Master‘s thesis

Ecological quality assessment for Pollurinn (Ísafjörður) by using biotic indices

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Declaration

I hereby confirm that I am the sole author of this thesis and it is a product of my own academic research.

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Arastou Gharibi

Abstract

Assessment of impacts is a fundamental element in environmental management and ecological indices are tools commonly used in this matter. Diversity indices such as Shannon-Weaver were commonly used for estimating these impacts. New developed biotic indices; AMBI and M-AMBI were applied for the first time as indicators of anthropogenic disturbances in Pollurinn of Skutulsfjörður (Ísafjordur) according to the European Water Framework criteria. The benthic macro-faunal community were studied to answer these questions: 1-What is the environmental health status in Pollurinn area? 2-Has the environmental health changed between the two sampling years? To answer these questions two sets of data from Náttúrustofa Vestfjarða study were used. Sampling regarding this study were conducted in 1997 and repeated in 2010. Indices delivered similar health classifications, categorizing sample stations between moderate and good. M-AMBI was used as the main indicator and its results showed: 1-Moderate health quality for the habitats close to the sewage outlets and good health quality for the central parts of Pollurinn and 2-About 21% quality deterioration in the area adjacent to the sewage outlets and about 10% improvement in central parts of the Pollurinn area.

Útdráttur

Mat á áhrifum er grundvallaratriði í umhverfisstjórnun og vistfræðilegir matstuðlar eru verkfæri mikið notuð í þeim tilgangi. Fjölbreytnistuðlar eins og Shannon-Weaver voru venjulega notaðir til að meta þessi áhrif. Nýlega þróaðir líffræðistuðlar; AMBI og M-AMBI hafa verið notaðir erlendis sem vísar á umhverfisáhrif manna. Hér voru þeir notaðir í fyrsta skipti í þessari rannsókn sem vísar á áhrif manna á umhverfisástands Pollsins í Skutulsfirði í samræmi við viðmið vatnatilskipunar Evrópu. Botndýrasamfélögin voru rannsökuð til að svara eftirfarandi spurningum: 1 – Hvert er umhverfisástand Pollsins? 2- Hefur ástand umhverfisins breyst milli sýnatökuára? Til að svara þessum spurningum voru notuð tvö gagnasöfn frá Náttúrustofu Vestfjarða. Sýni voru tekin árið 1997 og sýnataka endurtekin 2010. Matstuðlar sem voru notaðir gáfu svipaða niðurstöðu um umhverfisástand og var ástand mismunandi sýnatökustöðva flokkað sem meðalgott eða gott. Samkvæmt M-AMBI stuðlinum voru niðurstöður þessar: 1 – Meðalgott umhverfisástand stöðva nálægt útrásum skólps og gott umhverfisástand fyrir stöðvar í miðjum Pollinum.2 – Um það bil 21% lakara ástand á stöðvum nálægt útrásum og 10% betra ástand í miðjum Pollinum.

I dedicate this study to the people of the Westfjords

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1 Introduction

1-1 Coastal and marine management

Sustainable development as defined by the Brundtland Commission is such "a development that meets the need of the present without compromising the ability of future generation to meet their own needs" (WCED, 1987). The basic issue is to define the threshold in which an ecosystem can function and provide the community with services while guaranteeing next generations' potential right (Pinto et al., 2009). Environmental services such as bio-depuration and fishery can be sustained when sound methods and protocols are available to survey the status and trends in ecosystems. In this concern some concepts and legislations has been established.

The DPSIR (drivers–pressure–state–impact–response) approach has widely been used globally to deal with socio-environmental issues. Driving forces (D) of social and economic development exert Pressure (P) on the environment which may change its state (S) and resultant impacts (I) on ecosystems and human health necessitates managerial response (R) (Borja and Dauer, 2008). If the impact component of the DPSIR induces particular managerial responses, their ultimate goal of related environmental management is to execute the best possible practice based on relevant legislations adopted to infer socio-economic growth while protecting environment. The assessment of impacts despite of evaluating managerial functions, verifies the ecological quality of ecosystems and when compared to legislative standards determines the necessity of remediation actions. Legislations such as the European Water Framework Directive (WFD) and the European Marine Strategy Framework Directive 2008(56/EC) have been put in practice to establish the basis for assessing the ecological integrity of estuarine and coastal waters. In this concern the WFD (Directive 2000/60/EC) has established the concept of Ecological Quality Status (EQS) for the assessment of the ecological quality of all European coastal water

bodies. According to the WFD, the water quality status should be classified into one of the classes ranged among high, good, moderate, poor or bad and all European waters should reach the good to high status by 2015. One of the mutual approaches of the WFD and the DPSIR is the development of indices to determine the ecological quality of aquatic systems. Indices are considered useful tools in decision making processes while they (I) describe the cumulative pressure affecting the ecosystem, (II) can evaluate both the state of the ecosystem and the successfulness of managerial responses and, (III) facilitate the communication of complex issues to non-specialist audiences (Pinto et al., 2009). This paper takes advantage of a repeated study performed by Náttúrustofa Vestfjarða regarding the sewage impact, to implement a new methodology based on biotic indices.

1-2 Study area

Ísafjörður is a town located in the Westfjords region of Iceland in the inner part of Skutulsfjörður (fig. 1). This fjord is one of series of minor fjords of glacier origin connected to a major one directly open to the ocean named Ísafjardardjup. The town stretched along the coasts but the older part of it is situated on a peninsula-shaped land dividing the fjord into two parts: Pollurinn in the inner part is confined by the peninsula and is nourished by some small rivers from the surrounding mountains. The area is used as a harbor for fishing and recreational fleets and is connected to the open sea (outer part) by a narrower canal of about 400 meters width and 1.5 kilometers long. Along this canal, there is a small port which accommodates touristic, fishing boats and containerships. The town had a population of 2699 individuals in July 2010 which had a slight decrease (12%) from July 1997 that had 3046 inhabitants (Hagstofa Íslands, 2011). Traditionally, several effluents discharge untreated sewage into the coastal and intertidal areas. Discharges in the Pollurinn area are mainly originated from urban and fishing boats activities. Since this area is less affected by currents that usually provide rarefaction, the sewage material may be trapped and accumulate in the sediments. In terms of natural factors, the Pollurinn is ocean-dominated and rivers' flows are not at a rate that can reach the bottom or cause significant change in salinity which is stable in the range of euhaline (30-35).

Figure 1: Ísafjörður and Pollurinn in Skutulsfjörður. Photo by Magnús Einarsson.

Grain size, current energy fluctuation and depth are considered the main natural variables in the sample stations (Table 5). Closest station to the open part of fjord (station G) may be exposed to higher hydrological energy compared to the stations B and D that are more sheltered (fig. 2). As it is going to be discussed, the bottom hydrodynamic regime in sample stations is in doubt and needs to be considered.

Anthropogenic impact on this aquatic environment, if significant, is deemed to be due to urban waste especially because of the lack of wastewater treatment facilities, and other applications such as harbor activities. The main sewage effluents entering this area (fig. 2) are supposed to be the major agent of pollution in this water body, therefore all three sample stations are selected in a unidirectional pattern whereas station B is the closest and station G is the most distant from the discharge area. Load of pollution originated from these effluents are considered one of the major concerns of the community since the area is confined and harbored from the main body of fjord preventing the waste material to be efficiently diluted and may sink and accumulate on the sea bottom.

There has been a debate in the community that the sewage discharge in this semi-closed area may induce organic matter accumulation and negative environmental consequences on this eco-region consequently there has been a question about the effects of the sewage discharge on Pollurinn area's aquatic ecosystem.

It will be the municipality's function (managerial Response) to avoid or mitigate the impact of these point sources of pollution. Regarding this managerial decision making this study is aimed to objectively determine the ecological quality of this area and to compare it with the last decade. The last survey was implemented in 1997 and the results are published in a report by Helgason et al. (2002) and since that time (by 2010) no environmental assessments have been carried out in this area. This study is going to be implemented: (1) to assess the long-term effect of wastewater discharges (State due to Pressure in DPSIR approach) and other pollution sources, (2) to evaluate the potential changes in environmental quality from previous decade on, and (3) to start monitoring background in the region to let efficient managerial decisions are made for prospective developments (e.g. aquaculture, engineering works), this study is to use a set of appropriate environmental indices to evaluate the health quality of pollurinn-Ísafjordur aquatic habitats (Impact of Pressure in the DPSIR approach).

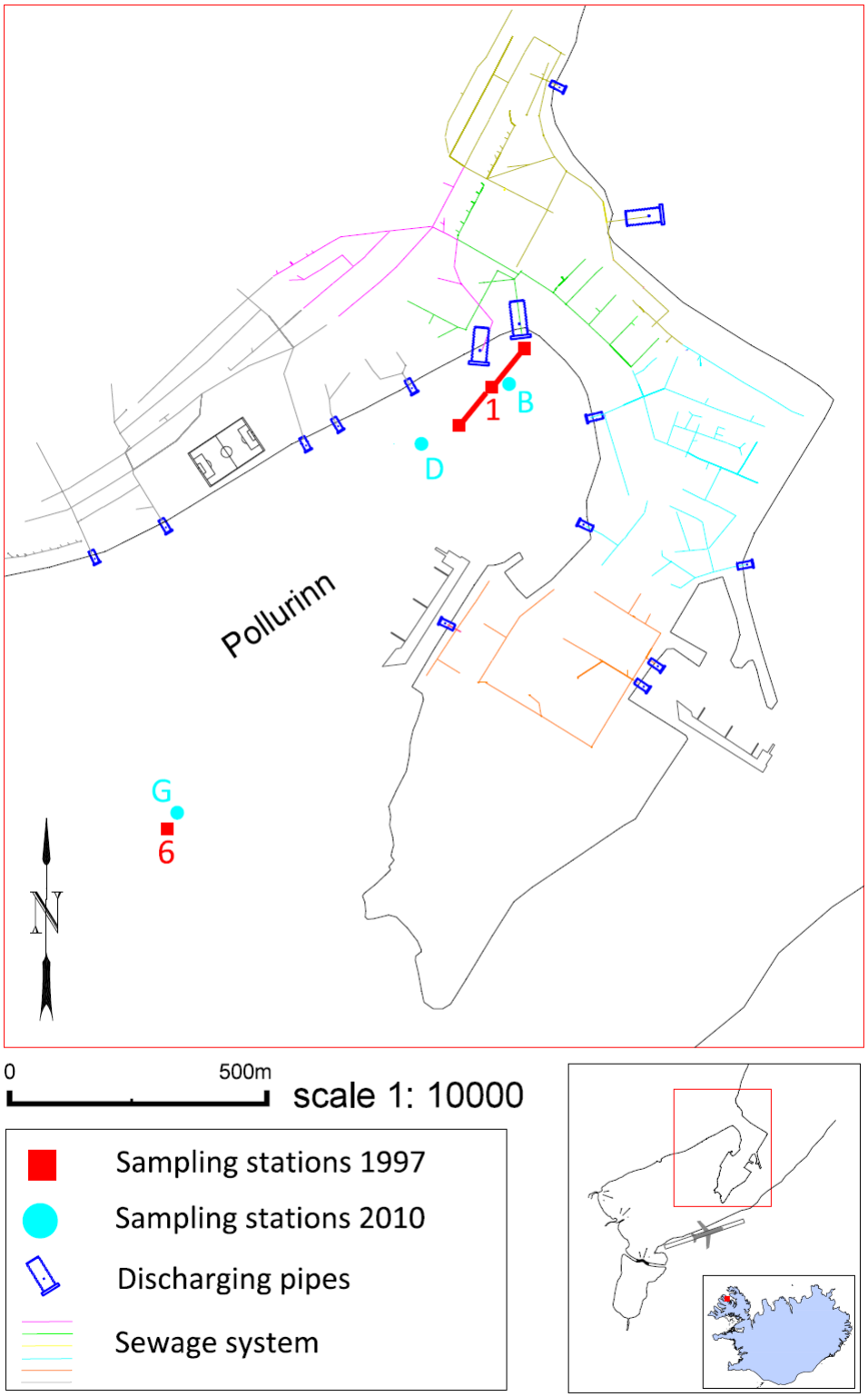

Figure 2: Ísafjörður and Pollurin area. Position of sampling stations of 1997 and 2010 studies. Sewage system provided by Ísafjarðarbær (modified).

1-3 Purpose of this study

Based on the WFD criteria, this Master's thesis is going to investigate and assess the biological integrity of the benthic macro-faunal community to give account of the ecological quality status and health condition of the Pollurinn water body. Species abundance, diversity and ecological status of the benthic macro-fauna communities were used as principal data to establish health status classification in each selected habitat in the area. In addition, historical records of adjacent pristine fjords will be reviewed to find unpolluted habitats with the same environmental feature of study habitats (e.g. depth and grain size) to be used as the basis for health classification with using M-AMBI software.

This study is going to answer two major questions: What is the environmental health status in Pollurinn area? Has the environmental health changed within the last 13 years?

2 Literature review

2-1 Marine- invertebrates and benthic indices

Benthic Sediment especially in coastal areas adjacent to urban areas is a place of dumping various sorts of wastes which may affect species in water column and sediment. These negative effects regardless of their degree and variety can be mirrored on the sediment and influence its inhabitants. Attention in applying macro-faunal species origins from their specific characteristics introducing them as useful biological indicators. Rosenberg et al, (2004) states that while most of the ecological impact and contamination load will finally end up on the sea bottom, studying sedimentary habitat and relative fauna is an effective way to assess the ecological changes in the sea. Having adjusted to tackle with predicted environmental fluctuations and inter-specific competition (Rosenberg et al., 2004), they have shown predictable responses to various natural and anthropogenic disturbances (Dauer, 1993; Montagna and Ritter, 2006). In addition several other supportive characteristics of macro-benthos usefulness is mentioned in recent studies such as; macro-benthic creatures have a sedentary nature unable to avoid deteriorating water-sediment threshold quality, they have a fairly long life-period therefore their reactions integrate water and sediment quality alteration over time (Reiss and Kröncke, 2005), they quickly respond to disturbances (Cruz-Motta and Collins, 2004), they have different degree of tolerance to stress (Bakalem et al, 2009), and they have a key role in nutrient cycling between sediment and water body (Dauvin et al., 2007). Given these attributes, macro-benthic taxa provide the basis for biological assessment methods that can be adjusted for application in different geographical areas (Weisberg et al., 1997; Borja et al., 2004b). Their application for estimating sedimentary and aquatic quality are mirrored in researches such as Pearson and Rosenberg, 1978; Gray and Mirza, 1979; Pearson et al., 1983; Hily, 1984; Rygg, 1985; Warwick, 1986; Majeed, 1987; Dauer, 1993; Grall and Glémarec, 1997, which have

provided scientific background to develop novel ecological quality (EcoQ) assessment methods after WFD (2000) implementation. Considering WFD obligation, these evaluative methods use benthic indices to translate community structure elements into a quality category (Rosenberg et al., 2004; Ruellet and Dauvin, 2007; Muxika et al., 2007) summarizing environmental status to a number, which allows for management decisions concerning environmental conditions (Borja and Tunberg, 2011). The WFD stated that the presence of sensitive and tolerant to pollution taxa, plus diversity and abundance of the benthic fauna, should be used for estimating the condition of sedimentary habitats (Leonardsson, et al., 2009). Some of the benthic indices developed after the WFD implementation, have been reviewed in some researches (see Ruellet and Dauvin, 2007; Pinto et al., 2009).

2-2 Environmental indicators

Borja and Dauer, (2008) have reviewed the essential characteristics of a proper index to be used to determine the impact component of DPSIR management approach.

1-Ecologically relevant; according to a well-proven theoretical model

2-Practical; data required for index calculations can be collected reliably and cost-effectively

3-Reference value measurable; the significance of indicator value can be assessed

4-Representative; policy decision feedbacks both current status and trends are measureable

5-Sensitivity; degradative and restorative inclination relevant to anthropogenic actions are traceable.

The essence for most recent ecological indices is a model described by Pearson and Rosenberg (1978) (P-R model); In a unidirectional gradient of disturbance (in original theme was organic enrichment) as intensity increases, in response, an individual according

to its abilities first, adjusts to the change and then will be replaced by a better adjusted individual which its dimension of tolerance to that kind of stress is higher. Considering this paradigm, individuals can be arranged based on their tolerance and sensitivity. Sensitive species occur in areas with no or low disturbance while tolerant species occur in disturbed environments (Rosenberg et al., 2004), similarly, sensitive species are mainly found in samples with high diversity and tolerant species are found mostly in samples with low diversity (Rygg, 2002). Thus, the benthic community's structure and abundance change gradually along gradients of disturbance (Leonardsson et al., 2009).

Apart from theoretical basis, other relative environmental indices' characteristics are concerned managerial practices and approaches which the best practices in terms of practicality, representativeness and sensitivity can be tracked in methods and approaches researchers have performed.

2-3 Use of benthic indices in ecological quality assessment

To estimate ecosystem status and condition, and considering the scope of the Water Framework Directive implementation, there has been a growing demand for reliable and robust ecological indices (Pinto, et al., 2009; Bigot et al, 2008).

The implementation of the water framework directive and the prospective goal of gaining good quality status for all European waters by 2015 have triggered debates regarding marine pollution issue and the development of different environmental indicators to estimate the Ecological Quality of aquatic areas. Benthic indices have been in the centre of attention and the subject of countless papers in terms of (I) developing new indicators to assess the Ecological Quality Status of marine and estuarine environments (e.g. Borja et al., 2000; Simboura and Zenetos, 2002 ; Rosenberg et al., 2004; Grall and Glémarec, 2005), (II) objectively reviewing the efficiency of indicators and comparing different classes of indices(Afli et al., 2008;Pinto, et al., 2009; Bakalem et al., 2009), (III) adjustment and

inter-calibration of already established indices in the areas other than their host habitat (Ruellet and Dauvin, 2007; Simboura and Argyrou, 2010).

Among the first used indices were diversity indices which, however were mostly developed in 50's, are still considered useful metrics for studying structure of the benthic invertebrates to evaluate the quality of coastal and marine habitats. Shannon diversity index (Shannon and Weaver, 1963; 1949), Simpson index (Simpson, 1949), Margalef richness (Margalef, 1958; 1968), Brillouin index (Brillouin, 1962), and Pielou's evenness (Pielou, 1966) have frequently been in use (e.g. Mojtahid et al., 2008; Muniz et al., 2011).

The most important function of proper benthic quality indicators is their ability to distinguish natural disturbances from anthropogenic perturbation (Carvalho et al., 2006; Dauvin, 2007; Elliott and Quintino et al., 2007), that results in their applicability to give a reliable account of ecosystem health status. For instance, this ability should be assessed in estuarine systems where the salinity fluctuation imposes a natural pressure on habitat's community and along the gradient of salinity from river basin to ocean-dominated area different community composition can thrive. In addition different grain size and depth may accommodate distinct species group (Rosenberg et al., 2004).

An essential part of assessing ecological integrity is the measurement of biological integrity, which in evaluating biological integrity benthic macro-invertebrates communities are the most emphasized biotic elements of aquatic ecosystems (Borja and Dauer, 2008) while they integrate the effects of different types of stress over time(Gray and Mirza, 1979; Borja et al., 2008a).

Since, the use of biotic indices is for the evaluation of the ecosystems' biological integrity, to do so, a method composed of proper metrics is necessary to incorporate biotic responses stimulated by disturbances from individuals to ecosystems. Several metrics conveying information about the ecosystem, when integrated, determine the general condition of the ecosystem (Pinto et al., 2009). When it comes to tracking the trace of pollution in a water body, the major privilege of a benthic index is its ability to distinguish natural (e.g. salinity changes) from man-induced perturbation (Dauvin, et al., 2007; Elliott and Quintino, 2007),

hence, the combination of metrics should form an index that is sensitive to anthropogenic stresses (or managerial actions) and can identify both suppressant and restorative responses (Borja and Dauer, 2008). There are several indices which are used globally and are considered useful in communicating biological integrity measuring different structural aspects of benthic community; abundance, species diversity, trophic status, complexity preservation, etc.

2-4 Different benthic indices

As soft bottom habitats are concerned, four main groups of benthic indices (Ruellet and Dauvin, 2007) whose their representative indicators has been successfully used in estimating the ecological quality of macro-benthic communities are mentioned below:

1-Indices mainly based on ecological groups. Are indices such as the AMBI (AZTI Marine Biotic Index), the BENTIX (a marine biotic index) (Simboura & Zenetos, 2002) and, BOPA (Benthic Opportunistic Polychaetes Amphipods Index) (Dauvin & Ruellet, 2007) which all of them divide species into different ecological groups.

2-Indices mainly based on trophic groups. ITI (Infaunal Trophic Index) suggested by Mearns and Word, (1982) categorizes species into different trophic groups.

3-Indices mainly based on diversity of the community. H' (Shannon-Weaver Index) based on the theory of information initially recommended by Shannon, (1949), BQI (Benthic quality Index) developed by Rosenberg et al. (2004), and based on P-R model (Pearson and Rosenberg, 1978).

4-Hybrid indices: based on a combination of other indices. M-AMBI (Multivariate AMBI) proposed by Muxika et al. (2007) is a biotic indicator which synthesizes ecological group-based AMBI index, diversity-based H' index and species richness value.

One of the most used benthic indices for soft bottoms (Reiss and Kröncke, 2005; Dauvin et al,. 2007; Zettler et al., 2007) is the AZTI's Marine Biotic Index (AMBI) (Borja et al., 2000) which was designed to ascertain the ecological quality of European coastal and estuarine waters by testing the response of soft-bottom benthic community to natural and anthropogenic disturbances in the ecosystem (Muxika et al., 2005). The theoretical basis of AMBI is the afore-mentioned P-R model (Pearson and Rosenberg, 1978) and the ecological strategies (*r*, *k* and *T*) stated by Pianka, (1970). Giving account of the benthic community health by providing a pollution classification, AMBI has been used for establishing Ecological Quality status (EQS) within the context of WFD (Borja et al, 2003). This index has been examined and been successful under different stress sources; wastewater discharge, chemical contaminants, oil platforms, metals, hypoxia, mud disposal, aquaculture, engineering works and etc, and has been applied on a worldwide scale: In Europe (see Muxica et al., 2005, Josefson et al., 2008; Borja et al., 2009; Borja and Tunberg, 2011) and also in Asia (Cai et al., 2003; Cheung et al., 2008), Africa (Afli et al., 2008; Bakalem et al., 2009; Bigot et al., 2008), South America (Muniz et al., 2005), North America (Borja et al., 2008b; Callier et al.,2009, Borja and Tunberg, 2011) and Greenland (Josefson et al., 2008).

However, AMBI may exhibit weakness in some regions where the salinity is low (oligohaline stretch) or highly variable. In low saline waters the number of species and diversity are usually lower than oceanic areas which this issue can have a limiting influence on this assessing method and the result values may introduce a low saline area as a disturbed area (Borja and Muxika, 2005).Also, In estuarine systems in which the salinity fluctuation is high (e.g. from low in riverine areas to high in oceanic areas) this misinterpretation may happen and a salinity gradient may be interpreted as a perturbation gradient. This issue was discussed as the "estuarine paradox" in a study by Elliott and Quintino, (2007).

Heip and Engels, (1974) have suggested that using only diversity indices might not give a reliable picture of benthic community status, on the other hand, some researchers have found inconsistencies when using AMBI alone, for instant as mentioned above, high

degrees of correlation between index values and relative environmental parameters were found in some studies (Muxika et al., 2005; Dauvin, 2007). Though, AMBI against diversity indices is less affected by sample size since by finding more species, values both, may increase or decrease.

In the WFD criteria besides disturbance-sensitivity factor, diversity and abundance proportions of the macro-faunal community should be addressed in benthic indicators' methodologies. To fulfill this implication, Muxika et al., (2007) proposed M-AMBI based on AMBI, species richness and Shannon diversity index. M-AMBI similar to AMBI has been tested in different regions (Borja et al, 2009a) and under different sources of disturbance (Bigot et al., 2008; Bakalem et al., 2009; Prato et al., 2009; Tataranni and Lardicci, 2010).

The application of M-AMBI for a habitat according to the WFD needs definition of the "Reference condition" values. The reference conditions for a specific aquatic area are corresponded to the quality of an undisturbed (no or very low anthropogenic impact) version of that area. Therefore for estimating these values pristine areas with similar features such as the same ranges of salinity, depth and grain size should be sampled and studied. For M-AMBI reference condition diversity and AMBI values of pristine stations need to be calculated. These principal values provide the bases for comparison of already impacted condition from unaffected condition consequently make ecological quality assessment of that study area possible.

AMBI and M-AMBI are both insensitive to the strong seasonal changes which are phenomena that may alter species abundance and structure in the habitat, but, in response of human-induced pressure they will follow the P-R model (Pearson and Rosenberg, 1978) for anthropogenic stresses such as organic enrichment (Borja and Tunberg, 2011).

2-5 Reference conditions

In the WFD, for marine quality assessment based upon benthic communities, apart from using an appropriate set of metrics, there has been an emphasis on the selection of reference conditions for each area typology (Muxika et al, 2007). Study habitats should be classified based on their specific natural features (e.g. salinity and depth dimensions), and then to be compared with their peer habitats (with the same features) where anthropogenic effects are absent or insignificant.

The WFD has recognized four approaches for extracting reference conditions (Muxika et al., 2007) for macro-invertebrates based indicators: (1) Comparing with pristine sites sampled at the same time (2) Historical data: these information need to be from the same natural features; the range of salinity classified based on items such as depth, (3) Models and (4) Expert judgment.

3 Methods

3-1 Sampling and analysis

An environmental study was carried out by Náttúrustofa Vestfjarða first in 1997 “Research on sewage pollution from seven fishing villages in Iceland” by Helgason et al. (2002) and repeated in 2010 (Eiriksson, T., Gallo C, and, Thórisson, B., unpublished data) . Data from the year 1997 are used in this paper. The sampling method implemented in the year 2010 is going to be explained here.

Sampling of this project was carried out in July 2010 and 16 stations were chosen in the surrounding sea area of Ísafjörður. Depths and coordinates of sites were recorded aboard a boat provided by Náttúrustofa Vestfjarða (fig. 3). For each station, 6 replicates were taken using a Van Veen grab with the cover surface of 200 cm^2. Among these replicates, 5 were designated for benthic analysis and the 6th replicate was taken for grain size analysis. For implementing this thesis three stations were chosen in the Pollurinn area, roughly in a linear order from the main sewage outlets and in correspondence with the stations of 1997 study (fig. 2).

The information regarding these sample stations of 2010 are available in table 1. These stations were taken in shallow water, between 7.5 to 15 meters depth, and named B, D and G with an increasing distance from sewage outlets. The stations taken for the study in 1997 were named 1 and 6. Station 1 is about the same place of station B but its three replicates were taken with a distance from each other. Station 6 is situated at the same place of station G (fig. 2).

Figure 3: Sampling in Pollurinn area in July 2010

Table 1: Information about samples of 2010

Station	Grab	Depth	Coordinate
B	200cm²	7,5m	N 66°04´25´´,W 23°07´33´´
D	200cm²	7,9m	N 66°04´21´´,W 23°07´45´´
G	200cm²	14,8m	N 66°03´57´´,W 23°08´19´´

The samples were carefully placed into hermetic plastic boxes and submerged with an 8 to 10 % formalin solution in order to preserve physical features of the collected animals. An appropriate amount of Natrii Boras (borax) where added to the samples to buffer the formalin's capacity of dissolving calcareous animal parts (e.g. Bivalvia). The formalin solution was removed after some days and samples materials were sieved through a 0.5 mm

mesh size stainless steel sieve. The on-sieve materials were stored in plexiglas boxes with 20% isopropanol solution.

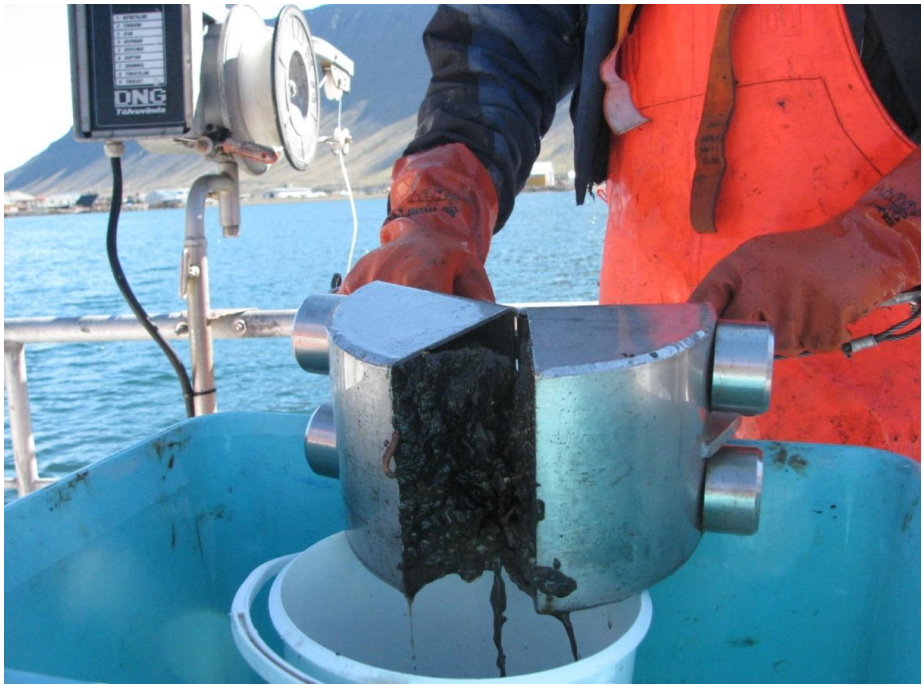

Figure 4: Grab content is placed in hermetic plastic box

Figure 5: Sampling facilities

Macro-faunal analysis was conducted on 3 replicates for each of the station taken in 2010. In order to speed up the preliminary sorting of the animals from the rest of the material, samples were fractioned in sub-samples. Samples usually were divided by two in a division box. For this reason, they were stirred in the division box until a homogeneous mixture was obtained so that the material becomes evenly separated. For all replicates this division was performed one time (0.5 sub) except for replicates B3 and D1 which were divided two times (0.25 sub) (table 2). Finally all species were analyzed applying a Leicha MZ12 stereoscope.

Table 2: Information about replicates (content was visualy estimated)

Station	Replicate	Content	Sub
B	1	70% black mud < 500 µm and 30% vegitation	0.5
	2	70% black mud < 500 µm and 30% vegitation	0.5
	3	60% black mud < 500 µm and 40% vegitation	0.25
D	1	80% black mud < 500 µm and 20% vegitation	0.25
	2	80% brown + black mud and 20% vegitation	0.5
	3	90% brown + black mud and 10% vegitation	0.5
G	1	100% brown + black mud < 500 µm	0.5
	2	100% brown + black mud < 500 µm	0.5
	3	100% brown + black mud < 500 µm	0.5

For this research another set of data was used. It belonged to a study which its sampling was done in August 1997 (Helgason et al., 2002). Samples of stations 1 and 6 were reviewed again to identify individuals to the lowest possible taxonomic level since some taxa were not identified to a level appropriate for the present study. Taxa should be identified down to their species level since their sensitivity and tolerance to pollution (e.g. sewage, physical disturbances) could be different even in the genus level (Rosenberg et al., 2004).

In cases which species identification was not met, taxa were sent to Líffræðistofnun Háskólans for review by Guðmundur V. Helgason. In addition some individuals due to being juvenile or damaged were not identifiable to the species level, in these cases they were presented in higher levels (e.g. family).

The numbers of individuals of each taxon in a sample were obtained by multiplying the number of subs and, individuals of the counted sub. In some cases in 1997 it was deemed advisable that instead of multiplication, to open the other sub(s) and to analyze them for the first time.

As this research's measurements included the AMBI and Multivariate-AMBI methods, the sampling standards should be according to the criteria expressed by the developers of these programs (Borja et al., 2000; Muxika et al., 2007a). In this respect, they suggested that at least 2 replicates should be taken for each station and the minimum sample area should be 0.25 m^2. Also, the proper sieve size was recommended to be 0.5 mm.

3-2 Grain size

The grain size analysis was carried out for the three stations of the 2010 study. For this reason, the observable vegetation was removed from sediment samples and then the soil was heated in an oven at 85°C for 24 hours to eliminate its moisture and to have organic particles dried. After removing from oven samples were weighted (first dried weight) and sieved on 63 μm mesh size which is the size threshold between sand and silt. Since, soil became stuck together after heating them in oven; the specimens were sieved under running water. The on-sieve material was heated again at the same temperature and period, and then weighted (second dried weight). By deducting the second dried weight from the first dried weight, the percentage of sand & gravel and silt & clay was calculated.

3-3 Abundance and diversity

By analyzing all found individuals the relative abundance for each sample and average for the stations were determined which led the way for diversity and ecological measurements. For assessing macro-benthic communities' diversity, several well-known diversity indices were applied and calculated by PRIMER 5 software. The idea of using different diversity indices was to apply them in a complementary framework; for a better interpretation and to have more tools for temporal and spatial comparisons.

Species richness (S) is the simplest and probably the most used diversity parameter signaling the relative wealth of a community, but maybe the most difficult in terms of definition whereas any richness measure is highly dependent on sample size; the wider the area of dragging, the more species are expected to be found in the samples (Peet, 1974). Species richness is quantified as the number of species in the relative sampling area. Though the exact composition of the benthic community due to limitation of sampling effort is not accessible, abundance of species can give valuable information about community's composition in the sampling area.

They are some diversity indices used in this study. All presented indices here (except Brillouin) are recognized as Heterogeneity (dual-concept diversity) indices which take both the number of species and evenness into account. The present Simpson formula is a statistically adjusted expression firstly developed by Simpson, (1949) and also suggested by Pielou, (1969) for finite sample size. The index estimates the probability that two randomly selected individuals belong to the same species. Shannon index (Shannon-Weaver, 1963; 1949) based on theory of information, is one of the most used diversity indices and takes into account both species richness and their distribution pattern (evenness). One of the Shannon-Weaver assumptions is that all species in the community are presented in data and population is considered infinite. This may put weight on uncertainty caused by inadequate sampling area. In the cases where randomness is not assured, Pielou, (1967) suggested the Brillouin formula (Brillouin, 1962) instead of Shannon index while the former one does not reflect the sample size. The tangible difference of Brillouin index with typical Heterogeneity indices is mirrored in the situation where a community with a noticeably greater number of individuals may gain a greater value even though the richness and evenness of the other community is higher (Peet, 1974). The bases for using a variety of indices are to have them cover each other's deficiency and to take into account all richness, evenness and dominance principals in diversity calculation for a more accurate explanation of communities' characteristics.

Among indices used, Simpson index has a low sensitivity to sample size while Shannon and Brillouin indices are identified as rather sensitive in this case. Shannon index had a

tendency toward emphasizing on richness and evenness, whereas , Simpson index weighted towards most abundant species, and is considered an index for concentration of dominance (Whittaker, 1965) and less sensitive to changes in species richness. In turn, Brillouin index relies mostly on richness. Pielou's evenness index was used to evaluate the stability and relationship to sample size of the Shannon diversity index which was applied as a parameter for estimating ecological status of benthic community and M-AMBI biotic index calculation.

Margalef's species richness index (Margalef, 1958) gives a proportion of species number found and individuals in the sample with dividing the species number by the natural log of the number of individuals sampled. This index emphasizes on species richness, is very sensitive to sample size (Magurran, 2004; Death, 2008) and is correctly calculated by using the total number of individuals rather than density; it is calculated for each replicate and then averaged so that value is obtained for the station (Gamito, 2010). For other diversity indices the abundance of three replicates were used for calculation because these indices consider proportions rather than absolute numbers.

For Shannon index, scale values between 0 to 5, first time suggested by Frontier, (1983) are assigned to evaluate biodiversity of a habitat. While 5 shows a high diversity and pictures an undisturbed system, in turn, 0 represents a highly disturbed area. Shannon index also has been used for ecological status determination in the context of WFD which is done by normalizing values based on the study habitat's natural variability. For instance, Vincent et al., (2002) has assigned values from 0-1 to 4-(5 and higher) for bad to high ecological conditions. The list of applied diversity indexes are represented in table 3.

Table 3: Diversity and biotic indices applied in this study and their associated variables and expression

Variables	Definition	Expression	Reference
S	Species number (Richness)	——	——
n_i	Number of individuals *i* th species	——	——
N	Total number of individuals	$\sum n_i$	——
J'	Pielou's evenness	$H'/\log_2(S)$	Pielou, (1966)
H'	Shannon-Wiener diversity index (Log2)	$-\sum[(n_i/N)x\mathrm{Log}_2(n_i/N)]$	Shannon & Wiener, (1949)
d	Margalef Species richness	$(S-1)/\mathrm{Log}(N)$	Margalef, (1958)
HB	Brillouin index	$\mathrm{Ln}(N!)-S\mathrm{Ln}(n_i!)/N$	Brillouin, (1962)
D	Simpson diversity index	$1-[\sum[n_i x(n_i-1)]/(Nx(N-1))]$, $0\leq D\leq 1$	Simpson, (1949)
AMBI	AZTI Marine Biotic Index	[(0x%GI)+(1.5x%GII)+(3x%GIII)+(4.5x%GIV)+(6x%GV)]/100	Borja et al. (2000)
M-AMBI	Multivariate-AMBI	AMBI, Reference condition (S,H')	Muxika et al. (2007)

3-4 Ecological grouping

Species inherent characteristics such as their original habitat and feeding behavior define their biological function. When it comes to anthropogenic disturbances, if the related stressors could change the physical feature of the habitat or could affect the community's structure, the species function and behavior might change too. However in addition, disturbances such as organic enrichment and inorganic pollution can directly change a species metabolism and become more than the threshold the species can tolerate. Distinct species in spite of their taxonomical differences can be grouped based on their response to man-induced stressors (Grall and Glemarec, 1997).

In order to apply AMBI index, all detected individuals were classified into one of the five ecological groups (EG) proposed by (Borja et al., 2000). It was mainly based on the ecological list presented in AMBI software version 4 (Borja et al., 2000), which was adopted by use of the previously suggested classifications in some researchers' works such as Glemarec and Hily, (1981) and, Grall and Glemarec, (1997) which had arranged these groups according to their sensitivity to an increasing disturbance gradient (Simboura et al., 2007).

In the whole, as the developers of the program suggested, the AMBI list's values are not supposed the matter of change; therefore these values are constant when applying AMBI in

different geographical areas or sources of disturbance. In this research some species which were not identified with certainty (e.g. *Pholoe cf minuta*), were assigned to a higher taxonomic level (e.g. *Pholoe* sp.). It was done in cases when the higher taxonomic group with similar grouping, was available in the list; in this manner the result did not change. This alteration usually happened in genus level or limitedly in higher taxonomic levels (e.g. Oligochaetes). On the other hand, unrecognizable taxa such as juveniles or damaged species were removed from the list as suggested by the developers of the method. Consequently most of the individuals were put in one of five ecological groups (EGs).These five groups according to Simboura and Argyrou, (2010) are outlined as:

Group I: Species very sensitive to organic enrichment and present in undisturbed circumstances, they consist of the specialist carnivores and some deposit-feeder tubicolous polychaetes.

Group II: Species indifferent to organic enrichment. Their presence is always in low densities and temporal fluctuation. This group includes suspension feeders, less selective carnivores and scavengers.

Group III: Species tolerant to increased levels of organic matter enrichment. These species may occur in undisturbed situations, but their growth is induced by organic enrichment. These include some of the surface deposit-feeder species (e.g. Spionids).

Group IV: Second-order opportunistic species. These are the small species with a short life cycle, adapted to live and proliferate in reduced sediment. They are the subsurface deposit feeders (e.g. Cirratulids).

Group V: First-order opportunistic species. These are the deposit feeders that proliferate in sediments near the surface.

While the above mentioned ecological grouping is formed based on species biological responds to external elements, their biological attributes such as feeding habits (e.g. Magnusson et al., 2003) and their origin habitats (e.g. Lejart & Hily, 2011) should be considered simultaneously. For this reason, literature reviewed as an effort to retrieve some

of taxa's relative biological traits which drove them to a certain behavior, tolerance range and consequently assigned them to a specific eco-group. Assigning all species to their respective eco-groups, their list was imported to AMBI software for ecological classification of relevant habitats.

3-5 Indices applied: AMBI and M-AMBI

AMBI is the most universally used biotic index within the WFD (Muxika et al., 2005; Zettler et al., 2007) and the newest version of its software (V.4.1) is downloadable on (http://ambi.azti.es/). The software provides a list of 5,900 taxa representative of soft bottom communities present at estuarine and coastal ecosystems stretched in a vast geographical scale from the North Sea to Mediterranean Sea in Europe and some parts of North and South American waters and etc. For application of this indicator package (AMBI and M-AMBI) two guidelines from Marine Pollution Bulletin were used (Borja et al., 2004a; Borja and Muxika, 2005).

AMBI and M-AMBI are applied to estimate the EcoQ status of sampled stations in pollurinn-skutulsfjordur. For each station's data set these indices deliver certain digits which need to be defined. Related indices values definition, disturbance classification and ecological status modes are shown in table 4.

For M-AMBI, the ecological factors including AMBI, Shannon diversity and species richness values corresponded to unpolluted condition (Bald et al., 2005; Borja and Tunberg, 2011) needed to be estimated for pristine sites to draw out reference conditions. For finding such sites historical data of Náttúrustofa Vestfjarða were reviewed.

Table 4: Biotic indices background values, associated classification and ecological condition

Biotic index	Index values	Classification	Ecological status	Classification Reference
AMBI	0.2<BI≤1.2	Normal	High	Borja et al. (2000)
	1.2<BI≤3.3	Slightly disturbed	Good	
	3.3<BI≤4.3	Moderately Disturbed	Moderate	
	4.3<BI≤5.0	Disturbed	Poor	
	5.0<BI≤5.5	Heavily disturbed	Poor	
	5.5<BI≤6.0	Heavily disturbed	Bad	
	6.0<BI≤7.0	Exteremely disturbed	Bad	
M-AMBI	BI>0.85	Normal	High	Muxica et al. (2007)
	0.55<BI≤0.85	Normal	Good	
	0.39<BI≤0.55	Disturbed	Moderate/Fair	
	0.20≤BI≤0.39	Heavily disturbed	Poor	
	BI<0.20	Heavily disturbed	Bad	

4 Results

4-1 Grain size

One of the natural parameters abundantly used for type-specific reference conditions is grain size. Benthic Habitats based on their grain size can accommodate different species (Kröncke and Bergfeld, 2001), therefore, parallel to the other natural and anthropogenic stressors sediment particles has a significant role in community's structure.

The 63µm mesh size was used for sieving. Since particles in the range of sand to gravel were not seen on the sieve, application of larger mesh sizes was found unnecessary. The grain size analysis revealed that silt & clay and fine sand were dominant in sampled benthic habitats. Station G almost situated in the central part of Pollurinn showed the greatest percentage of silt & clay whereas in station D about 350 meters from discharge points the percentage of sand was higher than silt & clay. The range of sediment particles for sampling stations of 2010 is presented below (table 5).

Table 5: Grain size results

Station	Grab	Sand & gravel	Silt &Clay
	cm²	(>63µm) %	(<63µm) %
B	200	38,6	61,4
D	200	66,2	33,8
G	200	23,0	77,0

4-2 Abundance of macro-benthic fauna and their diversity

In investigated stations and over the whole study period, 57 benthic species were identified and 16480 specimens were counted. In 1997, 28 species including about 11590 individuals were found in all samples, which were composed of 7700 copepods and nematodes, and around 3890 other individuals (table 7). Nematodes and copepods due to their small size could not be completely captured by the used sieve (500 μm) and their real densities were under-sampled. Therefore, they were removed from the diversity and biotic indices calculations, but, they were counted and their average abundance in stations is shown in tables 6 and 7. Species richness in stations 1 and 6; 16 and 17 respectively, showed a slight increase while abundance by 1297.7 individuals (ind.) in station 1 followed by 25 ind. in station 6 displayed a huge decline.

Replicates of station 1 taken in 1997 showed significant discrepancy in terms of abundance; the abundance of approximately 3390 ind. was followed by abundance of 56 ind. in replicate 2 and an increase to 448 ind. in replicate 3 resulted in an average abundance of 1297.7 with standard deviation of 1821. In station 1, Oligochaetas were dominant taxa with average abundance of 456 ind. accounting for 35% of total abundance and after them; *Capitella capitata* with abundance of 298 ind. had 23% of total station's abundance. It is noteworthy that 98% of total Oligochaetes 1342 ind. and 99.8 % of all *Capitella capitata* 892 ind. species of station 1 were found in replicate 1 of this station. This replicate was the closest one to the discharge area. Both of these two mentioned taxa are r-strategists; they are able to reproduce rapidly in the polluted areas. Also 3390 individuals (87%) of all taxa and 13 species (76%) out of 17 species of station 1, was belonged to this replicate.

In 2010 study stations, about 4900 individuals were found in all samples containing 1620 individuals other than copepods and nematodes which showed 58% decrease in total number of detected individuals compared to 1997 (table 6).

In 2010, from the whole number of species which were found in stations (43), 14 different species were found in station B, 19 different species in station D and 35 distinct taxa were detected in station G. This pattern was correlated with increasing distance from the main wastewater discharge area in Pollurinn which were amounted to 140 m in B, 340 m in D and 1170 m in G. In terms of abundance, the results represented 300.4 ind. in station B, 148.7 ind. in station D and 171.0 ind. in station G. In station B Oligochaetas with abundance of 132 ind. (44% of total) were dominant, and *Fabricia sabella* by 66.7 ind. (22%) and *Capitella capitata* with 38 ind. (12.6%) were after them. It is noteworthy that most *Fabricia sabella* recorded in B, were found on an Ulva leaf at replicate 2. In station D, *Pholoe* sp. by abundance of 46.7 ind. (31.4% of total) was dominant followed by *Polydora* sp. 33.3 ind. (22.3%) and *Chaetozone cf setosa* 17.3 ind. (11.6%). In station G there was no dominant species and total abundance were distributed between 4 major species of *Chaetozone cf setosa*, *Cossura longocirrata*, *Pholoe* sp. and *Polydora* sp. by densities; 25.3, 24.7, 21.7 and 20.7ind. and 14.8% to 12.1% of total station's abundance.

The abundances of different groups of taxa can be compared too. 22 Polychaeta families were found in 2010 while only 12 Polychaetes were detected in 1997. In turn, Gastropods represented 3 families in 1997 whereas in 2010 they had only one family.

In terms of abundance and species richness trends sampling stations of 1997 and 2010 showed general similarities. In both years, with increasing distance from the discharge area species richness showed increase, however between the two years, station G compared to station 6 (situated at the same coordinate) represented more than 2 times increase in number of species (35 and 17). Abundance in both 1997 and 2010 years had a descending order with increasing distance from the discharge area however the rate of decrease was very higher in 1997 compared to 2010.

Table 6: Abundance of macro-benthic taxa (number of individual) in sampling stations (2010).

Taxa	Station		
	B	D	G
Nematoda*	162,7	246,7	266,7
Nemertea	5,3	0,0	0,0
Mollusca			
Bivalvia	0,0	0,0	0,7
Cardiidae			
Cardium spp	0,0	0,0	3,0
Thyasiridae			
Thyasira cf flexuosa	0,0	0,0	2,7
Myidae			
Mya arenaria	1,3	0,7	0,3
Mya cf truncata	0,7	0,0	0,0
Mytilidae			
Mytilus edulis	0,0	2,0	0,0
Musculus discors	0,0	0,7	1,3
Tellinidae			
Macoma calcarea	0,0	0,7	3,7
Gastropoda			
Rissoidae			
Onoba aculeus	8,0	0,0	0,0
Annelida			
Clitellata			
Oligochaeta	132,0	2,0	3,3
Polychaeta			
Ampharetidae			
cf Ampharete spp	0,0	0,0	4,3
Flabelligeridae			
Brada cf villosa	0,0	0,0	1,3

Table 6: Abundance of macro-benthic taxa (number of individuals) in sampling stations (2010) continue.

Taxa	Station		
	B	D	G
Capitellidae			
Capitella cf capitata	38,0	5,3	1,0
Mediomastus fragilis	0,0	0,7	0,0
Cirratulidae			
Chaetozone cf setosa	0,0	17,3	25,3
Cossuridae			
Cossura longocirrata	0,0	6,7	24,7
Hesionidae			
Microphthalmus cf aberrans	0,0	5,3	1,3
Maldanidae			
Maldane sarsi	0,0	0,0	0,7
Opheliidae			
Ophelina acuminata	0,0	0,0	1,7
Orbiniidae			
Naineris quadricuspida	15,3	0,0	0,0
Scoloplos armiger	2,7	5,3	2,3
Oweniidae			
Owenia cf fusiformis	0,0	0,0	1,3
Paraonidae			
Aricidea cf suecica	0,0	0,0	0,3
cf Levinsenia gracilis	0,0	0,0	11,3
Pectinaridae			
Pectinaria spp	0,7	0,7	0,3
Phyllodocidae			
Phyllodoce spp	0,0	2,7	0,0
Polynoidae	0,0	0,0	0,7
Pholoidae			
Pholoe spp	0,7	46,7	21,7

Table 6: Abundance of macro-benthic taxa (number of individuals) in sampling stations (2010) continue.

Taxa	Station		
	B	D	G
Sabellidae			
Fabricia sabella	66,7	1,3	10,0
Scalibregmatidae			
Scalibregma inflatum	0,3	0,0	1,0
Apistobranchidae	0,0	0,0	0,7
Spionidae			
Polydora spp	3,3	33,3	20,7
Spio sp	0,0	0,0	0,7
Sternaspidae			
Sternaspis scutata	0,0	0,0	0,3
Syllidae			
cf Syllis spp	0,0	0,0	9,0
Terebellidae			
Terebellides stroemii	0,0	0,0	3,7
Arthropoda			
Crustacea			
Copepoda*	346,0	10,0	0,7
Isopoda			
Paramunnidae			
Pleurogonium spp	0,0	1,3	0,7
Cumacea			
Leuconidae			
Leucon spp	0,0	0,0	3,3
Amphipoda	0,0	2,0	0,7
Oedicerotidae juv	0,0	0,0	0,7
Lysianassidae	0,0	14,0	2,0
Insecta			
Chironomidae	2,7	0,0	0,0

**These taxa are not included in the biotic indices calculations*

Table 7: Abundance of macro-benthic taxa (number of individuals) in sampling stations (1997).

Taxa	Station	
	1	6
Nematoda*	415,3	27,5
Nemertea	34,7	0,0
Turbellaria	5,3	0,0
Mollusca		
Bivalvia		
Myidae		
Mya truncata	1,0	0,0
Mytilidae		
Mytilus edulis	21,7	1,0
Gastropoda		
Rissoidae		
Onoba aculeus	5,3	1,5
Skeneopsidae		
Skeneopsis planorbis	0,0	0,5
Turbinidae		
Margarites sp.	0,7	0,0
Annelida		
Clitellata		
Oligochaeta	456,0	1,5
Polychaeta		
Ampharetidae	0,0	0,5
Capitellidae		
Capitella cf capitata	298,0	0,0
Cirratulidae		
Chaetozone setosa	0,0	3,0
Paraonidae		
Aricidea suecica	0,0	2,5
Pectinaridae		
Pectinaria spp	5,3	0,5

Table 7: Abundance of macro-benthic taxa (number of individuals) in sampling stations (1997) continue.

Taxa	Station	
	1	6
Phyllodocidae	10,7	0,0
Eteone sp.	5,3	0,0
Polynoidae		
Harmothoe sp.	0,0	0,5
Pholoe spp	112,7	7,0
Orbiniidae		
Scoloplos armiger	22,3	1,0
Sabellidae		
Fabricia sabella	5,3	0,0
Serpulidae		
Spirorbis spp	21,3	0,0
Apistobranchidae	0,0	1,0
Spionidae		
Polydora spp	292,0	2,0
Arthropoda		
Crustacea		
malacostraca		
Leuconidae		
Eudorella sp.	0,0	0,5
Copepoda*	2132,7	1,5
Cumacea		
Leuconidae		
Leucon spp	0,0	1,0
Amphipoda		
Oedicerotidae juv		
Monoculodes sp.	0,0	0,5

**These taxa are not included in the biotic indices calculations*

4-3 Diversity indices

The data sets for calculating all indices use the average of three replicates as abundance whereas for Margalef's species richness abundance for each replicate was used separately. The Margalef diversity index, showed an increasing species richness rate with increase of distance from the main discharge point in both 1997 and 2010 though in 2010 this increase had a higher rate; 1.36 in B to 3.90 in G compared to 1.26 in station 1 to 2.58 in station 6 in 1997 (note that station G and 6 are situated at the same coordinate but taken in different years). With having a distance-increase rate for Pielou's evenness, the Shannon diversity, as expected, had a growing value for both years ranging from 2.2 to 4 in 2010 and 2.48 to 3.53 in 1997. Interestingly, Simpson index delivered the value of 0.91 for both G and 6 signaling no change in terms of diversity between sampling years. Species richness, Brillouin and Shannon diversity results showed value increase in this place between years, whereas Pielou's evenness and total abundance showed a declining pace between these two years. In the whole, the community, for both 1997 and 2010, in terms of species richness and diversity parameters improved when distance from the discharge area increased. Their pattern of behavior followed the P-R model (Pearson and Rosenberg, 1978). According to the Pielou's evenness results the species are less evenly distributed in the community in 2010 compared to 1997. Finally, the results showed a slight temporal diversity improvement when comparing 1997 to 2010 benthic communities.

Table 8 : Diversity indices and corresponding parameters results; (D) Abandance of stations, (d) Margalef, (J') Pielou's evenness, (Si) Simpson indices

Station	Rep	S	D	d	J'	Brillouin	H' (log2)	Si
2010								
B	3	14	277,6	1,36	0,58	1,46	2,20	0,70
D	3	19	154,1	1,99	0,71	1,92	3,03	0,82
G	3	35	169,3	3,90	0,78	2,50	3,98	0,91
1997								
1	3	16	1297,6	1,26	0,62	1,69	2,48	0,76
6	2	17	25	2,58	0,86	1,80	3,53	0,91

4-4 Ecological grouping

The ecological grouping (EG) of all detected species were determined with the exception of some taxa that could not be identified down to species level due to their size, damaged body or in cases where the taxon was not assigned to a group in the AMBI list of species. Species related EG, their abundance and percentage were determined for all stations in both two study years (tables 9 and 10).

The ecological classification simplifies the community structure to five EGs leads the way to study the reasons that they despite of different feeding habits and original habitat, in terms of tolerance to pollution function similarly. Tables 10 and 11 represented the dominant eco-groups in sampling stations. It was shown that EG (V), first-order opportunistic, was the major group in station B (2010) with about 62% of all individuals found there. The next abundant groups were EG (II) by 24%, EG (I) account for 8.6%, EG (III) by 4% and EG (IV) only 1% of total population in this station. These results showed a notable discrepancy in relative percentages demonstrating an unbalanced distribution of eco-groups in this station. In station D (2010) EG (II) and (IV) by about 38% of abundance were mutually dominant groups, the EG (I) was the third important group included about 10% of total individuals in this site. In station G, EG (IV) was introduced as main group by 42% which was followed by EG (II) and EG (III) by 33% and 14% of abundance.

It is noteworthy that by increasing distance from the discharge area (station B) to central parts (station G) of it, there found some trends in eco-groups' arrangements: (1) EG (V) was decreased from B to G, especially from the innermost site (station B) to station D this decrease was significantly high. (2) EG (III) had an increasing abundance with increasing distance from the discharge points. (3) EG (IV) was enhancing in presence by increasing distance from the sewage outlets. (4) EG (I) was never dominant, also, was never disappeared in sites and was the most abundantly found in sand dominated site; station D (table 11). In station1 of 1997, EG (V) was the leading group by 58% of abundance and EG (IV) was the second abundant group by 22%. The considerable fact was the disappearance

of sensitive to organic enrichment species (EG I) while had less than 1% (0.6%) of total individuals in the station. It is while that after 13 years in 2010 about 9% of the habitat's community was belonged to EG (I). In station 6 (1997) the EG (II) was dominant by 34% and EG (I) by 24% was the second rich group in terms of abundance. In general, EG (I), (II) and (III) had a growing rate by increasing distance from the inner parts while EG (V) and (IV) had a declining rate in this case (table 12).These outcomes were in accordance with P-R paradigm described by Pearson and Rosenberg, (1978).

Table 9: Benthic fauna's ecological group (Borja et al.,2000) and their Abundance and percentage in sampling stations (2010)

Taxa	Eco-group	Station B		Station D		Station G	
		Abundance	Percentage	Abundance	Percentage	Abundance	Percentage
Nemertea	III	5,3	1,9%				
Bivalvia	N.A					0,7	0,4%
Cardium sp.	III					3,0	1,8%
Thyasira flexuosa	III					2,7	1,6%
Mya arenaria	II	1,3	0,5%	0,7	0,5%	0,3	0,2%
Mya truncata	II	0,7	0,3%				
Mytilus edulis	III			2	1,3%		
Musculus discors	I			0,7	0,5%	1,3	0,8%
Macoma calcarea	II			0,7	0,5%	3,7	2,2%
Onoba aculeus	I	8	2,9%				
Oligochaeta	V	132	47,5%	2	1,3%	3,3	2,0%
Ampharete sp.	I					4,3	2,6%
Brada villosa	I					1,3	0,8%
Capitella capitata	V	38	13,7%	5,3	3,6%	1,0	0,6%
Mediomastus fragilis	III			0,7	0,5%		
Chaetozone setosa	IV			17,3	11,6%	25,3	15,2%
Cossura longocirrata	IV			6,7	4,5%	24,7	14,8%
Microphthalmus aberrans	II			5,3	3,6%	1,3	0,8%
Maldane sarsi	I					0,7	0,4%
Ophelina acuminata	III					1,7	1,0%
Naineris quadricuspida	I	15,3	5,5%				
Scoloplos armiger	III	2,7	1,0%	5,3	3,6%	2,3	1,4%

Table 9: Benthic fauna's ecological group (Borja et al.,2000)and their Abundance and percentage in sampling stations (2010) countinue

Taxa	Eco-group	Station B		Station D		Station G	
		Abundance	Percentage	Abundance	Percentage	Abundance	Percentage
Owenia fusiformis	II					1,3	0,8%
Aricidea cf suecica	I					0,3	0,2%
Levinsenia gracilis	III					11,3	6,8%
Pectinaria sp.	I	0,7	0,3%	0,7	0,5%	0,3	0,2%
Phyllodoce sp.	II			2,7	1,8%		
Polynoidae	II					0,7	0,4%
Pholoe sp.	II	0,7	0,3%	46,7	31,4%	21,7	13,0%
Fabricia sabella	II	66,7	24,0%	1,3	0,9%	10,0	6,0%
Scalibregma inflatum	III	0,3	0,1%			1,0	0,6%
Apistobranchidae	I					0,7	0,4%
Polydora sp.	IV	3,3	1,2%	33,3	22,4%	20,7	12,4%
Spio sp.	III					0,7	0,4%
Sternaspis scutata	III					0,3	0,2%
syllis sp.	II					9,0	5,4%
Terebellides stroemii	II					3,7	2,2%
Pleurogonium sp.	N.A			1,3	0,9%	0,7	0,4%
Leucon sp.	II					3,3	2,0%
Amphipoda	N.A			2	1,3%	0,7	0,4%
Oedicerotidae	N.A					0,7	0,4%
Lysianassidae	I			14	9,4%	2,0	1,2%
Chironomidae	III	2,7	1,0%				

Table 10: Benthic fauna's ecological group (Borja et al.,2000)and their Abundance and percentage in sampling stations (1997)

Taxa	Eco-group	Station1		Station 6	
		Abundance	Percentage	Abundance	Percentage
Nemertea	III	34,7	2,7%		
Turbellaria	II	5,3	0,4%		
Mya truncata	II	1,0	0,1%		
Mytilus edulis	III	21,7	1,7%	1	4,0%
Onoba aculeus	I	5,3	0,4%	1,5	6,0%
Skeneopsis planorbis	I			0,5	2,0%
Margarites sp.	N.A	0,7	0,1%		
Oligochaeta	V	456,0	35,1%	1,5	6,0%
Ampharetidae	N.A			0,5	2,0%
Capitella capitata	V	298,0	23,0%		
Chaetozone setosa	IV			3	12,0%
Aricidea suecica	I			2,5	10,0%
Pectinaria sp.	I	5,3	0,4%	0,5	2,0%
Phyllodocidae	N.A	10,7	0,8%		
Eteone sp.	III	5,3	0,4%		
Harmothoe sp.	II			0,5	2,0%
Pholoe sp.	II	112,7	8,7%	7	28,0%
Scoloplos armiger	III	22,3	1,7%	1	4,0%
Fabricia sabella	II	5,3	0,4%		
Spirorbis sp.	II	21,3	1,6%		
Apistobranchidae	N.A			1	4,0%
Polydora sp.	IV	292,0	22,5%	2	8,0%
Eudorella sp.	N.A			0,5	2,0%
Leucon sp.	II			1,0	4,0%
Monoculodes sp.	I			0,5	2,0%
Monoculodes sp.	I			0,5	2,0%

Table 11: Macrobentic taxa dominant ecological groups and percentages in sampling stations (2010)

Station	Dominant eco-group	Percentage (%)
B	V	62,2
	II	24
	I	8,6
	III	4
	IV	1,2
D	II	38,6
	IV	38,5
	I	10,4
	III	5,4
	V	4,9
G	IV	42,4
	II	33,0
	III	13,8
	I	6,1
	V	2,6

Table 12: Macrobentic taxa dominant ecological groups and percentages in sampling stations (2010)

Station	Dominant eco-group	Percentage (%)
1	V	58,1
	IV	22,5
	II	11,2
	III	6,6
	I	0,8
6	II	34,0
	I	24,0
	III	8,0
	IV	20,0
	V	6,0

If the most abundant species got arranged, it is observable that sub-surface deposit feeder Oligochaetes were the most influential group of first-order opportunistic in both sampling years. In station B, suspension feeder *Fabricia sabella*, belonging to indifferent to organic enrichment group, was the second important species in terms of abundance. In Station D similar to station 6, carnivorous *Pholoe* sp. belonging to EG (II) was the most abundant genus and the selective deposit feeder *Polydora* sp. classified in EG (IV) was the second abundant genus in mentioned stations (tables 13 and 14). Interestingly the dominant species in station B always lost their domination when moving toward central parts of Pollurinn and first-order opportunistic gave the way to second-order opportunistic and indifferent species. Probably this is the sign of a major change in sediment's bio-chemical attributes. Investigations such as the bio-chemical analysis (see Muniz et al., 2011), the bottom water energy regime versus species assortment (see Gamito & Furtado, 2009), trophic grouping relations (see Afli et al., 2008) could give valuable information regarding these changes.

For example, the communities' composition can be reviewed based on trophic habits and links between inhabitants. Such a linkage can be predator-prey relationship. For instance, Nemertea group showed a significant abundance decrease in closest sampled station to sewage outfall from 1997 to 2010 by 34.7 ind. to 5.3 ind. The change might be due to this group's feeding habit which preys on Oligochaetas EG (V) (Jennings and Gibson, 1969) and its potential ability to tolerate moderate organic enrichment EG (III). The results showed that Oligochaetes' abundance decreased from 456 in 1997 to 132 ind. in 2010.

Table 13: Dominant species and their ecological groups in sampling stations (1997) and their abundance

Station	Taxa	Eco-group	Abundance	Percentage
1	Oligochaeta	V	456	35,1%
	Capitella capitata	V	298	23,0%
	Polydora sp.	IV	292	22,5%
6	*Pholoe* sp.	II	7	28,0%
	Chaetozone setosa	IV	3,0	12,0%
	Aricidea suecica	I	2,5	10,0%

Table 14: Dominant species and their ecological groups in sampling stations (2010) and their abundance

Station	Taxa	Eco-group	Abundance	Percentage
B	Oligochaeta	V	132	47,5%
	Fabricia sabella	II	66,7	24,0%
	Capitella capitata	V	38	13,7%
D	*Pholoe* sp.	II	46,7	31,4%
	Polydora sp.	IV	33,3	22,4%
	Chaetozone setosa	IV	17,3	11,6%
G	*Chaetozone setosa*	IV	25,3	15,2%
	Cossura longocirrata	IV	24,7	14,8%
	Pholoe sp.	II	21,7	13,0%
	Polydora sp.	IV	20,7	12,4%

4-5 AMBI and M-AMBI

The AMBI values were determined for all stations by calculating the relative values for each replicate and then taking average of them. In all replicates eco-groups were sorted. Two types of groups were not assigned to an ecological class: Those who were not identified in the AMBI software and those who due to being juvenile or damaged could not be identified down to the species level. For 1997, in average only 5 percent of species were not specified in a particular ecological category; *Portlandia iris* and *Margarites* sp. were not recognized in software's list and Ampharetidae and Apistobranchidae families were not classified in any ecological group since they were juvenile. For 2010, *Pleurogonium* sp. was not assigned to an ecological group. Species presented as Apistobranchidae, Oedicerotidae and Polynoidae families however each of them were representative of a single species but were not nominally recognizable, so, were not assigned to specific eco-groups. In the whole, only about 2% of individuals were not ecologically categorized in the applied system. In AMBI's methodology it is declared that if the percentage of uncategorized species reaches 50% the result of the software is not reliable and the percentages higher than 20% should be evaluated with caution (Borja and Muxika, 2005). In this study's case the uncategorized animals are considerably lower than caution zone. Finally, lists of species were imported to the software and the results were delivered.

In 1997, the AMBI value for station 1 was 3.59 classifying the habitat as moderately disturbed with transitional to polluted benthic condition that represented a considerably unbalanced community. Based on WFD criteria the station had a "moderate" EQS describing a community dominated by taxa indicative of pollution (e.g. *Capitella capitata* and Oligochaeta) whereas some sensitive taxa were still existent (e.g. *Aricidea suecica and Pectinaria* sp.). Moving more than one kilometer away from the discharge area, station 6 presented a notably better habitat health representative of a slightly polluted habitat and a "good" ecological status (table 16 and 17). Though eco-groups (II) and (I) were dominant in this station but presence of species from group (IV) showed signs of a slightly biased situation.

The results of AMBI for 2010 revealed value of 4.46 for the closest station to the sewage outlets (about 150 m), indicating a habitat which in terms of standard ecological qualities was degraded and classified as “poor” according to the WFD criteria. In Station D, about 350 meters away from the discharge area, value of 2.93 showed a major improvement in the EQS of the respective benthic area from “poor” to “moderate” category. In station G, however, 1100 meters away from the discharge area the EQS almost had the same range of station D and the value of 3.09 (table 16 and 17). Both stations D and G were belonged to the same category; slightly polluted, demonstrating an almost healthy habitat with “good” EQS according to the WFD standards.

With comparing the results of these two years in sites with the same coordinate, some degrees of degradation in benthic community status was observed which demonstrated ecological deterioration in the sampled habitats. In this concern, 1&B habitat survey showed 19% increase in AMBI value which revealed a decrease in ecological quality showing a more polluted habitat from 1997 to 2010. Concurrently, 6&G habitat’s measurement represented a 40% increase in AMBI values which again, signified a worse quality status. It is noteworthy that the habitat 1&B did not fulfill the quality status criteria stated by WFD and therefore should be subsumed of managerial mitigation actions. On the other hand, habitat 6&B was classified as having a “good” EQS based on WFD norms demonstrating an acceptable habitat quality.

For M-AMBI, the reference conditions determination followed the methodology of Muxika et al., (2007). The data bank of Náttúrustofa Vestfjarða Bolungarvik with the focus on Westfjords area was searched for pristine stations with the characteristics associated with natural factors of sampling stations: depth between 5 to 20 meters, sediment type of mud-sand, and salinity range of euhaline. The appropriate stations were found in two published reports by Eiríksson et al, (2010).

Some stations which fulfilled the requirements mentioned above were chosen and their species lists were used to calculate Shannon diversity (H’), species richness (S) and AMBI values .The highest H', S and AMBI values were found in two stations which their information is defined in table 15.

Table 15: Information of stations used for determining reference conditions

Station	Place	Coordinate	Depth	Sediment	Year
H3	Hestfjordur	N 65°54.941' W 22°58.903'	10,3m	Sand-mud	2009
Grun B	Grunnavik	N 66°15.395' W 22°54.577'	16,8m	Sand-mud	2005

The WFD reference conditions criteria demonstrated that an aquatic habitat could reach a "high" EQS when sensitive species are present but no tolerant species are existent (Directive, 2000),accordingly, Borja et al., (2003) stated that the maximum value for AMBI, equal to zero, is acquired when only sensitive species are present in the habitat. In this study's concern, the major problem corresponded to deriving reference conditions was the considerable percentage of eco-group (III) and existence of some opportunistic species in benthic communities of selected reference sites resulting in AMBI reference values higher than standards for undisturbed areas. AMBI<1.2 correlated with a high quality status (table 4) were not obtained from results. This situation usually happened in originally enriched areas such as estuarine systems that other eco-groups such as tolerant or opportunistic are naturally present (Muxika et al., 2007).

Following the WFD reference standards (Directive, 2000), and Borja and Muxika, (2005) guidelines, the AMBI values associated with undisturbed habitats were not resulted. As a solution, the EG (IV) and EG (V) were removed from the list of chosen pristine stations based on Muxika et al., (2007) approach which was adopted for this kind of situation. Expert judgment (Borja et al., 2004b; Bald et al., 2005) were another alternative for the reference values determination in these circumstances. Based on the AMBI calculation following the Muxika et al., (2007) and expert judgment following Bald et al., (2005) the reference values for the relative pristine aquatic area were concluded. Values from 1 to 1.3 were delivered using the former approach whereas expert judgment suggested value of 1 for this condition. Therefore value of 1 was admitted for AMBI reference value. Consequently, the reference conditions for high EQS were determined as; H' of 4.1 (bits/individual), S of 47 and AMBI of 1.0. For bad EQS, Diversity values of zero and AMBI of 6 were adopted which were representative of an extremely polluted habitat.

The reference values were imported to the M-AMBI software version 4 (http://ambi.azti.es/) for gaining habitat's health classifications. This software with applying factor analysis (FA) and discriminant analysis (DA) (Bald et al., 2005) takes into account diversity information of relative habitats for quality measurements. The results of AMBI and M-AMBI with their associated disturbance classification and ecological status are shown in table 16 and 17.

Table 16: AMBI and M-AMBI indices results, associated classification and ecological condition for 1997

Indices and relative concepts	Stations (1997)	
	1	**6**
AMBI	3,59	1,85
Dominant eco-groups	V(58,1%),IV(22,5%)	II(37,8%),I(24,4%)
Benthic community health	Transitional to Polluted	Slightly polluted
Disturbance classification	Moderately disturbed	Slightly disturbed
Ecological status (WFD)	Moderate	Good
Shannon (H')	2,48	3,53
Species Richness (S)	16	17
M-AMBI	0,51	0,74
Disturbance classification	Disturbed	Normal
Ecological status (WFD)	Moderate/Fair	Good

As Muxika et al, (2007) suggested, and while the M-AMBI integrates diversity and ecological characteristics of the benthic communities, the M-AMBI was chosen as the main indicator for environment health determination. M-AMBI outputs similar to AMBI values represented an improving quality trend with increasing distance from the discharge area for both years 1997 and 2010. It is noteworthy that this trend is correlated with the results of chemical measurements in Pollurinn area in 1997 (Helgason et al., 2002). These analyses were not performed in 2010. Nitrogen content in water body was decreased from 0.53 mg/liter in station 1 habitat to 0.11 mg/liter in station 6 in the central parts. Phosphate and organic carbon had the same trend as well. These outcomes showed a significant association between the decrease in nutrients and increase in habitat's health condition.

Table 17: AMBI and M-AMBI indices results, associated classification and ecological condition for 2010

Indices and relative concepts	Stations (2010)		
	B	**D**	**G**
AMBI	4,46	2,93	3,09
Dominant eco-groups	V(61,2%),II(25%)	IV(41,6%),II(38.1%)	IV(44,4%),II(32,5%)
Benthic community health	Polluted	Slightly polluted	Slightly polluted
Disturbance classification	Disturbed	Slightly disturbed	Slightly disturbed
Ecological status (WFD)	Poor	Good	Good
Shannon (H')	2,20	3,03	3,98
Species Richness (S)	14	19	35
M-AMBI	0,40	0,63	0,81
Disturbance classification	Disturbed	Normal	Normal
Ecological status (WFD)	Moderate/Fair	Good	Good

Comparing station B&1 habitat's quality status in 1997 and 2010, and the quality status of the farthest station from the sewage outlets (station G & 6) showed a level of improvement signaling a slight restoration in EQS from 1997 to 2010. In terms of ecological classification no change happened in stations' health classification while comparing sampling years. Stations 1 and 6 with values of 0.51 and 0.74 were classified as "moderate" and "good" respectively, whereas congruously stations B with value of 0.4 was classified as "moderate" (though marginally and towards poor status) and stations D and G were categorized as "good" according to the WFD criteria representing a normal habitat and a rather healthy aquatic area.

Therefore, it seemed that in areas adjacent to the sewage outfall, between the discharge points to station D; 350 m overall, a degradation in habitat's health has occurred which according to the M-AMBI results, this deterioration was about 21% and in contrast, in the central parts of the fjord about 1 kilometer away from the outlets, an improvement in quality about 9% was observed (fig. 1).

5 Discussion

Using macro-fauna as bio-indicators of water and sediment pollution, is found to be more realistic than other approaches such as chemical analysis (Hall et al., 1997) since they accumulate long-term effects and influences of different sources of contamination (Denoyelle et al., 2010) and, respond rather quickly to anthropogenic and natural stresses (Pearson and Rosenberg, 1978; Dauer, 1993). In terms of macro-benthos' applicability, this study shows the same results observed in other benthic community assessment researches. The mixture of different benthic-based indices can differentiate impacted from un-impacted habitats. However benthic community indices are able to determine habitat degradation, they are not effective in identifying the agents of the environmental impact (Engle et al., 1994).

The pollutant particulates' distribution depends on the energy regime of the area. Energy regime is suggested to be highly correlated with grain size too (e.g. Newell et al., 2001). In this concern Muniz et al., (2011) states that energy profile just above sediment-water line decides the bottom sediment size; the lower the hydrodynamic energy, the smaller the grain size. The grain size sample results of this study (table 5) shows similar sediment composition (mud-sand) ranged from 77% silt-clay in central Pollurinn, 61% in discharge area represented by station B and, to the sandiest bottom composed of only 34% silt-clay in station D (fig. 1). These percentages can describe the energy regime at sediment-water interface in the sampled benthic habitats which represent a low energy area. As a result, accumulation of organic material is likely to happen. In addition, the hydrodynamic regime by controlling advection of food particles plays an important role particularly in detritus and suspension feeding population's presence and, directly or indirectly, affects the whole benthic communities' structure (Kröncke & Bergfeld, 2001).

To estimate any kinds of diversity measurements such as species richness, and also in ecological grouping for biotic indices taxa need to be sorted as species. Therefore, most of the contemporary biotic indicators use measures demanding individuals in their species level. For instance, it is demonstrated that for trophic classification of taxa, they need to be analyzed to their species level since the feeding habits can be different even in genus level (Kröncke & Bergfeld, 2001). On the other hand, Ruellet and Dauvin, (2007), suggest the development of benthic indices which applies higher taxonomic groups to decrease the cost of surveys. performing biotic measurements necessitates taxa as species therefore demand higher expertise which makes the process tangibly more costly than assessment methods based on higher groups (Rosenberg et al., 2004),besides, laboratorial misidentification is more likely to occur in the level of species (Schilling et al., 2006). For aquatic habitats' Health assessment, the WFD has stated sensitivity to organic enrichment concept as one of the parameters that should be considered in measurements which is an element particular of species level. Supra-specific levels (e.g. genus or family) since may have great sensitivity differential in lower level(s) (Rosenberg et al., 2004) cannot be used to fulfill this requirement. Also if this issue is considered from DPSIR approach, applying supra-specific levels cannot accomplish the "practicality" stage of adopting an appropriate environmental indicator; section (2-2) since a more cost-effective approach decreases the reliability of the indicator, obscuring its relative sensitivity and representativeness makes its application unjustified.

Some researchers suggest that for pollution assessments a sieve with mesh size of 1 mm is suitable (see Fitch and Crowe, 2010) whereas for benthic community composition review a smaller mesh aperture is necessary (Schlacher and Wooldridge, 1996). Also, several investigators argued that since an important part of targeted taxa are opportunistic polychaetes which are sometimes too small to be captured by 1 mm mesh size (Dauvin et al., 2007); the 0.5 mm sieve should be applied (Pinto et al., 2009). On the other hand, the amount of work for 1 mm sieve is much lesser than a 0.5 mm sieve (Rosenberg et al., 2004).

Traditionally, measurements of impacts on benthic communities used to be implemented mainly based on species richness and diversity indices (Ugland et al., 2008). It is while that, univariate indices such as Shannon diversity could not distinguish between anthropogenically disturbed and undisturbed areas in many cases (Bouchet and Sauriau, 2008) since they are dependent on parameters such as habitat type, sample size and seasonal variation (Simboura and Argyrou, 2010).

Interestingly The Shannon index though was not developed based on similar concept of biotic ecological-grouped AMBI but, delivered almost similar trends and pattern of behavior associated with a gradient of organic matter (Borja et al, 2000 and Muxika et al, 2007) which followed P-R paradigm (Pearson and Rosenberg, 1978). As a result, Shannon diversity values, based on Vincent et al., (2002) threshold assignment, represented similar EQSs as AMBI delivered and with the exception of station B of 2010, other eco-classifications are similar. Considering the suggestion of Zettler et al., (2007) that salinity gradient have an impact only on diversity indices but not on ecological-based indices, this results suggests that the lack of a salinity gradient between stations elicit diversity and biotic indices to draw out similar quality classification.

In this study, diversity indices such as Shannon and Simpson however could show the same trends of AMBI and were correlated with chemical analysis implemented in 1997 could not accurately discriminate between pristine and impacted areas. Some pristine habitats with the same natural features of study areas delivered similar diversity values (Eiríksson et al., 2010) classifying pristine areas in moderate health status. The diversity indices do not consider sensitivity and tolerance of the species which this issue can be one of the reasons of their inefficiency (Fitch and Crowe, 2010). They cannot reliably identify anthropogenic perturbation so they do not have sensitivity and representative characteristics of a proper index in DPSIR managerial approach. Therefore, diversity indices though informative are not used for habitat health measurements independently.

Since the WFD looks for anthropogenic changes in the EcoQ of habitats, natural variability should be considered in the methodologies used, including reference conditions assignment. In terms of natural and anthropogenic habitats' sedimentary characteristics,

man-induced organic enrichment and muddy sediments are usually associated with presence of species tolerant to excess organic load. If enrichment continues appearance of opportunistic species and hypoxia will be the results (Majeed, 1987). the presence of tolerant and opportunistic taxa could not necessarily be associated with impoverished habitats instead, might be related to naturally stressed environments (Dauvin et al., 2007; Fitch & Crowe, 2010) such as enriched estuarine systems (Gamito, 2009).

AMBI values derived from pristine areas are in the range of good EQS but not in the high interval as expected. For gaining high quality status from AMBI software the community needs to be dominated by species sensitive to organic enrichment (EG I); a condition that does not exist in these pristine habitats. In this study's case, the pristine sites with sediment type of mud-sand showed communities composed of EG (II) of 21- 55%, EG (III) of 4-18%, EG (IV) of 3-12% also Oligocheata (*Tubificoidec benedii*) opportunistic group were present in habitats. The presence of EG (II) including taxa indifferent to organic enrichment and EG (III) containing species tolerant to organic enrichment suggests that the related pristine habitats are organically enriched. In transitional water ecosystems or enriched estuaries the dominance of tolerant species may occur (Simboura and Argyrou, 2010). To tackle with this problem Teixeira et al. (2009) imply that decrease in EG (III) coefficient increases the efficiency of AMBI.

The AMBI software does not deliver low values representing a high health status. Application of Muxika et al., (2007) and Bald et al., (2005) approaches are the available solutions for reference condition determination in enriched estuaries case described by Gamito, (2009). Consequently, the values representative of a high quality status are calculated for pristine areas and are applied for health quality status of sites in Pollurinn area.

However, the AMBI is criticized by some researchers suggesting that this index may show some weaknesses in inner parts of some fjords due to low salinity and its fluctuations, inducing suppression in relative communities' diversity factors. These shortcomings do not seem to be influential in the present study's case since, this fjord does not have significant salinity changes and its range is fixed on euhaline. Concerning, M-AMBI, Ruellet and

Dauvin, (2007) however, suggest the complementary use of indices with different concepts (diversity, throphic and, ecological grouped) argue that inclusion of both species richness and H' in the M-AMBI calculation puts too much emphasis on diversity. Technically the usage of two diversity parameters cannot be interpreted as such a deficiency since mentioned diversity and ecological parameters are not communicating the same information and in the same scale, besides, although they should be correlated in trends (they are supposed to deliver similar EQSs) they do have different variance ratio, so, their effect in the whole value can be more or less different but not necessarily biased. M-AMBI has taken into account diversity, abundance and pollution-sensitivity factors highlighted in the WFD, yet, since this index is rather new (Muxika et al, 2007) the proposed shortcoming should be studied in a long-term period and throughout distinct geographical areas.

6 Conclusions and Recommendation:

6-1 Conclusion

Anthropogenic pollution in costal and marine areas is a complex issue that requires managerial responses. Several tools can be used to estimate impacts according to environmental legislations. The M-AMBI index was developed to implement the WFD criteria and it was used in this study to estimate the health status of Pollurinn area. This index has a well-known ecological background and it is as cost effective as the diversity indices which are used regularly. It can capture information regarding human induced actions and therefore in terms of representativeness and sensitivity is proper and fulfills the requirements for implementing the impact component of DPSIR approach. The results of 2010 show a moderate health quality for the areas close to the sewage outlets and good health quality for the central parts of Pollurinn. Comparison between 1997 and 2010 studies displays about 21% quality deterioration in the area adjacent to the sewage outlets and about 10% percent improvement in central parts of the Pollurinn.

Three different sets of indices were used in this study; diversity, ecological and multivariate grouped indices. Several diversity indices were applied and delivered almost similar temporal and spatial trends for analyzed stations (table 8). The results mirrored the fact that diversity indices however putting emphasis on different diversity aspects (dominance, species richness or evenness) communicated similar patterns. In the whole, diversity indices could illustrate the gradient of pollution but were not able to determine the degree of deterioration in a site since their resultant values of pristine and impacted habitats showed that they are not able to acceptably distinguish between natural and anthropogenic stressors.

AMBI could distinguish a pollution gradient from the sewage outlets to central parts of the Pollurinn area. For 2010, it revealed a poor health status in the station adjacent to discharge area and good quality condition for other two stations. Moreover, by comparing the results of 1997 and 2010 general quality degradation is observable.

The AMBI results of pristine sites could not deliver values associated with high health status. As suggested by variety of researches muddy benthic areas and natural organic enrichment accommodates opportunistic species which contribute to higher values signaling lower quality for a habitat. To gain values representing a high quality status (AMBI<1.2) the opportunistic species were deducted from the list of taxa and value of 1 was rendered by the software and adopted as the reference value. This result was concordant with expert judgment for such estuarine systems suggested by Bald et al. (2005) and Borja et al. (2004b) which was 1.

M-AMBI was used as an index to integrate diversity and ecological information of benthic habitats for determining the EQS of them. Due to improvement in diversity factors M-AMBI showed a restoration in the central parts of Pollurinn.

In terms of habitat classification by applying different indices just in station B, H' and M-AMBI delivered "moderate" EQS whereas AMBI rendered "poor" EQS for the station. For the rest of the reviewed stations, H', AMBI and M-AMBI gave out similar EcoQ classification. By comparing 1997 and 2010 results, all the indices have tracked degradation in areas close to the wastewater discharge points. However indices results indicated the fact that the area's quality status has remained in the "moderate" class but it is clear that the quality is declining; M-AMBI results showed 21% quality deterioration. While, even the "moderate" quality status is not acceptable in the WFD standard and necessitates mitigating or preventive managerial actions to improve the quality status to the range of "good", therefore the present declining quality trend may require emergent response.

6-2 Recommendations

Environmental management actions are performed in response to ecological assessment. The present assessment for Pollurinn does not show a significant degradation for the studied area but the concern somewhat remains there in base of cumulative effects. Two potential responses are sanitation system and redirecting a part of the discharge flow to the open sea (fig. 2) to have a better rarefaction and bio-decomposition. In this concern a pipe with appropriate length should be used in order to bypass the surface sea currents to avoid the wastewater to move back towards the beach.

Assessment´s accuracy depends on the available information and their interpretations. This is why a regular monitoring can give a better resolution in assessing ecological quality of concerned habitats by presenting temporal and spatial comparable data. Skutulsfjörður which is subject to anthropogenic stressors requires monitoring and timely actions to prevent deteriorations that may negatively influence other practices such as aquaculture and recreational activities in the entire fjord. In addition to macro-faunal community survey, chemical analysis of sediment and/or water column can give a better understanding of the ongoing environmental processes and provide information regarding the quality of organic matter sinking on the sea bottom (Kröncke & Bergfeld, 2001). The results of chemical analysis conducted in 1997 (Helgason et al., 2002) on the pelagic phase illustrated a significant correlation between distance from discharge area, nutrient concentration and health status of habitats. Different ways for benthic community data interpretation were proposed in this paper and the subject is still open for new researches. The M-AMBI biotic index approach seems an appropriate method for the EcoQ assessment of the mentioned area. To decrease the probability of deficiencies which may contribute to inaccuracy of this method, both sampling time and sampling tool can be improved. Sampling time for benthic community should be performed in the early summer to avoid the excessive presence of young individuals (Eiriksson, T, personal communication). Regarding sampling tool, the ideal grab should have a greater covering area in order to fulfill the AMBI's mentioned criteria of 0.25 m^2 which requires less sampling effort.

In addition, using a complementary method based on the troghic grouping of the benthic macro-invertibrates is another practical approach. A trophic index such as ITI (Ruellet and Dauvin, 2007) can provide additional information about the status of the benthic community. Classification of species based on their trophic groups although needs extra skills but does not need further work or expenses in taxonomic analysis or sampling methodologies. Therefore this study recommends the application of a trophic-grouped index in addition to diversity and ecological-grouped indices. This application provides another useful evaluative parameter which contributes to a higher accuracy in EQS determination. As an example, in estimating reference condition for M-AMBI, an extra set of information (trophic properties) can be helpful in differentiating human induced from natural stresses in a habitat.

Reference

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