Master's thesis in coastal ecology

Daniel Molnes Hole

Long term effects of activated carbon on benthic communities - 2021

Universitetet i Agder
Fakultet for teknologi og realfag
Institutt for naturvitenskaplige fag

Universitetet i Agder

Fakultet for teknologi og realfag

Institutt for naturvitenskaplige fag

Gimlemoen 25

4604 Kristiansand

http://www.uia.no

Sammendrag

Tynnsjikttildekking med aktivt kull har potensiale som metode for å utbedre felt forurenset med hydrofobe organiske forbindelser. Behandlingen ble utført på testfelt i Eidangerfjorden (95m) og i Ormerfjorden (30m). Disse to fjordene er en del av Grenlandfjord systemet som har en lang historie med forurenset bunnsediment. I denne studien ser man på langtidseffektene tynnsjiktstildekking med aktivt kull har hatt på samfunn av bentiske organismer ni år etter. Antallet av individer, arter og biomasse hadde blitt redusert betraktelig på begge testfelt sammenlignet med referanse-feltene. Forskjellene var spesielt merkbare i Ormerfjorden der den mest dominante arten, *Amphiura filiformis*, på referanse-feltet var helt fraværende på testfeltet. Etter 2010 var gruppen Echinoidea fraværende fra begge felt som var behandlet med aktivt kull. Indeksene som brukes til å undersøke den økologiske tilstanden i den Norske kystsonen med bakgrunn i vanndirektivet kunne ikke påvise den negative effekten som aktivt kull behandlingen hadde på de bentiske organismer. Langtidseffekten som aktivt kull kan ha på bentiske organismer må derfor blir nøye vurdert før det utføres en slik behandling i større skala. I tillegg er det behov for mer forsking for å forstå hvorfor behandlingen med aktivt kull påvirker bentiske organismer.

Abstract

Thin-layer capping with activated carbon (AC) has a potential as situ remediation method for sediments exposed to hydrophobic organic compounds (HOCs). This treatment was applied at two test fields; one in the Eidangerfjord (95 m) and the other in the Ormerfjord (30 m) in 2009. These two fjords are a part of the Grenland fjord system which have a long history of contaminated sediments. In this study we will address long-term effects of thin-layer capping with AC on the benthic community nine years later. Number of individuals, species and biomass was significantly reduced in both test fields compared to their corresponding reference fields. The differences were particularly notable in the Ormerfjord as the most dominant specie in the reference field, the brittle star *Amphiura filiformis,* was completely absent in the test field. After 2010 he faunal group Echinoidea was absent in both AC treated field. The indices used to assess the ecological condition in the water framework directive monitoring system for coastal waters in Norway did not reflect negative effects AC treatment had on the benthic communities. The long-term effects of AC on the benthic community should therefore be carefully evaluated before applying this treatment on a larger scale. More research is also needed to improve the understanding of why the AC treatment affects the benthic community.

Table of content

Preface

This has been an exciting and informative journey where I have done much and met many knowledgeable and nice people willing to share their knowledge and experiences. I got the chance to join the field sampling in the Grenland fjords October 2018 where we collected the samples used in this project. I have also been given training in species identification of benthic organisms and this was my favorite part of this project, and hope I get the opportunity to work with species identification in the future.

I would like to give a special thanks to my supervisor Hilde Cecilie Trannum at NIVA for coordinating and guiding me through this period. I would also like to give Rita Næss a special thanks for teaching me the ropes in the lab as well as teaching me the art of species identification. Thanks to Lise Ann Tveiten at NIVA for facilitating the lab for me, making sure I had everything I needed. I would also thank Juan Pardo at NIVA for being readily available for questions when I needed it. Otherwise, I would like to thank everyone at NIVA Grimstad for making the place a wonderful place to work at as well as allowing me to use their group rooms and laboratory. I would also take the opportunity to thank the crew on RV "Trygve Braarud" for allowing me to join as well as assisting during the field sampling. And finally, I would thank my parents Rune Molnes and Elin Hole which have supported me through this period as a student.

Grimstad, 18.05.2021

Hole, Daniel Molnes

1 Introduction

1.1 Benthos

1.1.1 Importance of benthic organisms in the marine environment

Marine sediments are the second largest habitat on earth, after the ocean water column and is home to a large part of the marine biodiversity (Gray and Elliott, 2009). The most common animals here are the Polychaete worms, other animal groups that can be frequently found are Bivalvia, Crustacea, Echinodermata, Gastropoda, nematodes (Gray and Elliott, 2009). These animals can be divided into subgroups based on size; microfauna (<63 µm), meiofauna (63 µm – 0.5 mm), macrofauna (0.5 mm - 5 cm) and megafauna (>5 cm) and of these groups macrofauna is the most studied (Gray and Elliott, 2009). The benthic fauna comprises mobile, sedentary, or sessile organisms, that can be found living on top of as well as within the sediments (Gray and Elliott, 2009). Soft bottom macroinvertebrates are invertebrates larger than 0.5 mm, they are mostly found living within the sediments (Gray and Elliott, 2009; Lardicci et al, 2004). They may live in permanent or semi-permanent tubes, or burrows, and depending on species and size they can penetrate several cm into the sediments (Berge et al., 2011).

The soft bottom macroinvertebrates feeds on detritus primarily from primary production, fecal pellets, animal carcasses, and other benthic organisms (Commito & Ambrose, 1985; Rygg, 1998; Snelgrove, 1998). By feeding on organic material that sinks to the bottom, they help convert plant material through secondary production (Snelgrove, 1998). This allows for the transfer of energy to other parts of the food chain if they become food for other animals, like fish, birds and mammals and other benthic organisms, and they also represent several trophic levels in the food chain (Hjelset et al., 1999; Barret et al., 2002; Pedersen et al., 2008; Gray & Elliott, 2009). There are several feeding strategies among soft bottom macroinvertebrates, two of the most common is suspension feeding[1] and deposit feeding[2]. Various species have certain

[1] Suspension feeding: Feeding on particles suspended in the water column

[2] Deposit feeding: Ingesting deposited particles, and sediment with organic material associated with these

preferences to particle types and sizes they ingest, while some others are relatively non-selective (Snelgrove, 1998). It will also vary how deep in the sediments deposit feeding takes place, this is also the case for how high up from the sediment suspension feeders get their particles (Snelgrove, 1998).

The reworking of the sediment that is caused by feeding and other activities like movement and burrowing is called bioturbation. Bioturbation can lead to sediment particles being moved vertically; this can result in subduction of organic matter (Lohrer et al., 2004). It also affects the sediment permeability and water content, this combined with the burrows may affect the water-sediment flux (Aller, 1988; Christensen et al., 2000; Graf & Rosenberg, 1997; Lohrer et al., 2004). The activity destabilizes chemical gradients in porewater and affects the availability of oxygen in the sediment, deeper burrows allow for deeper aerobic sediments (Kristensen, 2000; Lohrer et al., 2004) This also helps decomposition of dead organic matter by promoting the decomposition of it through enhancing microbial activities and growth rates (Kristensen, 2000; Lohrer et al., 2004; Gray & Elliott, 2009). This influences the rates of organic matter mineralization in the sediment and the fluxes of the inorganic nutrients back to the water column (Aller, 1988; Giblin et al., 1995; Riisgård et al., 1996; Graf & Rosenberg, 1997; Hansen & Kristensen, 1998; Christensen et al., 2000; Lohrer et al., 2004). These recycled nutrients can have a significant impact on the primary production, which in turn leads to more organic matter dropping to the sediments and more food for the benthic community (Giblin et al., 1995; Pilskaln et al., 1998; Welsh, 2003).

1.1.2 Disturbances of benthic communities

In ecology, disturbance can be defined as “any discrete event in time that disrupts ecosystem, community, or population structure and changes resources, substrate availability, or the physical environment” (White & Pickett, 1985). It has been shown that disturbances in a benthic area can lead to a varying degree of destruction or changes that makes the habitat unsuitable for some species, which will lead to changes in marine soft-bottom community structure (Johnson, 1970; Dauer & Simon, 1976; Thistle, 1981; Eckman, 1983; Probert, 1984; Hall et al., 1991; Thrush & Dayton, 2002; Gray & Elliott, 2009). For the benthic area disturbances can both be naturally occurring events and human activities, e.g. storms, wave movement, anoxia, red tides, fishing, organic pollution, oil pollution, dredging activities, depositing of sediment, and metal pollution (Dobbs & Vozarik, 1983; Ong & Krishnan, 1995; Yeo & Risk, 1979; Santos & Simon, 1980; Dauer & Simon, 1976; Grassle & Grassle, 1974; Pearson & Rosenberg, 1978; López-Jamar & Mejuto; 1988; Somerfield et al., 1995;

MacDonald et al ., 1996; Kaiser & Spencer, 1996; Boyd et al., 2000; Bolam & Rees, 2003; Trannum et al., 2004).

In instances where the disturbance results in the removal of species or a sufficient number of individuals, a process of secondary succession will commence (Pearson & Rosenberg, 1978; Britannica, T. Editors of Encyclopaedia, 2019). Examples of disturbances that can cause this are dredging activities, or depositing of sediment (López-Jamar & Mejuto, 1988; Somerfield et al., 1995; Boyd et al., 2000). The first species to recolonize after a disturbance are opportunistic species, where common traits in this group is having relatively small individuals, short life cycle, reproduce frequently, and high recruitment and death rates (Gray & Elliott, 2009). There will be overall few species with high numbers of individuals which are sedentary deposit feeders with shallow burrows, and the bioturbation they cause is usually limited to the upper layer of the sediment (Gray & Elliott 2009; Pearson & Rosenberg, 1978). If the disturbance is not continuous these species are replaced over time by better competitors which can rework the sediment at greater depths (Gray & Elliott, 2009). The better competitors are usually larger, have a slow development with a low death rate, longer-lived, and reproduce less frequently with planktonic larvae (Gray & Elliott, 2009). If all the benthic species are removed from an area during a disturbance the recolonization will only happen as horizontal migration by adult individuals from surrounding areas, or by larval recruitment from the water column (Bolam & Rees, 2003; Schratzberger et al., 2006). In cases where some adult individuals survive, the recovery time after a disturbance can be reduced as the adults can add their own offspring as well as rework the sediment which in turn can facilitate the colonization by other species (Thrush et al., 1992). In addition, the time it takes for a benthic community to recover after a disturbance usually increases with the size of the affected area (Zajac et al., 1998).

In scenarios where the sediments are contaminated the habitat can become less favorable for certain species (Gray & Elliott, 2009). Even if this kind of disturbance does not remove individuals initially and cause a secondary succession, it can apply a constant stress on the benthic organisms and induce changes in benthic communities over time (Pearson et al., 1983; Peeters et al., 2001; Hyland et al., 2003; Van Griethuysen et al., 2004). Some spices have a higher tolerance to certain types of contaminants and can live with them, other species may be more sensitive and can be reduced or disappear if concentration is high enough (Rygg, 1995; Guillaumot et al. 2018; Gray & Elliott, 2009). The benthic species composition will then change from sensitive to tolerant which usually leads to a reduction in biodiversity

(Rygg, 1995). In addition to changes in benthic community structures, bioaccumulation[3] can occur resulting in build-up of contaminates like HOCs (Gray & Elliott, 2009; Frid & Caswell, 2017; Tuomisto, 2019;). These contaminants can then be transferred through the food web and result in critical levels in higher trophic levels animals like birds, fish, seals and humans (Mitrou et al., 2001; Gray & Elliott, 2009; Frid & Caswell, 2017; Tuomisto, 2019).

1.1.3 Use of benthic organisms in environmental surveillance

The various species within the benthic community react differently to disturbances and contaminants (Rygg, 1995; Rygg, 2002; Gray & Elliott 2009; Rygg & Norling, 2013). As described previously this can lead to reduction or elimination of certain species, giving space for new species to occupy the sediments thus changing the species composition. In addition some of the benthic species are long-lived and many have low mobility, hence the species composition can then describe the condition of the benthic environment to a large degree (Gray & Elliott 2009; Direktoratsgruppa vanndirektivet, 2018). Therefore the benthic species composition can be used as an indicator on stress caused by various forms of disturbance and is used to estimate the environmental condition in several monitoring programs (Ros & Cardell., 1991; Rygg, 1995; Direktoratsgruppa vanndirektivet, 2018).

1.2 The Water Framework Directive

The Water Framework Directive (WDF) (Council Directive, 2000) is an EU directive for the management of all bodies of water in Europe. The directive is incorporated into the EEA agreement and is therefore binding for Norway as well. The main goal of the directive is to provide a framework for the determination of environmental goals that gives the most all-around protection of the water environment, and sustainable use for all bodies of water. This directive was implemented into the Norwegian national legislation in 2006. For every natural body of water, the environmental goal is to have “good” ecological and chemical conditions within the year 2021. This is checked by using a classification system in Norway. In the classification system all the water bodies will be evaluated for its ecological and chemical condition.(Direktoratsgruppa vanndirektivet, 2018)

[3] Bioaccumulation: Build-up over time of a chemical in a living creature

1.2.1 Ecological condition

The ecological condition includes five states of quality: “Very good”, “good”, “moderate”, “bad”, and “very bad”. “Very good” state represents an ecological condition which has no human impact. This is often referred to as the reference condition. Further decrease in quality states reflect the increasing deviation from the reference condition; the “Good” state reflects little deviation, the “Moderate” state reflects moderate deviation, the “Bad” state reflects a significant deviation, and finally “Very Bad” reflects a very large deviation from the natural condition.(Direktoratsgruppa vanndirektivet, 2018)

1.2.1.1 Biological quality elements

For the different water categories, the Water Framework Directive has specified which biological quality element to assess. The biological quality elements used to classify coastal waters in Norway are benthic fauna, macroalgae, eelgrass, and phytoplankton. Each of the biological quality elements have specific parameters and indices used to assess the body of water. Some of these parameters and indices only needs measuring and others need calculation as well. The location of the body of water will decide how the values are interpreted. In order to assess which quality state a biological quality element gets, the different parameters and index values are transformed into a ratio between 0 to 1, where 1 is representing the reference condition. This ratio is called the ecological quality ratio (EQR). The ratio is then normalized (nEQR) so it can be combined into an average value for a benthic quality element which decides the quality state. The quality states for the different biological quality elements are then compared and the worst quality state will then be used to decide the final ecological condition. (Direktoratsgruppa vanndirektivet, 2018)

1.2.1.2 Procedure to assess ecological condition

If the worst assessed quality status for the biological quality elements indicates “moderate”, “bad”, or “very bad” condition, then the ecological condition is set to this quality status and there is no need to use the supporting quality elements. However, if the biological quality elements suggest a “very good” or “good” ecological condition then the supporting quality elements have to be evaluated. Supportive physical-chemical parameters can downgrade the ecological condition to “good” or “moderate”, while the supportive hydro-morphological parameters can only downgrade from “very good” to “good”. Description of the supporting quality elements is described here. (Direktoratsgruppa Vanndirektivet, 2018)

1.2.2 Assessment of a body of water

If the ecological condition is “very good” or “good” and the chemical condition described in Direktoratsgruppa vanndirektivet (2018) is “good”, the environmental goal for that body of water is achieved. In this scenario the goal now is not to worsen the condition, "very good" should not become "good". If the goal is not achieved and/or there is a chance for it not to reach it by 2021, then measures must be implemented to achieve the environmental goal. These measures will be surveyed after implementation to see if it has achieved the desired effect. (Direktoratsgruppa vanndirektivet, 2018)

1.3 Contaminated sediments

Part of the contaminants released into the coastal marine environment tend to end up on the bottom. Contaminants like hydro organic compounds (HOCs), like environmental toxins, often have hydrophobic characteristics with an extraordinary ability to bind to particles, and in calmer hydrodynamic areas the particles deposit into polluted bottom sediments (Næs et al., 2004), where it can pose as long-term reservoirs due to these compounds being persistent. However, this is not the final resting place for these contaminants. Polluted sediments may end up being a secondary source of pollution after the primary sources has stopped polluting (Larsson, 1985). Contaminants absorbed to sediment normally develop an equilibrium with the dissolved fraction in the pore water and in the overlying surface water to be taken up by fish and other aquatic organisms. Contaminated sediments have been shown induce changes in benthic communities and thus pose a risk to aquatic sediments (Pearson et al., 1983; Peeters et al., 2001; Hyland et al., 2003; Van Griethuysen et al., 2004).

Dioxins is part of the HOCs group which also include polychlorinated biphenyls (PCBs) and many pesticides. Dioxins is a term including structurally related chemical groups such as polychlorinated dibenzo-p-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs), and many other chemicals (Tuomisto, 2019). Dioxins can be produced in nature through volcanoes and forest fires, but mainly it is by-products of industrial processes like smelting, bleaching of paper, and manufacture of pesticide (Frid & Caswell, 2017). Most of these compounds are persistent and not easily degraded by microbes, therefore they tend to accumulate in the environment (Tuomisto, 2019). Dioxins are much more soluble in lipids than in water and will easily accumulate in lipid or fatty tissues in animals (Frid & Caswell, 2017; Tuomisto, 2019). It’s also not easily metabolized by organisms and little is excreted through urine, therefore the concentration of dioxins in animals tends to build up over time causing bioaccumulation (Frid & Caswell, 2017; Tuomisto, 2019). The slow elimination of these compounds in nature will allow dioxins to accumulate trough the food chain and cause severe problems for animals high in the trophic levels, like seabirds and seals, and it could pose a risk to human health (Mitrou et al., 2001; Frid & Caswell, 2017; Tuomisto, 2019). In European and Norwegian legislation, dioxins and dioxin like compounds are classified as priority substances with low environmental regulatory limits (Vannforskriften, 2006; Council Directive, 2013).

1.3.1 Strategies of dealing with contaminated sediments

There are already several established strategies when dealing with contaminated sediments, these consist generally of dredging, conventional capping and monitored natural recovery (Förstner & Apitz, 2007; Perelo, 2010). There are however challenges when applying these strategies. Dredging is both expensive and time-consuming, it needs to be deposited somewhere, it resuspend contaminated sediment into the water column, and removes the benthic organisms (USEPA 2005; Ghosh et al., 2011; Fathollahzadeh et al., 2015). With conventional capping it is hard to guarantee that contaminates remain isolated in the long term, as several types of disturbances can affect the cap isolation ability (USEPA 2005; Ghosh et al. 2011). In addition conventional capping can be expensive, changes in sediment bathymetry may be unacceptable, and buries benthic organisms to a degree where it is hard or impossible to remerge, and if they are able to remerge they may bring contaminated material back to the sediment surface (Stronkhorst, 2003; Ghosh et al., 2011). Both of these strategies are highly disruptive for the benthic communities, and benthic species will need to recolonize the area which may take a long time (Ghosh et al., 2011). The benthic community are a major food source for other organisms in the ocean, and a long recovery time could have negative effects on commercial species like bottom feeding fish (Duineveld & Van Noort, 1986; Bolam & Rees, 2003) Monitored natural recovery involves leaving the contaminated sediment in place and let natural processes like sedimentation and biological and chemical processes deal with it, but this takes more time and are less predictable as contaminated sediment can easily spread with trough disturbances (USEPA 2005; Perelo 2010). The increased time can lead to contaminants posing a long-term health risk to humans and wildlife through bioaccumulation in the food web (USEPA 2005).

Due to the challenges with some of these strategies, other methods using thin-layer capping (1-10 cm) with active sorbents have been explored (Perelo 2010; Ghosh et al., 2011; Choi et al., 2016). Compared with conventional capping, thin-layer capping is cheaper, less disruptive to the soft-bottom fauna as some individuals can migrate vertically after capping making recolonization take shorter time, and it causes less changes in sediment bathymetry making it viable in more areas (Maurer et al., 1981; Maurer et al., 1982; Essink, 1999; Schratzberger et al., 2006; Wilber et al., 2007; Ghosh et al., 2011). The reason why thin-layer capping by itself is not being considered is due to its poor ability to isolate the contaminated sediments, both small disturbances and bioturbation may bring buried contaminants to the sediment surface (Thibodeaux & Bierman, 2003; USEPA 2005; Josefsson et al., 2010).

1.3.2 Thin-layer capping with activated carbon

Activated carbon (AC) is one of the active sorbents being explored with thin-layer capping (Ghosh et al., 2011; Choi et al., 2016). The reason is the ability of carbonaceous particles to attract and accumulate hydrophobic organic compounds (HOCs) (Ghosh et al., 2000; Ghosh et al., 2003). Carbon particles comes in different forms like coal, charcoal, soot, humic matter, and decayed remains of plants and animals, however these different forms have different sorption capacities for HOCs (Grathwohl, 1990; Karapanagioti et al., 2000; Salloum et al., 2002). The type and concentration of carbon found in sediments indicates how well HOCs is absorbed in the sediments and how much can be released to surrounding water and organisms.

Activated carbon is not found naturally and needs to be synthesized trough activation, where the material is filled with small pores that increase its surface area (Marsh & Rodríguez-Reinoso, 2006). This increased surface area allows for extremely high sorption capacities compared to other types of carbonaceous particles (Walters & Luthy, 1984; Luthy et al., 1997; Ghosh et al., 2003; Cornelissen et al., 2005; Marsh & Rodríguez-Reinoso, 2006). AC will effectively bind HOCs and thereby reducing its bioavailability to benthic organisms and its release into the water column (Rust et al., 2004; Zimmerman et al., 2004, 2005; Millward et al.2005; Cho et al., 2007, 2009; McLeod et al., 2008; Beckingham & Ghosh, 2011; Cornelissen et al., 2011, 2012; Josefsson et al., 2012; Kupryianchyk et al., 2013; Lin et al., 2014; Patmont et al., 2014; Samuelsson et al., 2015;). This binding effect will also bind the contaminants that can emerge trough the thin-layer cap trough disturbances and bioturbation (Thibodeaux & Bierman, 2003; USEPA 2005; Sun & Ghosh., 2007; Lin et al., 2014; Josefsson et al., 2010;). Newly deposited contaminated sediment will also get treated as the bioturbation and other natural processes mixes the sediment layers (Sun & Ghosh., 2007); Lin et al., 2014).

Thin-layer capping with AC has been suggested to be a less harmful method compared to the dredging and conventional capping on the benthic fauna (Ghosh et al., 2011), but so far there has been mixed results. Some studies have found little to no negative effects on the benthic fauna (Rakowska et al., 2012; Janssen & Beckingham, 2013). In a review with a collection of 82 tests, one-fifth of them found impacts to benthic organisms which resulted from AC exposure (Janssen & Beckingham, 2013). Other studies have observed negative impacts on some species like decrease in growth (Millward et al., 2005; Janssen et al., 2012; Nybom et al., 2012, 2015), survival (Kupryianchyk et al., 2011), lipid content (Jonker et al., 2004; Rust et al., 2004; Janssen et al. 2012; Nybom et al., 2012), and changes in behavior (Jonker et al., 2004; Nybom et al., 2012, 2015), reproduction (Nybom et al., 2012, 2015), and morphology (Nybom et al. 2015). Some studies on the benthic community level have also different results. One freshwater study reported an initial perturbation after exposure to AC which was one year later followed by recolonization and recovery (Kupryianchyk et al., 2012). Another freshwater study found no negative effects on the benthic community (Beckingham et al., 2013). In Trondheim Harbor, Norway they showed a decrease in both the number of species and their abundance for a marine benthic community one year after capping with powdered AC (Cornelissen et al., 2011).

1.4 Grenland

The Grenland fjords in southeast Norway the sediments have elevated concentrations of dioxins and mercury stemming from past industrial activities (Knutzen et al., 2003). The emission has ceased, but the contaminants is still an issue as it can be released from the sediments (Larsson, 1985; Fagerli et al., 2016). The fjord system did not have “good” chemical and ecological conditions even in 2015 and therefore do not met the standard set in the Norwegian water directive (Fagerli et al., 2016). Saloranta et al. (2008) modelled that treatment of the most contaminated areas (“hot spots”) has little effect compared to treating a larger area which covers a significant portion of the contaminated sediment. However, due to the large size that would need to be treated dredging is not feasible and capping with a thick enough layer is expensive and buries the benthic organism on the location. A large pilot study was set up in 2009 to test the effects and feasibility of thin-layer capping with various materials, one of them being thin-layer capping with powdered AC (Schaanning et al., 2011). One month, one year and four year after capping, the effects on benthic fauna and contaminant fluxes from the sediment was investigated (Cornelissen et al., 2012, 2016; Samuelsson et al., 2017; Raymond et al., 2020). The most recent investigation in this pilot study was carried out nine years after capping where Schaanning et al. (2021) reported on the effects of thin-layer capping with AC on contaminant fluxes.

1.5 Aim of this study

This thesis will address the long-term effects of thin-layer capping with AC on marine benthic macrofaunal communities nine years after capping. Findings from the previous investigations in this pilot study will be used to look at trends over the years. Central questions are how the test and reference field in both fjords differ with regards to restitution-rate, and if there are species or faunal groups which are particularly sensitive. This thesis will also look at the ability to assess potential effects of thin-layer capping with AC by using the biodiversity indices currently used to assess the benthic quality element in Norway as well as Pielou's index of evenness (J'). Species richness, total abundance, biomass as well as selected biodiversity indices from 2018 will be subject to statistical testing to identify the long-term effects of AC on benthic organisms. Further, multivariate statistics will be used to assess how the community composition has developed through time and how the fjords differ.

2 Method

2.1 Description of site

This project were performed in the Grenland fjords which are located in south-east Norway and consists of several smaller fjords (Figure 1).

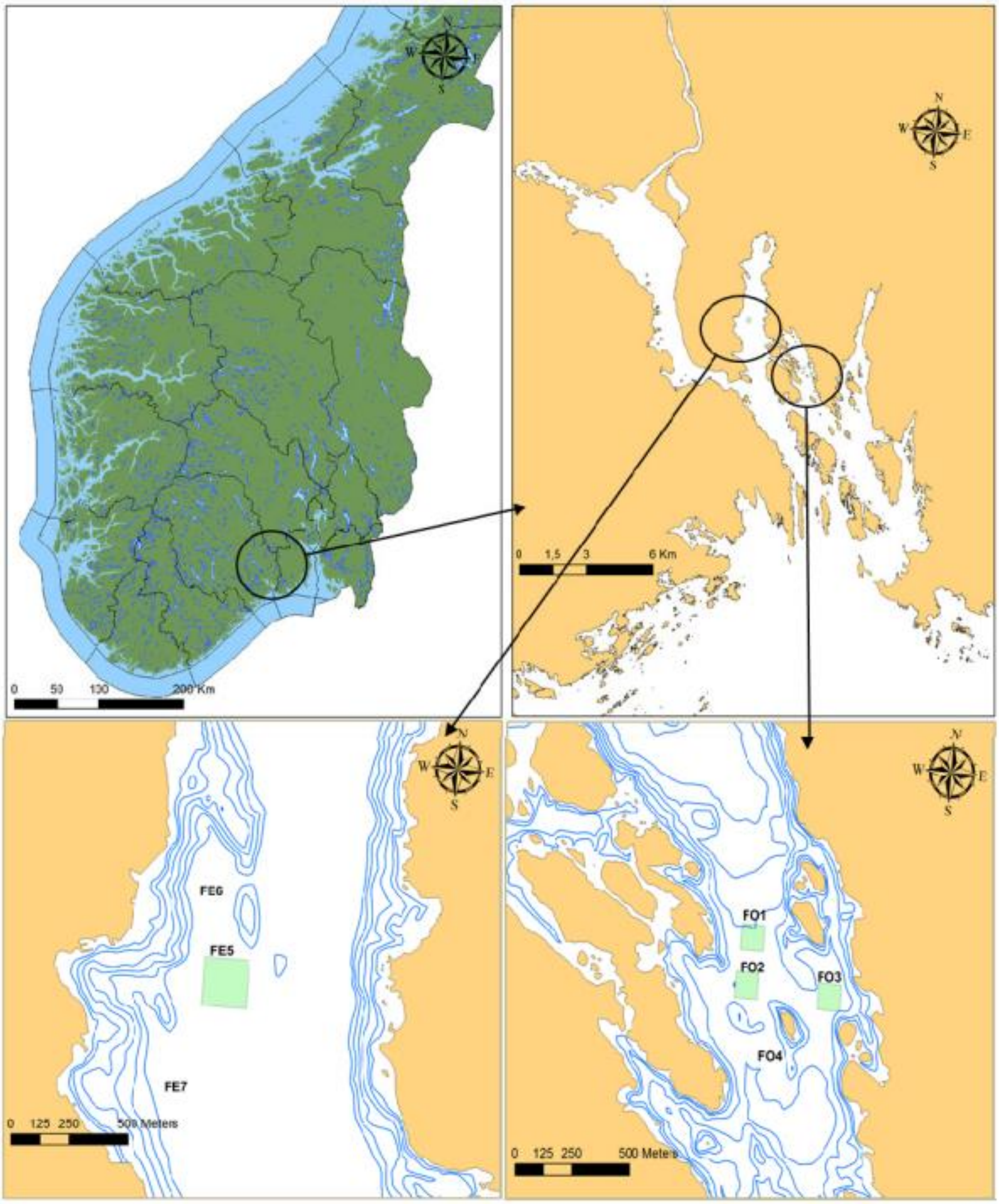

Figure 1: Combined map with Norway, the Grenland fjord system, and the test locations in the Eidangerfjord (low left) and the Ormerfjord (low right). Figure from Schaanning et al. (2011)

The northern and innermost part of the fjord system is mainly the Frierfjord basin with a depth of 98 m at the deepest (Alve, 2000). At the north-east end the Frierfjord receives a dominant runoff from the Skien river with an average of 270 m^3/sec annually, resulting in a brackish surface water layer of usually 3-6 m (Molvær, 1980). The Frierfjord is further connected to the outermost fjord system across a narrow sill of 23 m depth where dense out-flowing fjord water is exchanged with less dense coastal water in-flowing from the seaward fjord system (Molvær, 1980; Alve, 2000). The outer fjord system is further separated by a sill at 55 m depth from the Skagerak sea at the seaward end (Molvær, 1999; Samuelsson et al., 2017). The Eidangerfjord and the Ormerfjord are two of the branches in this outer fjord system, and this is where the test fields were established (Schaanning et al., 2011). The Eidangerfjord in the northern part of the outer fjord system situates the deepest locality for the test fields at about 80-95 m depth and has an accumulation type of bottom with (Samuelsson et al., 2017). The Ormerfjord is located adjacently south-east of Eidangerfjord where the test fields are located at 30 m depth, the seabed environment can be characterized as a transport bottom (Samuelsson et al., 2017). The test locations at 80-95m have 1-2mm aged and compacted sediment, while the test location at 30 m had 0.5mm, hence approximately three times more sedimented material is received by the deeper locality compared to the shallower location (Samuelsson et al., 2017).

2.2 History of Grenland

For centuries, the Frierfjord has received material from a growing industrialization along the Skien river. Initially from water-driven sawmills, later from the pulp and paper industries (Alve, 2000).

One of the major sources of pollution was Norsk Hydro magnesium processing plant at Herøya starting in 1951 which released dioxins and other chlorinated organics contaminations as by-products into the Frierfjord and caused high concentrations of dioxins in the ecosystem, also in the neighboring branches in the fjord system (Bradshaw et al., 2012).

The dioxins are by-products originating from the production of water free magnesium which involves several high temperature processes comprising carbon, chlorine and a catalyst, a treatment that brought 95% of the formed PCD/PCDD to the water phase, and further emitted into the innermost part of the Frierfjord using seawater scrubbers (Knutzen & Oehme, 1989; Oehme et al., 1989; Ruus et al., 2006). From the wastewater the magnesium factory enriched the sediments in the fjords with contaminations like; mercury (Hg), persistent organic pollutants (POPs) being polychlorinated dibenzofu- rans/dibenzo-p-dioxins (PCDF/PCDD), octachlorostyrene (OCS), hexachlorobenzene (HCB), while the sediments was contaminated by polychlorinated biphenyls (PCBs) from other activities in the area (Knutzen et al. 2003; Raymond et al., 2020).

The industry together with shipping and other human activities contaminated the sediments of the fjords with several hydrophobic organic contaminants, including furans and dioxins, where the main source contributed at the most with 12 kg PCDD/F-TEQ/year from the magnesium processing plant on Herøya by the Frierfjord between 1951 and 2002 (Trannum et al., 2021).

Restrictions and improved effluent treatment reduced the contaminant discharge during the mid-1970s and late 1980s, but high contaminant concentrations have remained in water, sediment and biota (Persson et al., 2002; Knutzen et al. 2003; Schlabach et al., 1998 as cited in Bradshaw et al 2012). Also after the main source of contamination ceased in 2002 by closing the magnesium factory, the accumulated dioxins from the entire period in the fjord sediments are regarded as a significant source of environmental pollution in the Grenland fjords (Schaanning et al., 2019).

Fjords are in effect sedimentation basins and by 1978, the Frierfjord was known as one of the most polluted fjords in Norway (Skei, 1978, 1981). Researching on Hg from a local chlor-alkali plant, it was found a two- to three-fold increase within the Frierfjord compared to the coastal water outside the fjord, indicating that the fjord pollution mainly was a local problem in the source area because of spontaneous sedimentation of a pollutant (Skei, 1981).

Related to environmental toxins in organisms, the Grenland fjord system is clearly the most researched in Norway, and this effort has provided the government a good scientific foundation for dietary advice, something that the top numbers of reassessments has confirmed (Økland, 2005). Condition assessment of the fjord areas and environmental toxins in fish and shellfish has been progressing since the early 1970s. Around 1990 the industry largely limited the emissions, which caused a notable reduction of environmental toxins in fish and shellfish. However the content level of environmental toxins, in particular dioxins, are still considered too high to lift the restrictions on dietary advice governed by the Norwegian Food Safety Authority (Ruus et al., 2013). The most recent official dietary advice, written in 2013, revised in 2019, still discourages consumption of fish and shellfish from the Grenland fjords (Norwegian Food Safety Authority, 2019).

2.3 Test fields and stations

In September 2009, the test sites were situated in the Ormerfjord and the Eidangerfjord. In the Ormerfjord equally sized fields at 10 000 m^2 were established at a depth of ca 30 m; FO1, FO2, FO3, and field FO4 for reference. Furthermore two 40 000 m^2 fields were established in the Eidangerfjord at the depths of typically 95m and 85m; FE5 for testing and FE6 for reference (Eek et al., 2011). Since the benthic community can change from year to year a reference field is needed in order to interpret how the treatments impacts the benthic community. To avoid distortion from trawling in the Eidangerfjord, the reference field FE6 was situated at a slightly shallower depth than the test field FE5. But then after the initial establishment, the trawling in the area ceased and an alternative reference field FE7 was introduced in 2010 in a more comparable water depth at 95 m (Samuelsson et al., 2017). Since no major differences between FE6 and FE7 was found by Raymond et al. (2020), the reference field FE6 at 80 m is considered sufficient.

The fields used in this study is FO4 and FE6 which are untreated fields used as reference, as well as FO3 and FE5 with 2 kg/m^2 AC amended to sediments from the same nearby location (Eek et al., 2011; Schaanning et al., 2011). The stations are all placed in the water region Skagerrak and have the water type Protected coast/fjord (S3) in the water directive (Shaanning et al., 2021). For the marine clay supply, PCDD/F extraction and analysis was used to ensure that non contaminated marine clay (1.5 × 106 kg dry weight (d.w.); bulk density 1.64 ± 0.02 kg L−1 (n = 10), water content 38−41%; total organic carbon (TOC) content 1.8%) could be extracted from 10−400 cm deep layers in the inner part of the Ormerfjord using a suction dredger (Cornelissen et al., 2012).

The AC that was amended by a ratio of 1:10 d.w./d.w. to the clay, had the properties of Jacobi Carbons, PB2 fine powdered, average particle size 20 µm, where 80% was smaller than 45 µm (Cornelissen et al., 2012; Trannum et al., 2021). To provide a sufficient density for the slurry comprising the marine clay and the AC, the salinity had to be increased by adding 1 kg NaCl per 40 kg clay d.w. (Cornelissen et al., 2012). The cap thickness measured one month after the deployment was for the treated fields; 11±6 mm at the FO3 with dredged clay and AC at 30 m, and 12±3 mm at the FE5 with dredged clay and AC at 95 m (Eek et al., 2011). Cornelissen et al. (2012) found that the final AC concentration in the treated fields FO3 and FE5 was 2% dry weight of sediments measured after nine months (Samuelsson et al., 2017).

With focus on the benthic macrofauna there have been four surveys in the Eidangerfjord and the Ormerfjord collecting samples using the same type of van Veen grab with sampling area

0.1 m^2. Happening 1 month, 14 months, 49 months and 110 months after capping, each with 3, 5, 5 and 4 replicate grab samples per field (Schaanning et al., 2019). More details on the tree proceeding missions can be found in Schaanning et al. (2014).

2.4 Field work

October 2018, four samples of benthic macrofauna were collected at each location using a van Veen grab (0.1 m^2), only grabs with a chamber volume 19 dm^3 where accepted. Details of each sample is shown in Table 1. Using seawater, the samples were sieved through 1 mm meshes, where visible specimens were manually collected using forceps to make the handling gentlest possible. A buffered solution of 10-20% formaldehyde stained with Rose Bengal was used to conserve the target residue in seawater, with an additional buffering of borax (20 g equivalent to one tablespoon). The samples were then stored for more than three months before the lab analysis began. In addition to the samples of the benthic macrofauna sediment cores were sampled with a Gemini-corer to find sediment fine fraction, TOC and total nitrogen (TN) in the top layer (0-1 cm). Water temperature and salinity were measured between 12.1-14.9 °C and 33.2-33.6 in the Ormerfjord, and between 7.0-12.0 °C and 34.0-34.6 in the Eidangerfjord.

2.5 Lab analysis

In the lab the benthic macrofauna samples were washed and put on fresh water for 24 hours to remove as much formaldehyde as possible. This prosses were done in a fume hood using gloves, lab coat and glasses.

The material in the samples were then sorted into the faunal groups Polychaeta, Bivalvia, Gastropoda, Crustacea, Ophiuroidea, and Echinoidea. The organisms not associated with any of these groups were put into Varia. These groups where then preserved on 80% ethanol for later analysis.

After sorting the fauna, it was identified to the lowest possible taxonomic level. The group Bivalvia, Gastropoda, Ophiuroidea, and Echinoidea were identified by Daniel M Hole. The species identification in these groups were then controlled by Rita Næss at NIVA Grimstad. The faunal group Polychaeta and Varia were identified by Rita Næss, and the group Crustacea were sent to a lab in Oslo to be identified by Marijana Stenrud Brkljacic.

Biomass was measured using wet weight (w.w.) for each species or lowest possible taxonomic level. Before the measurement was taken the individuals were put in fresh water then quickly dried using a filter paper. The tubes from tube building Polychaeta were removed and liquid inside sea urchins were drained prior to weighing. The individuals were then put in a pre-weighed container and weighed on a scale with a sensitivity of 0.0001 g.

After removal of inorganic carbon by acidification, TOC and TN were determined using carbon, hydrogen, and nitrogen analyses. Sediment fine fraction (% particles < 0.063 mm) was determined by wet sieving (Trannum et al, 2021).

Table 1. Geographical positions (WGS84 Decimal Degrees) and depths (m) for grab sampling per field station in Grenland 2018. Modified from Schaanning et al. (2019)

Latitude	Longitude	Field	Station	Depth	Date
59.07787	9.702787	FE6 Referanse	A	83	24.10.2018
59.07806	9.702621	FE6 Reference	B	82	24.10.2018
59.07825	9.702839	FE6 Reference	C	81	24.10.2018
59.07844	9.702869	FE6 Reference	D (E)	80	24.10.2018
59.07569	9.704359	FE5 AC	A	96	24.10.2018
59.07518	9.703392	FE5 AC	C	96	24.10.2018
59.07468	9.704189	FE5 AC	B (E)	96	24.10.2018
59.07475	9.702947	FE5 AC	D	95	24.10.2018
59.05666	9.7554	FO3 AC	D	25	24.10.2018
59.05636	9.755811	FO3 AC	A	26	24.10.2018
59.05636	9.755285	FO3 AC	B	26	24.10.2018
59.05626	9.754804	FO3 AC	C	27	24.10.2018
59.05366	9.751155	FO4 Reference	E	30	23.10.2018
59.053741	9.751053	FO4 Reference	B	31	23.10.2018
59.053696	9.751275	FO4 Reference	C	30.7	23.10.2018
59.053566	9.751506	FO4 Reference	D	30	23.10.2018

2.6 Data and analysis

All data was put inn Microsoft Excel for Windows and simple calculations was done here. Creation of the figures was done in RStudio using the packages “ggplot2” and “vegan”.

Data treatment and statistical analysis was done in:

-RStudio (using the package “vegan” and “car”)

-NIVAs programs for calculating some indices and nEQR

2.6.1 Bray-Curtis dissimilarity index

Dissimilarity measures are frequently used by ecologists between pairs of sites (Ricotta & Podani, 2017). Bray-Curtis dissimilarity index is useful especially in multivariate-analysis of large datasets to calculate how different two sites are with respect to their composition of species. By counting the numbers of different species representing each site, a ratio between the count of common species present at both sites to the total number of species at both sites indicates how different the sites are on a scale between 0 and 1 where 0 is identical and 1 is dissimilar (Quinn & Keough, 2002). The Bray-Curtis dissimilarity index in this thesis was calculated after the data had been transformed for fourth-root.

Formula for Bray-Curtis dissimilarity index:

$$Sim_{ab} = \left(\frac{\sum_{i=1}^{S} |x_{ia} - x_{ib}|}{\sum_{i=1}^{S} (x_{ia} + x_{ib})} \right)$$

X_{ia}= number of individuals of the ith species in location a, X_{ib}= number of individuals of the ith species in location b, S= total number of species.

2.6.2 Cluster analysis

A cluster analysis was performed on the samples taken in 2018 and another was preformed using the average value from the samples taken at the four locations over the years. The Bray-Curtis dissimilarity index was used to determine the similarities between samples which is then used to group then in a hierarchy pattern. The aim of cluster analysis is to find “natural groupings” of samples where each sample belonging to a group is more similar to other samples in the same group than to samples in different groups. By also applying hierarchical methods, the groups are arranged relative to other groups by the level of similarity or dissimilarity into a resulting dendrogram. In ecological work the cluster analysis is suited to show composition of species for different sites or for samples from the same site at different times. (Clarke & Warwick, 2001)

RStudio with the package “vegan” were used to calculate the Bray-Curtis dissimilarity index and make the dendrograms.

2.6.3 Non-metric multidimensional scaling (nMDS)

A non-metric MDS-ordination was performed for each fjord using all samples collected at the four locations over the years. The Bray-Curtis dissimilarity index was used to determine the similarities between samples which the analysis uses to plot a two-dimensional map with points representing the samples. The distance between points shows the degree of similarity, the closer two points are the more similar the samples are in respect to their species composition. A stress value is also calculated to give an indication to how well the points fit in the coordination system. The stress value will get a value between 0 and 1, where 0 is no stress between the points meaning they fit perfectly (Clarke & Warwick, 2001). Stress values under 0.1 is preferred, but values under 0.2 is considered good.

RStudio with the package “vegan” were used to calculate the Bray-Curtis dissimilarity index and make the nMDS plot.

2.6.4 Indicies and benthic quality element

There are five indices associated with the bentic quality element: the diversity indices Shannon-Wiener’s diversity index (H’) and Hurlbert’s diversity index (ES_{100}), the sensitivity indices Norwegian Sensitivity Index (NSI_{2012}) and Indicator Species Index (ISI_{2012}), the Norwegian Quality Index (NQI1) which is using both species diversity and sensitivity. These indices were calculated for each sample if possible, then in order to calculate the indices for each field the average value from the samples is used. These five indices will be used to find the quality state of the benthic quality element for the different locations with a process explained in Direktoratsgruppa Vanndirektivet (2018).

Shannon-Wiener’s diversity index (H’) (Shannon & Weaver, 1963)
Shannon-Wiener’s diversity index (H’) is used to describes the species diversity. The index uses number of individuals and species, and how the number of individuals is divided among the species. However, the species identity is not used when calculating the index. The index value increases as the number of species goes up, and the more even the individuals are spread among them. High values for this index are usually a good sign, and a value of 3.3 or up is required for the water type S3 to reach the class “good” or better.

Formula for Shannon-Wiener’s index (H’):

$$H' = -\sum_{i=1}^{S} p_i log_2 p_i$$

pi= ni/N, ni = number of individuals of the ith species, N = total number of individuals, S = total number of species.

Hurlbert's diversity index (ES_n) (Hurlbert, 1971)

Hurlbert's diversity index (ES_n) is another index used to describe the species diversity. This index calculates the expected number of species for n individuals, n cannot exceed the number of individuals that exist in the sample. The index uses number of individuals and species, and how the number of individuals is divided among the species. The species identities are not accounted for in this index either. A high value means there are expected to be many species in each sample which is looked at as a positive. A sample needs to have at least 100 individuals (ES_{100}) in order to use this index as a parameter to find the benthic quality element. A value of 20 or up is required for the water type S3 to reach the class "good" or better with this index.

Formula for Hurlbert's diversity index (ES_n):

$$ES_n = \sum_{i=1}^{S} \left(1 - \frac{\binom{N-N_i}{n}}{\binom{N}{n}}\right)$$

N = total number of individuals, Ni = number of individuals of the ith species, S = total number of species.

Norwegian Sensitivity Index (NSI_{2012}) (Rygg & Norling, 2013)

Norwegian Sensitivity Index (NSI_{2012}) is an index used to classifying the condition of an area using sensitivity values for several species. This index has been developed using data from the Norwegian fauna as basis. A total of 591 species have been assigned a sensitivity value. A value of 20 or up is required for the water type S3 to reach the class "good" or better with this index.

Formula for Norwegian Sensitivity Index (NSI_{2012}):

$$NSI = \sum_{i}^{S} \left(\frac{N_i * NSI_i}{N_{NSI}}\right)$$

Ni = total number of individuals of the ith species, NSI_i = NSI-value for species i (sensitivity score), N_{NSI} = number of individuals with a NSI_i value assigned to them.

Indicator Species Index (ISI_{2012}) (Rygg & Norling, 2013)

Indicator Species Index (ISI_{2012}) is an index used to classifying the condition of an area using sensitivity values for several species. This index only used the presence of species in order to calculate the index value. A value of 7.6 or up is required for the water type S3 to reach the class "good" or better with this index.

Formula for Indicator Species Index (ISI_{2012}):

$$ISI = \sum_{i}^{S} \left(\frac{ISI_i}{S_{ISI}}\right)$$

ISI_i = ISI-value for species i (sensitivity score), S_{ISI} = number of species present with a ISI_i value assigned to them.

Norwegian Quality Index (NQI1) (Rygg, 2006)

Norwegian Quality Index (NQI1) is an index using both species diversity and sensitivity values for several species. A value of 0.63 or up is required for the water type S3 to reach the class "good" or better with this index.

Formula for Indicator Species Index (ISI_{2012}):

$$NQI1 = \left[0.5 * \left(1 - \left(\frac{\mathrm{AMBI}}{7}\right)\right) + 0.5 * \left(\frac{\mathrm{SN}}{2.7}\right) * \left(\frac{\mathrm{N}}{\mathrm{N} + 5}\right)\right]$$

AMBI is an sensitivity index, SN is diversity indices, N = total number of individuals

Pielou's evenness index (J') was calculated as well to get a measurement of how even the number of individuals were distributed among the species. This index is not incorporated into the benthic quality element.

Pielou's evenness index (J') (Pielou, 1966)

Pielou's evenness index (J') is calculated using the Shannon-Wiener's diversity index (H') and is often presented together with it. Unlike H', Pielou's evenness index show only how the individuals is distributed among the species. The species identities are not accounted for in this index. Here the value calculated will differ between 1 to 0, were high values means individuals are equally distributed between the species, and low values means there are many individuals in some of the species and few in others.

Formula for Pielou's evenness index (J'):

$$J' = \frac{H'}{H'_{max}}$$

H' = Shannon-Wiener's index, H'max = highest possible value H' can get and is calculated as:

$$H'_{max} = -\sum_{i=1}^{S}\left(\frac{1}{S}\right)log_2\left(\frac{1}{S}\right) = log_2 S$$

S = total number of species.

H', ES_{100}, and J' indices was calculated using RStudio. NQI1, ISI_{2012}, NSI_{2012} was calculated by NIVA. NIVA was also responsible for transforming and normalizing the biological quality element indices to normalized ecological quality ratio (nEQR).

2.6.5 Statistical analysis

The difference in number of individuals, species, biomass and the calculated indices between the AC treated field and reference field in both the Eidangerfjord and the Ormerfjord were statistically tested using one-way ANOVA. In order to run this test, the data needs to be normally distributed, and the variance needs to be homogeneous across the groups. The normality was assessed using visual inspection and the Shapiro-Wilk test. The homogeneity of variances was assessed using Levene's test from the "car" package in R. Data not satisfying these assumptions were transformed using logarithm.

All the statistical analysis was done in RStudio.

3 Results

3.1 Community structure

In 2018 a total number of 2875 individuals and 74 species were collected from the four fields. The Eidangerfjord had both a higher number of individuals and species compared to the Ormerfjord. Average number of species were 37.5 and 50.25 in the Eidangerfjord and 11 and 21.5 per 0.1 m^2 in the Ormerfjord. Average number of individuals were 241.5 and 369.75 in the Eidangerfjord and 26.75 and 80.75 per 0.1 m^2 in the Ormerfjord. The biomass however showed no difference between the fjords, the Eidangerfjord had an average biomass of 2.91 and 10.8, and the Ormerfjord had 1.54 and 13.4 (g.w.w.) per 0.1 m^2. The number of individuals, species and biomass was significant lower in the AC treated field compared to the corresponding reference field in both fjords (Table 8). The difference was particularly big in the Ormerfjord. The fields within each fjord were more similar to one another than to the fields in the other fjord (Fig. 6). The reference and AC treated field in the Eidangerfjord were more similar to one another compared to the Ormerfjord. The group Echinoidea is also absent in the AC treated fields in both fjords. The J index had a significantly higher value in both AC treated fields compared to their corresponding reference fields (Table 8). The J values for the AC treated field in the Ormerfjord were particularly high at 0.86 indicating an even distribution of individuals among the species (Table 2).

In the Eidangerfjord the group Polychaeta dominated the number of individuals and species in both fields (Figs. 2, 3). The list over the most common species also shows the Polychaeta group is well represented in these locations and that the Ploychaeta *Spiophanes kroyeri* was the most dominant representing 32.6 % of the individuals in the reference field (FE6) and 22.6 % in the test field (FE5) (Table 5). *Spiophanes kroyeri* had twice the number of individuals in the reference field compared to the AC treated field, but the biomass per individual where much higher in the AC treated field (Table 6). The Polychaeta *Chaetozone setosa* on the other hand had a five times higher number of individuals in the AC treated field (Table 5). The list over the most common species also shows both fields share many of the same species. The proportion of different groups in number of individuals and species, looks very similar in both locations. The same is the case in biomass when excluding the group Echinoidea. The biomass was over three times higher in the reference field and two times higher if the most dominant biomass group in the reference field, Echinoidea is excluded (Figs. 4, 5). In the AC treated field, the group Polychaeta dominated the biomass. The overall state of the benthic quality element was classified as “good” for both fields, which is acceptable according to the water directive (Table 3). The H’, ES_{100}, NSI_{2012} and NQI1 indices all showed “good” and ISI_{2012} index got “very good” for both the refence field and AC treated field (Table 2). NSI_{2012}, NQI1 and ISI_{2012} indices were significantly higher in the reference field (Table 8).

In the Ormerfjord the group Ophiuridea dominated the number of individuals in the reference field (Figs. 2, 4). The main cause of this is *Amphiura filiformis* which made up 52.9% of the individuals found here (Table 4). However, in the AC treated field this species was completely absent. The group Echinoidea dominated the biomass in the reference field followed by the group Ophiuroidea (Fig. 4). The group Polychaeta had the highest number of species (Fig. 3). The second most common species in the reference field was *Hyala vitrea* making up only 6.2%. The AC treated field had a very low number of individuals, species and biomass compared to any of the other fields sampled in 2018. Biomass had the biggest difference between the reference field and AC treated field, where the reference field had more than eight times the biomass. The number of individuals and species was more than three times and almost twice as high in the reference field as well. The most common species was the Bivalve *Nucula nitidosa* which made up 26.2% of the individuals here with only seven individuals on average per sample. The Polychaeta *Nephtys incisa* and Gastropod *Hyala vitrea* was the second most common species making up 15.9% of the individuals here each. Both the reference field and the AC treated field were classified as “good” for the benthic quality element, making them acceptable according to the water directive as well (Table 3). Neither field had enough individuals to calculate the ES_{100} index. The NSI_{2012} and NQI1 index showed “good” and the H’ index showed “moderate” in both fields in the Ormerfjord (Table 2). The ISI_{2012} index got “very good” in the reference field and “good” in the AC treated field, and was the only index putting the two fields into different classes. However, there were no significant difference between the values in ISI_{2012} despite the different classifications (Table 8). The other indices used to classify the benthic quality element were not significant ether. The differences in number of individuals and species between the two fields were not clear when looking at the indices.

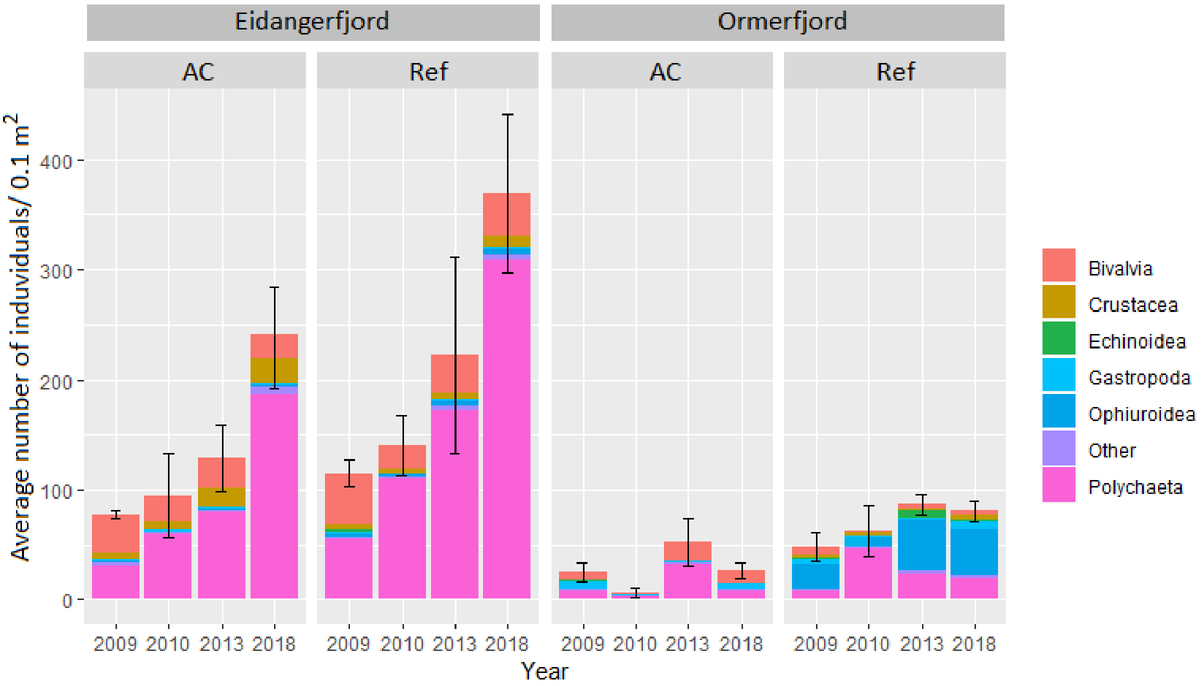

Figure 2: Average number of individuals per 0.1m^2 (± sd) in the Eidangerfjord and the Ormerfjord year 2009, 2010, 2013 and 2018, bars split into faunal groups.

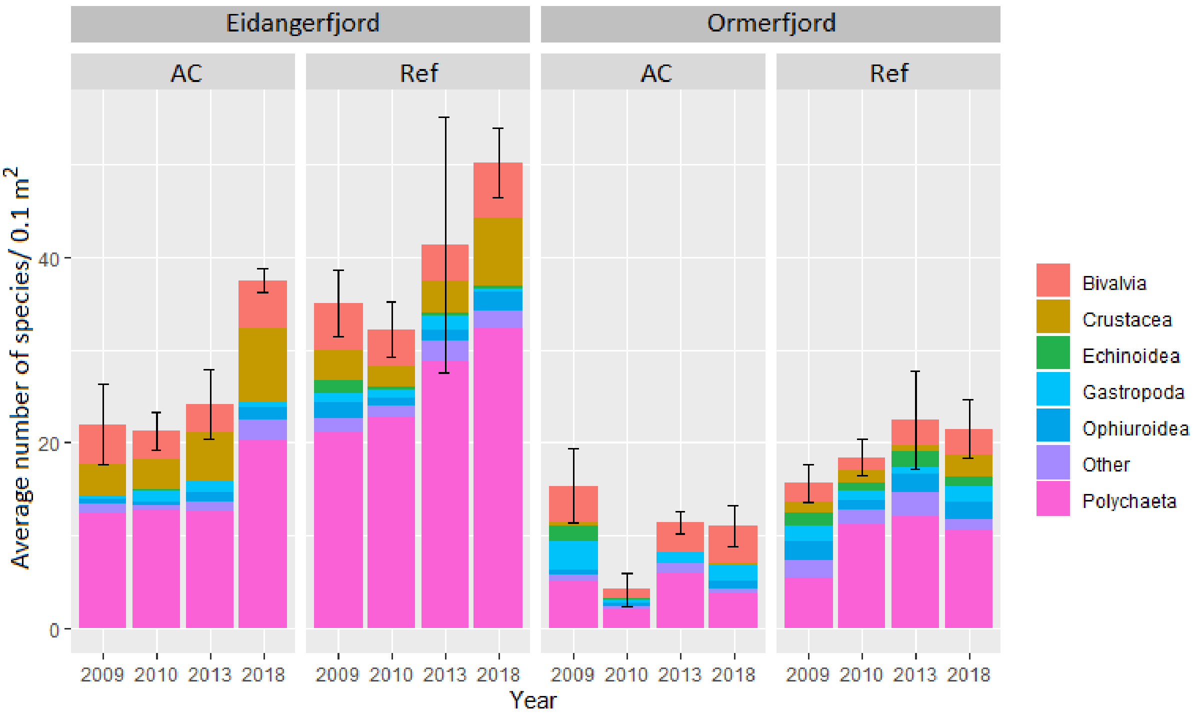

Figure 3: Average number of species per 0.1m^2 (± sd) in the Eidangerfjord and the Ormerfjord year 2009, 2010, 2013 and 2018, bars split into faunal groups.

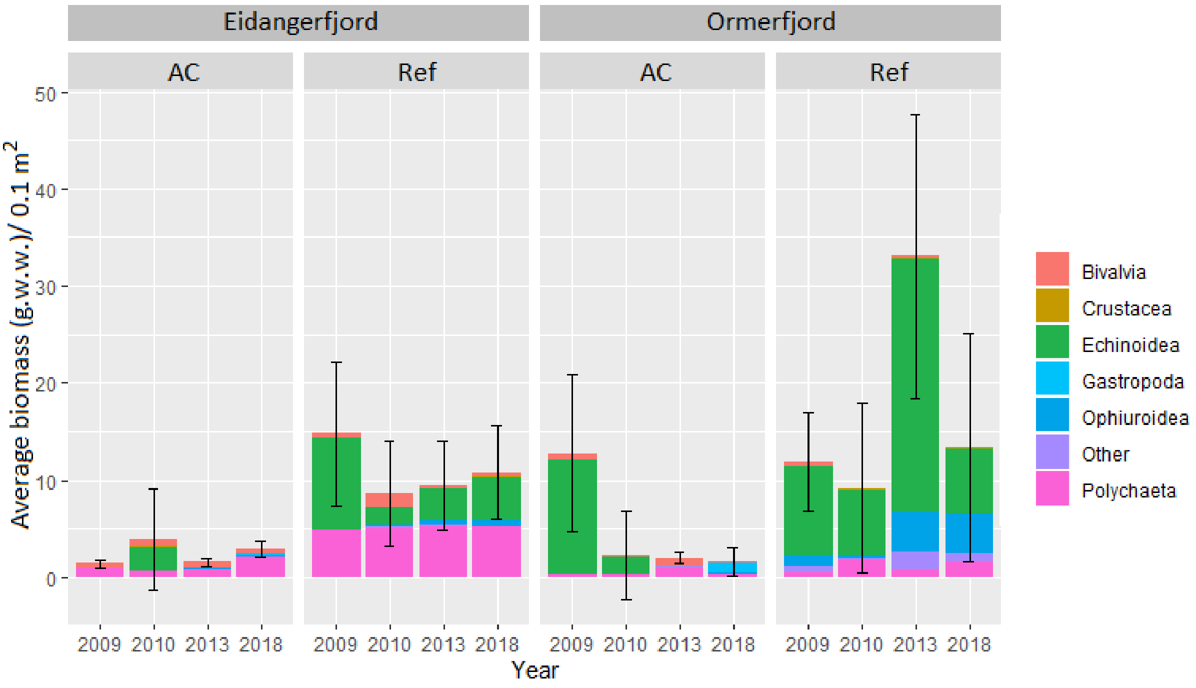

Figure 4: Average total macrofauna biomass (g.w.w.) per 0.1m^2 (± sd) in the Eidangerfjord and the Ormerfjord year 2009, 2010, 2013 and 2018, bars split into faunal groups.

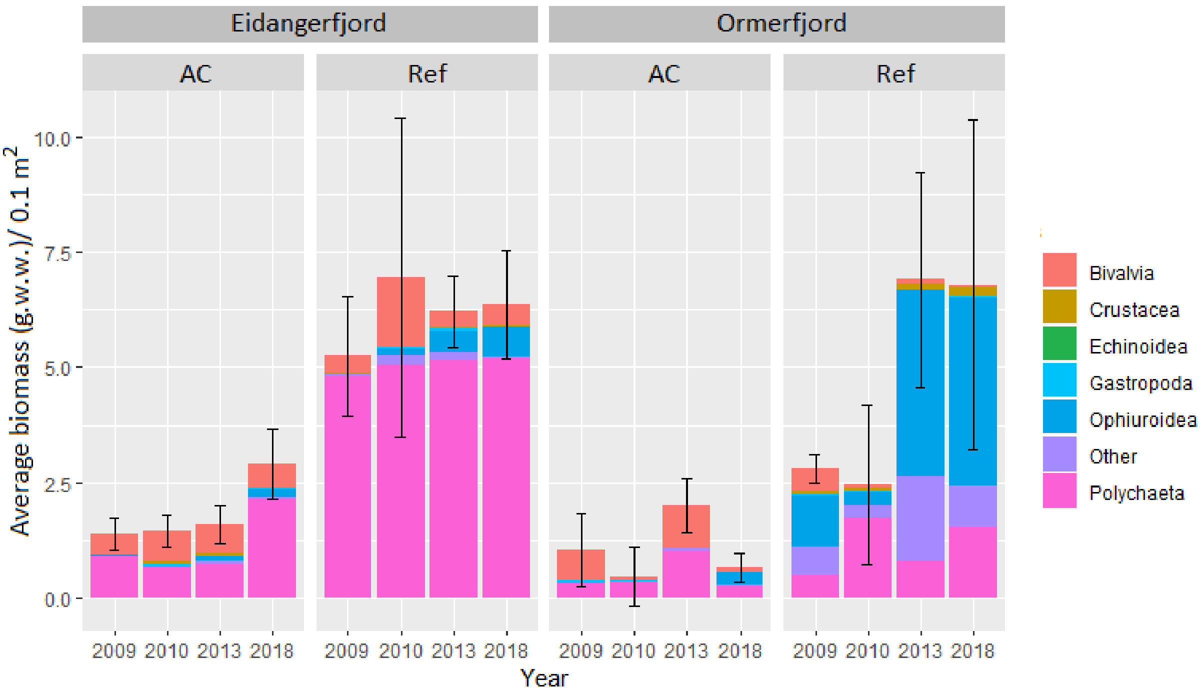

Figure 5: Average total macrofauna biomass (g.w.w.) per 0.1m^2 (± sd) in the Eidangerfjord and the Ormerfjord year 2009, 2010, 2013 and 2018, bars split into faunal groups. Group Echinoidea and one individual (Aporrhais pespelecani) from the Gastropoda group removed.

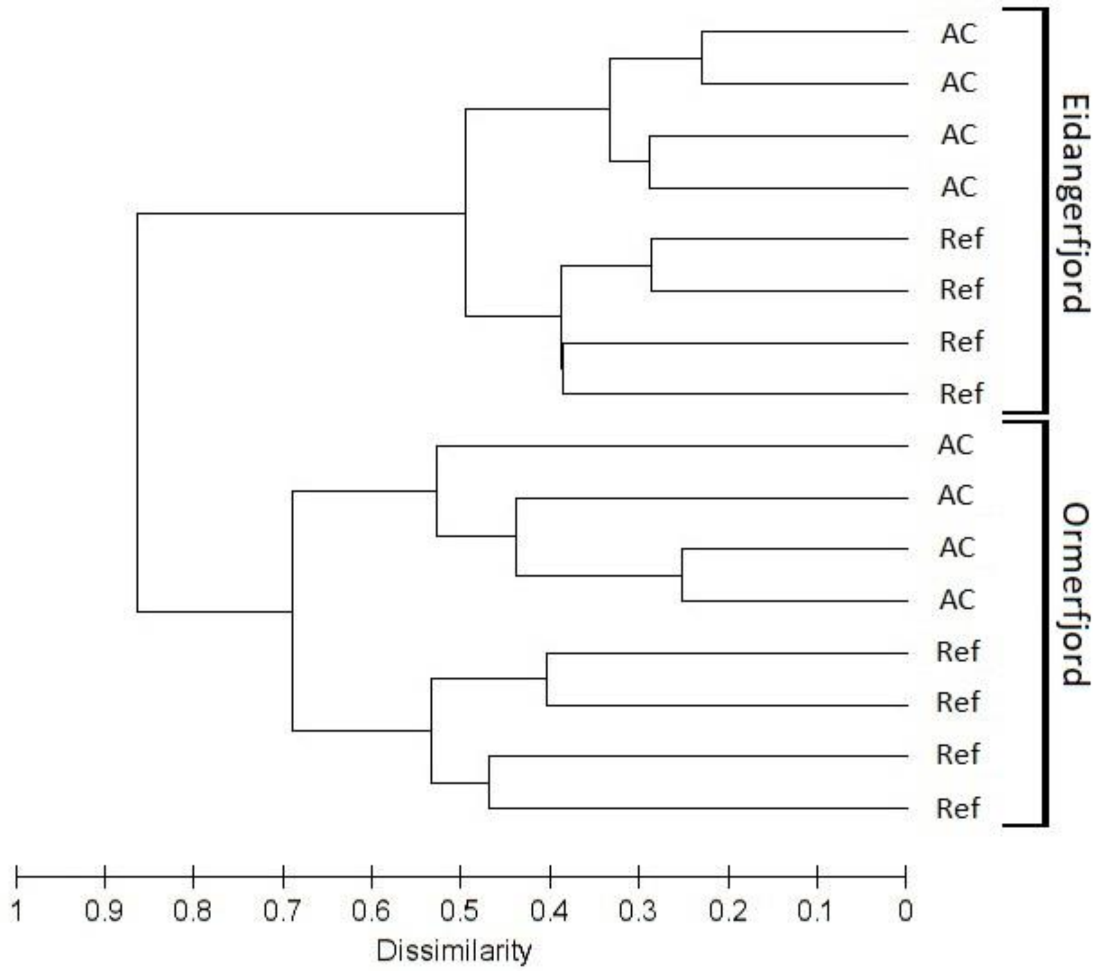

Figure 6: Cluster analysis of species composition in 2018. Horizontal axis shows the dissimilarities between samples. Vertical axis shows the different stations samples. Data transformed by fourth root. Bray-Curtis dissimilarity.

Table 2: Indices for AC treated and reference fields (mean) in 2018 for both fjords. S = number of species, A = number of individuals, H' = Shannon-Wiener diversity index, ES_{100} = Hurlbert's diversity index, ISI_{2012} = Indicator Species Index, NSI2012 = Norwegian Sensitivity Index, NQI1 = Norwegian Quality Index, J' = Pielou's evenness index. "very good" = bule, "good" = green, "moderate" = yellow.

Fjord	Treatment	S	A	H'	ES_{100}	ISI_{2012}	NSI_{2012}	NQI1	J'
Eidanger-fjord	AC	37.5	241.5	3.99	26.35	8.59	20.61	0.67	0.76
	Ref	50.25	369.75	3.93	26.97	9.38	22.40	0.71	0.70
Ormer-fjord	AC	11	26.75	2.97	-	8.25	24.14	0.72	0.86
	Ref	21.5	80.75	2.97	-	9.41	24.15	0.75	0.67

Table 3: Normalized Ecological Quality Ratio (nEQR) values for AC treated and reference fields (mean) in 2018 for both fjords. S = number of species, A = number of individuals, H' = Shannon-Wiener diversity index, ES_{100} = Hurlbert's diversity index, ISI_{2012} = Indicator Species Index, NSI2012 = Norwegian Sensitivity Index, NQI1 = Norwegian Quality Index. "very good" = bule, "good" = green, "moderate" = yellow.

Fjord	Treatment	S	A	nEQR H'	nEQR ES_{100}	nEQr ISI_{2012}	nEQR NSI_{2012}	nEQR NQI1	Avg. nEQR
Eidanger-fjord	AC	37.5	241.5	0.77	0.76	0.81	0.63	0.64	0.72
	Ref	50.25	369.75	0.74	0.75	0.84	0.70	0.68	0.74
Ormer-fjord	AC	11	26.75	0.55	-	0.74	0.77	0.70	0.69
	Ref	21.5	80.75	0.55	-	0.82	0.77	0.73	0.72

Table 4: Average number of individuals per 0.1m² with percentage of total number of individuals (A), biomass (g.w.w.) per 0.1m² (B), average biomass (g.w.w.) per individual (B/A) for the ten most common species found in the Ormerfjord year 2018. Bi=Bivalvia, C=Crustacea, G=Gastropoda, O=Ophiuroidea, P=Polychaeta.

Ormerfjord AC			
Species	A (%)	B (g.w.w.)	B/A (g.w.w.)
Nucula nitidosa (Bi)	7 (26.2)	0.0478	0.00683
Nephtys incisa (P)	4.25 (15.9)	0.217	0.0511
Hyala vitrea (G)	4.25 (15.9)	0.0110	0.00258
Corbula gibba (Bi)	2 (7.48)	0.00715	0.00358
Thyasira flexuosa (with juvenile) (B)	2 (7.48)	0.0119	0.00595
Abra nitida (B)	0.75 (2.80)	0.0422	0.0562
Diplocirrus glaucus (P)	0.75 (2.80)	0.00075	0.001
Prionospio fallax (P)	0.75 (2.80)	0.000238	0.000317
Callianassa subterranea (C)	0.5 (1.87)	0.00045	0.0009
Amphiura chiajei (with juvenile) (O)	0.5 (1.87)	0.0727 (no arm)	0.145
Ormerfjord Ref			
Species	A (%)	B (g.w.w.)	B/A (g.w.w.)
Amphiura filiformis (with juvenile) (O)	42.75 (52.9)	0.927 (no arm)	0.0217
Hyala vitrea (G)	5 (6.20)	0.0163	0.00326
Abyssoninoe hibernica (P)	3.5 (4.33)	0.0499	0.0143
Callianassa subterranea (C)	3.25 (4.02)	0.172	0.0530
Nephtys incisa (P)	3 (3.72)	0.0656	0.0219
Prionospio multibranchiata (P)	2.75 (3.41)	0.00215	0.000782
Diplocirrus glaucus (P)	1.75 (2.17)	0.0128	0.00729
Corbula gibba (Bi)	1.25 (1.55)	0.0213	0.0170
Cylichna cylindracea (G)	1.25 (1.55)	0.0127	0.0101
Pectinaria belgica (P)	1.25 (1.55)	0.0006	0.00048

Table 5: Average number of individuals per 0.1m² with percentage of total number of individuals (A), biomass (g.w.w.) per 0.1m² (B), average biomass (g.w.w.) per individual (B/A) for the ten most common species found in the Eidangerfjord year 2018. Bi=Bivalvia, C=Crustacea, N=Nemertea, O=Ophiuroidea, P=Polychaeta.

Eidangerfjord AC			
Species	A (%)	B (g.w.w.)	B/A (g.w.w.)
Spiophanes kroyeri (P)	54.5 (22.6)	1.47	0.0270
Chaetozone setosa (P)	43.5 (18.0)	0.207	0.00476
Paramphinome jeffreysii (P)	25.5 (10.5)	0.0288	0.00114
Thyasira equalis (with juvenile) (Bi)	16.5 (6.74)	0.411	0.0253
Aphelochaeta marioni (P)	14 (5.81)	0.156	0.0111
Heteromastus filiformis (P)	12.25 (5.08)	0.0170	0.00139
Leucon nasica (C)	6.75 (2.80)	0.00675	0.001
Nemertea indet (N)	6 (2.49)	0.0119	0.00198
Eudorella emarginata (C)	5.25 (2.18)	0.0066	0.00126
Prionospio cirrifera (P)	5 (2.07)	0.00695	0.00139
Eidangerfjord Ref			
Species	A (%)	B (g.w.w.)	B/A (g.w.w.)
Spiophanes kroyeri (P)	120.5 (32.6)	1.88	0.0156
Paramphinome jeffreysii (P)	42 (11.4)	0.0673	0.00160
Thyasira equalis (with juvenile) (Bi)	28.5 (7.72)	0.423	0.0148
Prionospio dubia (P)	23.75 (6.43)	0.0933	0.00393
Heteromastus filiformis (P)	20 (5.41)	0.0539	0.00269
Prionospio cirrifera (P)	16 (4.33)	0.0367	0.00229
Aphelochaeta marioni (P)	12.25 (3.32)	0.113	0.00922
Abyssoninoe hibernica (P)	11.25 (3.05)	0.306	0.0272
Chaetozone setosa (P)	8.25 (2.23)	0.0222	0.00269
Amphiura chiajei (with juvenile) (O)	6.25 (1.69)	0.221 (no arm)	0.0354

3.2 Trends between 2009-2018

Except for the number of species and biomass in 2009, the number of individuals, species and biomass was higher in the reference field over this time period in both fjords (Figs. 2, 3, 4). After 2010 the group Echinoidea was absent in the AC treated field in both fjords. The standard deviation in Figure 4 is very large in some cases and is caused by big variations in biomass between samples. This is mostly due to the presence of some individuals within the group Echinoidea, and when removing the largest individuals Figure 5 shows there is still a notable standard deviation in some of the cases. The MDS plot shows that there are more similarities between the samples within the sites than between the sites most of the years (Figs. 7, 8). Both fjords are separated in the cluster analysis over the years showing greater similarities within the fjords compared to between them (Fig. 9).

In the Eidangerfjord both the number of individuals and species seems to follow one another the reference and AC treated field over the years, with the reference field having more individuals and species (Figs. 2, 3). The group Polychaeta seems to be driving the changes in the number of species and individuals over the years for both fields. The reference field had higher biomass than the AC treated field every year (Fig. 4). The differences were more than ten times at most in 2009 and more than two times at the closest in 2010 which was due to a large sea urchin (*Brissopsis lyrifera*). In the reference field the biomass from the group Polychaeta was relatively unchanging over the years despite the number of individuals in this group increasing by more than five times over the years (Fig. 5). In the MDS plot the reference and test field is following one another in a parallel pattern while keeping the same distance to one another (Fig. 7). The most common species in both fields in the Eidangerfjord 2018, the Polychaeta *Sipphanes kroyeri*, had more individuals in the reference field every year but the biomass per individual was higher in the AC treated field (Table 6). Besides 2013 and 2018 in both fields being the most similar in the cluster analysis, there was no clear separation between the AC treated field and the reference field over the years (Fig. 9).

In the Ormerfjord more species were found in the reference field in 2010, 2013 and 2018, the difference was particularly big in 2010 where there were more than four times as much (Fig. 3). The group Crustacea was almost entirely absent from the AC treated field. There were fewer individuals found in the AC treated field every year, the biggest difference was in 2010 here as well with over nine times as many individuals in the reference field (Fig. 2). The group Ophiuroidea dominated the number of individuals in the reference field in 2009, 2013 and 2018, the group Polychaeta dominated in 2010. The reference field have the highest biomass in 2010, 2013 and 2018 with the group Echinoidea representing the biggest portion (Fig. 4). The biggest difference was in 2013 where the reference field had sixteen times higher biomass. This was largely due to a high representation of the group Echinoidea which was absent in the AC treated field this year. In 2009 the AC treated field had a similar number of species and biomass to the reference field; however, the number of individuals was almost half. The number of individuals, species, and biomass saw a decrease from 2009 to 2010, and from 2010 to 2013 the number of individuals and species increased in the AC treated field. From 2013 to 2018 the number of individuals decreased, and the number of species and biomass remained relatively unchanged. Overall, the AC treated field was reduced in biomass, number of individuals and species in 2010, 2013 and 2018, with 2010 being particularly reduced. The group Ophiuroidea was numerous in the reference fields but occurred rarely and with low numbers in the test fields. From 2009 to 2013 the number of individuals and species increased in the reference field and changed little between 2013 and 2018. When excluding the group Echinoidea from the biomass, the reference field shows little change in the biomass from 2013 to 2018. The standard deviation is big in the reference field indicating big differences in the biomass between the samples. Both the MDS plot and cluster analysis shows the reference field changed relatively little over the nine years compared to the test field (Figs. 8, 9).

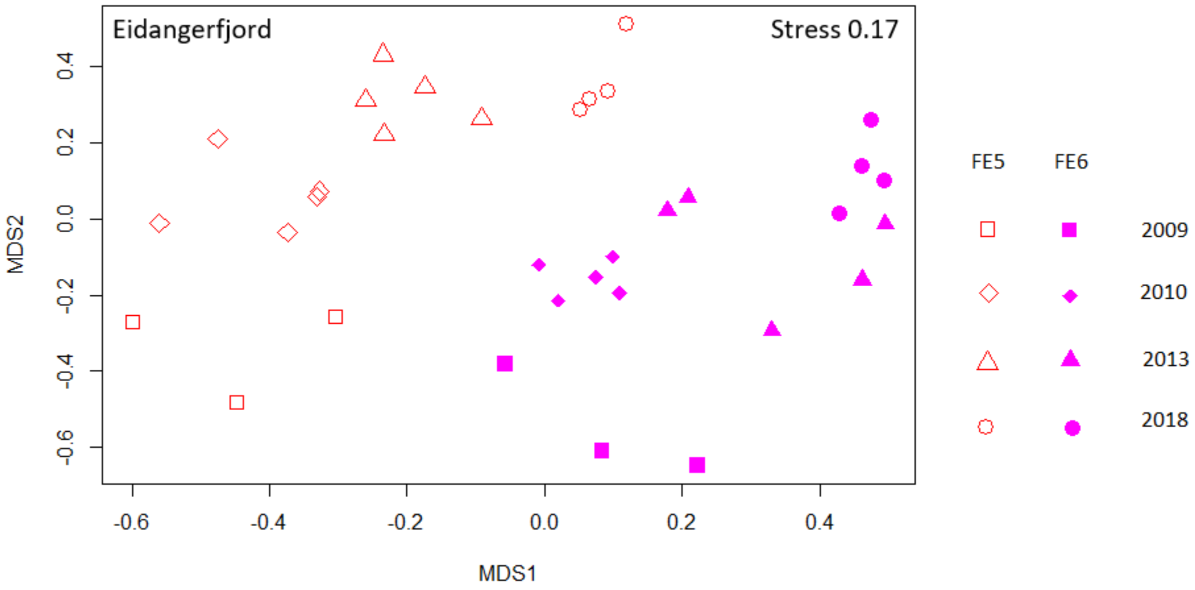

Figure 7: nMDS plot of the samples taken in the Eidangerfjord year 2009, 2010, 2013 and 2018. Stress-level of 0.17 is accepted. Data transformed with fourth root, Bray-Curtis dissimilarity.

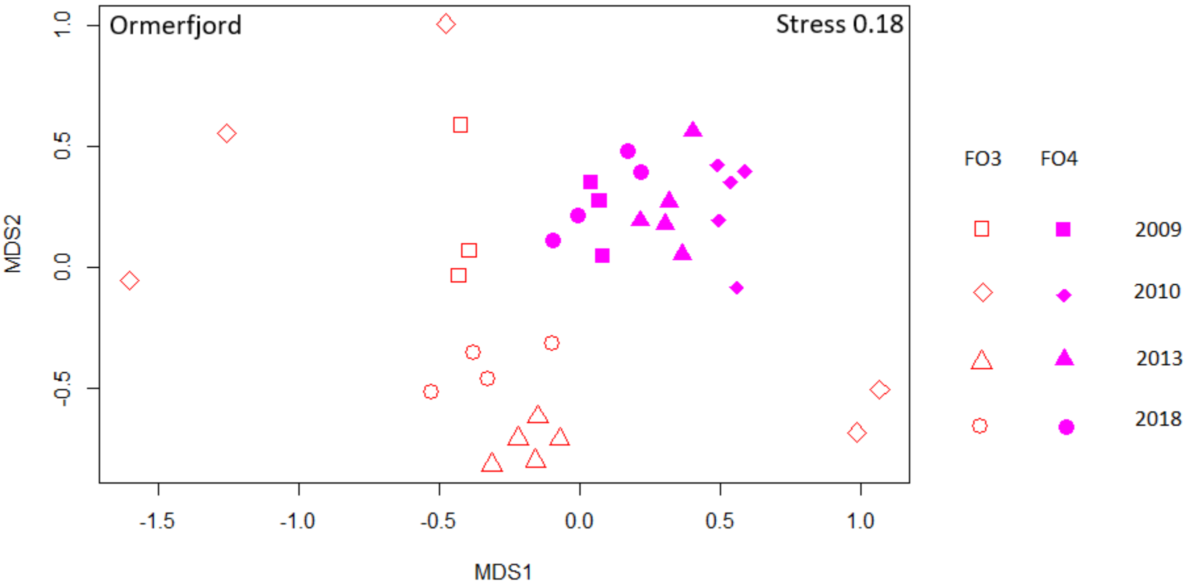

Figure 8: nMDS plot of the samples taken in the Ormerfjord year 2009, 2010, 2013 and 2018. Stress-level of 0.17 is accepted. Data transformed with fourth root, Bray-Curtis dissimilarity.

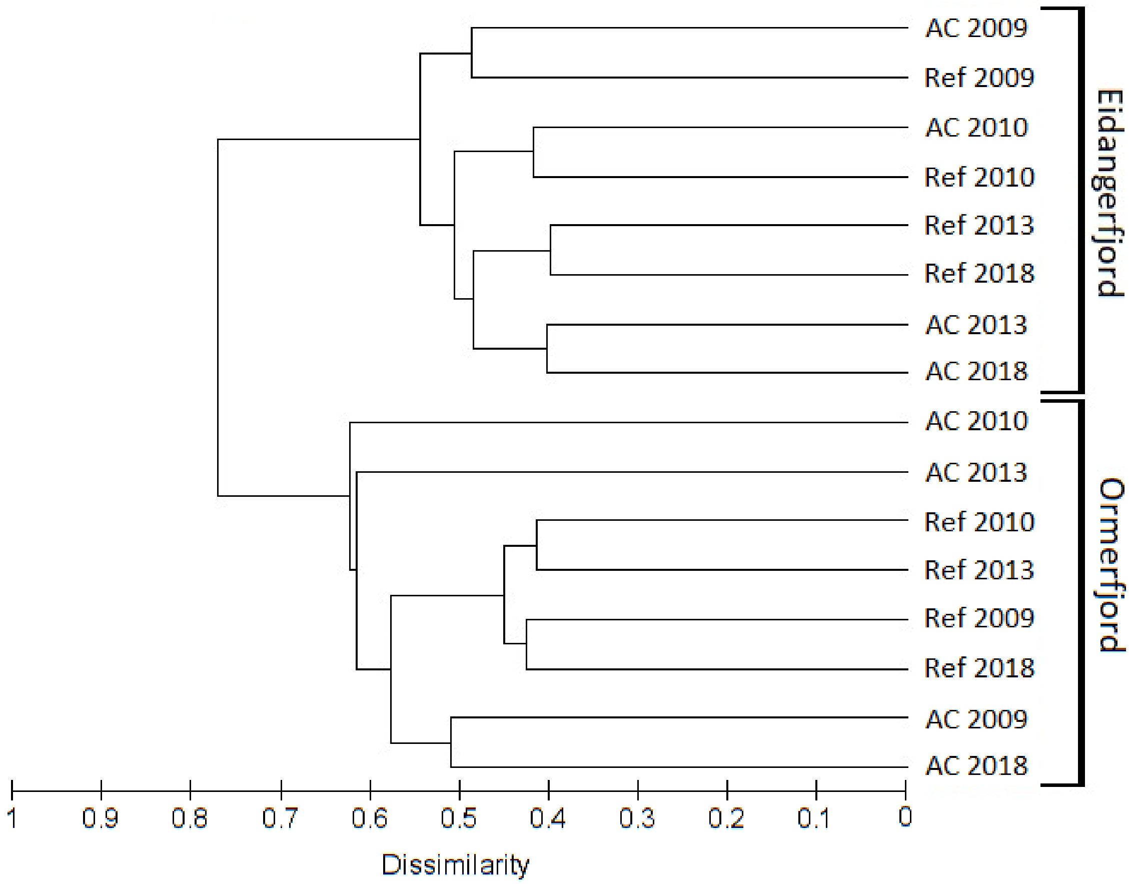

Figure 9: Cluster analysis of average species composition across all years. Horizontal axis shows the dissimilarities between samples. Vertical axis shows the different stations each year. Data transformed by fourth root. Bray-Curtis dissimilarity.

Table 6: Number of individuals, biomass and biomass/individual for the species Spiophanes kroyeri in the reference field and test field the Eidangerfjord year 2009, 2010, 2013, and 2018.

The Eidangerfjord					
Species	Site	Year	Biomass/Ind.	Ind.	Biomass
Spiophanes kroyeri	Ref	2009	0.006458	48	0,31
	AC	2009	0.01625	24	0,39
	Ref	2010	0.004118	68	0,28
	AC	2010	0.008	45	0,36
	Ref	2013	0.004856	104	0,505
	AC	2013	0.008679	53	0,46
	Ref	2018	0.015597	482	7,5176
	AC	2018	0.027005	218	5,887

3.3 Sediment parameters

In the Eidangerfjord the total organic carbon (TOC) was 35 mg/g in the AC treated field and 23.8 mg/g in the reference field, and the total nitrogen (TN) was 3.1 mg/g AC treated field and 2.0 mg/g in the reference field (Table 7). In the Ormerfjord the TOC was 28.8 mg/g in the AC treated field and 9.1 mg/g in the reference field, and the TN was 2.4 mg/g AC treated field and 0.9 mg/g in the reference field. The Eidangerfjord had a higher TOC and TN in both the AC treated and reference field compared to their corresponding fields in the Ormerfjord. The higher amount of TOC is likely due to the Eidangerfjord being an accumulation bottom type compared to a transport type bottom in the Ormerfjord. The TN in the Eidangerfjord was also higher. This indicates that there is more organic matter and food in the Eidangerfjord. There were also differences in TOC and TN between the AC treated and reference field, which indicated more organic matter and food availability in the AC treated fields.

The sediment fine fraction (% <0.063 mm) were 74 and 80 in the AC treated and reference field in the Eidangerfjord, and 91 and 77 in the AC treated and reference field in the Ormerfjord. TOC/TN ranged were ranged from 10.1 to 12.0 showing no notable difference between treatment or fjords.

Table 7: Sediment fine fraction (% <0.063 mm), total organic carbon (TOC), total nitrogen (TN), and TOC/TN ratio in top 0-1 cm of sediments at all locations in 2018.

Fjord	Treatment	Sediment fine fraction (%)	TOC mg/g	TN mg/g	TOC/TN
Eidangerfjord	AC (95 m)	74	35.0	3.1	11.3
	Ref (80 m)	80	23.8	2.0	11.9
Ormerfjord	AC (30 m)	91	28.8	2.4	12.0
	Ref (30 m)	77	9.1	0.9	10.1

Table 8: Summary of the one-way ANOVA tests done on variables between AC treated and reference fields in 2018 for both fjords. Values marked with "" indicate significant difference. H' = Shannon-Wiener diversity index, ES100 = Hurlbert's diversity index, ISI2012 = Indicator Species Index, NSI2012 = Norwegian Sensitivity Index, NQI1 = Norwegian Quality Index, J' = Pielou's evenness index.*

Fjord	Variable (transformation)	DF	F value	Pr(>F)
Eidangerfjord	Biomass (log)	1	22.32	0.00324*
	Species	1	35.05	0.00103*
	Individuals	1	9.076	0.0236*
	H'	1	0.216	0.659
	ES_{100}	1	0.127	0.734
	NQI1	1	34.77	0.00106*
	NSI_{2012}	1	126.6	2.95e-05*
	ISI_{2012}	1	10.07	0.0192*
	J'	1	6.945	0.0388*
Ormerfjord	Biomass (log)	1	10.4	0.018*
	Species	1	30.77	0.00145*
	Individuals	1	83.02	9.82e-05*
	H'	1	0	1
	NQI1	1	3.921	0.095
	NSI_{2012}	1	0.001	0.975
	ISI_{2012}	1	4.782	0.0714
	J'	1	17.31	0.00594*

4 Discussion

4.1 Fauna

The Eidangerfjord (80-95 m) had a more diverse benthic community than the Ormerfjord (30 m) for both the AC treated field and reference field in 2018. Nine years after capping with AC the benthic community in both fjords exposed to the AC treatment had significant fewer individuals, species and lower biomass compared to their corresponding reference fields (Table 8). A similar pattern was also seen in the previous years when benthic organisms were sampled (Figs. 2, 3, 4). In both fjords the reference field shows variation in number of individuals, species, biomass and composition when sampled over the years (Figs. 2, 3, 4, 7, 8). This is interpreted as natural variation and is the reason why capping with AC treatment is compared with a reference field rather than the state of the area before capping.

4.1.1 The Ormerfjord

Out of the two fjords the benthic community in the Ormerfjord had the strongest response to the AC treatment, this response was particularly strong one year after capping (Figs. 2, 3, 4). Higher diversity in a benthic community could increase the ecosystem resilience (Douglas et al., 2017). By having more species capable of performing important tasks in the sediment, the removal of some might have little effect on the ecosystem services the benthic community provides. The strong response in the Ormerfjord compared to the Eidangerfjord is probably due to the less diverse benthic community here. The MDS plot and cluster analysis shows that the reference field changed relatively little over the years compared to the AC treated field (Figs. 8, 9). This also shows that the benthic community in the AC treated field was very disturbed. In the Ormerfjord there was also differences in which group of benthic organisms dominated and which species was present. The group Ophiuroidea was numerous in the reference field but occurred in few times and in low numbers in the AC treated fields (Fig ind). The species *Amphiura filiformis* was the dominant species in the reference field in 2018 making up 52,9 % of the individuals found, while the same species was entirely absent in the AC treated field (Table 4). As *Amphiura filiformis* was found in the AC treated field before capping and was present in the clay capped control field, AC with capping is the only reasonable explanation (Samuelsson pers. com.; Raymond et al., 2020). The group Echinoidea was absent in the AC treated field in 2013 and 2018. The absence of the brittle star *Amphiura filiformis* and the group Echinoidea which are important bioturbators could have caused a slow recovery, as bioturbation can facilitate colonization as well as recruitment of other species (Thrush et al., 1992). *Nucula nitidosa* was the most dominating species in the AC treated field making up 26,2 % of the individuals found.

4.1.2 The Eidangerfjord

In the Eidangerfjord the effects of AC treatment on benthic communities seems to be more moderate compared to the Ormerfjord, but the effect is still significant (Table 8). As discussed previously this may be due to communities with a higher diversity have shown to increase the resilience to disturbances. The number of individuals and species in the AC treated field seems to follow the natural variation in the reference field (Figs. 2, 3). The same can also be observed in the MDS plot as the reference and AC treated field is following one another in a parallel pattern while maintaining approximately the same distance (Fig. 7). The similar variation between the fields coupled with a similarity in the proportions of the different groups of animals and the fields sharing 7 out of 10 species on the list of the most common spices could suggest a similar recruitment in the two locations. The two fields are not getting more similar over time indicating the AC treated field is still affected nine years later with little recovery since capping. However, we cannot exclude the possibility that these differences are due to natural variation between the two fields prior to the capping.

As the reference field is at 80 m depth and the AC treated field is at 95 m depth there could be a big difference between them. Samuelsen (2017) and Raymond (2020) have investigated this by adding a new reference at 95 m depth which were sampled in 2010 and 2013. The 95 m reference field had been previously trawled which can cause disturbances to benthic communities (Kaiser & Spencer, 1996). However, the field was assumed recovered from this trawling as some trawling sensitive species was found and it showed a similar increase in number of individuals compared to the reference field at 80 m indicating there is only natural variation going on and not recovery. They found no difference in the number of individuals or biomass, but there was a difference in the number of species. Using the reference field at 80 m could therefore give the impression that the difference is bigger than it really is. the Eidangerfjord in Figures 2 and 3 is also a god example on why capping with AC treatment is compared with a reference instead of the state of the area before capping, as the steady increase in both species and individuals over the years could be interpreted as recovery after capping, which one would see is not the case when looking at both. The group Polychaeta dominated the number of individuals, species and biomass in both fields in the Eidangerfjord in 2018 (Figs. 2, 3, 4). The group also make up most of the most common species here as well (Table 5). *Spiophanes kroyeri* was the most common species in both fields in the

Eidangerfjord 2018 However, the biomass per individual is almost twice as high in the AC treatment field while there is more than twice the number of individuals in the reference field. A similar trend could also be seen in the previous years. The Polychaeta *Chaetozone setosa* had a higher number of individuals in the AC treated field in contrast to most of the most common species found in this fjord. The group Echinoidea was absent in the AC treated field in 2013 and 2018.

4.2 Indices

The effects of capping with AC observed in this study was not well reflected in the benthic indices used in determining the ecological condition in the Norwegian water directive.

4.2.1 The Ormerfjord

In the Ormerfjord 2018 the AC treated field had 11 species and 26.75 individuals on average compared to 21.5 species and 80.75 individuals found in the reference field. Based on these values the number of individuals and species is considered to be very low in the AC treated field and low in the reference field. Despite this none of the indices used in the benthic quality element in Norway showed a significant difference between both fjords (Table 8). The ES_{100} index could not be calculated as it requires at least 100 individuals on average per sample to be so. The AC treated and reference field got the same value on the Shannon-Wiener index (H'), despite the AC treated field having both fewer individuals and species (Table 2). H' did classify the Ormerfjord to "moderate" condition and was the only benthic index used in the Norwegian water directive to classify the area below "good" (Table 3). NSI_{2012} and NQI1 both gave the classification "good". ISI_{2012} was the only index where the two fields were classified to different conditions, giving the AC treated field a "good" condition and the reference field "very good".

4.2.2 The Eidangerfjord

In the Eidangefjord there were 37.5 species and 241.5 individuals on average in the AC treated field compared to 50.25 species and 369.75 individuals on average in the reference field. The number of individuals and species here is closer compared to the Ormerfjord. The H' index classified both fields as "good" but gave AC treated fields a higher value, despite this field having notably less individuals and species compared to the reference field (Table 3). ES_{100}, NSI_{2012} and NQI1 gave both fields a "good" condition, while ISI_{2012} gave both a “very good” condition. ISI_{2012}, NSI_{2012} and NQI1 indices were significantly higher in the reference field (Table 8).

4.2.3 Evaluating the indices

Both fields in both fjords achieved “good” condition on the benthic quality element in 2018 according to the system used in the Norwegian water directive (Table 3). This does not match what has been observed in this study. Both fields in the Ormerfjord had a low number of individuals and species, particularly the AC treated field which was severely depleted. According to the Norwegian water directive the indices are best used as an indicator for eutrophication, increase in organic load and sedimentation (Direktoratsgruppa vanndirektivet, 2018). Under organic enrichment it is expected that the number of individuals increase while the number of species decreases, leaving larger numbers of some tolerant species (Pearson & Rosenberg, 1978). This is not the case in this study, as both the number of individuals and species is lower in the AC treated fields (Figs. 2, 3). This will specially affect the diversity indices H’ and ES_{100} since they both are calculated using number of individuals, species, and number of individuals in each species. The H’ index will increase as the number of species increase, but it also increases as the individuals are more evenly distributed among the species (Gray and Elliot, 2009). Looking at the J’ index we can see that both the AC fields have a higher value than their respective reference field. This is particularly true in the Ormerfjord where the AC treated field got 0.86 in evenness vs. 0.67 in the reference field. The big difference in evenness has managed to override the effects fewer species would have in the H’ index. The indices using species tolerance as part of the calculation (NSI_{2012}, ISI_{2012}, NQI1) were a little better suited to see the differences between the AC treated fields and reference fields. Yet they still failed to show the disturbed state the AC treated field in the Ormerfjord was in. As with the diversity indices these indices perform better in detecting responses to eutrophication. They are sensitive to an increase in certain individuals tolerant to this disturbance, which is not the case in the AC treated fields as the disturbance has in general reduced the number of individuals in both stations. In addition, calculating indices using species tolerance would require that some species be classified as tolerant or not tolerant species. This could make these indices unsuited to evaluate the state of certain areas as a species that are classified as tolerant because of its tolerance to some disturbances could be sensitive towards other disturbances. *Amphiura filiformis* is such a case as they are classified as tolerant species in the NSI_{2012} and an indifferent species in AMBI (Rygg & Norling, 2013), but in a study on what effects oil production and exploration had on the benthic communities it was found this species was very sensitive to oil pollution (Olsgard & Gray, 1995).

Classifying a body of water using indices and samples taken at one specific time can also be troublesome, as the benthic community can change from year to year as well throughout the year. This could give a wrong classification as indices can change based on the time samples were collected (Reiss & Kröncke, 2005). In this study the samples are collected around the same time of the year every time and the AC treated field is compared to a reference field so the effects of AC can be observed without relying on a classification using indices. However, it does raise the question on why the benthic quality element used in classifying the ecological condition of a body of water is determined using indices alone. Since the result this method gives is prone to seasonal variation, natural variation, evenness in a reduced community, and not accounting for species having a different response to various kinds of disturbances. One would think that if an area has very few species and individuals it would end up getting an ecological status fitting this state, but as things stands this is not the case. One could discuss their findings in the report, but it is not possible to change the classification of the benthic quality element as the indices is the sole deciding factor here. The only other option now is to drop benthic quality element when classifying a body of water if it is expected that indices is unsuited for the area and the potential disturbances that has affected the benthic fauna. This method of classifying the benthic quality element may have to be revised or at least allow professional judgment of the samples.

Much can be revealed just by looking at the number of species, individuals, biomass, and absence or presence of certain species. Assessing an area using these parameters as well could allow for more accurate evaluation of the ecological status. Of course allowing professional judgment is not without its flaws as people can have biases and overall different people could consider the same samples differently. New knowledge could change the way one professional would consider a sample, without the readers knowing that this new knowledge is applied and from what point it was applied to a potential series of samples. Another interesting question could be whether the indices had spotted the effects of the AC treatment if the sampling period had been during another time of the year and how the community structure in the AC field and reference field would change over a year. Are there more species affected by AC.

4.3 Benthic response

Negative effects like decrease in growth, lipid content, reproduction, behavior changes, and morphology have been reported on benthic organisms affected by AC(Jonker et al., 2004; Rust et al., 2004; Millward et al., 2005; Kupryianchyk et al., 2011; Janssen et al., 2012; Nybom et al., 2012, 2015;). Powdered AC were used in the Grenland fjords, small particle size could have a stronger negative effect on benthic organisms than larger particles (Nybom et al. 2012). Although negative effects were observed on several benthic species in this study, using number of individuals, species, and biomass makes it impossible to conclude exactly what response the different species had. However, by using a reference field it should be possible to see what species or which group of species was affected by the AC treatment.

4.3.1 *Amphiura filiformis*

The most notable effect of AC on a benthic species was seen on the brittle star *Amphiura filiformis*. While the species dominated the benthic community at the reference field in Ormenrfjorden (Table 4), it was completely absent from the corresponding AC treated field. *Amphiura filformis* is a common species in the north east Atlantic Ocean and can be found down to 200 m (Rosenberg et al., 1997; Rosenberg & Lundberg, 2004). It lives buried in the sediments with its disk located in a chamber at 6-10 cm from the sediment surface and can live 20 years or more (O'Connor et al., 1983; Solan & Kennedy, 2002). It stretches its arms out from the sediments to collect food mainly by feeding in suspended particles, but it can switch to deposit feeding in stagnant waters and areas with low water flow (Buchanan, 1964; Duchêne & Rosenberg, 2001; Solan & Kennedy, 2002). It is considered a functional important species due to its role in the sediment-water exchange processes and bioturbation (Solan & Kennedy, 2002; O'Reilly et al., 2006). *Amphiura filiformis* have been found to account for 80 % of the total flux of O2 into the sediment, where at least 67 % of this portion is diffusion across the additional sediment-water interfaces created by this species (Vopel et al., 2003). It was modelled that if this species was to go extinct in an area the overall bioturbation potential of the community could go down and cause a collapse (Solan et al., 2004). Bioturbation can cause the release of contaminants from the sediment to the overlying water (USEPA 2005; Thibodeaux & Bierman, 2003; Josefsson et al., 2010). However, in an AC treated field it would promote mixing with the underlying contaminated sediments as well as mixing with newly deposited sediment thereby increasing the effectiveness of the treatment (Sun et al., 2007; Lin et al., 2014). The disappearance of *Amphiura filiformis* in the presence of AC could therefore make this treatment less effective at reducing the bioavailability of contaminants underneath and above the cap.

As mentioned in in the index section *Amphiura filiformis* can have a varying response to different disturbances. It has shown to be sensitive to oil pollution (Olsgard & Gray, 1995) and metals like copper (Rygg, 1985). The regeneration rate of its arms is reduced in hypoxic conditions, while organic enrichment affected the regeneration positively (Nilsson, 1999). A massive increase in both abundance and biomass of this species between 1972 and 1988 in Skagerrak has been attributed to organic enrichment (Josefson, 1990). The species has been classified as an indifferent species in AMBI and a tolerant species in NSI_{2012} (Rygg & Norling, 2013).

The absence of *Amphiura filiformis* in the AC treated field in the Ormerfjord is poorly understood. As the species was found in abundance in a clay capped field without AC in the Ormerfjord in the previous years in this project (Samuelsson et al., 2017), it is relatively safe to say it is the AC and not the thin-layer capping causing the absence of this species. In a previous study it was found that the effects of AC seem to be the most severe when the AC particle size are small (Nybom et al. 2012). So one of the causes could be possible ingestion of AC particles, as the powdered AC used could overlap with the preferable particle size range in feeding activities. Feeding on these particles could cause multiple negative effects, first of energy will have to be spent collecting and transporting these particles to the mouth and as the particles are poor in nutrients they could starve. AC particles can have sharp edges causing mechanical damage when passing through the gut (Nybom, 2015). As AC can sorb essential nutrients (Jonker et al., 2004; Schreiber et al., 2005; Janssen et al., 2012), a decrease in uptake of nutrients from the gut might be possible. Reduced bioavailability of nutrients in the sediment as a result of this sorption is also a possible cause of the absence of *Amphiura filiformis*. Seeing as the species is primarily a suspension feeder (Buchanan, 1964), there are most likely other mechanisms affecting them as well. In the beginning of this section, it was mentioned that several studies have reported various effects of AC on benthic organisms. However, in this study *Amphiura filiformis* is absent in the AC treated fields and it is therefore impossible to tell how they respond to AC other than being absent. More research is needed to find out how they are affected by AC and why they are absent in the AC treated field.

4.3.2 Echinoidea

The group Echinoidea disappeared in the AC treated field in both fjords after 2010, although this group is not numerus in individuals, they can make up a large part of the biomass in a location (Fig. 4). The loss of this group could have negative effects on several ecosystem functions, as their bioturbation can enhance nutrient circulation and thereby enhance the ecosystem productivity (Lohrer et al., 2004; Lohrer et al., 2005). This could also lead to similar consequences as with *Amphiura filiformis*, which accounted for a large part of the total flux of oxygen into the sediment and its potential to enhance the effectiveness of the AC treatment.

4.3.3 *Spiophanes kroyeri*

The Polychaeta *Spiophanes kroyeri* was the most dominating species in both the reference field and AC treated field in the Eidangerfjord (Table 5). However, the average number of individuals per 0.1 m^2 in the AC treated field was 54.5, while this number in the reference field was 120.5 making the population here more than twice as dense. A similar trend of differences in densities could be seen for most of the dominant species in both fields, apart from the Polychaeta *Chaetozone setosa* which had a much higher number of individuals in the AC treated field.

Spiophanes kroyeri is considered a tolerant species by both AMBI and NSI_{2012} (Rygg & Norling, 2013). Some studies found it to be sensitive to metal pollution (Rygg, 1985; Trannum et al., 2004), while in one study the species seemed to be very tolerant to high levels of copper (Olsgard, 1999). The reduced number of individuals in the AC treated field across all years shows the species was negatively affected by the treatment. It might be worth noting that the biomass per individuals in the AC treated field were much higher across all years as well. The reasons behind the biomass differences are hard to determine using the results in this paper, but there could be some possible explanations. The TOC and TN was higher in the AC treated field while the number of individuals and biomass was higher in the reference field. The increased access to nutrition and reduced competition in the AC field could facilitate more growth for this species. However, if this was the case then why is the density so reduced in the AC field, perhaps there is a bottleneck at some stage in their life cycle reducing the number of individuals allowed grow up when AC is present. A closer look at the samples could be appropriate to see if there were any morphological differences between the groups.

4.4 Final thoughts

Long-term effectiveness of the thin-layer capping with AC treatment in this project was reported in Schaaning et al. (2021). It was found that the treatment still reduced uptake of dioxins in benthic organisms despite new contaminated sediments being deposited from surrounding areas. This shows the treatment is an effective way of reducing contaminant bioavailability. However, the benthic fauna, particularly in the Ormerfjord, responded negative to the powdered AC used in this project. The difference between the two fjords could indicate this treatment could be better suited in some areas. The benthic community has several important roles in ecosystem like affecting the oxygen concentration in the sediment, enhance microbial activities, increase fluxes of inorganic nutrients back to the water column, and act as food for other organisms to name a few (Lohrer et al. 2004; Pedersen et al., 2008; Gray and Elliott 2009). A depleted benthic fauna and removal of key species like *Amphiura filiformis* in the AC treated field in the Ormerfjord can therefore have a very negative impact on the ecosystem as a whole if the treatment were to be applied to a larger area. The use of this treatment must be carefully weighed against the possible long-term effects on the benthic community before applying it.

5 Conclusion

Thin-layer capping with powdered activated carbon mixed with clay negatively affected the benthic community both fjords nine years after capping. Number of individuals, species and biomass were reduced as a result of the AC treatment, the effects being most notable in the Ormerfjord (30 m). The stronger response in the Ormerfjord is likely due the fjord having a less diverse benthic community compared to the Eidangerfjord (80-95 m). The faunal group Echinoidea was absent in both AC treated field after 2010. The brittle star *Amphiura filiformis* vanished after the AC treatment in the Ormerfjord and has still not returned nine years later. The indices used to assess the benthic quality element in the water framework directive monitoring system for coastal waters in Norway did not reflect negative effects AC treatment has on the benthic communities.

The long-lasting effects of AC on the benthic community as well as the elimination of the important key specie *Amphiura filiformis* could impair several ecosystem functions like enhancing microbial activities and growth rates, converting dead organic material to meat and act as a food source fish. the Eidangerfjord were less negatively affected by the AC treatment compared to the Ormerfjord, this could indicate this treatment may be more suitable in some areas. As this treatment is effective at reducing contaminant release and bioavailability, a careful evaluation of the long-term effects on the benthic community in an area is highly recommended before applying this treatment on a large scale. This study also shows that the indices used is not suited to assess the benthic quality element when the benthic community is affected by this kind of disturbance. Other indices might be needed to correctly assess the effects on benthic community solely based indices, or a different approach involving new methods or professional judgment of samples could also be an appropriate part in assessing the benthic quality element.

More research is needed to get a better understanding of how and why AC affects some of these spices as much as it does. In this study the faunal group Echinoidea, the brittle star *Amphiura filiformis*, and the Polychaeta *Spiophanes kroyeri* have the most notable responses to AC. More research on the effects various sizes of AC particles have on benthic organisms will also be recommended as the particles used in this study were small and easily ingestible.

References

Aller, R. C. (1988). Benthic fauna and biogeochemical processes in marine sediments: The role of burrow structures. *Nitrogen Cycling in Coastal Marine Environments*, 301–338.

Alve, E. (2000). Environmental Stratigraphy. In R. E. Martin (Ed.), *Environmental Micropaleontology: The Application of Microfossils to Environmental Geology* (pp. 323–350). Springer US. https://doi.org/10.1007/978-1-4615-4167-7_15

Barrett, R. T., Anker-Nilssen, T., Gabrielsen, G. W., & Chapdelaine, G. (2002). Food consumption by seabirds in Norwegian waters. *ICES Journal of Marine Science*, *59*(1), 43–57. https://doi.org/10.1006/jmsc.2001.1145

Beckingham, B., Buys, D., Vandewalker, H., & Ghosh, U. (2013). Observations of limited secondary effects to benthic invertebrates and macrophytes with activated carbon amendment in river sediments. *Environmental Toxicology and Chemistry*, *32*(7), 1504–1515. https://doi.org/10.1002/etc.2231

Beckingham, B., & Ghosh, U. (2011). Field-Scale Reduction of PCB Bioavailability with Activated Carbon Amendment to River Sediments. *Environmental Science & Technology*, *45*(24), 10567–10574. https://doi.org/10.1021/es202218p

Berge, J. A., Borgersen, G., & Norling, K. (2011). *Potensielle bioturbatorer i deponiet ved Malmøykalven* (NIVA-rapport No. 6138–2011). Norsk institutt for vannforskning. https://niva.brage.unit.no/niva-xmlui/handle/11250/215380

Bolam, S. G., & Rees, H. L. (2003). Minimizing Impacts of Maintenance Dredged Material Disposal in the Coastal Environment: A Habitat Approach. *Environmental Management*, *32*(2), 171–188. https://doi.org/10.1007/s00267-003-2998-2

Boyd, S. E., Rees, H. L., & Richardson, C. A. (2000). Nematodes as Sensitive Indicators of Change at Dredged Material Disposal Sites. *Estuarine, Coastal and Shelf Science*, *51*(6), 805–819. https://doi.org/10.1006/ecss.2000.0722

Bradshaw, C., Tjensvoll, I., Sköld, M., Allan, I. J., Molvaer, J., Magnusson, J., Naes, K., & Nilsson, H. C. (2012). Bottom trawling resuspends sediment and releases bioavailable contaminants in a polluted fjord. *Environmental Pollution*, *170*, 232–241. https://doi.org/10.1016/j.envpol.2012.06.019

Bray, J. R., & Curtis, J. T. (1957). An Ordination of the Upland Forest Communities of Southern Wisconsin. *Ecological Monographs*, *27*(4), 325–349. https://doi.org/10.2307/1942268

Britannica, T. Editors of Encyclopaedia. (2019, August 2). *Secondary succession*. Encyclopedia Britannica. https://www.britannica.com/science/secondary-succession

Buchanan, J. B. (1964). A Comparative Study of Some Features of the Biology of Amphiura Filiformis and Amphiura Chiajei [Ophiuroidea] Considered in Relation to their Distribution. *Journal of the Marine Biological Association of the United Kingdom*, *44*(3), 565–576. https://doi.org/10.1017/S0025315400027776

Cho, Y.-M., Ghosh, U., Kennedy, A. J., Grossman, A., Ray, G., Tomaszewski, J. E., Smithenry, D. W., Bridges, T. S., & Luthy, R. G. (2009). Field Application of Activated Carbon Amendment for In-Situ Stabilization of Polychlorinated Biphenyls in Marine Sediment. *Environmental Science & Technology*, *43*(10), 3815–3823. https://doi.org/10.1021/es802931c

Cho, Y.-M., Smithenry, D. W., Ghosh, U., Kennedy, A. J., Millward, R. N., Bridges, T. S., & Luthy, R. G. (2007). Field methods for amending marine sediment with activated carbon and assessing treatment effectiveness. *Marine Environmental Research*, *64*(5), 541–555. https://doi.org/10.1016/j.marenvres.2007.04.006

Choi, Y., Thompson, J. M., Lin, D., Cho, Y.-M., Ismail, N. S., Hsieh, C.-H., & Luthy, R. G. (2016). Secondary environmental impacts of remedial alternatives for sediment contaminated with hydrophobic organic contaminants. *Journal of Hazardous Materials*, *304*, 352–359. https://doi.org/10.1016/j.jhazmat.2015.09.069

Christensen, B., Vedel, A., & Kristensen, E. (2000). Carbon and nitrogen fluxes in sediment inhabited by suspension-feeding (Nereis diversicolor) and non-suspension-feeding (N. virens) polychaetes. *Marine Ecology Progress Series*, *192*, 203–217. https://doi.org/10.3354/meps192203

Clarke, K. R., & Warwick, R. M. (2001). *Change in marine communities: An approach to statistical analysis and interpretation* (2nd ed.). PRIMER-E Ltd, Plymoth. http://plymsea.ac.uk/id/eprint/7656/

Commito, J., & Ambrose, W. (1985). Multiple trophic levels in soft-bottom communities. *Marine Ecology Progress Series*, *26*, 289–293. https://doi.org/10.3354/meps026289

Cornelissen, G., Amstaetter, K., Hauge, A., Schaanning, M., Beylich, B., Gunnarsson, J. S., Breedveld, G. D., Oen, A. M. P., & Eek, E. (2012). Large-Scale Field Study on Thin-Layer Capping of Marine PCDD/F-Contaminated Sediments in Grenlandfjords, Norway: Physicochemical Effects. *Environmental Science & Technology*, *46*(21), 12030–12037. https://doi.org/10.1021/es302431u

Cornelissen, G., Elmquist Kruså, M., Breedveld, G. D., Eek, E., Oen, A. M. P., Arp, H. P. H., Raymond, C., Samuelsson, G., Hedman, J. E., Stokland, Ø., & Gunnarsson, J. S. (2011). Remediation of Contaminated Marine Sediment Using Thin-Layer Capping with Activated Carbon—A Field Experiment in Trondheim Harbor, Norway. *Environmental Science & Technology*, *45*(14), 6110–6116. https://doi.org/10.1021/es2011397

Cornelissen, G., Gustafsson, Ö., Bucheli, T. D., Jonker, M. T. O., Koelmans, A. A., & van Noort, P. C. M. (2005). Extensive Sorption of Organic Compounds to Black Carbon, Coal, and Kerogen in Sediments and Soils: Mechanisms and Consequences for Distribution, Bioaccumulation, and Biodegradation. *Environmental Science & Technology*, *39*(18), 6881–6895. https://doi.org/10.1021/es050191b

Cornelissen, G., Schaanning, M., Gunnarsson, J. S., & Eek, E. (2016). A large-scale field trial of thin-layer capping of PCDD/F-contaminated sediments: Sediment-to-water fluxes up to 5 years post-amendment. *Integrated Environmental Assessment and Management*, *12*(2), 216–221. https://doi.org/10.1002/ieam.1665

Council Directive, 2000/60/EC. (2000). *Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 Establishing a Framework for Community Action in the Field of Water Policy*. https://eur-lex.europa.eu/legal-content/EN/LSU/?uri=CELEX%3A32000L0060

Council Directive, 2013/39/EU. (2013). *Directive 2013/39/EU of the European Parliament and of the Council of 12 August 2013 Amending Directives 2000/60/EC and 2008/105/EC as Regards Priority Substances in the Field of Water Policy*. https://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:L:2013:226:0001:0017:EN:PDF

Dauer, D. M., & Simon, J. L. (1976). Repopulation of the polychaete fauna of an intertidal habitat following natural defaunation: Species equilibrium. *Oecologia*, *22*(2), 99–117. https://doi.org/10.1007/BF00344711

Direktoratsguppa vanndirektivet. (2018). *Veileder 02:2018 Klassifisering av miljøtilstand i vann*. Direktoratsgruppen for gjennomføringen av vannforskriften (In Norwegian. 220 pp + appendix). https://www.vannportalen.no/veiledere/klassifiseringsveileder/

Dobbs, F. C., & Vozarik, J. M. (1983). Immediate effects of a storm on coastal infauna. *Marine Ecology Progress Series*, *11*(3), 273–279.

Douglas, E. J., Pilditch, C. A., Kraan, C., Schipper, L. A., Lohrer, A. M., & Thrush, S. F. (2017). Macrofaunal Functional Diversity Provides Resilience to Nutrient Enrichment in Coastal Sediments. *Ecosystems*, *20*(7), 1324–1336. https://doi.org/10.1007/s10021-017-0113-4

Duchêne, J.-C., & Rosenberg, R. (2001). Marine benthic faunal activity patterns on a sediment surface assessed by video numerical tracking. *Marine Ecology Progress Series*, *223*, 113–119. https://doi.org/10.3354/meps223113

Duineveld, G. C. A., & Van Noort, G. J. (1986). Observations on the population dynamics of amphiura filiformis (ophiuroidea: Echinodermata) in the southern north sea and its exploitation by the dab, Limanda limanda. *Netherlands Journal of Sea Research*, *20*(1), 85–94. https://doi.org/10.1016/0077-7579(86)90064-5

Eckman, J. E. (1983). Hydrodynamic processes affecting benthic recruitment1. *Limnology and Oceanography*, *28*(2), 241–257. https://doi.org/10.4319/lo.1983.28.2.0241

Eek, E., Cornelissen, G., Schaanning, M., Beylich, B. A., Evenstad, T. A., Haug, I., Kirkhaug, G., Storholt, P., & Bredeveld, G. (2011). *Evaluering av gjennomføring av testtildekking i Eidangerfjorden og Ormefjorden* (NGI-Rapport No. 20071139-00-120-R). https://www.ngi.no/download/file/7625

Eek, E., Schaaning, M., & Cornelissen, G. (2014). *Tynntildekking av forurensete sedimenter: Overvåking av fire testfelt i Grenlandsfjordene* (Oppdragsrapport M-219/2014). Miljødirektoratet. https://www.ngi.no/download/file/7616

Essink, K. (1999). Ecological effects of dumping of dredged sediments; options for management. *Journal of Coastal Conservation*, *5*(1), 69–80. https://doi.org/10.1007/BF02802741

Fagerli, C. W., Ruus, A., Borgersen, G., Staalstrøm, A., Green, N. W., Hjermann, D. Ø., & Selvik, J. R. (2016). *Tiltaksrettet overvåking av Grenlandsfjordene i henhold til vannforskriften. Overvåking for konsortium av 11 bedrifter i Grenland.* (NIVA-rapport No. 7049–2016). Norsk institutt for vannforskning. https://niva.brage.unit.no/niva-xmlui/handle/11250/2407419

Fathollahzadeh, H., Kaczala, F., Bhatnagar, A., & Hogland, W. (2015). Significance of environmental dredging on metal mobility from contaminated sediments in the Oskarshamn Harbor, Sweden. *Chemosphere*, *119*, 445–451. https://doi.org/10.1016/j.chemosphere.2014.07.008

Förstner, U., & Apitz, S. E. (2007). Sediment remediation: U.S. focus on capping and monitored natural recovery. *Journal of Soils and Sediments*, *7*(6), 351–358. https://doi.org/10.1065/jss2007.10.256

Frid, C. L. J., & Caswell, B. A. (2017). *Marine Pollution*. Oxford University Press.

Ghosh, U., Gillette, J. S., Luthy, R. G., & Zare, R. N. (2000). Microscale Location, Characterization, and Association of Polycyclic Aromatic Hydrocarbons on Harbor Sediment Particles. *Environmental Science & Technology*, *34*(9), 1729–1736. https://doi.org/10.1021/es991032t

Ghosh, U., Luthy, R. G., Cornelissen, G., Werner, D., & Menzie, C. A. (2011). In-situ Sorbent Amendments: A New Direction in Contaminated Sediment Management. *Environmental Science & Technology*, *45*(4), 1163–1168. https://doi.org/10.1021/es102694h

Ghosh, U., Zimmerman, J. R., & Luthy, R. G. (2003). PCB and PAH Speciation among Particle Types in Contaminated Harbor Sediments and Effects on PAH Bioavailability. *Environmental Science & Technology*, *37*(10), 2209–2217. https://doi.org/10.1021/es020833k

Giblin, A. E., Foreman, K. H., & Banta, G. T. (1995). Biogeochemical Processes and Marine Benthic Community Structure: Which Follows Which? In C. G. Jones & J. H. Lawton (Eds.), *Linking Species & Ecosystems* (pp. 37–44). Springer US. https://doi.org/10.1007/978-1-4615-1773-3_4

Goutte, A., Chevreuil, M., Alliot, F., Chastel, O., Cherel, Y., Eléaume, M., & Massé, G. (2013). Persistent organic pollutants in benthic and pelagic organisms off Adélie Land, Antarctica. *Marine Pollution Bulletin*, *77*(1), 82–89. https://doi.org/10.1016/j.marpolbul.2013.10.027

Graf, G., & Rosenberg, R. (1997). Bioresuspension and biodeposition: A review. *Journal of Marine Systems*, *11*(3), 269–278. https://doi.org/10.1016/S0924-7963(96)00126-1

Grassle, J. F., & Grassle, J. P. (1974). Opportunistic life histories and genetic systems in marine benthic polychaetes. *Journal of Marine Research. Sears Foundation for Marine Research, Yale University: New Haven*, *32*(2), 253–284.

Grathwohl, P. (1990). Influence of organic matter from soils and sediments from various origins on the sorption of some chlorinated aliphatic hydrocarbons: Implications on Koc correlations. *Environmental Science & Technology*, *24*(11), 1687–1693. https://doi.org/10.1021/es00081a010

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

Guillaumot, C., Fabri-Ruiz, S., Martin, A., Eléaume, M., Danis, B., Féral, J.-P., & Saucède, T. (2018). Benthic species of the Kerguelen Plateau show contrasting distribution shifts in response to environmental changes. *Ecology and Evolution*, *8*(12), 6210–6225. https://doi.org/10.1002/ece3.4091

Hall, S. J., Basford, D., Robertson, Raffaelli, D., & Tuck, I. D. (1991). Patterns of recolonisation and the importance of pit-digging by the crab Cancer pagurus in a subtidal sand habitat. *Marine Ecology Progress Series*, *72*, 93–102.

Hansen, K., & Kristensen, E. (1998). The impact of the polychaete Nereis diversicolor and enrichment with macroalgal (Chaetomorpha linum) detritus on benthic metabolism and nutrient dynamics in organic-poor and organic-rich sediment. *Journal of Experimental Marine Biology and Ecology*, *231*(2), 201–223. https://doi.org/10.1016/S0022-0981(98)00070-7

Hjelset, A. M., Andersen, M., Gjertz, I., Lydersen, C., & Gulliksen, B. (1999). Feeding habits of bearded seals (Erignathus barbatus) from the Svalbard area, Norway. *Polar Biology*, *21*(3), 186–193. https://doi.org/10.1007/s003000050351

Hurlbert, S. H. (1971). The Nonconcept of Species Diversity: A Critique and Alternative Parameters. *Ecology*, *52*(4), 577–586. https://doi.org/10.2307/1934145

Hyland, J. L., Balthis, L. W., Engle, V. D., Long, E. R., Paul, J. F., Summers, J. K., & Van Dolah, R. F. (2003). Incidence of Stress in Benthic Communities Along the U.S. Atlantic and Gulf of Mexico Coasts within Different Ranges of Sediment Contamination from Chemical Mixtures. *Environmental Monitoring and Assessment*, *81*, 149–161. https://doi.org/10.1023/A:1021325007660

Janssen, E. M.-L., & Beckingham, B. A. (2013). Biological Responses to Activated Carbon Amendments in Sediment Remediation. *Environmental Science & Technology*, *47*(14), 7595–7607. https://doi.org/10.1021/es401142e

Janssen, E. M.-L., Choi, Y., & Luthy, R. G. (2012). Assessment of Nontoxic, Secondary Effects of Sorbent Amendment to Sediments on the Deposit-Feeding Organism Neanthes arenaceodentata. *Environmental Science & Technology*, *46*(7), 4134–4141. https://doi.org/10.1021/es204066g

Johnson, R. G. (1970). Variations in Diversity within Benthic Marine Communities. *The American Naturalist*, *104*(937), 285–300. https://doi.org/10.1086/282662

Jonker, M. T. O., Hoenderboom, A. M., & Koelmans, A. A. (2004). Effects of sedimentary sootlike materials on bioaccumulation and sorption of polychlorinated biphenyls. *Environmental Toxicology and Chemistry*, *23*(11), 2563–2570. https://doi.org/10.1897/03-351

Josefson, A. B. (1990). Increase of benthic biomass in the Skagerrak-Kattegat during the 1970 s and 1980 s– effects of organic enrichment?. *Marine Ecology Progress Series. Oldendorf*, *66*(1), 117–130.

Josefsson, S., Leonardsson, K., Gunnarsson, J. S., & Wiberg, K. (2010). Bioturbation-Driven Release of Buried PCBs and PBDEs from Different Depths in Contaminated Sediments. *Environmental Science & Technology*, *44*(19), 7456–7464. https://doi.org/10.1021/es100615g

Josefsson, S., Schaanning, M., Samuelsson, G. S., Gunnarsson, J. S., Olofsson, I., Eek, E., & Wiberg, K. (2012). Capping Efficiency of Various Carbonaceous and Mineral Materials for In Situ Remediation of Polychlorinated Dibenzo-p-dioxin and Dibenzofuran Contaminated Marine Sediments: Sediment-to-Water Fluxes and Bioaccumulation in Boxcosm Tests. *Environmental Science & Technology*, *46*(6), 3343–3351. https://doi.org/10.1021/es203528v

Kaiser, M. J., & Spencer, B. E. (1996). The Effects of Beam-Trawl Disturbance on Infaunal Communities in Different Habitats. *Journal of Animal Ecology*, *65*(3), 348–358. https://doi.org/10.2307/5881

Karapanagioti, H. K., Kleineidam, S., Sabatini, D. A., Grathwohl, P., & Ligouis, B. (2000). Impacts of Heterogeneous Organic Matter on Phenanthrene Sorption: Equilibrium and Kinetic Studies with Aquifer Material. *Environmental Science & Technology*, *34*(3), 406–414. https://doi.org/10.1021/es9902219

Knutzen, J., Bjerkeng, B., Næs, K., & Schlabach, M. (2003). Polychlorinated dibenzofurans/dibenzo-p-dioxins (PCDF/PCDDs) and other dioxin-like substances in marine organisms from the Grenland fjords, S. Norway, 1975–2001: Present contamination levels, trends and species specific accumulation of PCDF/PCDD congeners. *Chemosphere*, *52*(4), 745–760. https://doi.org/10.1016/S0045-6535(03)00102-4

Knutzen, J., & Oehme, M. (1989). Polychlorinated dibenzofuran (PCDF) and dibenzo-p-dioxin (PCDD) levels in organisms and sediments from the frierfjord, southern Norway. *Chemosphere*, *19*(12), 1897–1909. https://doi.org/10.1016/0045-6535(89)90013-1

Koelmans, A. A., Poot, A., Lange, H. J. D., Velzeboer, I., Harmsen, J., & Noort, P. C. M. van. (2010). Estimation of In Situ Sediment-to-Water Fluxes of Polycyclic Aromatic Hydrocarbons, Polychlorobiphenyls and Polybrominated Diphenylethers. *Environmental Science & Technology*, *44*(8), 3014–3020. https://doi.org/10.1021/es903938z

Kristensen, E. (2000). Organic matter diagenesis at the oxic/anoxic interface in coastal marine sediments, with emphasis on the role of burrowing animals. In G. Liebezeit, S. Dittmann, & I. Kröncke (Eds.), *Life at Interfaces and Under Extreme Conditions* (pp. 1–24). Springer Netherlands. https://doi.org/10.1007/978-94-011-4148-2_1

Kupryianchyk, D., Peeters, E. T. H. M., Rakowska, M. I., Reichman, E. P., Grotenhuis, J. T. C., & Koelmans, A. A. (2012). Long-Term Recovery of Benthic Communities in Sediments Amended with Activated Carbon. *Environmental Science & Technology*, *46*(19), 10735–10742. https://doi.org/10.1021/es302285h

Kupryianchyk, D., Rakowska, M. I., Roessink, I., Reichman, E. P., Grotenhuis, J. T. C., & Koelmans, A. A. (2013). In situ Treatment with Activated Carbon Reduces Bioaccumulation in Aquatic Food Chains. *Environmental Science & Technology*, *47*(9), 4563–4571. https://doi.org/10.1021/es305265x

Kupryianchyk, D., Reichman, E. P., Rakowska, M. I., Peeters, E. T. H. M., Grotenhuis, J. T. C., & Koelmans, A. A. (2011). Ecotoxicological Effects of Activated Carbon Amendments on Macroinvertebrates in Nonpolluted and Polluted Sediments. *Environmental Science & Technology*, *45*(19), 8567–8574. https://doi.org/10.1021/es2014538

Lardicci, C., Castelli, A., & Tagliapietra, D. (2004). Soft Bottom Macrobenthos. In *Mediterranean Marine Benthos: A manual of methods for its sampling and study* (Biologia Marina Mediterranea, Vols. 11, Chapter 4, pp. 99–131).

Larsson, P. (1985). Contaminated sediments of lakes and oceans act as sources of chlorinated hydrocarbons for release to water and atmosphere. *Nature*, *317*(6035), 347–349. https://doi.org/10.1038/317347a0

Lin, D., Cho, Y.-M., Werner, D., & Luthy, R. G. (2014). Bioturbation Delays Attenuation of DDT by Clean Sediment Cap but Promotes Sequestration by Thin-Layered Activated Carbon. *Environmental Science & Technology*, *48*(2), 1175–1183. https://doi.org/10.1021/es404108h

Lohrer, A. M., Thrush, S. F., & Gibbs, M. M. (2004). Bioturbators enhance ecosystem function through complex biogeochemical interactions. *Nature*, *431*(7012), 1092–1095. https://doi.org/10.1038/nature03042

Lohrer, A. M., Thrush, S. F., Hunt, L., Hancock, N., & Lundquist, C. (2005). Rapid reworking of subtidal sediments by burrowing spatangoid urchins. *Journal of Experimental Marine Biology and Ecology*, *321*(2), 155–169. https://doi.org/10.1016/j.jembe.2005.02.002

López-Jamar, E., & Mejuto, J. (1988). Infaunal benthic recolonization after dredging operations in La Coruña Bay, NW Spain. *Cahiers de Biologie Marine*, *29*, 37–49.

Luthy, R. G., Aiken, G. R., Brusseau, M. L., Cunningham, S. D., Gschwend, P. M., Pignatello, J. J., Reinhard, M., Traina, S. J., Weber, W. J., & Westall, J. C. (1997). Sequestration of Hydrophobic Organic Contaminants by Geosorbents. *Environmental Science & Technology*, *31*(12), 3341–3347. https://doi.org/10.1021/es970512m

Macdonald, D. S., Little, M., Eno, N. C., & Hiscock, K. (1996). Disturbance of benthic species by fishing activities: A sensitivity index. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *6*(4), 257–268. https://doi.org/10.1002/(SICI)1099-0755(199612)6:4<257::AID-AQC194>3.0.CO;2-7

Marsh, H., & Rodríguez-Reinoso, F. (2006). *Activated Carbon*. Elsevier.

Maurer, D., Keck, R. T., Tinsman, J. C., & Leathem, W. A. (1981). Vertical migration and mortality of benthos in dredged material—part I: Mollusca. *Marine Environmental Research*, *4*(4), 299–319. https://doi.org/10.1016/0141-1136(81)90043-X

Maurer, D., Keck, R. T., Tinsman, J. C., & Leathem, W. A. (1982). Vertical migration and mortality of benthos in dredged material: Part III—polychaeta. *Marine Environmental Research*, *6*(1), 49–68. https://doi.org/10.1016/0141-1136(82)90007-1

McLeod, P. B., Luoma, S. N., & Luthy, R. G. (2008). Biodynamic Modeling of PCB Uptake by Macoma balthica and Corbicula fluminea from Sediment Amended with Activated Carbon. *Environmental Science & Technology*, *42*(2), 484–490. https://doi.org/10.1021/es070139a

Millward, R. N., Bridges, T. S., Ghosh, U., Zimmerman, J. R., & Luthy, R. G. (2005). Addition of Activated Carbon to Sediments to Reduce PCB Bioaccumulation by a Polychaete (Neanthes arenaceodentata) and an Amphipod (Leptocheirus plumulosus). *Environmental Science & Technology*, *39*(8), 2880–2887. https://doi.org/10.1021/es048768x

Mitrou, P. I., Dimitriadis, G., & Raptis, S. A. (2001). Toxic effects of 2,3,7,8-tetrachlorodibenzo-p-dioxin and related compounds. *European Journal of Internal Medicine*, *12*(5), 406–411. https://doi.org/10.1016/S0953-6205(01)00146-7

Molvær, J. (1980). Deep Water Renewals in the Frierfjord—An Intermittently Anoxic Basin. In H. J. Freeland, D. M. Farmer, & C. D. Levings (Eds.), *Fjord Oceanography* (pp. 531–537). Springer US. https://doi.org/10.1007/978-1-4613-3105-6_48

Molvær, J. (1999). *Grenlandsfjordene 1994-97 Undersøkelser av vannkjemiske forhold og vannutskiftning* (NIVA-rapport No. 756/99). Norsk institutt for vannforskning. https://core.ac.uk/display/52089961

Næs, K., Persson, J., Saloranta, T., Andersen, T., Berge, J. A., Hylland, K., Ruus, A., Tobiesen, A., Bergstad, O. A., & Knutsen, J. A. (2004). Dioksiner i Grenlandsfjordene – DIG. Oppsummering av forskningsprosjektet. In *94* (NIVA-Rapport No. 4876–2014). Norsk institutt for vannforskning. https://niva.brage.unit.no/niva-xmlui/handle/11250/212529

Nilsson, H. C. (1999). Effects of hypoxia and organic enrichment on growth of the brittle stars Amphiura filiformis (O.F. Müller) and Amphiura chiajei Forbes. *Journal of Experimental Marine Biology and Ecology*, *237*(1), 11–30. https://doi.org/10.1016/S0022-0981(98)00214-7

Norwegian Food Safety Authority. (2019). *Grenlandsfjordene—Advarsel om miljøgifter, Forurensning: Klorerte organiske forbindelser*. Miljøstatus. https://miljostatus.miljodirektoratet.no/tema/forurensning/advarsler-mot-fisk-og-sjomat/grenlandsfjordene---advarsel-om-miljogifter/

Nybom, I. (2015). *Activated carbon amendments for sediment remediation: Reduction of aquatic and biota concentrations of PCBs, and secondary effects on Lumbriculus variegatus and Chironomus riparius* [PhD Thesis]. Itä-Suomen yliopisto.

Nybom, I., Waissi-Leinonen, G., Mäenpää, K., Leppänen, M. T., Kukkonen, J. V. K., Werner, D., & Akkanen, J. (2015). Effects of activated carbon ageing in three PCB contaminated sediments: Sorption efficiency and secondary effects on Lumbriculus variegatus. *Water Research*, *85*, 413–421. https://doi.org/10.1016/j.watres.2015.08.044

Nybom, I., Werner, D., Leppänen, M. T., Siavalas, G., Christanis, K., Karapanagioti, H. K., Kukkonen, J. V. K., & Akkanen, J. (2012). Responses of Lumbriculus variegatus to Activated Carbon Amendments in Uncontaminated Sediments. *Environmental Science & Technology*, *46*(23), 12895–12903. https://doi.org/10.1021/es303430j

O'Connor, B., Bowmer, T., & Grehan, A. (1983). Long-term assessment of the population dynamics of Amphiura filiformis (Echinodermata: Ophiuroidea) in Galway Bay (west coast of Ireland). *Marine Biology*, *75*(2), 279–286. https://doi.org/10.1007/BF00406013

Oehme, M., Manø, S., & Bjerke, B. (1989). Formation of polychlorinated dibenzofurans and dibenzo-p-dioxins by production processes for magnesium and refined nickel. *Chemosphere*, *18*(7), 1379–1389. https://doi.org/10.1016/0045-6535(89)90029-5

Økland, T. E. (2005). *Kostholdsråd i norske havner og fjorder*. Bergfald & Co.

Olsgard, F. (1999). Effects of Copper Contamination on Recolonisation of Subtidal Marine Soft Sediments – an Experimental Field Study. *Marine Pollution Bulletin*, *38*(6), 448–462. https://doi.org/10.1016/S0025-326X(98)90202-8

Olsgard, F., & Gray, J. S. (1995). A comprehensive analysis of the effects of offshore oil and gas exploration and production on the benthic communities of the Norwegian continental shelf. *Marine Ecology Progress Series*, *122*, 277–306. https://doi.org/10.3354/meps122277

Ong, B., & Krishnan, S. (1995). Changes in the macrobenthos community of a sand flat after erosion. *Estuarine, Coastal and Shelf Science*, *40*(1), 21–33. https://doi.org/10.1016/0272-7714(95)90010-1

O'Reilly, R., Kennedy, R., & Patterson, A. (2006). Destruction of conspecific bioturbation structures by Amphiura filiformis (Ophiuroida): Evidence from luminophore tracers and in situ time-lapse sediment-profile imagery. *Marine Ecology Progress Series*, *315*, 99–111. https://doi.org/10.3354/meps315099

Patmont, C. R., Ghosh, U., LaRosa, P., Menzie, C. A., Luthy, R. G., Greenberg, M. S., Cornelissen, G., Eek, E., Collins, J., Hull, J., Hjartland, T., Glaza, E., Bleiler, J., & Quadrini, J. (2014). In situ sediment treatment using activated carbon: A demonstrated sediment cleanup technology. *Integrated Environmental Assessment and Management*, *11*(2), 195–207. https://doi.org/10.1002/ieam.1589

Pearson, T. H., Gray, J. S., & Johannessen, P. J. (1983). Objective selection of sensitive species indicative of pollution-induced change in benthic communities. 2. Data analyses. *Marine Ecology Progress Series*, *12*, 237–255. https://doi.org/10.3354/meps012237

Pearson, T., & Rosenberg, R. (1978). Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. *Oceanography and Marine Biology*, *16*, 229–311.

Pedersen, T., Nilsen, M., Nilssen, E. M., Berg, E., & Reigstad, M. (2008). Trophic model of a lightly exploited cod-dominated ecosystem. *Ecological Modelling*, *214*(2), 95–111. https://doi.org/10.1016/j.ecolmodel.2007.12.012

Peeters, E. T. H. M., Dewitte, A., Koelmans, A. A., van der Velden, J. A., & Besten, P. J. den. (2001). Evaluation of bioassays versus contaminant concentrations in explaining the macroinvertebrate community structure in the Rhine-Meuse delta, The Netherlands. *Environmental Toxicology and Chemistry*, *20*(12), 2883–2891. https://doi.org/10.1002/etc.5620201231

Perelo, L. W. (2010). Review: In situ and bioremediation of organic pollutants in aquatic sediments. *Journal of Hazardous Materials*, *177*(1), 81–89. https://doi.org/10.1016/j.jhazmat.2009.12.090

Persson, N. J., Gustafsson, Ö., Bucheli, T. D., Ishaq, R., & Broman, D. (2002). Soot-Carbon Influenced Distribution of PCDD/Fs in the Marine Environment of the Grenlandsfjords, Norway. *Environmental Science & Technology*, *36*(23), 4968–4974. https://doi.org/10.1021/es020072l

Pielou, E. C. (1966). The measurement of diversity in different types of biological collections. *Journal of Theoretical Biology*, *13*, 131–144. https://doi.org/10.1016/0022-5193(66)90013-0

Pilskaln, C. H., Churchill, J. H., & Mayer, L. M. (1998). Resuspension of Sediment by Bottom Trawling in the Gulf of Maine and Potential Geochemical Consequences. *Conservation Biology*, *12*(6), 1223–1229. https://doi.org/10.1046/j.1523-1739.1998.0120061223.x

Probert, P. K. (1984). Disturbance, sediment stability, and trophic structure of soft-bottom communities. *Journal of Marine Research*, *42*(4), 893–921. https://doi.org/10.1357/002224084788520837

Quinn, G. P., & Keough, M. J. (2002). *Experimental Design and Data Analysis for Biologists*. Cambridge University Press. https://books.google.no/books?id=VtU3-y7LaLYC

Rakowska, M. I., Kupryianchyk, D., Harmsen, J., Grotenhuis, T., & Koelmans, A. A. (2012). In situ remediation of contaminated sediments using carbonaceous materials. *Environmental Toxicology and Chemistry*, *31*(4), 693–704. https://doi.org/10.1002/etc.1763

Raymond, C., Samuelsson, G. S., Agrenius, S., Schaanning, M. T., & Gunnarsson, J. S. (2020). Impaired benthic macrofauna function 4 years after sediment capping with activated carbon in the Grenland fjords, Norway. *Environmental Science and Pollution Research*, *28*(13), 16181–16197. https://doi.org/10.1007/s11356-020-11607-0

Reiss, H., & Kröncke, I. (2005). Seasonal variability of benthic indices: An approach to test the applicability of different indices for ecosystem quality assessment. *Marine Pollution Bulletin*, *50*(12), 1490–1499. https://doi.org/10.1016/j.marpolbul.2005.06.017

Ricotta, C., & Podani, J. (2017). On some properties of the Bray-Curtis dissimilarity and their ecological meaning. *Ecological Complexity*, *31*, 201–205. https://doi.org/10.1016/j.ecocom.2017.07.003

Riisgård, H. U., Jürgensen, C., & Andersen, F. Ø. (1996). Case Study: Kertinge Nor. In *Eutrophication in Coastal Marine Ecosystems* (pp. 205–220). American Geophysical Union (AGU). https://doi.org/10.1029/CE052p0205

Ros, J. D., & Cardell, M. J. (1991). Effect on benthic communities of a major input of organic matter and other pollutants (coast off Barcelona, Western Mediterranean). *Toxicological & Environmental Chemistry*, *31*(1), 441–450. https://doi.org/10.1080/02772249109357719

Rosenberg, R., & Lundberg, L. (2004). Photoperiodic activity pattern in the brittle star Amphiura filiformis. *Marine Biology*, *145*(4), 651–656. https://doi.org/10.1007/s00227-004-1365-z

Rosenberg, Rutger, Nilsson, H. C., Hollertz, K., & Hellman, B. (1997). Density-dependent migration in an Amphiura filiformis (Amphiuridae, Echinodermata) infaunal population. *Marine Ecology Progress Series*, *159*, 121–131. https://doi.org/10.3354/meps159121

Rust, A. J., Burgess, R. M., McElroy, A. E., Cantwell, M. G., & Brownawell, B. J. (2004). Influence of soot carbon on the bioaccumulation of sediment-bound polycyclic aromatic hydrocarbons by marine benthic invertebrates: An interspecies comparison. *Environmental Toxicology and Chemistry*, *23*(11), 2594–2603. https://doi.org/10.1897/03-355

Ruus, A., Bakke, T. H., Bjerkeng, B., & Knutsen, H. (2013). *Overvåking av miljøgifter i fisk og skalldyr fra Grenlandsfjordene 2012* (Miljødirektorat Rapport:M-8/2013 NIVA-rapport:6571-2013). Norsk institutt for vannforskning. https://niva.brage.unit.no/niva-xmlui/handle/11250/198622

Ruus, A., Berge, J. A., Bergstad, O. A., Knutsen, J. A., & Hylland, K. (2006). Disposition of polychlorinated dibenzo-p-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs) in two Norwegian epibenthic marine food webs. *Chemosphere*, *62*(11), 1856–1868. https://doi.org/10.1016/j.chemosphere.2005.07.064

Ruus, A., Berge, J. A., Hylland, K., Bjerkeng, B., Bakke, T., & Næs, K. (2006). Polychlorinated Dibenzo-p-Dioxins (PCDDs) and Dibenzofurans (PCDFs) in the Grenland Fjords (Norway)—Disposition, Levels, and Effects. *Journal of Toxicology and Environmental Health, Part A*, *69*(1–2), 185–200. https://doi.org/10.1080/15287390500259509

Rygg, B. (1995). *Indikatorer for miljøtilstand på marin bløtbunn. Klassifisering av 73 arter/taksa. En ny indeks for miljøtilstand, basert på innslag av tolerante og ømfintlige arter på lokaliteten* (NIVA-rapport No. 3349–95). Norsk institutt for vannforskning. https://niva.brage.unit.no/niva-xmlui/handle/11250/208532

Rygg, B. (1998). *Endringer i trofinivå og individtetthet hos bløtbunnsfaunaen langs kysten av Sør-Norge* (NIVA-rapport No. 3885–98). Norsk institutt for vannforskning. https://niva.brage.unit.no/niva-xmlui/handle/11250/209990

Rygg, B. (2002). *Indicator species index for assessing benthic ecological quality in marine waters of Norway* (NIVA-Report No. 4548–2002). Norwegian Institute for Water Research. https://niva.brage.unit.no/niva-xmlui/handle/11250/211748

Rygg, B. (2006). Developing indices for quality-status classification of marine soft-bottom fauna in Norway. In *33* (NIVA-Report No. 5208–2006). Norwegian Institute for Water Research. https://niva.brage.unit.no/niva-xmlui/handle/11250/213219

Rygg, Brage. (1985). Effect of sediment copper on benthic fauna. *Marine Ecology Progress Series*, *25*, 83–89.

Rygg, Brage, & Norling, K. (2013). *Norwegian Sensitivity Index (NSI) for marine macroinvertebrates, and an update of Indicator Species Index (ISI)* (NIVA-Report No. 6475–2013). Norwegian Institute for Water Research. https://niva.brage.unit.no/niva-xmlui/handle/11250/216238

Salloum, M. J., Chefetz, B., & Hatcher, P. G. (2002). Phenanthrene Sorption by Aliphatic-Rich Natural Organic Matter. *Environmental Science & Technology*, *36*(9), 1953–1958. https://doi.org/10.1021/es015796w

Saloranta, T. M., Armitage, J. M., Haario, H., Næs, K., Cousins, I. T., & Barton, D. N. (2008). Modeling the Effects and Uncertainties of Contaminated Sediment Remediation Scenarios in a Norwegian Fjord by Markov Chain Monte Carlo Simulation. *Environmental Science & Technology*, *42*(1), 200–206. https://doi.org/10.1021/es070622l

Samuelsson, G. S., Hedman, J. E., Elmquist Kruså, M., Gunnarsson, J. S., & Cornelissen, G. (2015). Capping in situ with activated carbon in Trondheim harbor (Norway) reduces bioaccumulation of PCBs and PAHs in marine sediment fauna. *Marine Environmental Research*, *109*, 103–112. https://doi.org/10.1016/j.marenvres.2015.06.003

Samuelsson, G. S., Raymond, C., Agrenius, S., Schaanning, M., Cornelissen, G., & Gunnarsson, J. S. (2017). Response of marine benthic fauna to thin-layer capping with activated carbon in a large-scale field experiment in the Grenland fjords, Norway. *Environmental Science and Pollution Research*, *24*(16), 14218–14233. https://doi.org/10.1007/s11356-017-8851-6

Santos, S. L., & Simon, J. L. (1980). Response of soft-bottom benthos to annual catastrophic disturbance in a south Florida estuary. *Marine Ecology Progress Series*, *3*(4), 347–355.

Schaanning, M., & Allan, I. (2012). Field experiment on thin-layer capping in Ormefjorden and Eidangerfjorden, Telemark. Functional response and bioavailability of dioxins 2009-2011. In *31 + appendix* (NIVA-Rapport No. 6285–2012). Norwegian Institute for Water Research. https://niva.brage.unit.no/niva-xmlui/handle/11250/215761

Schaanning, M. T., Beylich, B., Gunnarsson, J. S., & Eek, E. (2021). Long-term effects of thin layer capping in the Grenland fjords, Norway: Reduced uptake of dioxins in passive samplers and sediment-dwelling organisms. *Chemosphere*, *264*, 128544. https://doi.org/10.1016/j.chemosphere.2020.128544

Schaanning, Morten, Beylich, B., Raymond, C., & Gunnarsson, J. S. (2014). *Thin layer capping of fjord sediments in Grenland. Chemical and biological monitoring 2009-2013* (NIVA-Report No. 6724–2014). Norwegian Institute for Water Research. https://niva.brage.unit.no/niva-xmlui/handle/11250/284003

Schaanning, Morten, Beylich, B., Samuelsson, G., Raymond, C., Gunnarsson, J., & Agrenius, S. (2011). *Field experiment on thin-layer capping in Ormefjorden and Eidangerfjorden; Benthic* *community analyses 2009-2011* (NIVA-Rapport No. 6257–2011). Norwegian Institute for Water Research. https://niva.brage.unit.no/niva-xmlui/handle/11250/215691

Schaanning, Morten, Trannum, H. C., Beylich, B., Raymond, C., & Størdal, I. F. (2019). *Undersøkelser av kjemisk utlekking og biota på testfelt på sjøbunnen i Grenlandsfjordene 2018-2019* (NIA-rapport No. 743–2019). Norsk institutt for vannforskning. https://niva.brage.unit.no/niva-xmlui/handle/11250/2643250

Schratzberger, M., Bolam, S., Whomersley, P., & Warr, K. (2006). Differential response of nematode colonist communities to the intertidal placement of dredged material. *Journal of Experimental Marine Biology and Ecology*, *334*(2), 244–255. https://doi.org/10.1016/j.jembe.2006.02.003

Schreiber, B., Brinkmann, T., Schmalz, V., & Worch, E. (2005). Adsorption of dissolved organic matter onto activated carbon—The influence of temperature, absorption wavelength, and molecular size. *Water Research*, *39*(15), 3449–3456. https://doi.org/10.1016/j.watres.2005.05.050

Shannon, C. E., & Weaver, W. W. (1963). *The Mathematical Theory of Communication*. University of Illinois Press.

Skei, J. (1981). The Entrapment of Pollutants in Norwegian Fjord Sediments—A Beneficial Situation for the North Sea. In *Holocene Marine Sedimentation in the North Sea Basin* (pp. 461–468). https://doi.org/10.1002/9781444303759.ch32

Skei, J. M. (1978). Serious mercury contamination of sediments in a Norwegian semi-enclosed bay. *Marine Pollution Bulletin*, *9*(7), 191–193. https://doi.org/10.1016/0025-326X(78)90177-7

Snelgrove, P. V. R. (1998). The biodiversity of macrofaunal organisms in marine sediments. *Biodiversity and Conservation*, *7*(9), 1123–1132. https://doi.org/10.1023/A:1008867313340

Solan, M., Cardinale, B. J., Downing, A. L., Engelhardt, K. A. M., Ruesink, J. L., & Srivastava, D. S. (2004). Extinction and Ecosystem Function in the Marine Benthos. *Science*, *306*(5699), 1177–1180. https://doi.org/10.1126/science.1103960

Solan, M., & Kennedy, R. (2002). Observation and quantification of in situ animal-sediment relations using time-lapse sediment profile imagery (t-SPI). *Marine Ecology Progress Series*, *228*, 179–191. https://doi.org/10.3354/meps228179

Somerfield, P. J., Rees, H. L., & Warwick, R. M. (1995). Interrelationships in community structure between shallow-water marine meiofauna and macrofauna in relation to dredgings disposal. *Marine Ecology Progress Series*, *127*, 103–112. https://doi.org/10.3354/meps127103

Stronkhorst, J., Ariese, F., van Hattum, B., Postma, J. F., de Kluijver, M., Den Besten, P. J., Bergman, M. J. N., Daan, R., Murk, A. J., & Vethaak, A. D. (2003). Environmental impact and recovery at two dumping sites for dredged material in the North Sea. *Environmental Pollution*, *124*(1), 17–31. https://doi.org/10.1016/S0269-7491(02)00430-X

Sun, X., & Ghosh, U. (2007). PCB Bioavailability Control in Lumbriculus Variegatus through Different Modes of Activated Carbon Addition to Sediments. *Environmental Science & Technology*, *41*(13), 4774–4780. https://doi.org/10.1021/es062934e

Thibodeaux, L. J., & Bierman, V. J. (2003). Peer Reviewed: The Bioturbation-Driven Chemical Release Process. *Environmental Science & Technology*, *37*(13), 252A-258A. https://doi.org/10.1021/es032518j

Thistle, D. (1981). Natural Physical Disturbances and Communities of Marine Soft Bottoms. *Marine Ecology Progress Series*, *6*(2), 223–228. https://doi.org/10.3354/meps006223

Thrush, S. F., & Dayton, P. K. (2002). Disturbance to Marine Benthic Habitats by Trawling and Dredging: Implications for Marine Biodiversity. *Annual Review of Ecology and Systematics*, *33*(1), 449–473. https://doi.org/10.1146/annurev.ecolsys.33.010802.150515

Thrush, S. F., Pridmore, R. D., Hewitt, J. E., & Cummings, V. J. (1992). Adult infauna as facilitators of colonization on intertidal sandflats. *Journal of Experimental Marine Biology and Ecology*, *159*(2), 253–265. https://doi.org/10.1016/0022-0981(92)90040-H

Trannum, Hilde C., Raymond, C., Næss, R., Borgersen, G., Gunnarsson, J. S., & Schaanning, M. T. (2021). Long-term response of marine benthic fauna to thin-layer capping with powdered activated carbon in the Grenland fjords, Norway. *Science of The Total Environment*, *776*, 145971. https://doi.org/10.1016/j.scitotenv.2021.145971

Trannum, Hilde Cecilie, Olsgard, F., Skei, J. M., Indrehus, J., Øverås, S., & Eriksen, J. (2004). Effects of copper, cadmium and contaminated harbour sediments on recolonisation of soft-bottom communities. *Journal of Experimental Marine Biology and Ecology*, *310*(1), 87–114. https://doi.org/10.1016/j.jembe.2004.04.003

Tuomisto, J. (2019). Dioxins and dioxin-like compounds: Toxicity in humans and animals, sources, and behaviour in the environment. *WikiJ Med*, *6*, 8.

USEPA. (2005). Chapter 6: Dredging and Excavation. In *Contaminated Sediment Remediation Guidance for Hazardous Waste Sites*. United States Environmental Protection Agency. https://www.epa.gov/nscep

Van Griethuysen, C., Van Baren, J., Peeters, E. T. H. M., & Koelmans, A. A. (2004). Trace metal availability and effects on benthic community structure in floodplain lakes. *Environmental Toxicology and Chemistry*, *23*(3), 668–681. https://doi.org/10.1897/02-583

Vannforskriften. (2006). *Forskrift om rammer for vannforvaltningen* (FOR-2006-12-15-1446). Lovdata. https://lovdata.no/forskrift/2006-12-15-1446

Vopel, K., Thistle, D., & Rosenberg, R. (2003). Effect of the brittle star Amphiura filiformis (Amphiuridae, Echinodermata) on oxygen flux into the sediment. *Limnology and Oceanography*, *48*(5), 2034–2045. https://doi.org/10.4319/lo.2003.48.5.2034

Walters, R. W., & Luthy, R. G. (1984). Equilibrium adsorption of polycyclic aromatic hydrocarbons from water onto activated carbon. *Environmental Science & Technology*, *18*(6), 395–403. https://doi.org/10.1021/es00124a002

Welsh, D. T. (2003). It's a dirty job but someone has to do it: The role of marine benthic macrofauna in organic matter turnover and nutrient recycling to the water column. *Chemistry and Ecology*, *19*(5), 321–342. https://doi.org/10.1080/0275754031000155474

White, P., & Pickett, S. T. A. (1985). *Natural Disturbance and Patch Dynamics: An Introduction*. 3–13. https://doi.org/10.1016/B978-0-08-050495-7.50006-5

Wilber, D. H., Clarke, D. G., & Rees, S. I. (2007). Responses of benthic macroinvertebrates to thin-layer disposal of dredged material in Mississippi Sound, USA. *Marine Pollution Bulletin*, *54*(1), 42–52. https://doi.org/10.1016/j.marpolbul.2006.08.042

Yeo, R. K., & Risk, M. J. (1979). Intertidal Catastrophes: Effect of Storms and Hurricanes on Intertidal Benthos of the Minas Basin, Bay of Fundy. *Journal of the Fisheries Board of Canada*, *36*(6), 667–669. https://doi.org/10.1139/f79-096

Zajac, R. N., Whitlatch, R. B., & Thrush, S. F. (1998). Recolonization and succession in soft-sediment infaunal communities: The spatial scale of controlling factors. *Hydrobiologia*, *375/376*, 227–240. https://doi.org/10.1023/A:1017032200173

Zimmerman, J. R., Ghosh, U., Millward, R. N., Bridges, T. S., & Luthy, R. G. (2004). Addition of Carbon Sorbents to Reduce PCB and PAH Bioavailability in Marine Sediments: Physicochemical Tests. *Environmental Science & Technology*, *38*(20), 5458–5464. https://doi.org/10.1021/es034992v

Zimmerman, J. R., Werner, D., Ghosh, U., Millward, R. N., Bridges, T. S., & Luthy, R. G. (2005). Effects of dose and particle size on activated carbon treatment to sequester polychlorinated biphenyls and polycyclic aromatic hydrocarbons in marine sediments. *Environmental Toxicology and Chemistry*, *24*(7), 1594–1601. https://doi.org/10.1897/04-368R.1

List of tables

List of figures

Appendix A: Species in Grenland 2018

List of species from all samples taken in the Grenland fjords 2018. Table page 1 of 3.

Species \ Fields:	FE5_A	FE5_B	FE5_C	FE5_D	FE6_A	FE6_B	FE6_C	FE6_D	FO3_A	FO3_B	FO3_C	FO3_D	FO4_B	FO4_C	FO4_D	FO4_E	Occurences	Tot. Ind.
Paramphinome jeffreysii	33	1	45	22	9	94	32	33									8	269
Aphrodita aculeata					1												1	1
Bylgides sarsi							1										1	1
Enipo kinbergi													1				1	1
Gattyana cirrhosa															1		1	1
Harmothoe sp.						1											1	1
Chaetoparia nilssoni	2		2			2	1	2									5	9
Nereiphylla lutea						1											1	1
Sige fusigera						1		1									2	2
Pholoe baltica								2									1	2
Pholoe pallida					1			2									2	3
Pholoe sp.	2						2										2	4
Oxydromus flexuosus							1	1									2	2
Glyphohesione klatti															1		1	1
Pilargis papillata						1											1	1
Exogone verugera				1		1											2	2
Ceratocephale loveni	3	7	4	3	1	3	2	2									8	25
Eunereis longissima		1					1	1									3	3
Nephtys incisa			1			1			3	4	3	7	3	4	4	1	10	31
Nephtys paradoxa				1													1	1
Nephtys sp.							1										1	1
Glycera alba	1	1	3		3	7	3	4					1				8	23
Goniada maculata	1		1			3	3	2					1		2		7	13
Abyssoninoe hibernica		1		1	11	12	11	11	2				4	5	2	3	11	63
Lumbrineridae indet											1						1	1
Lumbrineris aniara								1									1	1
Ophryotrocha sp.								1									1	1
Phylo kupfferi						1											1	1
Apistobranchus tullbergi		2		3													2	5
Levinsenia gracilis			1	3	3		6	5									5	18
Paradoneis eliasoni				2		1	1										3	4
Laonice sarsi							1	1									2	2
Prionospio cirrifera	12	2	5	1	11	41	3	9									8	84
Prionospio dubia		3			14	40	14	27					1				6	99
Prionospio fallax	2	5	1	2	1	2		5	1	1	1						10	21
Prionospio multibranchiata														4	2	5	3	11
Pseudopolydora aff. paucibranchiata	1							1									2	2
Spiophanes kroyeri	79	35	51	53	98	104	126	154						2		1	10	703
Magelona minuta									1								1	1
Chaetopterus variopedatus														1			1	1
Aphelochaeta marioni	16	12	16	12	19	11	6	13									8	105
Chaetozone setosa	46	30	42	56	13	7	3	10						1	1		10	209
Tharyx killariensis	2	2	3	3		1		1									6	12
Brada villosa			1		1												2	2
Diplocirrus glaucus	1	1			3	2	2	3		1		2	4	1		2	11	22
Polyphysia crassa					4	1	2	3						1	2	1	7	14
Scalibregma inflatum	2	3	7	4	2	1						1		1		3	9	24
Ophelina norvegica							1										1	1
Heteromastus filiformis	13	10	13	13	16	18	20	26	1					1			10	131

Euclymeninae indet							3	4									2	7
Maldanidae indet														1			1	1
Praxillella affinis					1	2		2									3	5
Rhodine loveni			1		2	2	3	3									5	11
Galathowenia oculata	1	2	2	3	20	1	2	1									8	32
Pectinaria belgica														1	1	3	3	5
Pectinariidae indet							1										1	1
Ampharete octocirrata						2											1	2
Eclysippe vanelli					2		1	4									3	7
Lysippe fragilis							1										1	1
Sosane wahrbergi		2	5	3	5	16	2	1									7	34
Sosane wireni							1										1	1
Eupolymnia nebulosa					1												1	1
Lysilla loveni													1				1	1
Paramphitrite tetrabranchia						1											1	1
Pista cristata							4	2									2	6
Pista lornensis	1		2														2	3
Polycirrus norvegicus																1	1	1
Polycirrus plumosus					1												1	1
Streblosoma bairdi					1	2	2	5									4	10
Terebellides stroemii	2	2	2		1			1							1	1	7	10
Trichobranchus roseus											2			1	1		3	4
Chone sp.	3		4		1												3	8
Euchone papillosa	1	2															2	3
Jasmineira caudata						1											1	1
Ostracoda indet		2		2													2	4
Cumacea indet		1															1	1
Eudorella emarginata	1	7	5	8				1									5	22
Eudorella truncatula			2	1													2	3
Leucon (Leucon) nasica	8	4	9	6		1											5	28
Leucon sp.		1			1												2	2
Diastylis cornuta						1											1	1
Tanaidacea indet	1	3	4	2													4	10
Gnathia maxillaris	1		2	4													3	7
Idotea sp.								1									1	1
Eugerda tenuimana					1	1		1									3	3
Ampelisca gibba					2	2	1	2									4	7
Ampelisca sp.					1												1	1
Leucothoe lilljeborgi					1	2		1									3	4
Eriopisa elongata		1		4	1	1		3							2		6	12
Arrhis phyllonyx	1	1		1													3	3
Westwoodilla caecula	1				1	3		1								1	5	7
Harpinia crenulata	1	1	1			2		2									5	7
Harpinia pectinata						1											1	1
Harpinia sp.				3		2											2	5
Pardaliscidae					1												1	1
Aoridae indet					1												1	1
Autonoe longipes					1												1	1
Processa canaliculata													1		1	2	3	4
Callianassa subterranea	1								2				1	3	2	7	6	16

Table page 3 of 3

Liocarcinus depurator															1		1	1
Amphiura chiajei	1			1	3	3	7	6		1	1					1	9	24
Amphiura filiformis								1					36	43	46	31	5	157
Amphiura sp. juvenil		1	1		1		3	2				1	1			14	8	24
Ophiura sp. juvenil				1													1	1
Brissopsis lyrifera							1	1						2		1	4	5
Echinocardium cordatum														1	1		2	2
Hyala vitrea				3			1		5	5	4	3	8		11	1	9	41
Aporrhais pespelecani									1								1	1
Raphitoma sp.												1					1	1
Hermania scabra		1															1	1
Philine quadripartita																1	1	1
Philine sp.												1					1	1
Cylichna cylindracea													1		1	3	3	5
Caudofoveata indet														2			1	2
Chaetoderma nitidulum						1		2									2	3
Ennucula tenuis					1	2											2	3
Nucula nitidosa									6	3	18	1	1		1		6	30
Nucula sp.																1	1	1
Yoldiella philippiana	2	2	2	2	5		1	10									7	24
Yoldiella sp.					1							1					2	2
Limidae indet					1												1	1
Mendicula ferruginosa	1	1		1		2	2	4									6	11
Thyasira equalis	11	10	7	9	14	23	18	24									8	116
Thyasira flexuosa										4	2					1	3	7
Thyasira sarsii	2																1	2
Thyasira sp.					6							1					2	7
Thyasira sp. juvenil	6	5	6	11	7	7	7	8			1						9	58
Tellimya tenella							2	1					1				3	4
Parvicardium minimum							3										1	3
Abra nitida	1	4	4	3			8		1	1		1	1		1		10	25
Corbula gibba									1	1	4	2	2	1		2	7	13
Cuspidaria cuspidata																1	1	1
Tropidomya abbreviata	1							1									2	2
Cerianthus lloydii				1													1	1
Nemertea indet	9	4	6	5	2	4	1	3						1	2	1	11	38
Golfingia (Golfingia) vulgaris vulgaris												1			1		2	2
Nephasoma (Nephasoma) minutum				1	1												2	2
Onchnesoma squamatum				1													1	1
Onchnesoma steenstrupii steenstrupii			1														1	1
Phoronida indet												2					1	2
Chaetognatha indet			2			1											2	3
Total numbe of individuals	**273**	**173**	**264**	**256**	**299**	**443**	**318**	**419**	**24**	**21**	**37**	**25**	**69**	**77**	**88**	**89**		
Total number of species	**38**	**37**	**36**	**39**	**49**	**51**	**46**	**55**	**12**	**10**	**11**	**15**	**19**	**21**	**24**	**26**		

Appendix B: Biomass in Grenland 2018

Measured biomass for all species in all samples collected in the Grenland fjords 2018. Table page 1 of 3.

Species \ Field:	FE5_A	FE5_B	FE5_C	FE5_D	FE6_A	FE6_B	FE6_C	FE6_D	FO3_A	FO3_B	FO3_C	FO3_D	FO4_B	FO4_C	FO4_D	FO4_E	Sum
Polychaeta	0.0369	0.0363	0.0341	0.0079	0.0311	0.0444	0.0462	0.0673	0.0002	0.0006	0.0004	0.0008	0.0004	0.0607	0.0009	0.0006	**0.3688**
Ampharete octocirrata						0.0037											**0.0037**
Eclysippe eliasoni					0.0057		0.0019	0.0066									**0.0142**
Lysippe fragilis							0.0034										**0.0034**
Mugga wahrbergi		0.0006	0.0011	0.0006	0.0006	0.0042	0.0002	0.0002									**0.0075**
Sosane wireni							0.0006										**0.0006**
Paramphinome jeffreysi	0.0399	0.0012	0.0513	0.0226	0.0101	0.1552	0.0519	0.0518									**0.3840**
Aphrodita aculeata					0.0179												**0.0179**
Apistobranchus tullbergi		0.0006		0.0015													**0.0021**
Heteromastus filiformis	0.0204	0.0106	0.0167	0.0204	0.0369	0.0493	0.0464	0.0829	0.0007					0.0004			**0.2847**
Chaetopterus variopedatus														1.6279			**1.6279**
Aphelochaeta marioni	0.2618	0.1361	0.1147	0.1111	0.2061	0.1403	0.0329	0.0727									**1.0757**
Chaetozone setosa	0.2251	0.1663	0.1883	0.2488	0.0313	0.0241	0.0078	0.0256						0.0014	0.0044		**0.9231**
Tharyx killariensis	0.0054	0.0038	0.0054	0.0059		0.0012		0.0002									**0.0219**
Ophryotrocha								0.0006									**0.0006**
Brada villosa			0.0002		0.0469												**0.0471**
Diplocirrus glaucus	0.0011	0.0142			0.0147	0.0127	0.0049	0.0176		0.0011		0.0019	0.0141	0.0233		0.0136	**0.1192**
Glycera alba	0.0185	0.1463	0.2956		0.0973	0.0627	0.0124	0.3739					0.1098				**1.1165**
Goniada maculata	0.0164		0.0002			0.0309	0.0493	0.0471					0.0186		0.0289		**0.1914**
Oxydromus flexuosus							0.0186	0.0352									**0.0538**
Abyssoninoe hibernica		0.0234		0.0054	0.6429	0.1556	0.2459	0.1798	0.0279				0.0539	0.0778	0.0196	0.0482	**1.4804**
Lumbrineridae											0.0031						**0.0031**
Lumbrineris aniara								0.0074									**0.0074**
Magelona minuta									0.0003								**0.0003**
Euclymeninae							0.0089	0.0071									**0.0160**
Maldanidae														0.0227			**0.0227**
Praxillella affinis					0.0008	0.0183		0.0164									**0.0355**
Rhodine loveni			0.0179		0.0442	0.1477	0.0801	0.2904									**0.5803**
Nephtys							0.0369										**0.0369**
Nephtys incisa			0.0609			0.1121			0.0763	0.1885	0.1381	0.4651	0.0119	0.0966	0.1002	0.0536	**1.3033**
Nephtys paradoxa				0.0094													**0.0094**
Ceratocephale loveni	0.0128	0.0362	0.0291	0.0069	0.0032	0.0382	0.0035	0.0096									**0.1395**
Eunereis longissima		0.0033					0.0057	0.0254									**0.0344**
Ophelina norvegica							0.0096										**0.0096**
Phylo kupfferi						0.9527											**0.9527**
Galathowenia oculata	0.0014	0.0039	0.0026	0.0363	0.0655	0.0023	0.0019	0.0003									**0.1142**
Levinsenia gracilis			0.0002	0.0021	0.0011		0.0031	0.0022									**0.0087**
Paradoneis eliasoni				0.0004		0.0005	0.0004										**0.0013**
Pectinaria belgica														0.0014	0.0003	0.0007	**0.0024**
Chaetoparia nilssoni	0.0016		0.0008			0.0025	0.0003	0.0086									**0.0138**
Nereiphylla lutea						0.0027											**0.0027**
Sige fusigera						0.0006		0.0017									**0.0023**
Glyphohesione klatti															0.0015		**0.0015**
Pilargis papillata						0.0012											**0.0012**
Bylgides sarsi							0.0008										**0.0008**
Enipo kinbergi													0.0185				**0.0185**
Gattyana cirrhosa															0.0028		**0.0028**
Harmothoe						0.0015											**0.0015**
Chone	0.0006		0.0007		0.0001												**0.0014**

Table page 2 of 3

Euchone papillosa	0.0003	0.0052															**0.0055**
Jasmineira caudata						0.0008											**0.0008**
Polyphysia crassa					2.1333	0.2499	0.5887	1.2701						1.1428	1.4566	0.8479	**7.6893**
Scalibregma inflatum	0.0071	0.0075	0.0267	0.0914	0.0142	0.0319						0.0047		0.0274		0.0456	**0.2565**
Pholoe	0.0058						0.0045										**0.0103**
Pholoe baltica								0.0048									**0.0048**
Pholoe pallida					0.0011			0.0098									**0.0109**
Laonice sarsi							0.0008	0.0065									**0.0073**
Prionospio cirrifera	0.0206	0.0017	0.0046	0.0009	0.0085	0.1299	0.0018	0.0065									**0.1745**
Prionospio dubia		0.0062			0.0387	0.1934	0.0662	0.0747					0.0031				**0.3823**
Prionospio fallax	0.0013	0.0021	0.0001	0.0011	0.0006	0.0018		0.0046	0.0000	0.0004	0.0005						**0.0126**
Prionospio multibranchiata														0.0033	0.0011	0.0042	**0.0086**
Pseudopolydora paucibranchiata	0.0012							0.0006									**0.0018**
Spiophanes kroeyeri	1.2855	0.9027	1.2153	2.4835	1.8686	2.9467	1.2076	1.4947						0.0323		0.0007	**13.4376**
Exogone verugera				0.0004		0.0004											**0.0008**
Eupolymnia nebulosa					0.0093												**0.0093**
Lysilla loveni													0.0582				**0.0582**
Paramphitrite tetrabranchia						0.0004											**0.0004**
Pista cristata							0.4068	0.1947									**0.6015**
Pistella lornensis	0.0044		0.0048														**0.0092**
Polycirrus norvegicus																0.0533	**0.0533**
Polycirrus plumosus					0.0051												**0.0051**
Streblosoma bairdi					0.2459	0.5467	0.6811	1.1171									**2.5908**
Terebellides stroemi	0.0026	0.0134	0.0084		0.0069			0.0009							0.0034	0.0015	**0.0371**
Trichobranchus roseus											0.0079			0.0161	0.0227		**0.0467**
Cumacea		0.0003															**0.0003**
Tanaidacea	0.0006	0.0003	0.0003	0.0003													**0.0015**
Ampelisca					0.0005												**0.0005**
Ampelisca gibba					0.0023	0.0016	0.0005	0.0017									**0.0061**
Aoridae					0.0004												**0.0004**
Autonoe longipes					0.0003												**0.0003**
Callianassa subterranea	0.0002								0.0018				0.0216	0.2171	0.2850	0.1657	**0.6914**
Eugerda tenuimana					0.0000	0.0002		0.0000									**0.0003**
Diastylis cornuta						0.0065											**0.0065**
Gnathia	0.0001		0.0012	0.0017													**0.0030**
Idoteidae								0.0004									**0.0004**
Eudorella emarginata	0.0020	0.0051	0.0067	0.0126				0.0002									**0.0266**
Eudorella truncatula			0.0003	0.0002													**0.0005**
Leucon		0.0009			0.0005												**0.0014**
Leucon nasica	0.0079	0.0030	0.0078	0.0083													**0.0270**
Leucon nasica						0.0002											**0.0002**
Leucothoe lilljeborgii					0.0012	0.0021		0.0008									**0.0041**
Eriopisa elongata		0.0014		0.0027	0.0003	0.0009		0.0010							0.0020		**0.0083**
Arrhis phyllonyx	0.0074	0.0088		0.0087													**0.0249**
Westwoodilla caecula	0.0008				0.0057	0.0105		0.0019								0.0005	**0.0194**
Pardaliscidae					0.0001												**0.0001**
Harpinia				0.0022		0.0005											**0.0027**
Harpinia crenulata	0.0004	0.0002	0.0004			0.0001		0.0002									**0.0013**
Harpinia pectinata						0.0003											**0.0003**
Liocarcinus depurator															0.1959		**0.1959**

Processa canaliculata													0.0041		0.0026	0.0023	**0.0090**
Ostracoda		0.0007		0.0012													**0.0019**
Chaetognatha			0.0029			0.0003											**0.0032**
Cerianthus lloydii				0.0012													**0.0012**
Brissopsis lyrifera							8.6403	9.1700						18.0282		1.0267	**36.8652**
Echinocardium cordatum														1.1343	6.3074		**7.4417**
Amphiura Arm	0.1965	0.0003	0.0163	0.2384	0.1483	0.4864	0.4964	0.6897		0.4920	0.3052	0.0008	2.0175		3.5397	2.0361	**10.6636**
Amphiura		0.0004	0.0004		0.0002		0.0007	0.0001				0.0006	0.0000			0.0032	**0.0056**
Amphiura														4.9704			**4.9704**
Amphiura chiajei	0.0941			0.1209	0.1402	0.1924	0.2785	0.2734		0.1379	0.1528					0.0030	**1.3932**
Amphiura filiformis								0.0007					0.7310	1.0888	0.9946	0.8952	**3.7103**
Ophiura				0.0002													**0.0002**
Parvicardium minimum							0.0032										**0.0032**
Corbula gibba									0.0006	0.0008	0.0162	0.0110	0.0052	0.0017		0.0782	**0.1137**
Cuspidaria cuspidata																0.0267	**0.0267**
Tropidomya abbreviata	0.0003							0.0268									**0.0271**
Tellimya tenella							0.0079	0.0038					0.0037				**0.0154**
Limidae					0.0000												**0.0000**
Ennucula tenuis					0.0012	0.0208											**0.0220**
Nucula																0.0000	**0.0000**
Nucula nitidosa									0.0304	0.0087	0.1036		0.0068		0.0010		**0.1505**
Pectinidae							0.0004										**0.0004**
Abra nitida	0.0012	0.0028	0.0725	0.0039			0.0071		0.0417	0.0917		0.0353	0.0005		0.0032		**0.2599**
Mendicula ferruginosa	0.0027	0.0034		0.0070		0.0083	0.0096	0.0186									**0.0496**
Thyasira	0.0522	0.0103	0.0564	0.0171	0.0422	0.0355	0.0200	0.0264			0.0018	0.0132					**0.2751**
Thyasira equalis	0.4764	0.3874	0.2893	0.3555	0.2442	0.5434	0.3218	0.4591									**3.0771**
Thyasira flexuosa											0.0326					0.0052	**0.0378**
Thyasira sarsii	0.2964																**0.2964**
Yoldiella					0.0013							0.0012					**0.0025**
Yoldiella philippiana	0.0082	0.0079	0.0141	0.0104	0.0143		0.0023	0.0374									**0.0946**
Caudofoveata														0.0941			**0.0941**
Chaetoderma nitidulum						0.0154		0.0066									**0.0220**
Aporrhais pespelecani									3.5488								**3.5488**
Cylichna cylindracea													0.0063		0.0080	0.0363	**0.0506**
Hyala vitrea				0.0059			0.0023		0.0121	0.0120	0.0105	0.0093	0.0246		0.0370	0.0035	**0.1172**
Hermania scabra		0.1239															**0.1239**
Philine												0.0227					**0.0227**
Philine quadripartita																0.0072	**0.0072**
Raphitoma												0.0066					**0.0066**
Nematoda			0.0000	0.0001	0.0002	0.0012	0.0004	0.0008									**0.0028**
Nemertea	0.0221	0.0034	0.0124	0.0096	0.0089	0.0039	0.0006	0.0044						0.0767	3.2314	0.0007	**3.3741**
Phoronida												0.0067					**0.0067**
Golfingia vulgaris												0.1306			0.1817		**0.3123**
Nephasoma minutum				0.0002	0.0003												**0.0005**
Onchnesoma squamatum				0.0024													**0.0024**
Onchnesoma steenstrupi			0.0012														**0.0012**
Total biomass	**3.1402**	**2.0821**	**2.5619**	**3.8673**	**6.2013**	**7.3970**	**13.4231**	**16.2397**	**3.7408**	**0.9337**	**0.7727**	**0.7105**	**3.1098**	**28.7454**	**16.4319**	**5.3604**	**114.7180**