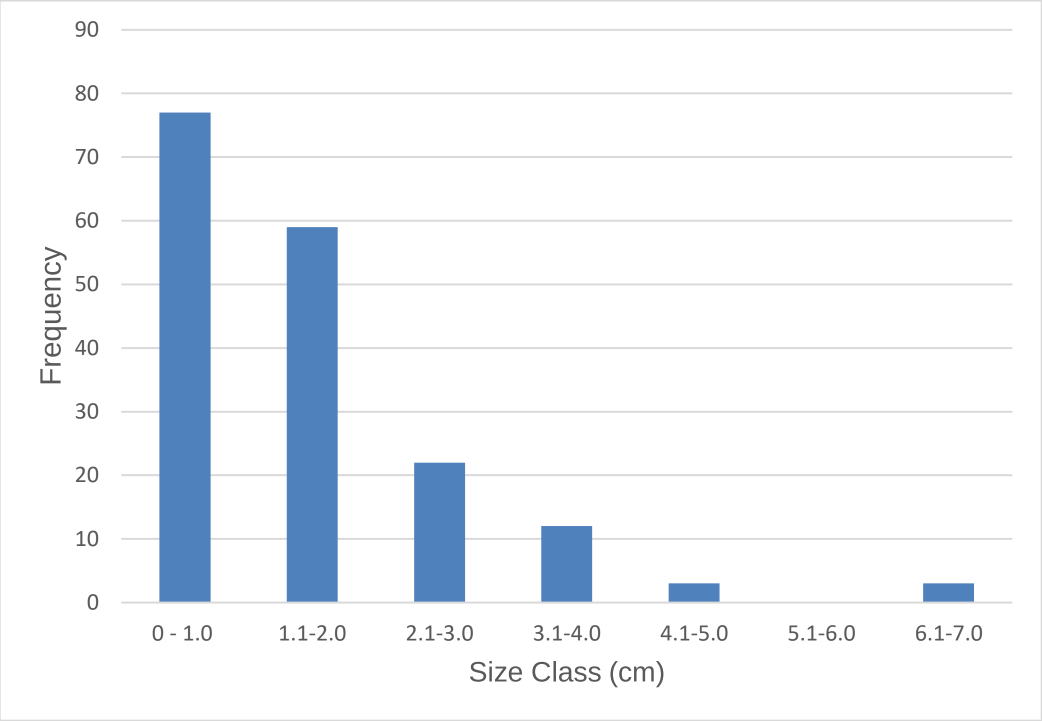

Figure 16 The size class frequency for the ragworm *H. diversicolor.*

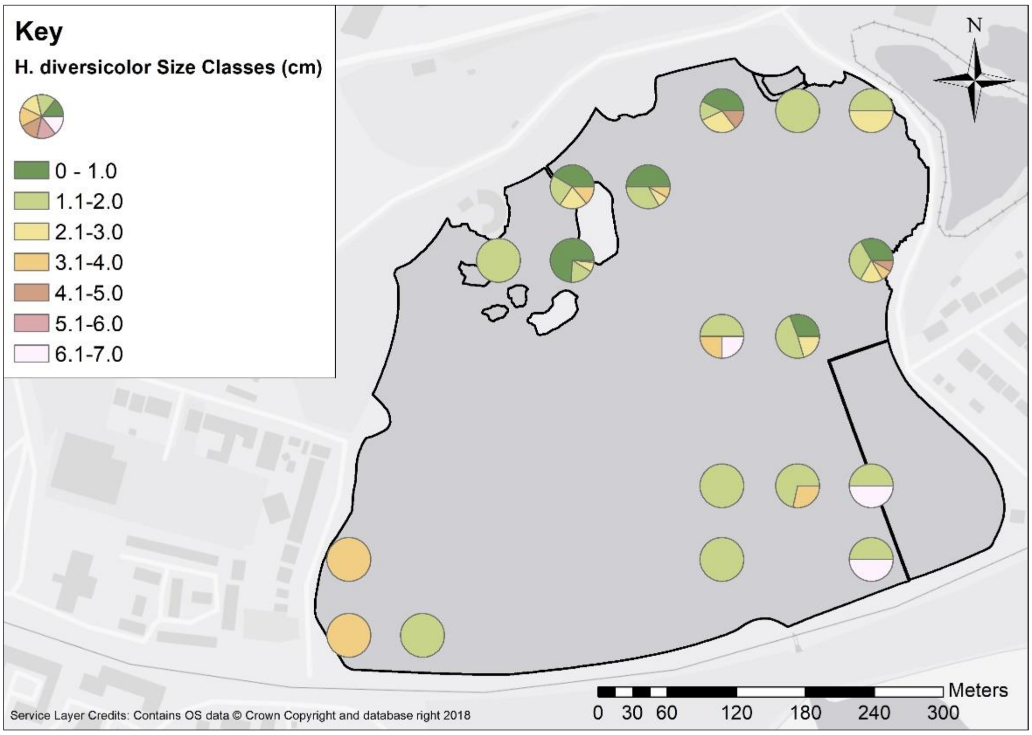

Figure 17 The size class distribution for the ragworm *H. diversicolor.*

## 3.9 Pelagic Fauna

Fauna caught in the fish traps were dominated by *P. varians* and *P. microps* (see Table 6). This differed from the most recent previous survey which was dominated by three-spined sticklebacks *Gasterosteus aculeatus* (Linnaeus 1758) and recorded a greater species richness with six fish species (Harrison et al. 2016). However, a comparison cannot be sensibly drawn due to differing survey methodology, time of year and location within the lagoon. Additionally, it should be noted that the lagoon underwent an accidental drain on a neap tide on May 8th, 2018 and took an additional 4 days to return to average volume. Anecdotal accounts of the lagoon described it as a large muddy puddle with some of the marginal sediment emersed (Rockley Watersports, personal communication). During the four days it was below average water height, the weather was consistently sunny, and the air temperature was as high as 21°C. Such conditions may have led to the mass mortality of some species, including fish who would have been forced into a smaller area and higher densities in unusually hot and potentially hypersaline conditions due to evaporation.

Table 6 Pelagic species and abundances of Poole Park lagoon.

| Scientific Name | Common Name | June | July | TOTAL |
|---|---|---|---|---|
| PHYLUM CRUSTACEA | | | | |
| *Carcinus maenas* | European green crab | 11 | 8 | 19 |
| *Palaemon varians* | Common ditch shrimp | 261 | 293 | 554 |
| PHYLUM CHORDATA | | | | |
| *Gasterosteus aculeatus* | Three-spined stickleback | 0 | 6 | 6 |
| *Pomatoschistus microps* | Common goby | 229 | 228 | 457 |
| **TOTAL ABUNDANCE:** | | 501 | 535 | **1,036** |

There was no significant correlation with invertebrate density with *C. maenas*, *P. varians* or *P. microps* abundance overall (see Table 7). This is unsurprising, given the time that has elapsed between their respective survey dates. *P. varians* abundance was highest at Site 1 for both surveys and lowest at Site 18 for June and absent at Sites 11 and 20 for July. *P. microps* were most abundant at Sites 8 and 13 for June and Site 15 for July. It was least abundant at Site 4 for June and absent at Sites 3, 8, and 9 for July. *C. maenas* were found at seven sites in June but only four in July. Abundances for the three species was not significantly different between surveys. *G. aculeatus* was found only in July in low abundances.

Table 7 Regression analyses for overall invertebrate density and pelagic abundance of Poole Park lagoon. Site 16 was a consistent outlier in all regressions.

| **Invertebrate Density and Pelagic Abundance of Poole Park Lagoon** | | | | | | |
|---|---|---|---|---|---|---|
| **Variable** | **Regression** | **$R^2$** | **DF** | **F** | **p** | **Unstandardized Co-efficient B** |
| *C. maenas* Total | Linear | 1.5 | 1, 18 | 0.276 | 0.605 | -892.732 |
| *G. aculeatus* Total | Linear | 2.5 | 1, 18 | 0.464 | 0.505 | 2256.277 |
| *P. microps* Total | Linear | 0.5 | 1, 18 | 0.082 | 0.778 | 50.042 |
| *P. varians* Total | Linear | 3.2 | 1, 18 | 0.601 | 0.448 | -111.297 |

In June, *P. microps* were found in almost equal abundances in the high OM area and low OM area (see Figure 18). *P. varians* abundance was considerably higher in the high OM area. *C. maenas* were in higher abundances in the low OM area. In July *P. microps* abundance was more than four times greater in the low OM area than the high OM area. Conversely *P. varians* abundance was almost twice as great in the high OM area than the low OM area. *G. aculeatus* were found only at three sites in the northeast of the lagoon in the low OM area with none found in June.

The traps were notably weedier with filamentous green algae in July when recovered, particularly in the high OM area. In June, no fouling was observed apart from Site 18 in Reedbed 2 by the large freshwater Pipe L8. July's survey followed a period of drought conditions in the UK, with record-breaking hot weather and just over 20% of the average rainfall for June-July in southwest England (Shukman 2018). Furthermore, a larger proportion of *P. microps* caught in July were larger, approximately 3-4cm, some with dorsoventral swelling in the pelvic area. This swelling could indicate fish that were in breeding condition. In July, several of the traps contained small proportions of dead *P. microps* with visible lacerations, possibly due to attempted predation by *C. maenas* outside the traps. Many of *P. varians* caught in July were much smaller (<2cm) than those caught in June. The decrease in *P. varians* and increase in *P. microps* and vice versa in both survey areas in July suggests that predation of *P. varians* by *P. microps* may be a factor in determining their distribution; particularly noting the differences in size classes.

### 3.9.1 Temporal Differences in Pelagic Abundance

There was no significant difference between *P. microps* abundance (F = 3.467, df = 38, p = 0.987), between *P. varians* abundance (F = 0.782, df = 38, p = 0.648), between *C. maenas* abundance (F = 0.160, df = 38, p = 0.607), or between *G. aculeatus* abundance (F = 14.124, df = 19, p = 0.110) in June and July.

### 3.9.2 Spatial Differences in Pelagic Abundance

There was a significant difference between *P. microps* abundance (F = 0.730, df = 38, p = 0.015) and between *P. varians* abundance (F = 3.130, df = 38, p = 0.011) in the area of high OM Content (Sites 1-10) and low OM Content (Sites 11-20). There was no significant difference between *C. maenas* abundance (F = 11.417, df = 26.939, p = 0.057) and between *G. aculeatus* abundance (F = 14.124, df = 19, p = 0.110) in the area of high OM Content and low OM Content.

Figure 16 The spatial distribution of pelagic fauna in a) June, b) July and c) overall.

# 4. Discussion

This study aimed to determine the primary abiotic and biotic factors responsible for the spatial distribution of lagoon macrozoobenthic communities and species. Following a spatially comprehensive survey of biodiversity and abiotic variables, it was found that species richness and abundance were not correlated with proximity to the Sluice Gate ($p = 0.86$ and $p = 0.68$ respectively) as predicted.

Species richness was significantly positively correlated with MGS and Distance from SW Shore, and significantly negatively correlated with OM Content, SDepth and Silt Content (see Table 3). Invertebrate density was significantly positively correlated with MGS and Distance from SW Shore, and significantly negatively correlated with OM Content, Salinity and Silt Content (see Table 2). Considering all abiotic factors, a BIOENV procedure yielded no statistically significant combination of variables that explained the faunal assemblages but identified SDepth as the abiotic factor that fit the best (Rho = 0.13, significance value = 0.46).

There were no significant correlations between pelagic species and invertebrate density. However, this should be interpreted with caution as the sampling periods were seven months apart and during different seasons (late Autumn and early summer respectively) and faunal communities are likely to exhibit temporal patterns not reflected in this study. On a broad spatial scale, it is evident that invertebrate density and richness are influenced primarily by physico-chemical factors within the lagoon ecosystem.

## 4.1 Sediment Factors in Poole Park Lagoon

Sediment factors, such as MGS, Silt Content, OM Content, and SDepth, are key variables in determining the spatial distribution of Poole Park lagoon’s biodiversity. This is also seen in Seymers lagoon and Poole Harbour and is a common primary or secondary factor in soft-bottomed ecosystems (Bone 2017; Herbert et al. 2010; Labrune et al. 2008; Jayaraj et al. 2008, van Hoey et al. 2004; Weston 1988). Grain size and organic matter content is indicative of other unmeasured associated variables such as interstitial space available, hydrogen sulphide content, and oxygen availability, which will also influence the distribution of infaunal invertebrates. Without measuring these variables, it is impossible to be certain that the sediment variables are truly responsible for determining spatial distribution of invertebrates and not one or a combination of several unknown but co-linear variables. However, the sediment variables measured in this study are likely to be representative of other associated variables, like hydrogen sulphide content, and these compounding factors will be jointly influential in what species can colonise the sediments. Organic enrichment, represented in this study by OM Content (LOI%), can lead to low dissolved oxygen and the by-products of decomposition such as hydrogen sulphide and ammonia (Gray et al. 2002). Low concentrations of $H_2S$ is lethally toxic to invertebrates and, combined with hypoxic conditions, can lead to mass mortality from physiological stress (Vismann 1991). Siltier sediments are often associated with greater organic matter content due to similar settling velocity and greater surface area of finer particles adsorbing organic carbon (Burone et al. 1993; Tyson 1995; Hedges et al. 1993).

### 4.1.1 Deposition in a Tidally Restricted Lagoon

The spatial distribution of significantly associated sediment variables, as interpolated map layers on ArcMap, show a consistent central to south west area that is associated with the maximum or minimum values for sediment variables, also visually demonstrating their co-linearity (see Figure 7). The CSW area is associated with high Silt Content and greater MGS, high OM Content and greater SDepth, and a hydrogen sulphide odour when sediment samples were taken. As Poole Park lagoon is tidally restricted, it is highly likely that wind-induced mixing is a crucial factor determining the distribution and sorting of sediment grain sizes and the areas that accumulate organic matter, such as floating algal mats or fallen leaves (Barnes 1994). Given prevailing wind direction is south-westerly, linear regressions were performed to find associations between the southwestern corner of the lagoon and measured abiotic variables (Weather Online 2018). OM Content, Silt Content, and SDepth were significantly negatively correlated with Distance from the SW

Shore, indicating that fetch built up over the lagoon from the southwest direction leads to greater velocity of wave action in the northeast corner. The large size and wide shape of Poole Park lagoon satisfies the “low aspect ratio” criteria that favours increased wind fetch and turbulence (Bamber et al. 2001; Covey 1999).

Waves can be seen in the northeast corner on windy days, with sea foam blowing over the adjacent road on particularly blustery days. The decreasing water depth from southwest to northeast would further promote the development of waves and the interaction between the benthic habitat and wind-induced mixing. Lower velocity in the CSW has led to greater deposition of siltier sediments and the accumulation of organic matter and thus deeper sediments overall. The Train Track that borders the south edge of the lagoon is acts as a wind break, preventing south westerly winds from dispersing accumulated floating organic matter in the southwest corner. Deposition of fine sediments and accumulation of organic matter in central regions is also demonstrated in the Mellah lagoon, Algeria, and in several Scottish lagoons (Magni et al. 2015; Covey 1999). Therefore, it is likely that the interaction between lagoon physiography and bathymetry has facilitated the deposition of silt and organic matter in the CSW and led to greater disturbance in the northeast. Thus, these areas are relatively species poor and low in abundance.

## 4.2 Aquatic Factors in Poole Park Lagoon

Aquatic variables measured in this study, apart from salinity and density, were not found to be significantly associated with overall invertebrate biodiversity. Temperature ranged by 2.6°C across the lagoon and appeared relatively homogenous on a horizontal spatial scale, regardless of proximity to the Sluice Gate or freshwater Pipes. The northeast was overall warmer and significantly positively correlated with distance from the southwest corner, likely due to its shallower depth and increased time exposed to sunlight at that time of year.

Salinity is negatively correlated with density and reduced salinity may negatively impact the numeric abundance of invertebrates that are able to tolerate hyposaline conditions. The salinity of the lagoon was relatively homogenous with a range of 2.5‰, apart from three perimeter sites which exhibited lower salinities around 24‰. One of these sites is adjacent to freshwater Pipe L2. However, salinity was measured at the water's surface, thus the salinity in the benthic environment and interstices is unknown.

### 4.2.1 Stratification in a Tidally Restricted Lagoon

Stratification in tidally restricted lagoons is common, and so the degree of vertical mixing for salinity is not known for Poole Park (Barnes 1980). Wind-induced mixing and wave action is likely to play an important role determining various abiotic factors in the absence of a current in tidally-restricted lagoons (Bamber et al. 2001). Thus, shallower and high energy areas such as the northeast of the lagoon are likely to be well-mixed. Salinity in Poole Park rises sharply following a flush event, from 6‰ to 27‰ in one case, with the interim salinity dependent on freshwater input (rainfall) (Harrison et al. 2016). The salinity gradient within Poole Park lagoon was relatively stenohaline and this homogeneity is likely due to wind-induced mixing. Thus, fluctuations in salinity may be more relevant to Poole Park on a temporal scale than on a spatial scale. Well-mixed waters also tend to be species poor as niches associated with hypo or hypersalinity are not available for colonisation by lagoon specialists, which must instead compete with estuarine generalists. Lagoons that maintain a salinity regime of 18-24‰ are more likely to develop specialist fauna dominated by *H. diversicolor* and Corophiid amphipods (Ouisse et al. 2011; Bamber et al. 2001; Robertson 1993). As Poole Park lagoon is large and wide, it's capacity to support a diverse invertebrate assemblage is limited (Bamber et al. 2001; Covey 1999). This limitation is compounded by its restricted and infrequent flushing regime and freshwater input of surface water discharge, making it vulnerable to eutrophication and hyposalinity events.

It is likely that salinity is a secondary factor in determining invertebrate distribution. While hypoxic-sulfidic sediments determine macrozoobenthic organisms on a spatial scale and will remain relatively stable, salinity is likely to determine distribution on a temporal scale with weather-dependent freshwater input and flush events resulting in a stochastic salinity regime (Dauvin et al. 2017). For example, proximity to Sluice Gate and freshwater Pipes may influence distribution on a secondary spatial scale during flush and rain events. This stochastic dynamism is typical of coastal lagoons (Barnes 1999). The absence of evidence in this study that proximity to the sluice gate is not correlated with abiotic gradients is not evidence that they don’t occur (Altman and Bland 1995). Indeed, horizontal salinity gradients may occur immediately post flush, and sediments may experience sorting during the velocity incurred by ebb and flood currents during a spring tide drain and flush event. The sedimentary factors often show a narrow gradient change perpendicular to the sluice which may be indicative of sorting during flush events. However, flushes are infrequent compared to the consistent wave energy derived from prevailing SW winds. Thus, sluice gate gradients may exist only in the short term.

Though salinity was not recorded to be lower than average at these sites during this study, the faunal assemblages at Sites 37, 38, 42 and 43 are indicative of an environment that undergoes hyposaline events. The dominance of tolerant *H. diversicolor* and opportunistic Chironomid larvae are species that are typically found near freshwater inputs and can tolerate salinities as low as 6‰ (Ouisse et al. 2011; Bamber 2001; Fritzsche and von Oertzen 1995). Sites 37, 38 and 43 are adjacent to Pipe L8 and Site 42 is adjacent to Pipe L2 (see Figure 2), which both discharge surface water run-off from the local urban area (Harrison et al. 2016).

## 4.3 Relationships Between Abiotic Factors and Overall Assemblage

There were no statistically significant correlations between any combination of abiotic factors and macrozoobenthic assemblages, following a BIO-ENV procedure. Lagoons are hostile habitats with stochastic abiotic factors and a longitudinal study would likely reveal patterns and trends that this study has not. As this was a snapshot study, conditions in the days and weeks prior to the data collection would have been instrumental in determining the assemblages recorded. Additionally, the spatial scale at which assemblages vary may occur at a finer scale than was surveyed. Taking more than one core sample per sample site for biotic data would improve the reliability of the data by providing the opportunity to analyse assemblages within the same site. What many lagoon studies fail to consider is the interactions between macrozoobenthic species and how their behaviours, such as predation, reproduction and interference, can lead to differences in assemblage composition. While macrozoobenthic assemblages are good indicators of local habitat quality, the relative proportions in which species occur may be down to interspecific interaction.

### 4.3.1 Relationship Between Salinity and Benthic Primary Productivity

One of the primary issues within lagoons is eutrophication and related dystrophic events. Benthic primary productivity is particularly high in shallow areas with low turbidity. During spring and summer, macroalgae can be seen growing on the bottom of the lagoon in most places, including along northeast edge (personal observation). Initially, this rapid growth increases benthic invertebrate biodiversity by increasing habitat complexity and food availability. However, as algal mat coverage and biomass peaks in late summer, oxygen availability in the benthic environment decreases (Ouisse et al. 2011). *Cladophora* spp. and other macroalgal species are nitrogen limited and thus rely on biologically available nitrogen to grow (Peckol et al. 1994). Salinity is known to affect biogeochemical pathways in nitrogen removal processes and transformations, leading to less nitrogen removed from coupled processes and greater nitrogen removed from direct processes in polyhaline conditions than oligohaline conditions (Hines et al. 2015). Short term fluctuations and variations in intensity of salinity fluctuations is also known to affect the community structure of denitrifying microbes but not the functional capacity of the community (Zaghmouri et al. 2018). Therefore, the microbial community is relatively resistant to physiological stress that changes in salinity may incur, or alternatively are adept at colonizing sediments with better adapted species or clades when salinity does change.

However, increases in porewater salinity does lead to desorption of ammonium ($NH_4^+$) from sediments into the overlying water column, making it available to macroalgal species (Giblin et al. 2010; Weston et al. 2010; Rysgaard et al. 1999). Additionally, where hydrogen sulphide and organic matter content in sediments is elevated, DNRA (dissimilatory nitrate reduction to ammonium) rates are higher and denitrification rates are lower, leading to greater amounts of dissolved ammonium (Song et al. 2014; An and Gardner 2002). Bioturbation effects from infaunal invertebrates, including *H. diversicolor* and *C. glaucum*, causes the release of ammonium from sediments (Murray et al. 2017; Solan et al. 2008; Ieno et al. 2006; Mermillod-Blondin et al. 2005). Where time between flushing events is relatively long, more nitrogen is removed via microbial processes (Joye and Anderson 2008). Subsequently, tidally restricted lagoons can accumulate biologically available ammonium promoting macroalgal blooms observed in Poole Park lagoon, leading to eutrophic conditions and degraded biodiversity.

## 4.4 SIMPROF Groupings and Ecotones

The SIMPROF groupings represent a spectrum of habitat suitability according to their sediment values and corresponding faunal assemblages. At one end is Group A with the lowest sediment factor values and lowest abundance and species richness, followed by Group C which had moderate sediment factor values and much higher abundances and species richness. Group D follows and as the sediment factor values increase, abundance and species richness begin to decline and while still relatively diverse with species representation from all phyla, the dominant species shifts to Chironomid larvae, indicating the decline in habitat quality. Finally, Group B represents the other end of the spectrum, with the maximum sediment factor values and lowest abundance. Although Group B has the same number of species as D, annelids are almost absent and there is a dominance of motile species. Interpreting the SIMPROF results as a spectrum as opposed to discrete groupings provides greater insight into habitat preferences for individual species and assemblages. However, when these groupings are plotted against interpolated sedimentary data layers, sites from all groups do not fit neatly into predicted areas based on the average sediment factor values for those groups. This emphasises the need for long term data to help explain spatial and temporal patterns and that although the SIMPROF groupings serve as a guide, lagoon assemblages and abiotic factors continue to be relatively stochastic and unstable.

Pearson and Rosenberg (1978) wrote extensively about the organic enrichment of soft-bottomed macrofaunal communities. Where organic input is highest, sediments are almost devoid of fauna, with low species richness and abundance (Rybarcyzk et al. 1996; Peterson et al. 1994). Abundance then rises rapidly as organic content becomes more moderate. Here, sediments are not hostile so can be colonised by opportunists drawn to the organic food source. Moderate nutrient input can increase primary productivity and thus boost macrozoobenthic abundance; macrozoobenthic biomass doubled in a 15-year period in the Dutch Wadden Sea due to organic enrichment (Gray 1992; Beukema and Cadee 1986). The SIMPROF groupings in this study could represent successional communities in ecological gradients, and frequently overlap. Ecotones and coenoclines are common transitional waters, particularly tidally restricted lagoons, where greater ranges in environmental gradients, such as salinity and temperature, limit species richness (Reizopolou et al. 2013; Basset et al. 2013; Bazairi et al. 2003).

The Intermediate Disturbance Hypothesis postulates that moderate magnitudes of stress on an ecosystem promotes biodiversity (Connell 1978). In the case of coastal ecosystems, and particularly lagoons, spatially heterogeneous and stochastic environmental variables facilitate niches which can be colonised by different species (Ouisse et al. 2011; Bamber 2001). Salinity and temperature in this study are spatially homogenous but inevitably will vary spatially on a temporal scale, which will affect macrozoobenthic assemblages. The magnitude of variation is beyond the scope of speculation in this study but has been demonstrated by Harrison et al. (2016). However, the spatial distribution and dominance of species can provide clues as to the historical long-term salinity and temperature trends.

## 4.5 Spatial Patterns of Invertebrate Species

### 4.5.1 Annelids

Annelid worms were overall absent from the CSW and CNE and were dominant at sites adjacent to freshwater input, predominantly due to a high proportion of euryhaline *H. diversicolor. Polydora* sp. was located at 14 sites in the east and north, with none located in the CSW and southwest. Their absence in the CSW suggests that the redox layer is largely non-existent and thus unsuitable for infaunal colonization. *Tubificoides* sp. was present at just four sites, again avoiding the CSW and CNE. Densities for this oligochaete worm were very low. This is unusual as this is an opportunistic worm, characteristic of polluted, organically enriched and hypoxic sediments (Giere 2006; Giere et al. 1999; Bagheri and McLusky 1982). However, although it is a pioneer species of such impoverished habitats, it is relatively slow-growing and long-lived which may make it vulnerable to dystrophic events, leaving relict populations to recolonize areas where it is locally extinct (Giere 2006). This can take over a year to match previous population levels (Bolam and Whomersley 2003).

The single individual of *F. enigmaticus* was found within its calcareous tube likely having broken off from the adjacent hard structure. The single individual of *P. mucosa*, a carrion feeder, will travel along the sediment surface to scavenge on carrion of dead crabs, molluscs and worms (Lee et al. 2004). Found at Site 9 in the CSW, there is likely ample prey items that have perished in the hostile conditions.

What is interesting is the conspicuous absence of opportunistic polychaete *Capitella capitata* (Fabricius 1780). A typical coloniser of organically enriched and disturbed sediments and present in Brownsea and Seymers lagoon, it was surprising not to find it within Poole Park lagoon where these habitat characteristics have been demonstrated (Gray and Elliott 2010; Tsutsumi 1990; James and Gibson 1980). In Harrison et al.’s (2016) survey, *C. capitata* was also absent from the lagoon. It is present in Parkstone Bay where the sluice gate opens into Poole Harbour but in very low densities. In Herbert et al.'s (2010) survey of the Poole Harbour invertebrates, *C. capitata* was found at 18 sites. At each site five samples were taken and at sites where *C. capitata* was found, it was often not present in all five samples. Thus, the sampling effort in this study may not have been sufficient to detect *C. capitata* in low densities. Additionally, despite year-round breeding and spawning, colonization capacity may be low due to the infrequent flushing events facilitating the movement of planktonic larvae from the harbour and the lack of wader species which would facilitate avian dispersal (Herbert et al. 2018; Bolam and Fernandes 2002; Shull 1997; Warren 1976). Competitive exclusion with other annelids may also play a role; sudden dystrophic events will lead to colonization of *C. capitata* but chronically stressed environments, such as lagoons, are dominated by stress-adapted species such as *H. diversicolor* (Warwick 1986; Pearson and Rosenberg 1978). Low abundances of *C. capitata* were observed in Poole Harbour despite expectations that they would be present in greater numbers due to algal mat enrichment. However, their peak abundance was thought to occur in the summer, when algal growth in the harbour was greatest, and were subsequently outcompeted (Thornton 2016). Thus, the relative abundance of *C. capitata* may be an indication of the successional stage of lagoonal assemblages following a disturbance event.

### 4.5.2 Crustaceans

*M. insidiosum* and *M. gryllotalpa* were the most abundant crustaceans and were absent at just eight sites and 20 sites respectively, though at the majority of sites occurred in very low abundances. Both live in tubes constructed on vegetation, in the sediment or on a hard substrate such as shells or man-made structures and thus can avoid hypoxic-sulfidic sediments that would limit other infaunal species, such as annelids (Barnes 1994). *M. insidiosum* can be frequently found outside of its tube, avoiding disturbed or hostile conditions in the sediment (Fricke et al. 2015). Furthermore, its tube construction can irrigate the sediment surface and thus improve oxygenation in the immediately local area, particularly as *M. insidiosum* generates currents into its tube to feed using its pleopods, drawing in oxygenated water and flushing out hydrogen sulphide (Fricke et al. 2015; Dixon and Moore 1997). Peracarid crustaceans are known to break diffusive boundary layers in hypoxic sediments, facilitating reoxygenation with bioturbation (Lindstrom and Sandberg-Kilpi 2008).

*M. palmata* is distributed fairly randomly and in very low abundances. Anibal et al. (2007) found that mudflat topography determined the presence of *M. palmata,* with concave areas more associated with the amphipod. In this study, sites where *M. palmata* is present are generally deeper than surround sites, indicating a concave area. Several sites where *M. palmata* are present are adjacent to hard structures that may be colonised by the reef-building tubeworm *F. enigmaticus*, with which they are known to be associated with (Obenat et al. 2006). *G. locusta* occurred at just four sites in very low abundances. Higher temperatures can negatively affect survival and fecundity of *G. locusta*, thus the hot summer temperatures combined with the loss of macroalgal habitat and food source in autumn may have contributed to its low abundance (Cardoso et al. 2018; Neuparth et al. 2002). Although organically enriched sediments provide a food source for *G. locusta*, competition with abundant benthic species may have limited population recovery (Costa et al. 2005).

### 4.5.3 Molluscs

The gastropod species in the lagoon were all incredibly low in abundance. The dominance of Hydrobiid snail *E. ventrosa* over *P. ulvae* observed in this study is not uncommon in tidally restricted lagoons (Barnes 1999; Barnes and Gandolfi 1998; Barnes 1994). The high abundance of empty shells indicates that these species were previously abundant in the lagoon but are no longer. *E. ventrosa* is clustered predominantly at sites around Reedbed 2 by Pipe L2. The lower salinity of this area is closer to the optimal feeding salinity of 20‰ for *E. ventrosa* and their presence in this area also indicates food availability (Barnes 1999). At Site 3, both Hydrobiid species were present and peaked in abundance (although still very low), indicating adequate food availability in this area.

*M. arenaria* was found at Site 34 and at two perimeter sites adjacent to the Sluice Gate. All *M. arenaria* individuals were <1cm in size, suggesting one of two things; that conditions are unfavourable for rapid growth or that they were recently recruited juveniles (<1 year). *M. arenaria* is characteristically patchy on both spatial and temporal scales and has several predators in the lagoon including the oystercatcher *Haematopus ostralegus* (Linnaeus 1758), the flounder *Platichtys flesus* (Linnaeus 1758), and the crab *C. maenas*. Larval and juvenile development is also negatively affected by bioturbation from the lugworm *Arenicola marina* (Linnaeus 1758), also present in the lagoon (Harrison et al. 2016; Strasser 1999; Strasser et al. 1999). However, *M. arenaria* was found at Sites 11 and 12 along the south edge by the Sluice Gate which was an area where lugworm casts were present.

*C. glaucum* was found at one site in the southwest corner but the majority were found in three sites in the northern area. The high energy of the north is likely favourable for these filter feeding bivalves. Abundance peaks at Site 48, adjacent to a surface water outflow, which may provide sufficient water flow for suspension feeding. Specimens were a mixture of juveniles and young adults (<2cm). The filtration activity of the non-native tubeworm *F. enigmaticus* may reduce seston availability for *M. arenaria* and *C. glaucum*, limiting their capacity to colonize the lagoon, thus limiting their abundance.

### 4.5.4 Chironomid Larvae

Chironomid egg masses are oviposited onto the water surface by the imago female, which are then distributed throughout the water body by currents and winds, so their distribution can be relatively random (Pinder 1995; Tokeshi 1995; Schmid 1993). However, the CSW is an area of deposition so the absence of Chironomid larvae suggests that, despite their resilience to hypoxic-sulfidic sediments, the CSW is too hostile for colonisation (Kanaya 2014; Kanaya 2005). Top down control from predation by *P. varians* in the CSW may also play a role (Roberts 1995).

## 4.6 Invertebrate Size Classes

### 4.6.1 *Monocorophium insidiosum*

Nearly all sites where *M. insidiosum* was present had very similar size class proportions, except those sites with very low abundances which were dominated by one or two size classes. This supports the idea that temperature and salinity, which can affect fecundity and growth rates in this species, are relatively homogenous throughout the lagoon on a temporal scale (Prato and Biandolino 2006). Higher temperatures enable *M. insidiosum* to become sexually mature at a smaller size, with first oviposition occurring at 3.3mm at 20°C after 29 days and 4.0mm at 10°C after 99 days (Nair and Anger 1979). Thus, most individuals in Poole Park lagoon are not likely to be sexually mature and will have recruited in late summer/ early autumn earlier in the year of study.

Immature females delay the onset of maturity in November and December. Sheader (1978) found that *M. insidiosum* only has two main generations per year and can live up to 12 months, which is due to the latitudinal differences in seasonality and temperature. The mature generation that produced the immature cohort will die over winter and late summer's immature brood will become sexually mature in spring to reproduce, with immature females delaying the onset of maturity in November and December (Prato and Biandolino 2006; Sheader 1978). Fluctuations in salinity and temperature outside the optimal ranges for breeding can result in egg loss (Kevrekidis 2004). Thus, the extreme salinity ranges brought about by the infrequent flushing regime and high temperatures in summer may negatively impact reproductive success in *M. insidiosum*. Determining reproductive success and variations in fecundity and growth rates in *M. insidiosum* within Poole Park lagoon are beyond the scope of this study.

### 4.6.2 *Hediste diversicolor*

The ragworm *H. diversicolor* has a greater variety of size classes in the north than in the south, where sites are dominated by one or two size classes. Male *H. diversicolor* can grow up to 12cm but females only grow up to 7cm. As no ragworms exceeding this size were found in this study, the sex ratio of *H. diversicolor* is likely to be heavily biased towards females in Poole Park lagoon, which has been seen in populations in north-eastern England (Olive and Garwood 1981). Colour differences indicate sexual maturity in *H. diversicolor* as they lose their rusty orange coloration and become greener. However, this was not observed in measured specimens as their preservation removed any pigmentation. Females begin to mature about 7cm in length and both sexes die shortly after spawning (Budd 2008). Thus, there may be low abundances of larger size classes as *H. diversicolor* reproduces only once before dying. In England, *H. diversicolor* generally has a single short spawning and recruitment period, stimulated by a rise in temperature (Olive and Gardwood 1981; Wharfe 1977; Dales 1950).

The northern sites which have low abundances of larger size classes and high abundances of smaller size classes indicate a recent recruitment of juvenile individuals. These sites may indicate a long-term breeding population with individuals present in most size classes representing different generations. It should be noted that these populations are adjacent to Pipe L8 and usually dominate the associated invertebrate assemblages, suggesting that the fluctuations in salinity discourage the colonisation of less tolerant species. This facilitates *H. diversicolor* to fully exploit habitat and food availability. With reduced competition, *H. diversicolor* can spend more energy on reproductive effort.

The dominance of one or two size classes in the south may indicate increased competition, predation, and hostile conditions that would increase metabolic requirements and reduce energy available for coelom development and reproduction. Size classes were overall relatively small, between 1.1cm and 3.0cm, with Sites 13 and 20 including specimens of the largest recorded size class 6.1-7.0cm. In polluted sites, *H. diversicolor* individuals are generally smaller (Durou et al. 2007). Thus, an absence of the larger size classes may indicate that they reached sexual maturity at a smaller size, reproduced and died.

## 4.7 Pelagic Fauna

There was no significant correlation with any of the pelagic fauna and abundance of macrozoobenthic species. Given that the survey periods were at least seven months apart, temporal and seasonal differences will occur in pelagic and macrozoobenthic populations (Carvalho et al. 2011; Como and Magni 2009; Bachelet 2000). Thus, the populations observed in June and July are unlikely to reflect the population dynamics in November. Absence of correlations does not mean that correlations and relationships do not occur; predation of macrozoobenthic species by *P. microps, P. varians* and *C. maenas* is well known (Pockberger et al. 2014; Rainbow and Smith 2013; Roberts 1995; Escaravage and Castel 1990; Anderson 1985).

*P. varians* was significantly associated with higher organic matter sites and *P. microps* was significantly associated with lower organic matter sites. *P. varians* is predominantly a detritivore but is also a primary and secondary consumer of algae, mysid shrimp, nematodes, mosquito and midge larvae, and annelids (Rainbow and Smith 2013; Roberts 1995; Escaravage and Castel 1990; Anderson 1985). They are tolerant of hypoxic conditions and their detritus-based diet may explain their dominance over *P. microps* in the area of high organic matter. Many of the *P. varians* individuals in this area were quite small and thus they may feed preferentially on particulate detritus.

*P. microps* feeds on polychaetes, molluscs, amphipods, and *N. vectensis* (Pockberger et al. 2014). It dominated sites of low organic matter where hypoxic conditions are less likely to occur as it is stressed by hypoxia (Peterson and Peterson 1990). Hypoxic conditions also influence reproductive behaviour and may negatively affect reproductive success (Reynolds and Jones 1999). Furthermore, macrozoobenthic abundance at sites of high organic matter are on average lower than sites of moderate to low organic matter. Thus, *P. microps* preferentially feeds at these sites. The low organic matter sites are also in closer proximity to hard structures associated with *F. enigmaticus* reefs and algal growth, where productivity is higher and habitat complexity increases protection from predators (Schwindt et al. 2001). *P. microps* preferred the habitat complexity offered from standing vegetation in Arne and Brownsea lagoon as opposed to homogenous soft-bottomed habitat (Wheeler 2013). However, *F. enigmaticus* reefs are preferentially visited by piscivorous birds due to higher prey densities (Bruschetti et al. 2009). *P. microps* could be seen in and around the reefs and concrete columns in high densities.

There was no temporal or spatial correlation between populations of *G. aculeatus* and of *C. maenas* due to their low abundances. *C. maenas* was associated with proximity to complex habitats, including the Concrete Walkway and Reedbed 2. This is likely due to the increased prey density.

## 4.8 Applications

### 4.8.1 Comparative Baseline and Long-Term Monitoring

This study could be used as a comparative baseline for future surveys in the lagoon if the methodology is replicated. In the autumn and winter of 2018, works are proposed in the lagoon, including a prolonged drain down and dredging of sediments for additional islands to be created. A boardwalk that extends into the lagoon centre may be constructed. A comparison of the lagoon fauna and environmental variables pre and post works would give an indication if the ecosystem is recovering. However, it should be undertaken at the same time of year (November), so the comparison is meaningful and accurate (Bamber et al. 2001). Ideally, a suite of baseline surveys would be more useful, taken at least quarterly, to understand trends and patterns, prior to comparison. Therefore, this should be considered for the monitoring of the lagoon going forward, particularly to aid site-specific and adaptive management and to improve knowledge on a key site in the regional Poole

Harbour lagoon network. If comparisons are drawn between future surveys and the one undertaken as part of this study, it should be noted that Poole Park lagoon is still considered a degraded habitat and is essentially a bar to raise, not to aspire to.

It should also be noted that lagoons are, by their very definition and nature, patchy, stochastic and stressed ecosystems, and subsequently it can be challenging to obtain baselines and filter out background variation and random events (Stringell et al. 2013; Perez-Ruzafa et al. 2007; Bamber et al. 2001). Lagoonal communities can show a naturally high variability in response to changes in abiotic variables which could be perceived as negative change. If a site is judged to display poor ecological performance, the attempted suppression of these pressures may negatively impact lagoon specialists. Thus, a rigorous and regular monitoring regime is crucial to minimise the influence of random events on long-term trends (Stringell et al. 2013; Bamber et al. 2001). However, it is acknowledged that sampling frequency for Poole Park may fall below the recommended quarterly minimum due to the potential financial constraints of the local authority responsible for its management (Lucas et al. 2006). It can be argued that no real baseline or ‘ideal’ environmental parameters exists within lagoons given their highly variable nature and cycles of dystrophic events and recolonization (Stringell et al. 2013; Gamito et al. 2005).

Increases in precipitation in winter and decreases in summer, changes in wind direction prevalence and speed, and increased storminess are all likely to occur in the coming years due to climate change (Fakhry et al. 2013; Nolan et al. 2012; IPCC 2007). This will affect the salinity regimes and associated macrozoobenthic assemblages of Poole Park lagoon, potentially increasing the range variability and the length of time spent at these extreme values. Changes in wind may alter the spatial distribution of finer sediments and increase turbidity. Overall temperatures in the UK are predicted to increase with milder winters (increase of up to 2.5°C) and considerably hotter summers (increase of up to 4.2°C) (Murphy et al. 2010). Thus, the seasonal ranges in temperature and salinity may increase significantly, potentially leading to more frequent and intense dystrophic events in the summer and thus slower recolonization over the winter. Abiotic data in this study will serve as a useful comparison to data collected in future years, particularly within the context of climate change.

### 4.8.2 Value for Avifauna

Brownsea Island lagoon is a site of significant importance for nesting sandwich terns *Thalasseus sandvicensis (*Latham 1787), common terns *Sterna hirundo*, (Linnaeus 1758), black-headed gulls and occasionally Mediterranean gulls supports *Larus melanocephalus* (Temminck 1820). Additionally, it provides crucial feeding habitat for internationally important species such as avocets and spoonbills. Brownsea lagoon supports these species due to the accessible depth for waders. Poole Park is 0.55m at its shallowest, too deep for wading birds. At the time of writing, proposed works for the lagoon include islands with graduated plateaus that will provide shallower areas, but it is unknown how successfully these will be colonised by macroinvertebrate prey items and if they will be of a suitable depth for utilisation by waders. However, subsidence has occurred on some of the existing islands, creating a littoral area that has been used by feeding oystercatchers (personal observation). Without intervention, the subsistence may continue over time and the littoral area will become too deep. The islands have been problematic for several years due to erosion from use by swans and geese, and various attempts have been made to keep them off. Methods have included lining the islands with reeds and shoring up the subsidence with wooden fencing. The persistence of swans and geese has thwarted these attempts however, and a permanent solution is yet to be enacted.

Furthermore, Poole Park experiences greater disturbance from human activity than the other Poole Harbour lagoon sites and may not be considered as a suitable feeding ground. Pickess and Underhill-Day (2002) found that Poole Harbour waders will preferentially feed in areas of lower disturbance, even if the benthic biodiversity is poorer, suggesting less disturbance is prioritised over richer feeding grounds. However, common sandpipers *Actitis hypoleucos*

(Linnaeus 1758), ruff *Calidris pugnax* (Linnaeus 1758), redshank *Tringa tetanus* (Linnaeus 1758), black-tailed godwits have been observed feeding on the mudflats during a drain down, so the creation of a permanent littoral mudflat could see a greater abundance and frequency of these species (Birds of Poole Harbour 2018).

Though at present Poole Park has a limited capacity to accommodate waders, it supports several fish prey species for piscivorous birds, including sand smelt *Atherina presbyter* (Cuvier 1829), bass *Dicentrarchus labrax* (Linnaeus 1758), herring *Clupea harengus* (Linnaeus 1758), flounder *P. flesus*, and species found in this study *P. microps* and *G. aculeatus* (Harrison et al. 2016). Little egret feed on *A. presbyter, P. microps* and *G. aculeatus*, cormorants feed on *A. presbyter*, kingfishers *Alcedo atthis* (Linnaeus 1758) feed on *G. aculeatus* and potentially *P. microps* as it falls within its preferred prey size range, and common and sandwich terns feed on *A. presbyter* (Farinos-Celdran et al. 2018; Wheeler 2013; Reynolds and Hinge 1996; Hafner et al. 1982). Thus, Poole Park's value as a supplementary feeding habitat for these piscivorous species is greater than for wader species. Its greatest habitat value is likely to be as a sheltered roosting site, where it supports large numbers of Mediterranean gulls, black-headed gulls, common gulls *Larus canus* (Linnaeus 1758), herring gulls *Larus argentatus* (Pontoppidan 1763), lesser black-backed gulls *Larus fuscus* (Linnaeus 1758), and greater black-backed gulls *Larus marinus* (Linnaeus 1758), all of which are designated with an amber or red UK conservation status and protected by The Wildlife and Countryside Act 1981 (Birds of Poole Harbour 2018).

Some of the islands proposed by Borough of Poole council will use a stony substrate to encourage terns to nest. Site fidelity among terns can be low, so discovery and colonisation of the new islands if appropriate could be relatively soon (Ratcliffe et al. 2000; Lloyd et al. 1991). Inaccessible from the lagoon shore, the islands would need protection from watercraft disturbance, particularly as the terns' breeding season in the UK coincides with the occupation of Poole Park lagoon in the summer by recreational boating company Rockley. Management of this could include a sign-posted exclusionary zone around the islands and by informing water users of the terns' whereabouts before they leave the shore (Burger and Leonard 2000).

The proposed dredging by Borough of Poole council will create deeper areas within the lagoon which could facilitate stratification depending on their location, promoting thermoclines and haloclines. Areas demonstrating salinity and temperature gradients could promote greater biodiversity by creating niche habitats appropriate for lagoon specialists (Bamber et al. 2001).

## 4.9 Limitations and Considerations

Sediment cores can provide a wealth of information about the benthic and infaunal habitat, such as granulometry and pore water chemistry. However, in-situ conditions, such as depth of oxygenated layer and how the grains are sorted, are not always measured when they could be particularly relevant in determining richness and abundance (Dauvin et al. 2017; Holland and Elmore 2008). The sediment samples are homogenised before subsamples are taken and processed by the Mastersizer laser diffractometer, so intact cross-sections of the sediment are destroyed and may not be representative of the ambient environment that resident organisms inhabit (Snelgrove and Butman 1994). For future studies, qualitative data should be noted for each sample where possible, including approximate depth of oxygenated and de-oxygenated layer, how the grains are vertically sorted, if the sample has a hydrogen sulphide odour, and where visible organic matter has accumulated. This information will augment data obtained from the laboratory and help explain the spatial distribution of infaunal invertebrates. For example, if ragworms were not present and it was noted that the oxygenated layer was particularly thin, it is likely the habitat was not suitable for them. Furthermore, infaunal species will exhibit sediment depth preferences depending on their tolerance to variable interstitial conditions, which may prevent competition (Gamenik et al. 1996; Thiermann et al. 1996; Dubilier et al. 1995; Vismann 1990). The data for their relative vertical distribution is also missed when they are sieved from sediment cores. There is a need for lab-based studies on the ecology and biotic interactions of lagoon benthic macroinvertebrates and such a study would shed light on their vertical distributions and depth preferences, with the data supporting field-based surveys.

Two metrics of particle size that have been used in similar soft-bottomed studies were used to quantify granulometry in this study to see if there was any difference in functionality when describing sediments; median grain size (MGS) and percent silt content (Silt Content) (Henkel and Politano 2017; Seiderer and Newell 1999). Information is filtered when continuous data, such as particle size, are broken into categories, such as particle size classes which were used to inform the Silt Content metric in this study (Steel et al. 2013). Indeed, Magnusson (1997) wrote that most ecological categories are arbitrary, and the breaks in grain size classes used in the study may not be functionally relevant when trying to determine associations with the macroinvertebrate assemblages. It is challenging to quantify particle sizes in a biologically relevant way when many lagoon macroinvertebrate species not only have multiple feeding strategies and dietary preferences, but the vertical particle sorting and invertebrate distribution data are lost during laboratory processing. Many particle size metrics, such as Silt Content and others such as dominant size class, are derived from the Wentworth scale (Wentworth 1949). In this study MGS and Silt Content produced very similar and significant values and outputs when used in statistical analyses and display very similar patterns as interpolated layers on a map. Their significant associations with density, species richness, and relative abundance of species made ecological sense within soft-bottomed lagoon ecosystems, therefore for future lagoon studies either metric is suitable. Within MGS and Silt Content are size classes that are not wholly represented by these simplified metrics, so it is still prudent to look closely at the continuous granulometry data in conjunction with biodiversity metrics, in case any obvious patterns can be identified that are not evident in statistical analyses or the simplified metrics.

It is challenging to build a broader picture of the recent temporal trends for the aquatic variables without data, as proximity to flush days and rainfall will influence the salinity and temperature regimes on a local and whole lagoon scale. Therefore, interpretation of these variables is limited. For future related studies, it would be prudent to obtain salinity and temperature data not just from the water surface, as in this study, but also benthic and interstitial salinity and temperature. It is well known that salinity and temperature undergo stratification in lagoon ecosystems, particularly if the water is not well-mixed as it may be in the tidally-restricted Poole Park lagoon; surface temperature and salinity may have had little influence over benthic and infaunal invertebrates (Barnes 1980; Bamber et al. 2001). Obtaining temporal data will help to identify trends but also identify where the system lags; that is the delayed response of macrozoobenthic species to changes in the environment. Thus, the macrozoobenthic assemblages recorded in this study may be indicative of long-term trends or a change in environmental variables that occurred prior to the survey date. This could lead to inaccurate interpretation of the relationships between abiotic variables and biodiversity metrics in this study.

# Chapter 2 - The Starlet Sea Anemone in Poole Park Lagoon

## 5. Introduction

### 5.1 Non-Native Species

Non-native species are one of the major drivers of biodiversity loss globally (IUCN 2000). Carlton (1987) defined non-native species as “introduced species are those taxa transported by human activity to regions where they did not exist in historical times”, historical times referring to 5000 years before the present. Non-native marine species are usually introduced via the shipping industry, either by fouling or ballast water, or deliberate or incidental introduction from the shellfish industry, particularly oysters from Japan and North America (Eno et al. 1997). Their impact can range from undetected to the domination and displacement of indigenous species to increasing biodiversity of indigenous species; the interactions are often complex and understudied. For example, the hybridized common cord-grass *Spartina anglica* dominates saltmarshes and reduces feeding grounds for waders (Davidson et al. 1991). However, *S. anglica* saltmarsh also provides crucial high tide roosts for wintering and passage waders (Morrison 2004).

At least 150 non-native marine species have been recorded in the UK, though this is likely an underestimation (Roy et al. 2007). Non-native taxa that are represented most frequently are red algae (19%), molluscs (18%), annelids (15%) and crustaceans (14%) (Eno et al. 1997). In addition to requiring a vector to facilitate their colonization of new territories, the host site is usually one of low species diversity with vacant niches to exploit (Ribera and Boudouresque 1995). Harbours and estuaries are common sites for the colonization of non-native species; ships often dock there, and they are typically low energy which promotes local settlement of larvae or propagules. The expanse of hard structures associated with developed harbours and estuaries are ideal sites for attachment of sessile species such as mussels and barnacles.

Poole Harbour is one such site. With an area of $13.9^2$ miles and a relatively narrow mouth, a full flush can take up to three days with only 22-45% of the harbour volume flushed on a spring tide (May and Humphreys 2005). It's shallow depth and limited flushing regime leads to warmer temperatures than the open sea, producing ideal conditions for opportunistic non-native species. Disturbance to sediments from pump-scoop dredging may also provide colonization opportunities for non-native invertebrates, allowing them to utilize space and resources once dominated by indigenous fauna (Clarke et al. 2018).

There are at least 12 known non-native species in Poole Harbour and have previously represented up to 60% of wet weight of assemblages (Harrison et al. 2016; Maggs and Magill 2014; Herbert et al. 2010; Eno et al. 1997). This is higher than the mean of 10.8 non-native species in marinas and harbours on the south coast of England (Arenas et al. 2006). Their interactions with native fauna and socio-economic effects are not fully understood, though there are some exceptions. The Manila clam *Ruditapes philippinarum* (Adams and Reeve 1850) is indigenous to western Pacific coasts but was introduced to the harbour in 1988 following successful commercial trials, and by 1994 was being harvested by local fishermen (Humphreys et al. 2015; Utting and Spencer 1991). The harbour's favourable physico-chemical parameters have enabled the clam to naturalise and the clam fishery had a value of £1.5 million in 2014 (Franklin et al. 2012).

### 5.1.2 Non-Native Species in Lagoons

The impounded nature of lagoons provides ideal shelter for non-native species that can withstand the stochastic salinity and temperature regimes. Species such as the red alga *Gracilaria vermiculophylla* (Papenfuss 1967), the sand gaper *M. arenaria*, and the Australian tubeworm *F. enigmaticus* have been identified in lagoons in Poole Harbour (Harrison et al. 2016; Maggs and Magill

2014). These species have not yet had any proven negative impacts on native fauna but demonstrate the capacity with which they can colonise an impounded lagoon. The cryptic bivalve *M. arenaria* is an important prey species for the flounder *P. flesus*, the curlew *Numenius arquata* (Linnaeus 1758) and the oystercatcher *H. ostralegus*, all known to utilise Poole Harbour lagoons for feeding (Harrison et al. 2016; Strasser 1999; Zwarts and Wanink 1989). The red alga *G. vermiculophylla* and tubeworm *F. enigmaticus* have the capacity to increase habitat complexity and biodiversity within lagoons, which are typically soft-bottomed homogenous environments (Heiman and Micheli 2010; Thomsen 2010; Nyberg et al. 2009; Bianchi and Morri 1995).

## 5.2 The Starlet Sea Anemone

The starlet sea anemone *Nematostella vectensis* (Stephenson 1935) is a small (≤1.5cm) infaunal cnidarian that thrives in muddy soft-bottomed substrate where it preys on other benthic invertebrates. Unlike many other anthozoans, *N. vectensis* has not faced selective pressure to develop a variety of neurotoxins owing to its limited prey selection consistent with the low species diversity of lagoons, having many genes that may enable it to rapidly refill its toxin stores (Moran and Gurevitz 2006). *N. vectensis* also has chemoreceptors that complement its mechanoreceptors, with the detection of prey-derived n-acetylase sugars reducing the vibration-induced action potential necessary to fire nematocysts. The detection of such sugars also increases the length of hairs that detect vibrations from prey movement, further increasing the chance of successful prey capture (Watson et al. 2009). Such adaptations ensure maximum exploitation of prey abundance.

*N. vectensis* can reproduce asexually and sexually, maximising its colonisation capacity. Transverse fission rates are higher at lower salinities and, with increased food availability (Hand and Uhlinger 1995), sexual reproduction occurs at full marine salinity (Hand and Uhlinger 1992). Sixty-one per cent of UK individuals are derived from one genotype suggesting a dominance in clonal reproduction, indicative of a species that has recently colonised an area (Pearson et al. 2008). Sexual reproduction has been induced under laboratory conditions at 20°C and has been recorded as occasionally occurring in the field during summer and autumn when temperature and salinity would be at its highest (Williams 1976; Frank and Bleakney 1978). Such reproductive plasticity is advantageous in a hostile lagoon environment.

The common ditch shrimp *P. varians*, *N. vectensis'* currently only known predator in the UK, was thought not to co-occur in the same sites as *N. vectensis* (Barnes 1994; Posey and Hines 1991). However, all three lagoons *N. vectensis* is present in in Poole Harbour also have records of the presence of *P. varians* (Bone 2017; Harrison et al. 2016; Herbert et al. 2010). In Brownsea Lagoon, *P. varians* and *N. vectensis* co-occured with densities of 42 and 3048 per $m^2$ respectively within the same benthic sample (Herbert et al. 2010). As *P. varians* does co-exist with *N. vectensis*, there is an opportunity for its predator-prey relationship to be quantified.

*N. vectensis* has been used in genome research of which there is an abundance of literature, but there is a notable absence of literature detailing its ecology in UK lagoons (Darling et al. 2005). It has been listed as Vulnerable on the IUCN Red List since 1983 and was subsequently listed under the Wildlife and Countryside Act 1981 as a protected species in 1988 (Sheader et al. 1997; World Conservation Monitoring Centre 1996). However, the last assessment was performed in 1996 with the annotation 'needs updating'. Sheader et al. (1997) postulated that *N. vectensis* was non-native having been introduced to Europe from North American populations via the oyster trade. The North American populations are unlikely to have colonised British coasts owing to their limited dispersal range, unless the British population is relict and indicative of a larger historical range. Therefore, a relict population would show appropriately divergent genetic distances.

Reitzel et al. 2007 analysed the DNA of British and North American *N. vectensis* specimens which confirmed that British populations have been introduced from western Atlantic populations. British populations possess significantly lower genetic diversity than western Atlantic populations, due to reproducing asexually as opposed to sexually. Barfield (2016) noted in his review article that while non-native species in the UK are eligible for protection under the Wildlife and Countryside Act (1981), *N. vectensis* satisfies only half of the designation criteria. Indeed, its lagoon habitat faces numerous threats and is at risk of habitat loss, but lagoons already receive protection in the UK and the designation of an inhabiting species to bolster lagoon protection is redundant.

It's protection in the UK, its Vulnerable status and the data on which these designations are based are outdated and in need of review. However, before such a review it would be prudent to quantify *N. vectensis'* effect on native lagoon fauna and lagoon benthic ecology, particularly its role as a passive predator and prey item for *P. varians*. It is known to consume small annelids, Hydrobiid snails, copepods, and has been observed feeding on Chironomid larvae in the field (Sheader et al. 1997; Welstead and Shardlow 1999).

Occurring only rarely in the now extinct Blue Lagoon in Poole Harbour in the late 90s, *N. vectensis* can now be found in high densities in all Poole Harbour lagoons, excluding Arne.

## 5.3 Rationale

Although salinity is important in determining distribution of invertebrates within lagoon habitats, including *N. vectensis*, sediment factors are also known to be a primary factor. Additionally, there is a paucity of data about the potential top-down control of its distribution from predator-prey interactions with *P. varians* also known to inhabit Poole Harbour lagoons. With reference to its revised status, it would be judicious to analyse its role within invertebrate assemblages through the non-native lens and critically appraise its potential interactions with native lagoon ecology.

### 5.4 Aims and Objectives

The study aims to semi-quantitatively assess the role of *N. vectensis* within native lagoon assemblages and to determine the abiotic factors that determine its distribution.

1) Utilising the abiotic and biotic data collected for Chapter 1
2) Identify relationships between abiotic variables and *N. vectensis* by:
    a) Using ArcGIS to visually identify spatial trends
    b) Conducting statistical analyses
3) Identify, if any, relationships between *N. vectensis* and overall macrozoobenthic assemblages by:
    a) Analysing assemblage composition
    b) Using existing literature to understand interactions
4) Identify, if any, relationships between *N. vectensis* and pelagic fauna by:
    a) Conducting appropriate statistical analyses
    b) Using existing literature to understand interactions

## 6. Materials and Methods

Numeric abundance data for *N. vectensis* was obtained from sediment cores taken on November 7th and 8th and were counted under a stereo microscope in laboratory conditions. Samples were fixed in 70% Industrial Methylated Spirit (IMS) for identification and quality control purposes. Body size data was not obtained for the *N. vectensis* due to the propensity for specimens to contract when fixated, preventing accurate measurement (Sheader et al. 1997). To analyse its interaction with environmental variables and pelagic fauna, abiotic factors were measured, and fish traps were used to obtain species data. Further details can be found in Materials and Methods in Chapter 1.

## 7. Results

*N. vectensis* was significantly negatively correlated with OM Content, Salinity, and Silt Content and significantly positively correlated with MGS (see Table 8,

Figure 20). *N. vectensis* density in the CSW portion of the lagoon was relatively low, not exceeding 500 individuals per m$^2$. Densities increase in the north west and east side of the lagoon, peaking at sample Site 34 with 9800 per m$^2$ (see Figure 19). This was identified as an outlier for regression analyses. When the outlier was removed and analysis re-run for SDepth and *N. vectensis* density they were significantly negatively correlated (F (df = 1, 46) = 4.316, p = 0.043).

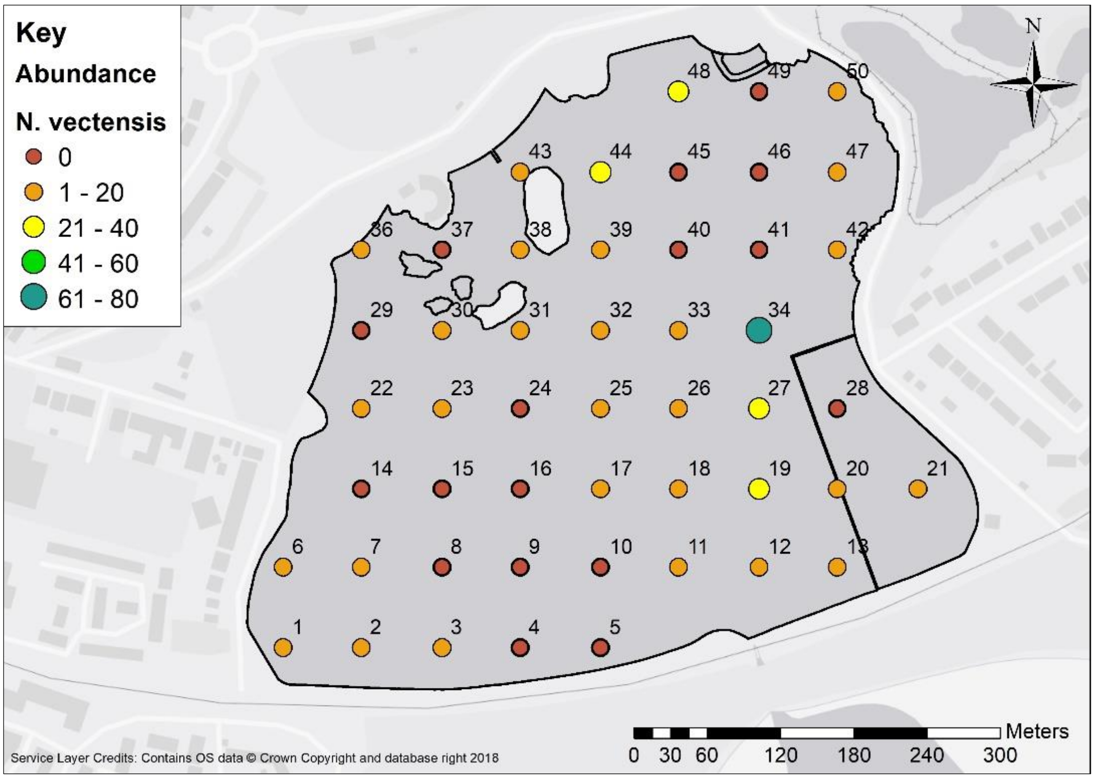

Figure 17 The spatial distribution of the density of *N. vectensis.*

The relative abundance of *N. vectensis* within an assemblage did not exceed 50% and the mean relative abundance was only 13% as *N. vectensis* was not present at 17 out of 49 sample sites. Only 11 out of 49 sites did *N. vectensis* exceed 20% relative abundance and species richness and overall invertebrate abundance at these sites ranged from 2 to 9 and 3 to 141 respectively. *N. vectensis* was the dominant taxa at seven sites but at only three of these sites was it the sole dominant taxa. *N. vectensis* was dominant at sites which had low invertebrate abundances with only two of the seven sites having higher than average (48) invertebrate abundances.

Table 8 Regression analyses for *N. vectensis* density and abiotic factors. Significant p values in bold. Site 34 was a consistent outlier in all regressions.

| ***N. vectensis* Density and Abiotic Factors of Poole Park Lagoon** | | | | | | |
|---|---|---|---|---|---|---|
| **Variable** | **Regression** | **$R^2$** | **DF** | **F** | **p** | **Unstandardized Co-efficient B** |
| OM Content | Linear | 11.1 | 1, 47 | 5.857 | **0.019** | -176 |
| Silt Content | Linear | 12.1 | 1, 47 | 6.496 | **0.014** | -31 |
| MGS | Linear | 9.2 | 1, 47 | 4.767 | **0.034** | 9.812 |
| SDepth | Linear | 7.5 | 1, 47 | 3.817 | 0.057 | -836 |
| Temperature | Linear | 0 | 1, 47 | 0.003 | 0.954 | 32 |
| Salinity | Quadratic | 18.8 | 2, 46 | 5.308 | **0.008** | **-**2232 |
| WDepth | Linear | 0.1 | 1, 47 | 0.045 | 0.832 | 340 |
| Distance from SW Shore | Linear | 10.1 | 1, 47 | 5.285 | **0.026** | 3 |
| Distance from E Shore | Linear | 10.7 | 1, 47 | 5.635 | **0.022** | -4 |
| Distance from N Shore | Linear | 5.3 | 1, 47 | 2.646 | 0.111 | -2 |
| Distance from Sluice | Linear | 0.9 | 1, 47 | 0.426 | 0.517 | -1.451 |

Of the 17 sample sites that *N. vectensis* was not present at, all but one (Site 49) had lower than average (46) overall invertebrate abundances and all but three (Sites 9, 14, 49) had lower than average (4) species richness and so were characterised by particularly poor biodiversity. The sites where *N. vectensis* was not present was dominated by amphipods and Chironomid larvae by >75% except for Site 41. These 17 sites are in two main clusters; five sites in the CNE and nine sites in the CSW (see Figure 19). Other sites included a pair at the lagoon edge in the northwest direction and a single site in the southeast inside the Model Boating Area. These 17 sites share a similar distribution pattern with overall invertebrate density.

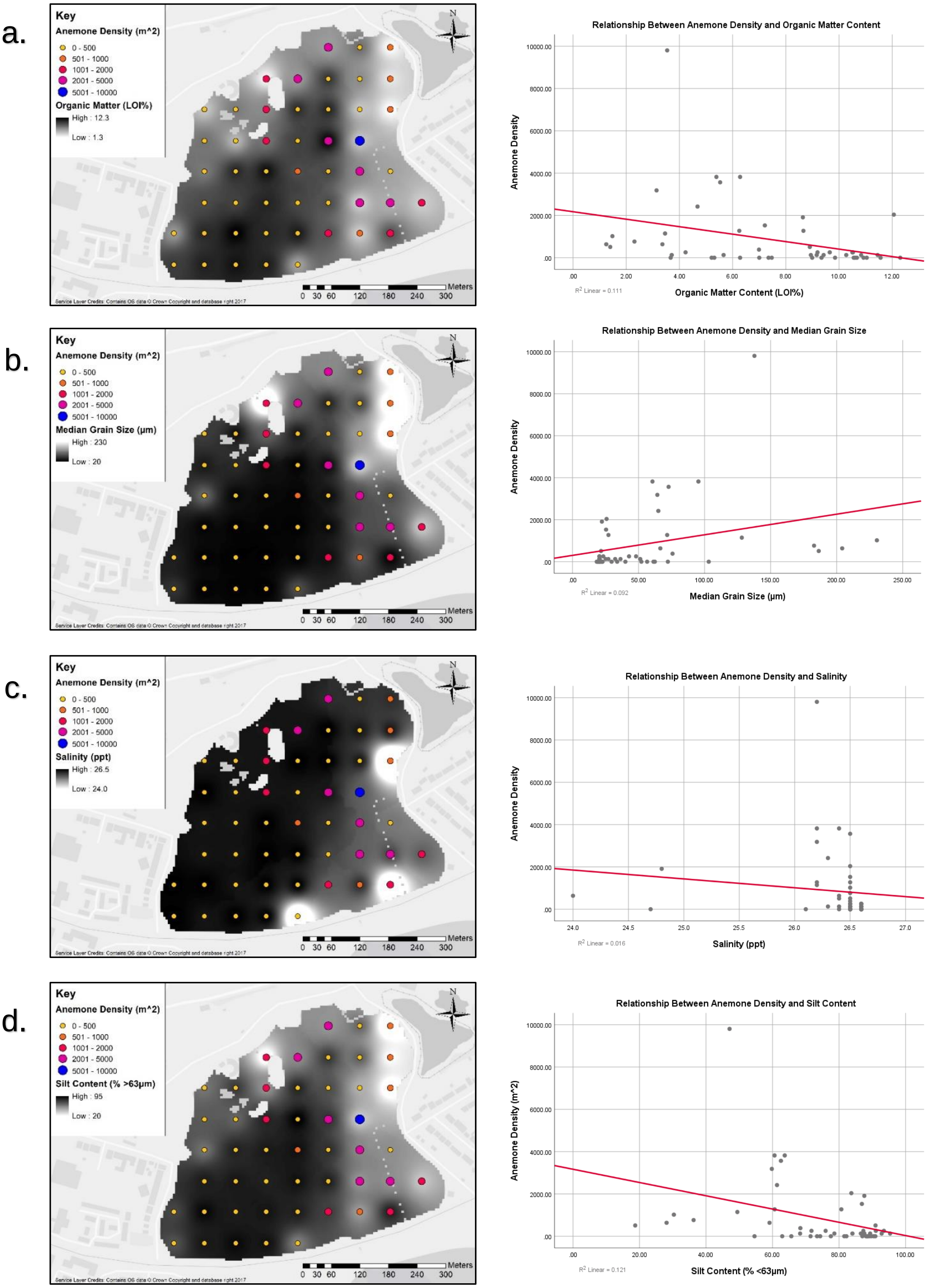

Figure 18 *N. vectensis* density with significantly associated abiotic variables, including maps of the spatial distribution of abiotic gradients and *N. vectensis* density and scatter graphs demonstrating relationships. a) OM Content, b) MGS, c) Salinity and d) Silt Content.

## 7.1 *N. vectensis* Density and Pelagic Fauna

*N. vectensis* was not significantly correlated with any pelagic fauna, though this should be interpreted cautiously given the length of time that has elapsed between survey periods (see Table 9).

Table 9 Regression analyses for *N. vectensis* and pelagic abundance in Poole Park Lagoon. Site 16 was a consistent outlier for regression analyses.

| ***N. vectensis* Density and Pelagic Abundance of Poole Park Lagoon** | | | | | | |
|---|---|---|---|---|---|---|
| **Variable** | **Regression** | **$R^2$** | **DF** | **F** | **p** | **Unstandardized Co-efficient B** |
| *C. maenas* Total | Linear | 0.2 | 1, 18 | 0.039 | 0.845 | 69.287 |
| *G. aculeatus* Total | Linear | 0.9 | 1, 18 | 0.162 | 0.692 | -2742469 |
| *P. microps* Total | Linear | 5.1 | 1, 18 | 0.974 | 0.337 | 34.328 |
| *P. varians* Total | Linear | 2.8 | 1, 18 | 0.524 | 0.479 | -21.246 |

*N. vectensis* was the dominant taxa at just seven sites, not exceeding 50%. At four of these sites, dominance is equally shared with one other, sometimes two, species due to extremely poor species richness and abundance at those sites. Prey species richness and abundance was greater at sites where *N. vectensis* was solely dominant compared to sites where it shared dominance.

## 7.2 Biotic Interactions

A scatterplot of *N. vectensis* and amphipod abundance shows a positive linear correlation, with anemone abundance increasing with amphipod abundance (see Figure 19). A scatterplot of the proportional percentage of Amphipods and Anemones within the overall invertebrate assemblage of each site shows as negative linear correlation with the proportion of *N. vectensis* increasing as the proportion of amphipods decrease (see Figure 20). Amphipod percentage dominance is generally highest at sites where *N. vectensis* is absent. *N. vectensis* density is significantly positively correlated with overall invertebrate density (F = (df = 1, 47) = 145.64, p = <0.05) (see Figure 21).

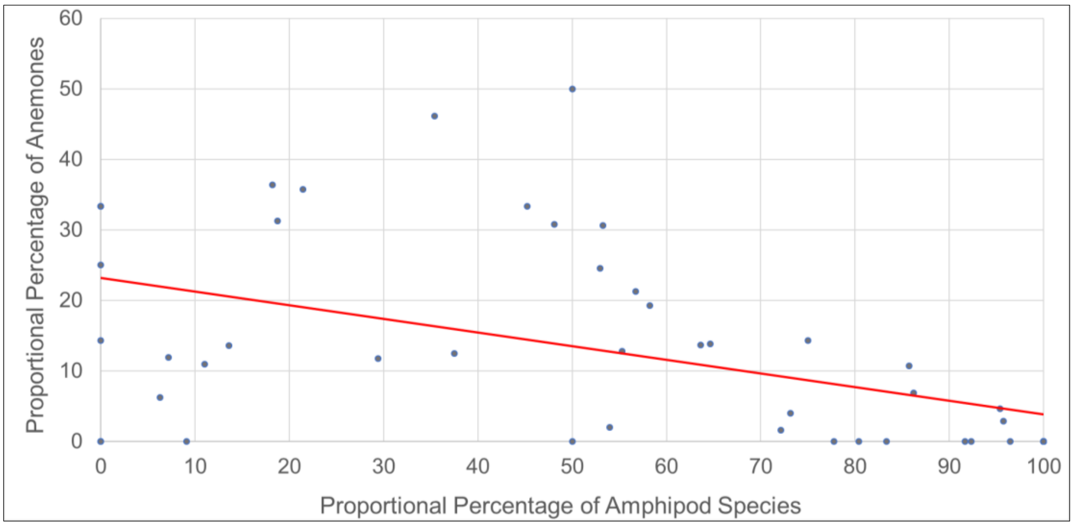

Figure 19 The negative linear relationship between the proportions of *N. vectensis* and amphipod abundance.

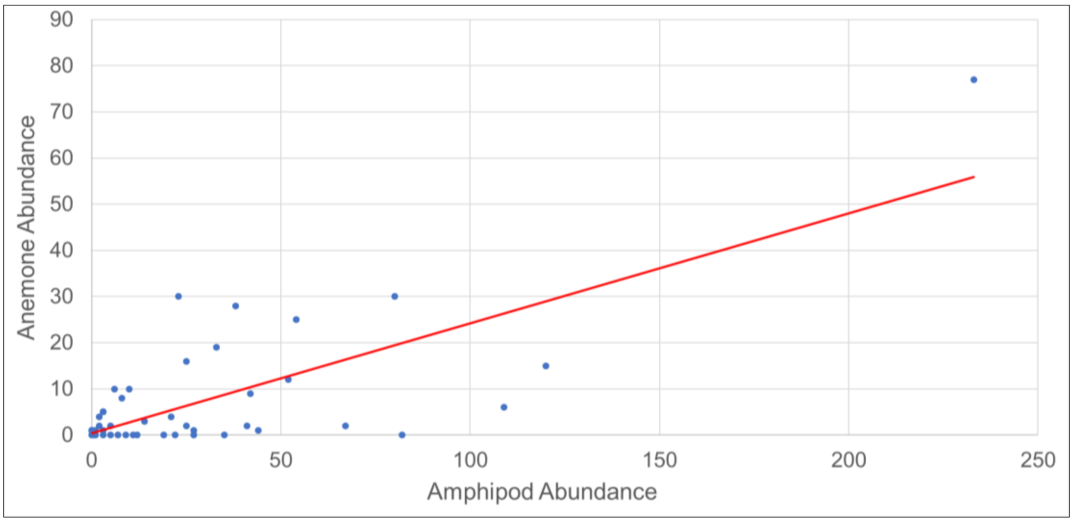

Figure 20 The positive linear relationship between *N. vectensis* and amphipod abundance.

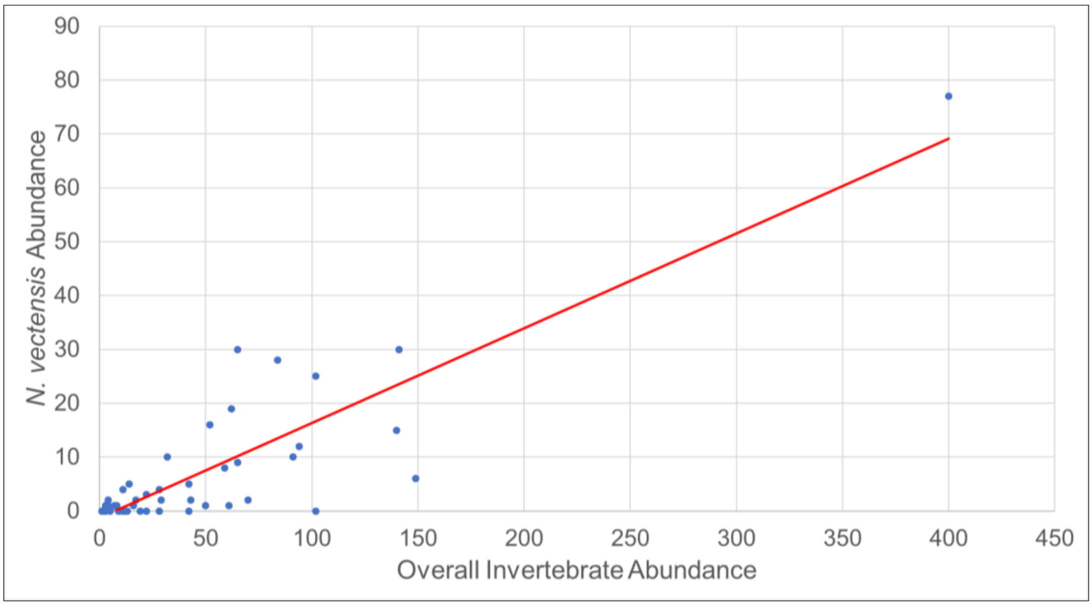

Figure 21 The positive linear relationship between overall invertebrate and *N. vectensis* abundance.

# 8. Discussion

This study aimed to determine the variables responsible for the spatial distribution of *N. vectensis* and assess its role within native assemblages. *N. vectensis* was significantly negatively associated with OM Content, Salinity, and Silt Content and significantly positively correlated with MGS. It was also positively correlated with macrozoobenthic density.

*N. vectensis* is known to preferentially colonise very fine sediments, so the significant correlations with particle size factors is likely because of their co-linearity with OM Content (Williams 2003). Fine sediments and increased OM Content often co-occur because of the low energy environment necessary for their deposition (Burone et al. 2003). Increased OM Content would lead to hypoxic-sulfidic conditions creating a hostile environment unsuitable for colonisation (Gray et al. 2002; Rybarczyk et al. 1996). Though *N. vectensis* was also found in the coarser sediments of the lagoon, the largest particle size did not exceed 230µm, contained at least 18% silt and would still be suitable for burrowing.

*N. vectensis* is a euryhaline species and so the significant association with salinity is interesting (Sheader et al. 1997; Barnes 1994). The salinity regime of the lagoon at the time of survey was relatively homogenous with a narrow range of 2.5‰, and only three sites were below 25‰; most of the lagoon was between 25.9‰ and 26.6‰. As salinity and *N. vectensis* are both significantly correlated with macrozoobenthic density it is possible that salinity and macrozoobenthic density are co-linear. Thus, this euryhaline anemone could be reliant on how the salinity regime dictates the spatial distribution of its prey species. An alternative explanation is that within ecological scales the association with salinity is irrelevant and it is unlikely the euryhaline anemone is affected by a relatively minor fluctuation in salinity.

*N. vectensis* was significantly positively correlated with macrozoobenthic density likely because of an increase in prey availability (Sheader et al. 1997). Increases in food intake increase the frequency of transverse fission, a common method of reproduction in English specimens, thus high prey densities will cause rapid population growth (Hand and Uhlinger 1995; Hand and Uhlinger 1992).

*N. vectensis* was abundant overall but was not found at all sites at which prey species also occurred. As *N. vectensis* is a small (<1cm) burrowing infaunal cnidarian, the absence of a well oxygenated layer at the sediment-water interface may cause the anemone to evacuate the sediment or suffer paralysis due to the inhibition of peristalsis which is vital to its locomotive behaviour (Williams 2003). Hypoxic-sulfidic sediments, like those in the central southwest of Poole Park lagoon, are unlikely to be suitable for colonisation and will explain why *N. vectensis* is not present even when suitable prey species such as amphipods are. There is a second cluster of five sites in the northeast with no *N. vectensis* which may occur due to the high energy environment preventing settlement.

In sites where *N. vectensis* was particularly abundant (N = >20), annelid worms were also moderately abundant. This is likely due to the shared preference for sediments that are not high in OM Content. The bioturbation by these worms could also facilitate the thixotropic and well oxygenated sediment favoured by *N. vectensis* (Williams 2003; Sheader et al. 1997). Furthermore, one *N. vectensis* individual identified under the stereo microscope was partially ingesting a *Polydora* sp. individual, indicating that worms of a certain size are vulnerable to predation by *N. vectensis*. However, where worms dominate by >50%, the proportion of other species decreases, including *N. vectensis*. This may be because of disturbance, such as dislodging *N. vectensis* from the sediment, predation by larger *H. diversicolor* individuals or even interspecific competition for prey with *H. diversicolor*.

Three of the most abundant sites, including superabundant Site 34, were adjacent to the Concrete Walkway. Sediment conditions here were favourable with low to moderate OM Content and MGS between 51µm and 145µm, enabling burrowing activity. Another factor potentially responsible for increased abundance at these sites is the proximity to the *F. enigmaticus* reefs that have colonised the Concrete Walkway, and the increase in prey associated with this. However, the tubeworm reefs are also associated with higher abundances of the only recorded predator of *N. vectensis; P. varians.* The predator-prey dynamics between the two species are unclear and could be scale dependent; juvenile *P. varians* could be prey for *N. vectensis* before preying on *N. vectensis* itself when it is a larger size.

The other two abundant (N = >20) sites were in the north, adjacent to the northeast cluster where no *N. vectensis* individuals were found. Though this is an area of high energy, south-westerly prevailing winds would be disrupted by the islands to the southwest of the two abundant sites reducing disturbance.

This study found no correlation between the abundance of *N. vectensis* or *P. varians*, suggesting that the shrimp has a negligible effect on *N. vectensis* density. However, the pelagic survey was not as spatially comprehensive as the macrozoobenthic survey and conducted several months apart so this should be interpreted cautiously. *P. varians* is omnivorous and will feed on detritus and macroinvertebrates though there is a paucity of data available on its diet (Barnes 1994). *Palaemon elegans* (Rathke 1837), a morphologically similar shrimp species that inhabits similar ranges, will consume Chironomid larvae, small crustaceans and *H. diversicolor* (Janas and Baranska 2008). *P. varians* may preferentially feed on species that occur at greater densities, such as the amphipods *M. insidiosum* and *M. gryllotalpa*. As previously mentioned, predation on *N. vectensis* may also be size dependent as smaller *P. varians* individuals could be vulnerable to predation by *N. vectensis* even if it they are too large to be consumed by it.

*N. vectensis* and amphipod abundance were positively correlated, and their percentage proportions were negatively correlated. This indicates that favourable habitat conditions allow these species to proliferate, but the increase in the percentage proportion of *N. vectensis* may have a negative effect on the proportion of their prey species due to increased rates of predation. However, this does not consider interspecific and interference competition from other species that will be colonising the favourable sediments and should be investigated further.

Due to the physal pinching method of transverse fission of *N. vectensis*, their abundance may be underestimated due to the discarding of budded specimens that had not developed into physiologically identifiable individuals – particularly as fixation induces contraction (Sheader et al. 1997). Additionally, *N. vectensis* can migrate vertically into floating macroalgal mats using adhesive rugae so benthic sampling may underrepresent *N. vectensis*, though it is unlikely to be by a significant margin (<5%) (Barnes 1994). It is undetermined if such vertical migration is incidental from accidental attachment or deliberate because of negative chemotaxis to hydrogen sulphide (Williams 2003). For future studies, it would be prudent to obtain a hand sample of overlying algal mats to rinse migratory invertebrates from.

*N. vectensis* within Poole Park lagoon appears to have a negligible effect on native macrozoobenthic assemblages. However, Poole Park is still a degraded habitat and *N. vectensis* may have greater top-down influence if invertebrate densities were greater. Its influence on disturbance of sediment is low due to its small size and shallow burial, only relocating when interstitial conditions become too hostile or it is displaced by wave action or the burrowing activity of another species. Upon reburial, sediment is not displaced in a conspicuous manner akin to lugworm activity. Instead *N. vectensis* uses peristaltic contractions and adhesive rugae to bury itself and is unlikely to disturb or displace other infaunal species (Williams 2003).

*N. vectensis* has a formidable capacity to kill other macrozoobenthic organisms with its rapidly refillable neurotoxin reservoir and chemosensors that enhance the likelihood of prey capture, though some organisms captured will be too large for ingestion (Moran and Gurevitz 2006; Watson et al. 2009). Therefore, there is a chance that it has a disproportionately negative effect on vulnerable species due to its indiscriminate method of prey capture, however this is beyond the scope of this study. Organisms captured incidentally but not consumed by *N. vectensis* may become available to other scavengers or predators and the anemone may enhance food availability to other carnivorous species. Transverse fission of *N. vectensis* increases with food availability (Hand and Uhlinger 1995). A population explosion of this sessile predator in areas of high prey densities could lead to a local sudden and rapid decline in abundance of certain species, altering the assemblage dynamics and potentially negatively affecting higher trophic levels reliant on the same prey species. Microcosm experiments would be necessary to quantify the effects of this small but potentially formidable anemone on microspatial macrozoobenthic assemblages. However, further surveys should not focus on *N. vectensis* as a vulnerable protected species but as a non-native and understudied species.

Historically, Poole Harbour has previously hosted lagoonal specialists within the estuary itself (*C. glaucum*) and it is possible that some brackish areas of the main harbour may act as refugia to populations of lagoonal specialists such as *N. vectensis* (Herbert et al. 2010). Although no individuals have been found in saltmarshes in the UK to date, *N. vectensis* can be found in saltmarsh sites on coasts of North America (Sheader et al. 1997). There are sites in Poole Harbour where saltmarshes, wetland scrapes, and lagoon habitats can be found within the same area. As sea defences fail against sea level rise, agricultural land has become intertidal wetland habitat where waders, such as ruff and redshank, will frequent to roost and feed (Birds of Poole Harbour 2018). The presence of avian predators indicates sufficient colonisation of aquatic invertebrates and warrants further study; such sites include Lytchett Fields, the flooded agricultural land in northwest Lytchett Bay, and Arne Moors (personal communication). Another site of interest is Holton Pools, a wetland scrape created in saltmarsh habitat that exhibits physiographic lagoonal properties and supports several species of wader (personal observation).

# 9. Conclusion

This study surveyed the spatial distribution of macrozoobenthic assemblage, pelagic fauna and abiotic variables to determine which factors dictated distribution of species, abundance and whole communities. The starlet sea anemone *N. vectensis* received particular attention due to both its cryptic and protected statuses in the UK. There were no significant correlations with proximity to Sluice Gate and species richness, abundance, overall assemblages or any of the abiotic factors. However, correlations with salinity may occur temporally, particularly during and post flush events. Species richness and abundance was significantly correlated with OM Content, Silt Content and MGS. Overall assemblages were not significantly correlated with any factor, and richness and abundance were not correlated with pelagic fauna. Thus, sediment factors are the predominant variables in determining spatial distribution of species and their abundances. This is likely due to the hostile conditions associated with high OM Content and increased food availability associated with moderate OM Content.

The epifaunal starlet sea anemone is also significantly negatively correlated with sediment factors, likely due to the physiological stress associated with hypoxic conditions in high OM Content sediments. Its effect on native fauna is beyond the scope of this study but it is postulated that in high prey densities it could negatively affect certain species with its indiscriminate and relatively effective method of prey capture. Further research should focus on predator-prey relationships and competition of this cryptic cnidarian with native fauna.

Poole Park is currently a degraded habitat, but the creation of islands may reduce turbidity in the northeast and change areas of deposition. This study will serve as a comparative baseline for surveys post-works. It is an important site in the Poole Harbour regional lagoon network, particularly for piscivorous and roosting coastal birds, though currently has little value for wading species that are at risk of habitat loss. Sites such as Lytchett Fields and Holton Pools exhibit lagoonal properties and should be the subject of further study, in addition to Arne Moors which has the capacity to become lagoonal in the future.

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# Appendices

Table A1. Abiotic measurements from all sample sites.

| Sample Site | Silt Content (%) | Median Grain Size (μ) | Organic Matter (%LOI) | Water Depth (m) | Sediment Depth (m) | Salinity (‰) | Temperature (°C) |
|---|---|---|---|---|---|---|---|
| 1 | 76.0 | 48.1 | 9.7 | 0.7 | 0.5 | 26.6 | 2.5 |
| 2 | 89.3 | 20.6 | 10.8 | 0.7 | 0.7 | 26.5 | 2.5 |
| 3 | 92.8 | 20.4 | 11.5 | 0.8 | 2.0 | 26.6 | 2.5 |
| 4 | 90.9 | 19.0 | 10.9 | 0.9 | 2.0 | 26.6 | 2.5 |
| 5 | 65.6 | 62.3 | 5.2 | 0.9 | 1.9 | 24.7 | 1.5 |
| 6 | 78.6 | 36.4 | 5.7 | 0.7 | 0.1 | 26.6 | 2.5 |
| 7 | 93.5 | 20.5 | 10.5 | 0.8 | 0.5 | 26.6 | 2.6 |
| 8 | 88.4 | 19.5 | 12.3 | 0.9 | 1.4 | 26.5 | 2.6 |
| 9 | 90.4 | 18.3 | 10.6 | 0.9 | 1.3 | 26.6 | 2.6 |
| 10 | 88.1 | 20.6 | 11.0 | 1.0 | 1.3 | 26.6 | 2.6 |
| 11 | 80.7 | 27.2 | 8.7 | 0.8 | 1.0 | 26.2 | 2.7 |
| 12 | 59.1 | 66.5 | 3.4 | 0.7 | 0.6 | 26.4 | 2.6 |
| 13 | 87.6 | 22.3 | 8.6 | 0.9 | 0.7 | 24.8 | 1.3 |
| 14 | 88.6 | 21.2 | 9.0 | 0.9 | 0.4 | 26.5 | 2.6 |
| 15 | 89.6 | 20.2 | 9.3 | 0.9 | 1.4 | 26.5 | 2.7 |
| 16 | 87.0 | 22.2 | 10.7 | 1.0 | 1.3 | 26.6 | 2.6 |
| 17 | 91.1 | 20.3 | 9.2 | 1.0 | 1.2 | 26.6 | 2.5 |
| 18 | 87.6 | 25.1 | 8.9 | 0.9 | 0.8 | 26.3 | 2.9 |
| 19 | 59.8 | 64.2 | 3.1 | 1.0 | 0.5 | 26.2 | 2.6 |
| 20 | 61.3 | 65.0 | 4.7 | 0.9 | 0.2 | 26.3 | 3.1 |
| 21 | 49.4 | 128.2 | 3.5 | 0.9 | 0.1 | 26.2 | 3.1 |
| 22 | 68.4 | 75.7 | 7.0 | 0.9 | 0.6 | 26.5 | 2.9 |
| 23 | 87.3 | 23.3 | 10.5 | 0.9 | 1.4 | 26.5 | 2.8 |
| 24 | 89.1 | 22.8 | 11.6 | 1.0 | 1.3 | 26.6 | 2.6 |
| 25 | 91.0 | 21.6 | 8.9 | 1.0 | 1.3 | 26.5 | 2.6 |
| 26 | 84.0 | 27.2 | 9.2 | 1.0 | 0.3 | 26.4 | 3.0 |
| 27 | 63.7 | 60.5 | 6.3 | 0.9 | 0.1 | 26.2 | 2.7 |
| 28 | 62.9 | 61.3 | 5.3 | 0.9 | 0.2 | 26.1 | 2.9 |
| 29 | 74.3 | 33.8 | 7.3 | 0.8 | 0.2 | 26.6 | 3.0 |
| 30 | 71.7 | 42.8 | 4.2 | 0.9 | 0.5 | 26.5 | 2.8 |
| 31 | 86.9 | 25.4 | 7.2 | 1.3 | 0.5 | 26.5 | 2.7 |
| 32 | 95.4 | 20.4 | 9.4 | 1.0 | 0.9 | 26.5 | 2.9 |
| 33 | 83.7 | 25.8 | 12.1 | 1.1 | 0.2 | 26.5 | 2.9 |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| 34 | 47.1 | 137.7 | 3.5 | 0.8 | 0.2 | 26.2 | 2.7 |
| 36 | 68.3 | 51.3 | 3.7 | 0.7 | 0.2 | 26.5 | 2.9 |
| 37 | 81.6 | 29.6 | 7.0 | 1.1 | 0.4 | 26.5 | 2.9 |
| 38 | 60.6 | 71.7 | 6.2 | 1.0 | 0.1 | 26.5 | 2.9 |
| 39 | 86.6 | 32.3 | 10.3 | 1.1 | 0.2 | 26.6 | 3.1 |
| 40 | 77.5 | 52.1 | 10.6 | 1.2 | 0.3 | 26.6 | 3.1 |
| 41 | 54.6 | 103.1 | 3.7 | 0.7 | 0.1 | 26.5 | 3.3 |
| 42 | 28.1 | 204.2 | 1.2 | 0.6 | 0.2 | 24 | 3.1 |
| 43 | 30.3 | 230.4 | 1.5 | 1.0 | 0.1 | 26.5 | 2.9 |
| 44 | 62.5 | 72.7 | 5.5 | 1.0 | 0.3 | 26.5 | 3.0 |
| 45 | 82.2 | 40.0 | 9.8 | 1.2 | 0.2 | 26.6 | 3.2 |
| 46 | 73.4 | 56.5 | 6.3 | 1.1 | 0.2 | 26.5 | 3.4 |
| 47 | 36.3 | 182.9 | 2.3 | 0.8 | 0.3 | 26.5 | 3.5 |
| 48 | 60.6 | 95.3 | 5.4 | 0.9 | 0.5 | 26.4 | 3.6 |
| 49 | 71.4 | 72.2 | 7.5 | 0.6 | 0.1 | 26.4 | 3.7 |
| 50 | 18.7 | 186.4 | 1.4 | 0.6 | 0.1 | 26.4 | 3.7 |

Table A2. Abundance of pelagic fauna at all sample sites.

| Site | June | | | | July | | | | TOTAL |
|---|---|---|---|---|---|---|---|---|---|
| | ***C. maenas*** | ***P. varians*** | ***G. aculeatus*** | ***P. microps*** | ***c. maenas*** | ***P. varians*** | ***G. aculeatus*** | ***P. microps*** | |
| 1 | 0 | 40 | 0 | 3 | 0 | 45 | 0 | 2 | 90 |
| 2 | 0 | 20 | 0 | 7 | 0 | 5 | 0 | 5 | 37 |
| 3 | 0 | 20 | 0 | 9 | 0 | 40 | 0 | 0 | 69 |
| 4 | 0 | 20 | 0 | 1 | 0 | 2 | 0 | 1 | 24 |
| 5 | 0 | 15 | 0 | 10 | 0 | 30 | 0 | 3 | 58 |
| 6 | 0 | 5 | 0 | 9 | 0 | 10 | 0 | 2 | 26 |
| 7 | 1 | 10 | 0 | 25 | 0 | 10 | 0 | 20 | 66 |
| 8 | 2 | 20 | 0 | 30 | 0 | 20 | 0 | 0 | 72 |
| 9 | 0 | 10 | 0 | 7 | 0 | 10 | 0 | 0 | 27 |
| 10 | 0 | 10 | 0 | 10 | 1 | 20 | 0 | 10 | 51 |
| 11 | 1 | 25 | 0 | 10 | 3 | 0 | 0 | 30 | 69 |
| 12 | 1 | 20 | 0 | 10 | 0 | 10 | 0 | 18 | 59 |
| 13 | 3 | 5 | 0 | 30 | 3 | 15 | 0 | 15 | 71 |
| 14 | 0 | 10 | 0 | 15 | 1 | 5 | 0 | 25 | 56 |
| 15 | 2 | 5 | 0 | 15 | 0 | 10 | 0 | 40 | 72 |
| 16 | 0 | 5 | 0 | 10 | 0 | 20 | 0 | 10 | 45 |
| 17 | 1 | 10 | 0 | 10 | 0 | 20 | 0 | 5 | 46 |
| 18 | 0 | 1 | 0 | 3 | 0 | 16 | 1 | 11 | 32 |
| 19 | 0 | 5 | 0 | 5 | 0 | 5 | 3 | 25 | 43 |
| 20 | 0 | 5 | 0 | 10 | 0 | 0 | 2 | 6 | 23 |
| TOTAL | 11 | 261 | 0 | 229 | 8 | 293 | 6 | 228 | 1036 |

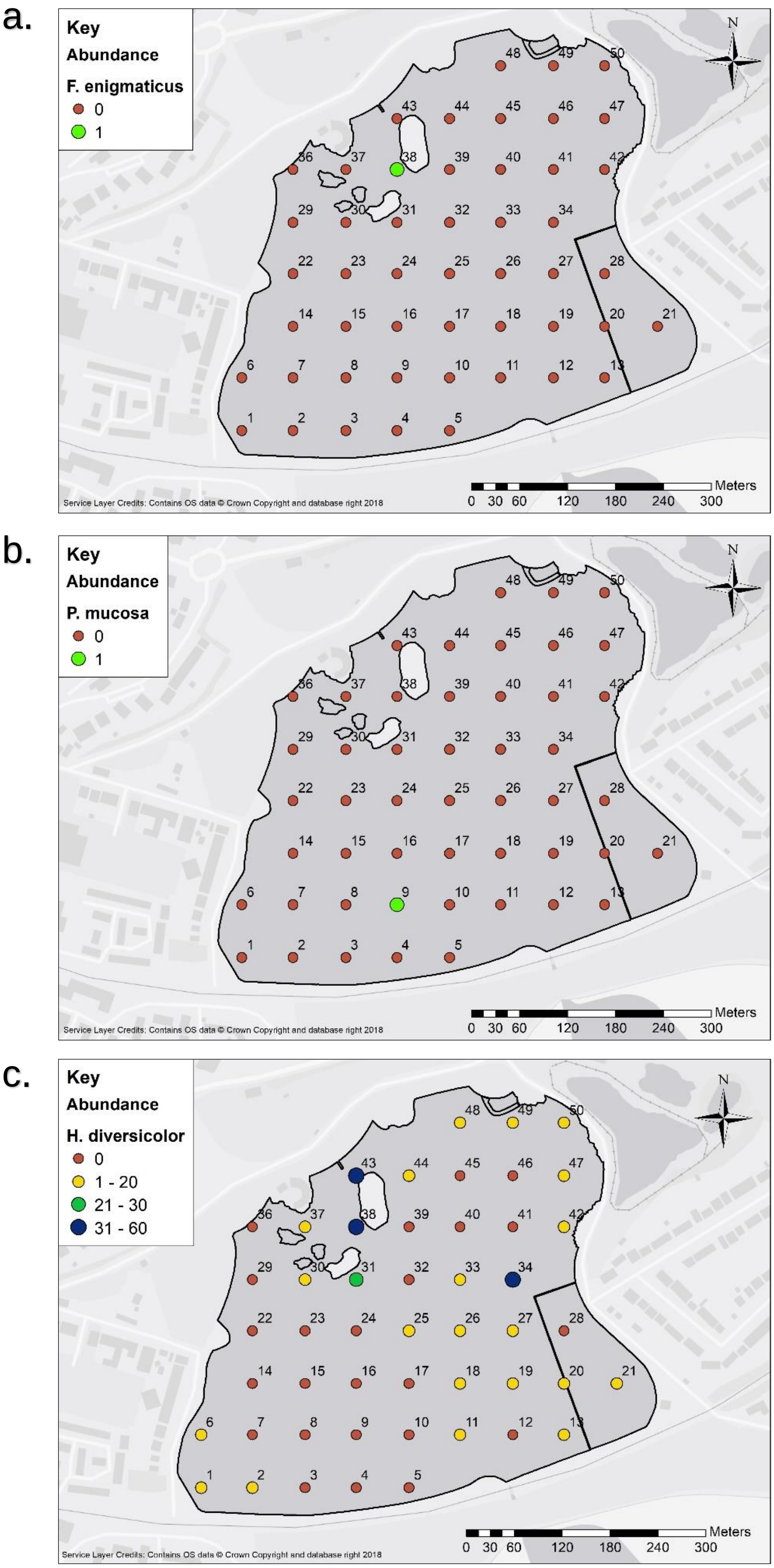

Figure A1 The spatial distribution of a) *F. enigmaticus,* b) *P. mucosa* and c) *H. diversicolor.*

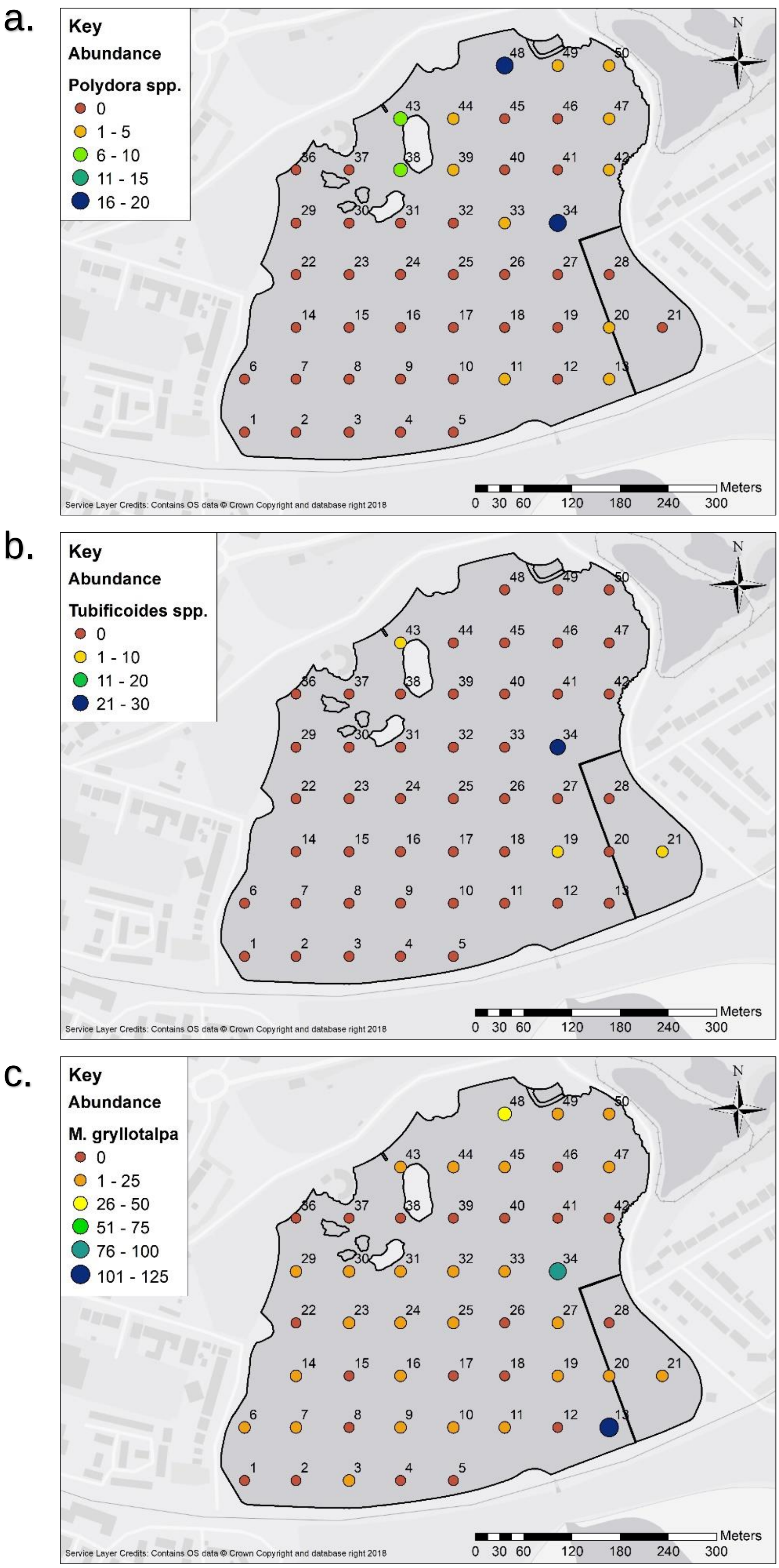

Figure A2 The spatial distribution of a) *Polydora* sp., b) *Tubificoides* sp. and c) *M. gryllotalpa.*

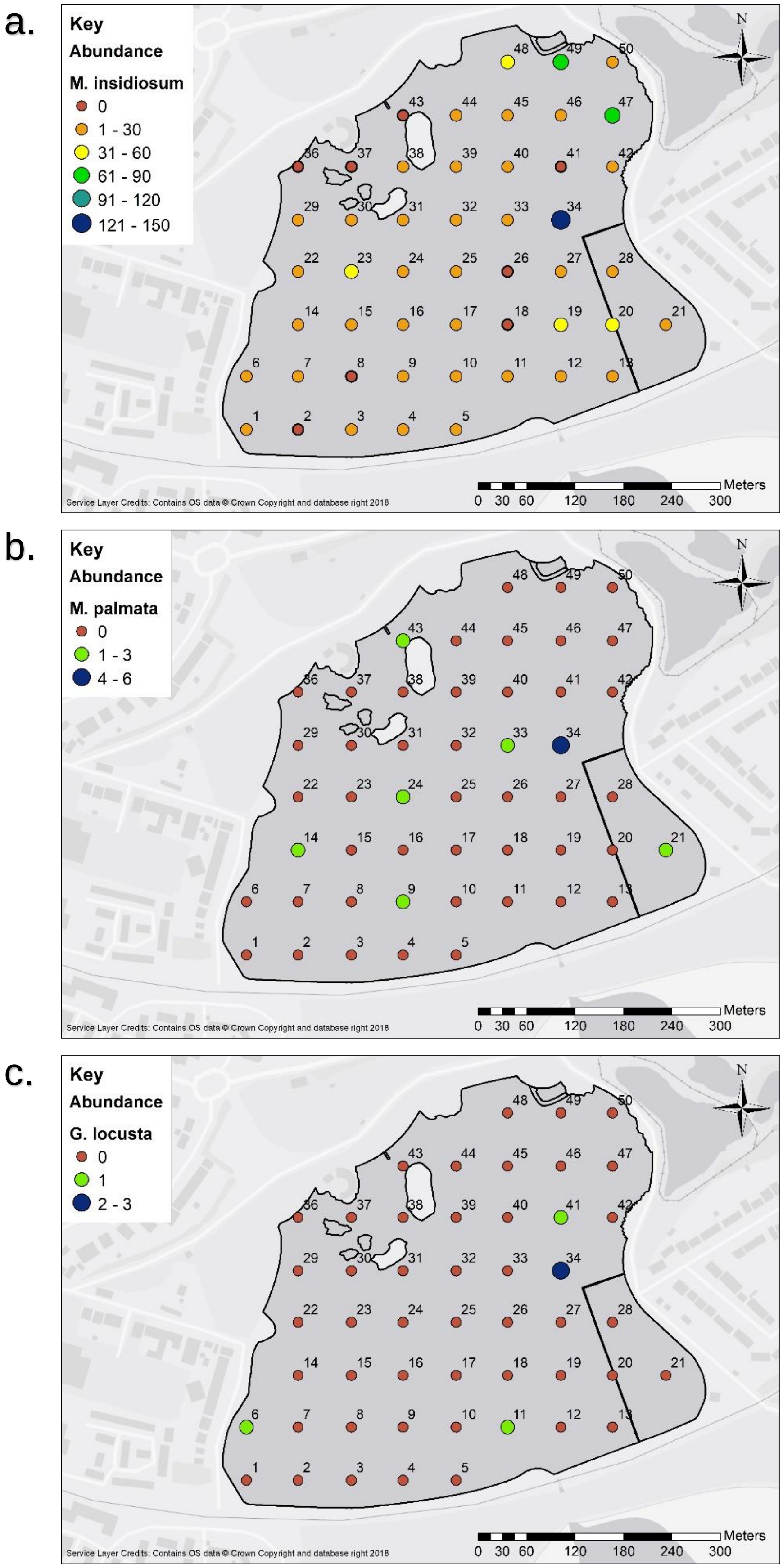

Figure A3 The spatial distribution of a) *M. insidiosum,* b) *M. palmata* and c) *G. locusta*

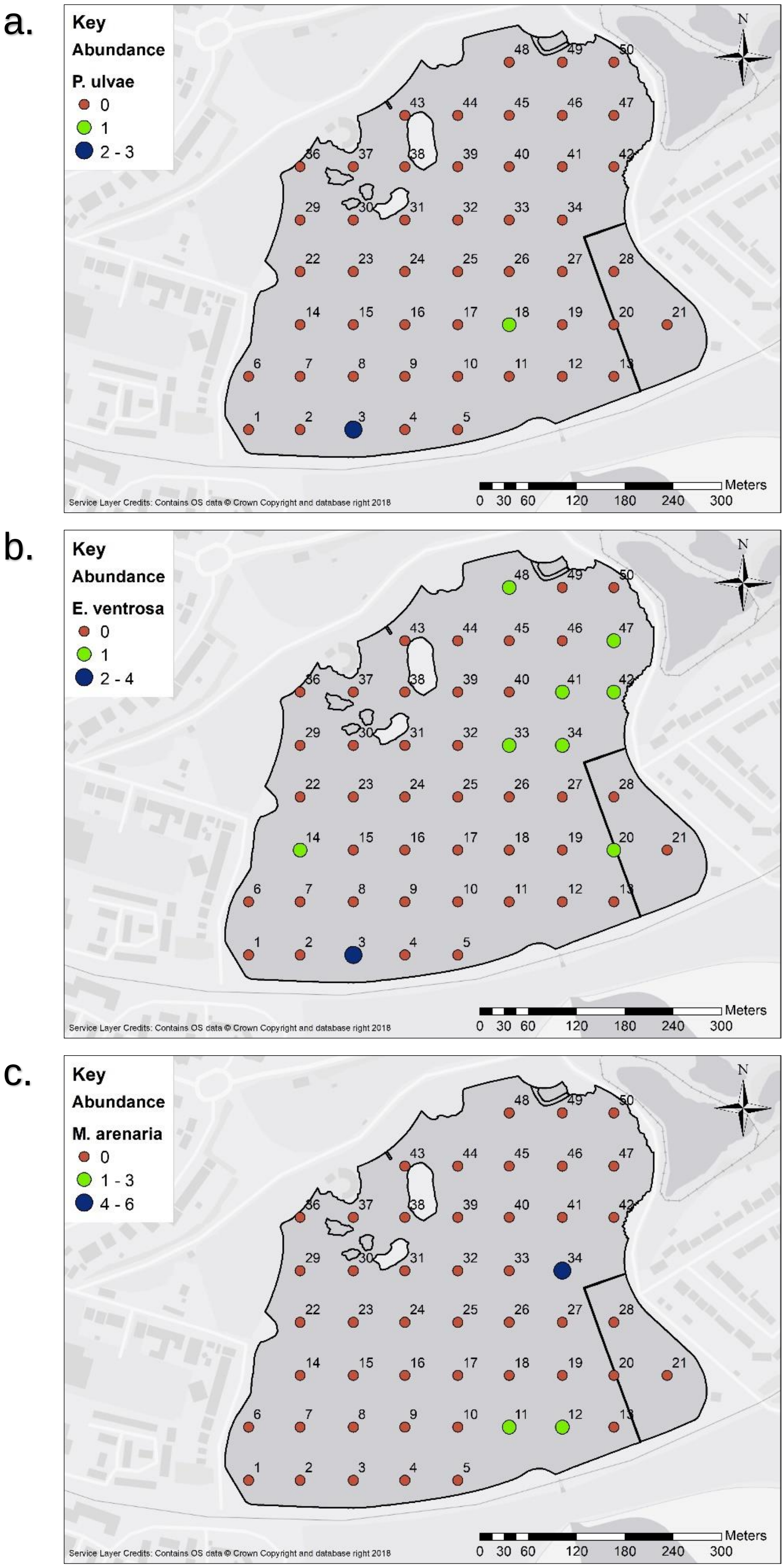

Figure A4 The spatial distribution of a) *P. ulvae,* b) *E. ventrosa* and c) *M. arenaria.*

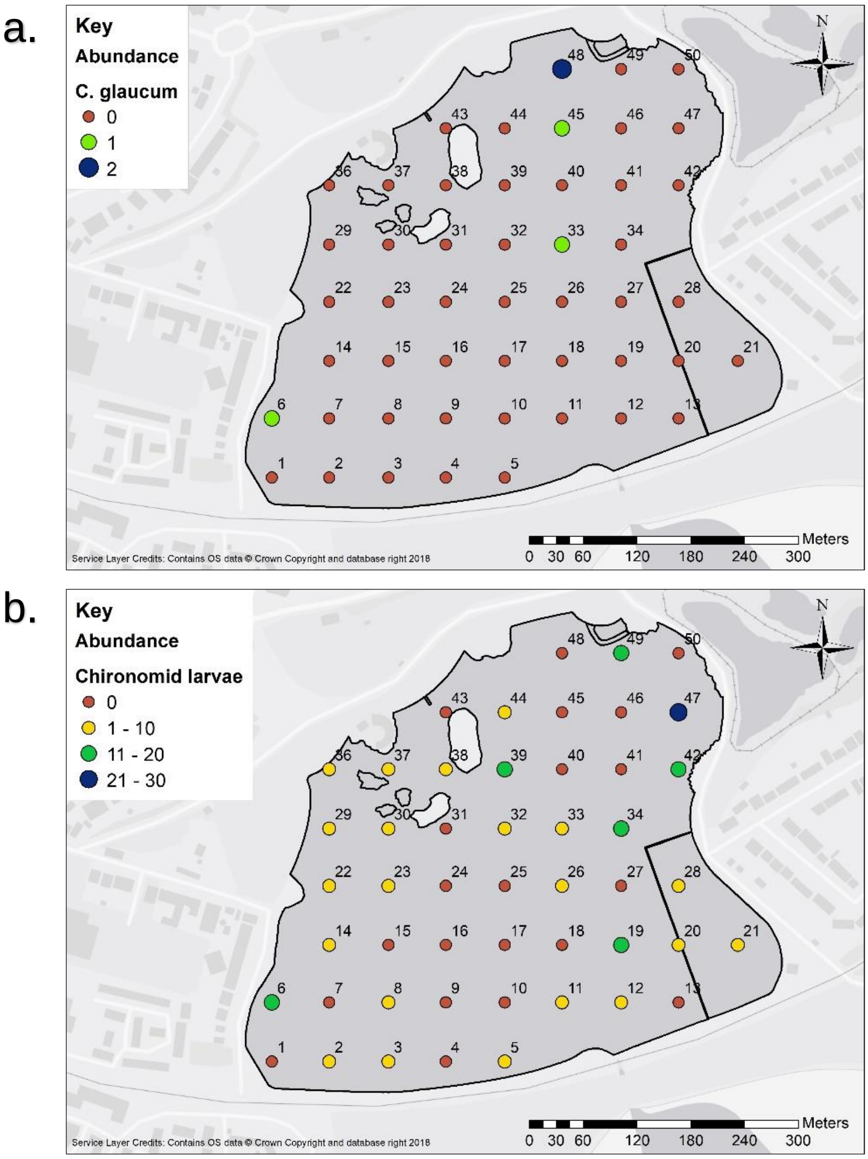

Figure A5 The spatial distribution of a) *C. glaucum* and b) Chironomid larvae.