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Meiofauna as a Tool for Marine Ecosystem Biomonitoring

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1. Introduction

Meiofauna are the metazoan component of the benthos, and also include large protozoans (e.g. foraminifera). They are defined by their body size (44-1000 µm) and are the most diversified element of the marine biota: as many as 24 of the 35 animal phyla have meiobenthic representatives which live in meiofauna, whether for all their life or just temporarily. It is the most abundant benthic group in the marine realm, and is thought to be closely connected to other faunal compartments of the benthic system. The function of meiofauna in marine benthic systems seems to be much more complex than previously supposed, and requires investigation to clarify their ecological importance in the benthic domain (see Balsamo et al., 2010 for review).

The aims of this paper are: to review advances in the use of meiofauna as a bio-indicator for the monitoring of marine ecosystems; and to highlight future perspectives of this approach. In particular, the use of the two most abundant and diverse meiofaunal groups (Foraminifera and Nematoda) will be considered.

2. Meiofauna as an environmental bio-indicator in marine ecosystems

The use of meiofauna as a biological indicator is a more recent development than the utilization of macrofauna in the assessment and monitoring of aquatic ecosystems (Coull & Chandler, 1992). The advantages of the former are numerous and strongly emphasized by Kennedy & Jacoby (1999), while some of the arguments traditionally advanced against their use underline difficulties in identification, the high rate of sampling frequency and the microscopic size of these organisms. However, new technologies and tools, such as standardized methodologies, electronic identification keys, molecular approaches and the creation of new indices, currently allow for and promote the use of meiofauna in ecological studies (see Giere, 2009, for review).

3. Two representative meiofaunal groups: Foraminifera and Nematoda

Among the meiobenthic protozoans, Foraminifera (class Foraminifera, *phylum* Granuloreticulata) are the most abundant and diverse of the shelled microorganisms in the oceans (Sen Gupta, 1999). The *phylum* Nematoda, meanwhile, is the most plentiful (often >50% of the total meiofauna, up to >90% in deep-sea sediments) and diverse metazoan meiofauna taxon (Boucher & Lambshead, 1995; Giere, 2009).

Foraminifera play a significant role in global biogeochemical cycles of inorganic and organic compounds, making them one of the most important groups on Earth (Yanko et al., 1999). Furthermore, many foraminiferal taxa secrete a carbonate shell that is readily preserved, and so record evidence of environmental stresses and changes over time. They are commonly small and abundant compared to other hard-shelled taxa and easy to collect, providing a highly reliable database for statistical analysis, even when only a limited volume of samples is available. Because of their widespread distribution, short life and reproductive cycles, high biodiversity, and specific ecological requirements, foraminifera may respond to environmental changes (e.g. Alve, 1995; Murray & Alve, 2002; Yanko et al., 1994). Moreover, with their high number of species and genera - around three to four thousand of the former (Murray, 2007) - benthic foraminifera are more likely to contain a variety of specialists that are sensitive to environmental change. For all of these reasons, they are particularly sensitive and can thus be successfully used for their value as bio-indicators of environmental change in a wide range of marine environments (Armynot du Châtelet & Debenay, 2010; Frontalini & Coccioni, 2011). The use of benthic foraminifera as bio-indicators of environmental quality can be investigated in terms of population density and diversity, assemblage structure, reproduction capability, test morphology - including size (dwarfism), prolocular morphology, ultrastructure, pyritization, abnormality, and the chemistry of the test. The study of pollution effects on benthic foraminifera and their use as proxies began in the 1960s (Boltovskoy, 1965; Resig, 1960; Watkins, 1961), and has been increasingly developed in recent decades as a result of environmental research (for reviews, see Alve, 1995; Boltovskoy et al., 1991; Frontalini & Coccioni, 2011; Murray & Alve, 2002; Nigam et al., 2006; Yanko et al., 1994).

The ecological value of nematodes is related not only to their notable quantitative importance in the benthic domain, but also to their pivotal role within the trophic chains of aquatic ecosystems and the stabilizing effects of shores (Platt & Warwick, 1980). The advantages cited above of using foraminifera as bio-indicators could be extended to nematodes (see Heip et al., 1985; Vanaverbeke et al., 2011), and it is for this reason that this *phylum* was recently proposed as an indicator with which to assess the ecological quality of marine ecosystems according to the Water Framework Directive (WFD, Directive 2000/60/EC) (Moreno et al., 2011). The nematode assemblage is generally studied in terms of density, diversity, assemblage structure, trophic guilds, life history strategies, body size and biological trait analysis. The Index of Trophic Diversity (ITD; Heip et al. 1984) and the Maturity index (MI), which is based on the ecological characteristics and reproductive strategies of nematodes (Bongers, 1990; Bongers et al., 1991), are the two indices that are more commonly applied in ecological studies of nematode assemblages.

Moreno et al. (2011), in analyzing the most frequently used indices in the ecological assessment of nematodes, have suggested that the taxonomic approach (presence/absence of specific indicator genera) reveals the best correspondence between environmental status and biological response, whereas, among the synthetic descriptors, c-p % composition and diversity (Shannon index, H′) can be used to evaluate ecological quality status efficiently.

4. Foraminiferal and nematode responses to different pollution sources and disturbances

Human activities, including industry, agriculture, mining, dredging, and dumping introduce large amounts of pollutants into marine areas, causing permanent and significant disturbance to and a major impact on ecosystems. Pollution may also occur in offshore environments, such as drilling rigs and oil platforms. When present in sufficient quantities, and under certain conditions, pollutants influence the biota living within and at the sediment interface. The benthic community generally responds to adverse ecological conditions, primarily by undergoing: i) local extinctions; ii) compositional biocenosis and trophic group changes; iii) assemblage modifications, which include changes in abundance and diversity; iv) dwarfism (Lilliput effect); v) changes in reproduction capability; and vi) cytological, biological and morphological variations (Fig. 1). Indeed, many studies of the effects on foraminiferal and nematode communities of a variety of disturbances have been carried out in different parts of the world (see, Fig. 2).

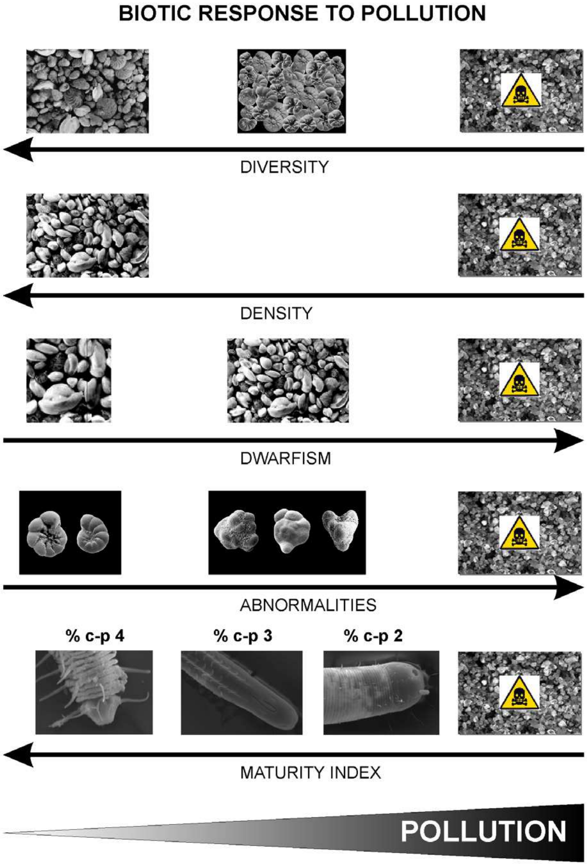

Fig. 1. Schematic response of Foraminifera and Nematoda to pollution.

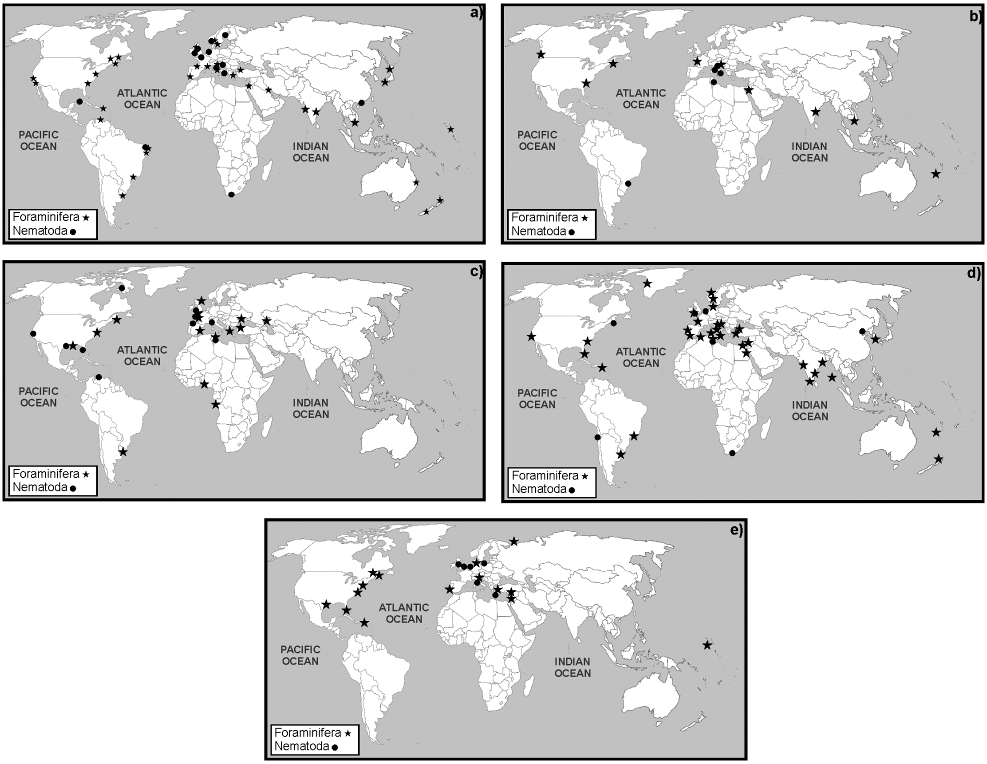

Fig. 2. Global distribution of the most significant studies involving the responses of benthic foraminifera and nematodes to: a) sewage discharge; b) aquaculture; c) hydrocarbons; d) trace elements; and e) other types of stress (e.g. thermal pollution and physical disturbance)

4.1 Sewage discharge and organic enrichment

The input of organic matter may have deleterious effects on marine ecosystems and their benthic community by inducing high nutrient levels. This can ultimately lead to oxygen deficiency, also known as eutrophication. Eutrophication in shelf environments is mainly due to the nutrient load of rivers, which is in turn related to river flow, the catchment area, and the industrialization and urbanization of the drainage basin. Both the quantity and the quality (labile - more degradable; and refractory - more difficult to metabolize) of the organic material have an impact on the benthic communities.

The main food sources (not taking into account the endosymbiotic species) for benthic foraminifera are organic matter (mainly the labile fraction) and the entire bacterial community that proliferates thereon. Moreover, when present in low quantities, like in deep-sea environments, the organic matter acts as a limiting factor. In contrast, it exerts a strong influence in marine coastal areas, where periodically enhanced production and the riverine input of nutrients may lead to an increase in the standing crop of the benthic foraminifera (e.g. Watkins, 1961). According to Alve (1995), the increase in organic matter in the so-called defined 'hyper-trophic' zone may stimulate the growth of large populations of benthic foraminifera. Alternatively, an excess of organic material can negatively influence the benthic community. In particular, dysoxic-anoxic conditions can be established where the oxygen demand to metabolize the organic matter exceeds its supply. In fact, most studies have revealed that around the areas of organic matter supply (e.g. outfall and pipe), a barren-abiotic zone might occur in response to the development of anoxic conditions (Alve, 1995). An additional benthic foraminiferal response to organic matter pollution includes the modification of the original assemblages. In particular, an increased dominance of agglutinated foraminiferal taxa at pollution sites has been found. This effect is probably the result of the taphonomic process, which is related to the dissolution induced by the input of large quantities of organic matter that have altered the properties of the sediments. It has been inferred that both increasing the concentration of the organic matter and the lower availability of oxygen may progressively exclude or favour some taxa. As a consequence, a species-specific response and a change in assemblage composition are to be expected according to the degree of eutrophication (Van der Zwaan, 2000). In fact, some taxa, including *Buliminella elegantissima* and *Bulimina marginata denudata,* were overwhelmingly dominant in the living assemblages close to the outfall area for the Los Angeles County sewage system (Bandy et al., 1964). It has also been documented that some taxa migrate upwards in sediment when oxygen concentrations diminish, apparently mirroring the oxygen gradient (Alve & Bernhard, 1995). More recently, on the basis of bibliographic analyses, Frontalini & Coccioni (2011) synthetized the degree of tolerance of some benthic foraminiferal taxa, particularly *Hopkinsina pacifica*, *Nonionella turgida*, *Valvulineria bradyana* and *Uvigerina peregrina,* along with several species belonging to the genera *Bulimina*, *Buliminella*, *Fursenkoina*, *Bolivina* and *Epistominella*, which were regarded as being the most tolerant to organic matter enrichment. Changes in density and assemblage composition are not the only response by benthic foraminifera to organic matter, with a reduction of species diversification also being reported in the vicinity of many outfalls (e.g. Resig, 1960). The development of an azoic area, high concentrations of sulphides, and reducing conditions are documented around the point sources (mainly pulp and paper mills) of refractory organic matter. In more restricted environments (e.g. fjords), where the connection with the open sea is limited, anoxia may occur very rapidly. The effect of increased human-induced organic flux was documented in short cores in Drammensfjord and Frierfjord (Norway), where marked changes in assemblage composition, diversity and abundance were reported (Alve, 1991a, 1999). An additional, important contribution, which exemplifies the effect of eutrophication over time, was provided by Barmawidjaja et al. (1995), who accurately described the distribution of benthic foraminifera in a 57-long core drilled in front of the Po Delta. Significant benthic foraminiferal changes and steps were observed. These changes were ascribed to: human-induced alterations of the main outflow canals of the Po River (1840 and 1880); a steady increase of the nutrient load (from 1900 onwards); the intensification of eutrophication (1930); and the first signs of anoxic events (1960). Since 1880, the original assemblages have been gradually replaced by more stress tolerant versions (e.g. dominated by *N. turgida*). Another substantial change occurred in 1930, when opportunistic stress tolerant species (*H. pacifica*, *Bolivina seminuda* and *Quinqueloculina stalkeri*) became dominant. The authors regarded *H. pacifica* (peaking in 1960) as the most stress tolerant taxon. Foraminiferal assemblages changes over both time and space have been reported in the Osaka Bay by Tsujimoto et al. (2006a,b), who reported for the eutrophicated inner part of the bay an assemblages dominated by agglutinated forms (*Trochammina hadai* and *Eggerella advena*). The same authors documented marked foraminiferal assemblages over the past 50 years, where the dominance of agglutinants, which was linked to the increase of eutrophication from the 1960s to the 1970s, was followed by a decrease of these forms in response to the imposition of improved environmental regulations. On the basis of a comparison of foraminiferal assemblages from samples collected in 1983 and 2001 in the northern sector of the Lagoon of Venice, Albani et al. (2010) reported unchanged conditions for 50% of the lagoon. They also documented the effectiveness of the purification plant that had been operating since the 1980s, as well as improvements in the water quality in the area near Porto Marghera. The enrichment of organic matter has also been regarded as responsible for increasing the number of abnormal tests and the emergence of large and protruding proloculi (e.g. Seiglie, 1971). It has been also reported that adult specimens are smaller in nutrient-rich, but oxygenated, environments, probably in response to rapid growth and high reproductive rates (Yanko et al., 1999).

In nematode assemblages, a general increase in abundance (in particular of deposit feeders) and decreases in diversity can be observed in correspondence with sewage outfalls or organic enrichment, although responses are not always unequivocal (Armenteros et al., 2010; Austen & Widdicombe, 2006; review by Coull & Chandler, 1992; Fraschetti et al., 2006; Sandulli & De Nicola-Giudici, 1990, 1991; Schratzberger & Warwick, 1998; Schratzberger et al., 2008; Somerfield et al., 2003) (Fig. 2a). In a laboratory experiment, Schratzberger & Warwick (1998) tested the response of two estuarine nematode assemblages (from organic-poor sandy and organic-rich muddy estuaries, respectively) to the intensity and frequency of organic enrichment. In the sand microcosms, the nematode assemblage changed more drastically than in the muddy versions: diversity and species' richness decreased significantly, including in response to low doses, whereas in the muddy estuaries this only occurred at medium and high levels of organic loading. The authors concluded that sand nematodes inhabiting a less organically enriched environment have a lower ecological tolerance to this type of stress than their mud counterparts. Accordingly, communities of different sediment types may be affected differently by organic disturbance.

A multivariate approach has revealed significant differences between the controls and both of the organically enriched treatments. Some indicator species of the different disturbance levels have also been identified. In sand microcosms with low dose levels, there was an initial increase in abundance of *Chromadora nudicapitata, Daptonema normandicum* and *D. hirsutum*, followed by an increase of *D. hirsutum* and a contextual decrease of *D. setosum* and *Odontophora longisetosa.* High levels of organic enrichment were characterized by a reduction in abundance of *O. longisetosa*, *D. setosum* and *Chromaspirina inglisi*. In contrast, in the mud microcosms, the discriminating low-abundance species were *Ptycholaimellus ponticus* and *D. procercum,* whereas *Terschellingia communis*, *T. longicaudata*, *Metachromadora vivipara* and *Sabatieria breviseta* decreased only in high dose levels. Some of these species are well-known for their physiological and behavioural adaptations to poorly oxygenated environments. These changes include a low respiratory rate, slow movements and the presence of intracellular inclusions of insoluble metal sulphide depositions (Nicholas et al., 1987; Warwick & Gee, 1984; Warwick & Price, 1979). Schratzberger & Warwick (1998) suggested that many small doses of a same amount of organic matter have a milder effect on assemblage structure than when administered in fewer but larger doses. The presence of some of the species' indicators of organic enrichment referred to by Schratzberger & Warwick (1998) has also been reported by other authors in relation to organically disturbed sediment. A recent experiment in Cienfuegos Bay (Armenteros et al., 2010) highlighted nematode abundance and diversity decreases as well as alterations in taxonomic and trophic structure. The authors suggested that the accumulation of by-products of bacterial metabolism (i.e. ammonia and hydrogen sulphide) is more deleterious for nematodes than oxygen *per se*. This is because nematodes have developed various behavioural (e.g. migration to 'oxygen islands') and/or physiological mechanisms (e.g. symbiosis with bacteria, low metabolic rate) to cope with hypoxic and even temporal anoxic events. Species-specific responses of nematode assemblages to organic enrichment, already documented by Schratzberger & Warwick (1998) and Schratzberger et al. (2008), have also been recorded by Armenteros et al. (2010), who report *S. pulchra*, *S. parasitifera*, *T. communis*, *T. gourbaultae* and *T. longicaudata* to be discriminant species of different levels of organic enrichment. The genera *Sabateria* and *Terschellingia* are among the dominant taxa in both the Pina Basin, an urbanized embayment on the coast of Pernambuco, Brazil (Somerfield et al., 2003), and some Ligurian harbours that are characterized by organic enrichment (Moreno et al., 2008, 2009). Fraschetti et al. (2006) documented a decrease in nematode abundance in a rocky subtidal area influenced by sewage outfall, but there was no reduction in the number of genera or changes to their taxonomic composition. Interestingly, the MI did not change significantly between the impacted and the control sites: its values were high overall, as was the detected percentage of c-p 4 and the epistrate feeders. This finding suggested that sewage discharge did not affect nematode assemblages in terms of favouring colonizers.

4.2 Aquaculture

Aquaculture has undergone a dramatic expansion worldwide, inducing a growing interest in and concern for its potential impact on coastal marine environments. The first effect is the build-up of faeces or pseudofaeces (biodeposition) on the benthic habitat just below the culture area. This can alter particle-size, organic content, and nitrogen-cycling, which can easily induce eutrophication and a decrease in oxygen penetration in the sediment (see, for review, Mirto et al., 2002; Netto & Valgas, 2010).

Only a few studies have been carried out on the foraminiferal assemblages that are specifically affected by aquaculture. These were mainly conducted in the Atlantic Ocean (Bouchet et al., 2007; Clark, 1971; Schafer et al., 1995; Scott et al., 1995), the Pacific Ocean (Debenay et al., 2009), the Red Sea (Angel et al., 2000), the China Sea (Debenay & Luan, 2006), the Japan Sea (Tarasova & Preobrazhenskaya, 2007), the Adriatic Sea (Vidović et al., 2009) and the Bay of Bengal (Jayaraju et al., 2008) (Fig. 2b). The high input of easily degradable organic matter and the wide variety of chemical and biological products used in aquaculture may introduce into the water body persistent and potentially toxic residues. Accordingly, benthic foraminifera may be negatively affected by aquacultural activities. In particular, foraminiferal population dynamics were investigated at an outfall site of land-based salmonid aquaculture (Clam Bay, Nova Scotia) by Clark (1971), who reported a strongly inverse relationship between local foraminiferal density and the discharge of fish meal. Moreover, a reduction of foraminiferal abundance, but no marked compositional changes in the assemblages, has been identified under salmon cages (finfish aquaculture) as a response to increased organic matter sedimentation (Scott et al., 1995). Aquaculture facilities have also produced localized anoxic areas which occur under fish cages, as seen in eastern Canada. This leads to less diversity and an increase of non-calcareous foraminifera (Schafer et al., 1995). The mechanisms responsible for these changes are probably not linked directly to an increased food supply, but can be found in the changing of sediment properties, the sedimentation rate and the low values of oxygen concentrations. In particular, a strongly positive correlation between the Foraminiferal Abnormality Index (FAI) (*sensu* Coccioni et al. (2005)) and: (1) the quantity of easily oxidized material deposited at the bottom of shrimp ponds; and (2) the sediment oxygen demand has been documented in New Caledonia (Debenay et al., 2009). These findings were corroborated by the poorly diversified assemblages, which were mainly dominated by *Ammonia tepida* and *Quinqueloculina seminula*, indicating very restricted conditions and major environmental stress. It has also been suggested that a very high FAI could be a potential indicator of great accumulations of native organic matter, leading to a high sediment oxygen demand. Meanwhile, changes in assemblage composition and decreased density have been observed under scallop cages (Minonosok Bay, Sea of Japan) by Tarasova & Preobrazhenskaya (2007). Higher proportions of agglutinated species as a response to shrimp farming and rice culture were also documented in the Mekong Delta (Vietnam) by Debenay & Luan (2006). Moreover, two sediment transects were investigated below a commercial fish farm in the Gulf of Eilat (Red Sea) by Angel et al. (2000), who found higher abundances of agglutinated species in the ''hypertrophic'' zone adjacent to the fish cages. The same author documented abnormal specimens, mainly *Peneroplis planatus*, in the zone, but it was unclear whether they had formed as a result of adverse conditions related to the fish farm. The impact of fish farming on benthic foraminiferal assemblages was evaluated in Drvenik Veliki Island (Croatia) by Vidović et al. (2009), who reported alterations in their composition but no changes in diversity. The same authors suggested that the presence, absence or relative abundance of species could be a possible indicator of organic enrichment due to fish farm activities.

Although free-living nematodes were assumed to tolerate sediment organic enrichment, field observations pointed to the general impact of fish farms on nematode abundance, namely a decrease in the levels thereof (La Rosa et al., 2001; Mazzola et al., 2002; Mirto et al., 2002). When it comes to the biomass, however, Duplisea & Hargrave (1996) did not find any variations below the cages. In contrast, Mirto et al., (2002) documented an increase in body weight as an effect of fish farm biodeposition, which was due to the dominance of some large, but tolerant, Comesomatidae genera (mainly *Pierrickia*, *Dorylaimopsis*, *Sabatieria*). The MI from fish farm sediment provided more congruent results, revealing an index decrease which was parallel to the biodeposition increase. This result also means that there is a good tool with which to detect the resilience of nematode assemblages. In contrast, the ITD did not highlight any clear impact of this type of disturbance (Mirto et al., 2002), with the same authors also reporting unambiguous responses to fish-farm disturbance, including in respect of the diversity parameters of the nematode community (*k*-dominance curves, richness, H′ and J). Different results were documented in a bluefin tuna fish farm in Vibo Marina, Italy (Western Mediterranean Sea) by Vezzulli et al. (2008), who recorded unclear variations in H′, J and the MI between cages and control stations, probably due to the limited impact of this type of farm. However, the analysis of the community at the genus level gave more consistent results in both of the investigations. Mirto et al. (2002) found that some nematode genera were highly sensitive to biodeposition (*Setosabatieria*, *Latronema* and *Elzalia*), and disappeared almost completely in farm sediments, whereas other tolerant genera mainly increased their dominance (*Sabatieria*, *Dorylaimopsis* and *Oxystomina*). Vezzulli et al. (2008) documented a dominance of *Tricoma*, *Desmoscolex*, *Quadricoma* and *Halalaimus* at their control station, and *Daptonema*, *Marylynnia*, *Sabatieria* and *Terschellingia* at the fish farm stations. Consequently, given that the indices cited above are not always sensitive enough to detect fish farm disturbance, the authors suggested that the identification of sensitive/tolerant nematode genera is the best indicator when it comes to noting even early benthic community changes below fish cages. The impact of intensive fish farming on the benthic environment seems to be greater than that of mussel farming, since mussels feed on natural resources (suspended particles) and are not sustained by any additional intensive feeding. In this respect, Danovaro et al. (2004) have documented the limited effect of a mussel farm on meiofaunal structure. This was due to the farm's minimum impact on sediment oxygen penetration and the biochemical compositions of the sediment organic matter. Yet several other investigations have reported that mussel biodeposition can have a serious effect on farm sediments, leading to severe organic matter accumulation and consequential reducing conditions, possibly also inducing significant changes in meiofaunal structure and, in particular, nematode assemblages (Mahmoudi et al., 2008; Mirto et al., 2000; Netto & Valgas, 2010). Many factors can influence these seemingly conflicting results, such as the culturing method used, the density of the cultivated mussels, the water depth and the hydrographical conditions in the system under investigation (Danovaro et al., 2004). In particular, H', richness and J significantly decreased in the mussel farm, which was probably related to changes in the sediment beneath the cages, which is characterized by a higher silt-clay percentage (Mahmoudi et al., 2008; Netto & Valgas, 2010). Indeed, nematode diversity is significantly affected by even slight granulometric variations (see Semprucci et al., 2010a and references therein). Multivariate analyses have highlighted that *Mesacanthion diplechma* was highly sensitive to the impact of a mussel farm, whereas *Paracomesoma dubium*, *T. longicaudata* and *T. communis* were very resistant thereto (Mahmoudi et al., 2008). A comparable opportunistic assemblage was reported by Netto & Valgas (2010), with an increase being mainly found in the tolerant genera of *Terschellingia*, *Sabatieria* and *Daptonema*. Among the functional parameters of the assemblage analyzed, the trophic diversity resulted lower below the cages, and selective deposit feeders or microvorous species were dominant (Netto & Valgas, 2010), probably because of the high microbial densities (Mirto et al. 2000). Meanwhile, the MI values were low due to a significant increase in opportunistic species, in particular the c-p 2 class (Netto & Valgas, 2010).

4.3 Hydrocarbons

In marine systems, the major sources of hydrocarbon contamination are oil exploration and production, natural seeps, atmospheric input, tanker accidents, industrial discharge, and urban run-off (Beyrem et al., 2010).

The effect of hydrocarbons on benthic foraminiferal assemblages in field and experimental studies has been evaluated in only a few papers (e.g. Alve, 1995; Denoyelle et al., 2010, in press; Ernst et al., 2006; Lockin & Maddocks, 1982; Mayer, 1980; Mojtahid et al., 2006; Morvan et al., 2004; Murray, 1985; Sabean et al., 2009; Vénec-Peyre, 1981; Whitcomb, 1978; Yanko & Flexer, 1991; Yanko et al., 1994) (Fig. 2c). A culture experiment on *Ammonia beccarii* and *Allogromia latilocollaris* exposed to crude oil revealed an inhibition in both reproduction and growth, while exposure to oil distillates induced narcosis, resulting in the death of all specimens (Whitcomb, 1978). The effect of a crude oil spill on the benthic foraminiferal assemblages on the coast of Brittany (France), which was caused by the Amoco Cadiz accident, was evaluated by Vénec-Peyre (1981), who reported morphological abnormalities but no changes in diversity or density. Meanwhile, minor or no negative effects of petroleum operations were in evidence in several platforms in Louisiana and the North Sea (Lockin & Maddocks, 1982; Murray, 1985). In contrast, a marked negative effect on benthic foraminiferal assemblages and a reduction of density and diversity were recorded in the Caspian Sea (Mayer, 1980) and the Odessa Bay (Yanko & Flexer, 1991). In order to address the response of benthic foraminifera to the 'Erika' oil spill, Morvan et al. (2004) evaluated at monthly/bimonthly intervals a site situated on the tidal mudflat in the Bay of Bourgneuf (France). Although a clear link between the occurrence of abnormalities and oil pollution was not found, an impoverished fauna in terms of density and species' richness was documented. The exposure of *A. tepida* at different concentrations of oil mixed in seawater revealed that the number of juveniles per reproduction event was lower in contaminated than in control cultures (Morvan et al., 2004). This experiment clearly documented the potential impact of oil on foraminiferal test shape, cytology and reproduction processes. Accordingly, in a laboratory experiment on the impact of oil on intertidal faunas, it was shown that the toxicity of oil components may lead to an increased mortality of benthic foraminiferal faunas Ernst et al. (2006). The effect of the discharge of oily drill cuttings on benthic foraminiferal assemblages was evaluated in the outer continental shelf off Congo (Africa) by Mojtahid et al. (2006). Different foraminiferal zones were determined according to pollution intensity. Low foraminiferal densities were recognized at the immediate vicinity of the discharge point, whereas the zone slightly further away from the disposal sites was characterized by very high foraminiferal densities and dominated by opportunistic taxa like *Bulimina aculeata*, *Bulimina marginata*, *Spiroplectinella sagittula* (reported as *T. sagittula*), *Trifarina bradyi* and *Bolivina* spp. Furthermore, foraminiferal densities seem to decrease along with the dominance of the opportunistic taxa. The environmental impact of weathered crude oil on benthic foraminifera in an Atlantic coastal salt marsh was evaluated by Sabean et al. (2009), who demonstrated the negative impact of oil on foraminiferal assemblages, as testified by a dramatic increase in abnormalities in *Miliammina fusca* when compared with the non-oiled control. Meanwhile, benthic foraminifera and macrofauna were evaluated as bio-indicators of an oil-based drill mud disposal site off Congo (Denoyelle et al., 2010). Poor faunas, dominated by some very tolerant taxa at the most polluted sites, were found, whereas greatly increased densities thereof, dominated by opportunistic taxa, were identified slightly further away from the disposal site. A chronic bioassay method has been developed by Denoyelle et al. (in press), who incubated the foraminifera *A. tepida* for 30 days in natural seawater with different concentrations of Fuel Oil no. 2. It was found that increased concentrations of this pollutant induced a significant decrease in the percentage of individuals displaying both pseudopodal activity and newly built chambers. This response clearly varies in terms of the function of the concentrations of the added pollutants. Although the impact of hydrocarbon pollution is far from being fully understood, increased mortality and abnormality and decreased density and diversity are among the effects on benthic foraminiferal assemblages.

Variable responses in terms of nematode abundance after oil contamination have been reported in the relevant literature: abundance may be relatively unaffected (Boucher, 1980; Elmgren et al., 1983; Fricke et al., 1981; Gee et al., 1992), or, in contrast, can significantly fall (Beyrem et al., 2010; Carman et al., 1995; Danovaro et al., 1995, 1999; Elmgren et al., 1980; Giere, 1979; Mahmoudi et al., 2002, 2005). The phenomenon of rapid recovery is present only in some cases (Fig. 2c). These controversial results may be due to: different dosages and the toxicity of hydrocarbons; variable responses to the disturbance of the species; the different bioavailability of hydrocarbons with sediment type; and the hydrodynamic conditions. Furthermore, different responses to fuel oil contamination may also be a consequence of community chronic exposure, which alters biota sensitivity to these contaminants (Mahmoudi et al., 2005). Accordingly, a benthic community from a contaminated area is more tolerant to hydrocarbon contamination than a comparable community from a less contaminated environment (Carman et al., 2000). Beyrem et al. (2007) documented that if the presence of diesel alone does not seem to affect total nematode abundance, the diesel-metal combination may cause significant abundance decreases as a result of the higher production of mucus-exopolymers by diesel. Exopolymers are in fact known to have a strong affinity for a variety of metals, and are readily consumed by benthic organisms, thus increasing the exposure of animals to metals (see Beyrem et al., 2007 and references therein). A reduction of species' diversity has been also reported by several other authors (Boucher, 1980; Boucher et al., 1981; Danovaro et al., 1995; Mahmoudi et al., 2002, 2005), probably due to a general increase in the mortality of the most sensitive species (Beyrem & Aïssa, 2000; Carman et al., 2000; Mahmoudi et al., 2002, 2005). For instance, a few days after the Amoco Cadiz oil spill, no significant impact of hydrocarbons on nematode assemblages was detected (Renaud-Mornant et al. 1981). However, after several months, a significant decline in nematode diversity was both revealed and related to a change in the structure of the community: pristine species associated with sandy sediments were clearly replaced by species which were typical of muddy sediments. Little data on the impact of hydrocarbon contamination on nematode trophic structure or trophic diversity are available, although significant alterations have rarely been found, probably due to the limited influence of oil pollution on nematode trophic structure (Danovaro et al., 1995, 2009; Schratzberger et al., 2003). Nevertheless, an alteration of single trophic guilds may be recorded. For instance, Danovaro et al. (1995) reported a decrease of group 1B after oil contamination. This group is likely to be directly affected by oil toxicity as its members ingest tar particles and oil emulsion during feeding (Danovaro, 2000). Surprisingly, group 2B was found to be dominant, or even increasing in abundance, after the oil spill (Danovaro, 2000; Schratzberger et al., 2003). Moreover, the importance of group 2A may rise as a result of the increased microphytobenthic biomass (Carman et al., 1995; Danovaro, 2000). Species - or genus - specific responses were reported by several authors. Danovaro et al. (1995) found that genera such as *Chromaspirina*, *Hypodontolaimus*, *Oncholaimellus*, *Paracanthonchus*, *Setosabatieria* and *Xyala* disappeared immediately after the Agip Abruzzo oil spill, although they recovered rapidly, thus appearing to be opportunists. In contrast, genera such as *Daptonema* and *Viscosia* appeared to be more tolerant to hydrocarbons. In their microcosm experiment, Mahmoudi et al. (2005) found that *Chaetonema* sp. was highly sensitive to diesel contamination; *Pomponema* sp. and *Oncholaimus campylocercoides*, meanwhile, were diesel-sensitive, while *Hypodontolaimus colesi*, *D. trabeculosum* and *D. fallax* were opportunistic and *Marylynnia stekhoveni* diesel-resistant. However, two of these species, *M. stekhoveni* and *O. campylocercoides*, may present with variable levels of sensitivity (see Beyrem et al., 2010), highlighting the fact that the impact of hydrocarbon on marine assemblages requires further investigation. As for *O. campylocercoides*, it is worth noting that Boufahja et al. (2011a) have recently demonstrated that the biometry, life cycle and fecundity of this species may be useful indices for the biomonitoring of hydrocarbon pollution in marine ecosystems.

4.4 Trace elements

The term 'heavy metal' is widely utilized but inadequately described in the scientific literature (see Duffus, 2002). In this paper, the term 'trace elements' will be used instead of 'heavy metals'. 'Trace elements' is a collective term, which refers to any metallic element that has a relatively high density. While most trace elements are biologically essential at very low concentrations, they become toxic to marine organisms above a specific threshold (Kennish, 1992). Although trace elements can be introduced into the environment by natural causes, the major input thereof are anthropogenic in origin, e.g. mining sites, foundries and smelters, the purification of metals, combustion by-products, traffic and coal, natural gas, paper, and chloro-alkali activities. One of the chief problems associated with trace elements is their persistence; unlike organic pollutants they do not decay, thus proving that they have a high potential for bioaccumulation and biomagnification.

Over the last four decades, many studies, conducted in different environmental settings, have focused on the response of benthic foraminifera to trace element pollution (e.g. Alve, 1991b, 1995; Armynot du Châtelet et al., 2004; Coccioni, 2000; Coccioni et al., 2005, 2009; Frontalini & Coccioni, 2008, 2011; Frontalini et al., 2009, 2010; Yanko et al., 1998, 1999) (Fig. 2d). These studies have revealed that this kind of pollution, which may cause pathological processes in the foraminiferal cell, plays an important role in the development of abnormal (teratological) tests. It may also lead to: changes in foraminiferal density and diversity; alterations in assemblage composition; size variation; and structural modification, including in megalospheric and dwarf specimens (for a review, Alve, 1995; Frontalini & Coccioni, 2011; Yanko et al. 1994). It has also been suggested that the presence of morphological abnormalities in benthic foraminiferal tests could be a powerful *in situ* bio-indicator of trace element pollution. Accordingly, Coccioni et al. (2005) developed the FAI to index and compare the levels of morphological abnormality occurring at different sites. More recently, and on the basis of bibliographic analysis, Frontalini & Coccioni (2011) were able to synthetize the inferred sensitive or tolerant response of many foraminiferal species and genera to pollution. Although most of these inferences are not yet supported by the results of laboratory experimentation, a high degree of tolerance to trace elements has been inferred for several taxa, including *Ammonia*, *Cribroelphidium*, *Haynesina*, *Brizalina* and *Bolivina*, as also reported by other researchers (Armynot du Châtelet & Debenay, 2010; Armynot du Châtelet et al., 2004; Carnahan et al., 2008, 2009; Yanko et al., 1999). In recent years, several studies have focused on the response of particular benthic foraminiferal species to selected trace elements in controlled laboratory conditions (e.g. Gustafson et al., 2000; Le Cadre & Debenay, 2006; Nigam et al., 2009; Saraswat et al., 2004). Laboratory culture experiments, through which the benthic foraminiferal response to various elements and concentrations of pollutants can be observed over time, represent the most effective and direct method with which to assess the effect of a single parameter on benthic foraminiferal assemblages. In fact, these experiments provide continuous and accurate observations on the benthic foraminiferal response under controlled conditions, whereby a single parameter can be altered, keeping the rest constant. In this way, the benthic foraminiferal response to specific parameters can be directly characterized. This leads to benthic foraminiferal culture studies wherein foraminifers are subjected to specific pollutants and their particular response thereto is thus documented. In particular, Gustafson et al. (2000) reported a decrease in benthic foraminiferal density when exposed to Tri-n Butyltin. Meanwhile, in a monospecific experiment involving the near-shore benthic foraminiferal species *Rosalina leei,* which was subjected to different concentrations of Hg, Saraswat et al. (2004) documented that the growth rate, as well as the maximum size, decreased considerably in the specimens subjected to gradually higher concentrations of mercury. Test abnormalities also developed at the same time. Moreover, the specimens were subjected to progressively increasing concentrations of Hg to see the further effects thereof: although their growth ceased, the specimens were still living at concentrations as high as 260 ng/l. The effect of graded concentrations of Cu on *A. beccarii* and *A. tepida* has been analyzed by Le Cadre & Debenay (2006) at the morphological level. The two species were sensitive to low, but survived high, concentrations of this trace element. Increasing such concentrations leads to greater delay before the production of new chambers, thus lengthening the period of time before reproduction and reducing the number of juveniles. The proportion of abnormal tests also rose. Moreover, cellular ultrastructure modifications of abnormal specimens exposed sub-lethal contaminations, while thickening of the organic lining, the proliferation of fibrillar vesicles, increases in the number and volume of lipidic vesicles, the disruption of the plasma membrane, increased numbers of residual bodies, and the detection of sulfur within the cells were also found. It has been suggested that the latter is the result of a detoxification mechanism involving the production of a metallothionein-like protein. More recently, Nigam et al. (2009) documented the response of the benthic foraminifera *R. leei* to the sudden or gradual addition of Hg into the media. They reported that when Hg was added suddenly, the specimens did not show any change in morphology during the initial 40 days. However, later on, test deformities developed at higher concentrations, or complete mortality occurred within 20 days at concentrations as high as 300 ng/l. Additional changes were documented in the rate of reproduction, the number of juveniles produced and the survival rate of the juveniles, with growth found to be inversely proportional to mercury concentration. When Hg concentration was increased gradually, irregularities in the newly added chambers were noticed only in the specimens subjected to very high levels of mercury, while growth was found to be inversely proportional to the concentration of this trace element.

Nematodes appear to be good biological indicators of trace element contamination, with it being documented that they may be even more sensitive than other meiofaunal taxa, such as copepods (Somerfield et al., 1994). Several field investigations and laboratory experiments on the effects of trace elements within nematode assemblages have been carried out and reported on the literature (e.g. Austen & McEvoy, 1997; Austen & Somerfield, 1997; Beyrem et al., in press; Boufahja et al., 2011b; Derycke et al., 2007; Guo et al., 2001; Gyedu-Ababio et al., 1999; Hedfi et al., 2007, 2008; Hermi et al., 2009; Mahmoudi et al., 2002, 2007; Millward & Grant, 1995; Somerfield et al., 1994) (Fig. 2d). Nematode abundance may be altered in different ways by different trace elements (Austen & McEvoy, 1997; Hedfi et al., 2007, 2008; Heip et al., 1984; Hermi et al., 2009; Boufahja et al., 2011b; Mahmoudi et al., 2002). In contrast, nematode diversity is very sensitive to these metals and significantly decreases after exposure (Austen & Somerfield, 1997; Boufahja et al., 2011b; Hedfi et al., 2007, 2008; Mahmoudi et al., 2002; Millward & Grant, 1995; Somerfield et al., 1994; Tietjen, 1980). The chemical form, as well as the type of trace element, is important in determining the toxicity effect on nematode assemblages (see Coull & Chandler, 1992 for review). The uptake of metals (e.g. Cu and Zn) by nematodes primarily occurs through cuticular mucous secretions, and may be very different in different species, even of the same genus (Howell, 1982, 1983). However, laboratory studies have shown that the effect of trace elements depends not only on the nature of the element, but also on some environmental factors e.g. temperature, salinity and trophic availability (Coull & Chandler, 1992). Nematode assemblage responses to different types of contaminants have been reported by several authors. Somerfield et al. (1994) and Austen & Somerfield (1997) documented that some species, such as *Ptycholaimellus ponticus*, *S. pulchra*, *Molgolaimus demani* and *Axonolaimus paraspinosus*, can tolerate a wide range of trace elements and may thus have evolved some tolerance adaptations. *Terschellingia* species survived well in all the microcosm treatments including those containing the highest metal doses (Austen & Somerfield, 1997). *Tripyloides gracilis* increased its presence in the most affected sediments (Austen & Somerfield, 1997; Tietjen, 1980). In accordance with the work of Somerfield et al. (1994), toxicity tests on the entire nematode assemblage from the severely contaminated estuary of Restronguent Creek (UK) (Millward & Grant, 1995) revealed that it was more resistant to Cu due to an enhanced Cu tolerance in some dominant species (e.g. *T. marinus*, *A. spinosus, D. setosum*, *Eleutherolaimus* sp., *Theristus acer*). Millward & Grant (1995) demonstrated for the first time that pollution-induced community tolerance (PICT) may be used as a tool to evaluate the biological impact of a chronic pollutant on the marine benthic system. Interesting results are also

Trace Elements	**Intolerant/sensitive**	**Opportunistic**	**Resistant**
Ni	*Leptonemella aphanothecae*	*Daptonema normandicum*, *Neochromadora trichophora*, *Odontophora armata*	*Oncholaimus campylocercoides*, *Bathylaimus capacosus*
Cu	*Microlaimus affinis*, *Monoposthia mirabilis*	-	-
Hg	*Araeolaimus bioculatus*	*Marylynnia stekhoveni*	*Prochromadorella neapolitana*
Cr	*Leptonemella aphanothecae*	*Daptonema normandicum*, *Sabatieria longisetosa*	*Bathylaimus capacosus*, *Bathylaimus tenuicaudatus*
Pb	*Calomicrolaimus honestus*	*Oncholaimus campylocercoides*	-
Zn	*Hypodontolaimus colesi*	*Xyala* sp., *Viscosia franzii*	*Oncholaimus campylocercoides*
Co	*Oncholaimellus mediterraneus*, *Oncholaimus campylocercoides*, *Neochromadora trichophora*	*Spirinia gerlachi*, *Viscosia franzii*, *Promonhystera* sp.	*Marylynnia stekhoveni*
Co/Zn	-	*Viscosia franzii*, *Sabatieria pulchra*	*Marylynnia stekhoveni*

Table 1. Some nematode sensitive/tolerant species proposed as possible preventive indicators of a contaminated sea (Beyrem et al., in press; Boufahja et al., 2011b; Hedfi et al., 2007, 2008; Hermi et al., 2009; Mahmoudi et al., 2007)

reported in some field surveys in the Mediterranean Sea: Mahmoudi et al. (2002) found significant, negative correlations between Cu, Pb and Zn and the biomass and diversity of nematodes, even though the trace element concentrations in the sediments of the Bou Ghrara Lagoon (Tunisia) were lower than envisaged in international norms. Similarly, Semprucci et al. (2010b) found an MI decrease in relation to a peak of Pb in concentration values between the effect-range low (ERL) and the median (ERM) criteria proposed by the NOAA. Nematodes from sediments from the Swartkops estuary in Port Elizabeth, South Africa were investigated by Gyedu-Ababio et al. (1999); the relationship between density, genera, community structure and environmental parameters, including the sediment concentrations of seven heavy metals (Mn, Ti, Cr, Pb, Fe, Sn and Zn), were analyzed. A combination of H' and the MI proved to be very useful for assessing polluted or stressed sites. The nematode community structure at polluted and low/no pollution sites, respectively, differed significantly. *Monhystera* and *Theristus* were found to be colonizers, and these genera were thus regarded as indicators of polluted sediments. Extensive microcosm experiments carried out along the Tunisian coastal zone have tested the nematode assemblage in respect of different trace elements, thus highlighting a set of nematode sensitive/tolerant species (Table 1). The application of the microcosm bioassay approach in these studies has enabled a clear range of sensitivity of nematode assemblages to several trace element doses to be depicted. In a laboratory study on the individual and combined effects of Co and Zn, Beyrem et al. (in press) inferred that there was an antagonistic interaction between these metals: the simultaneous presence of Co and Zn seems to have a lesser impact on nematode species' composition. The clear species–specific responses to the different types of contaminants found in these studies support the possible use of these taxa in the effective biomonitoring of trace elements in coastal marine ecosystems.

4.5 Other types of anthropogenic disturbance

Although the impact induced by organic matter and trace element pollution is a major concern, marine and transitional environments can actually also be influenced by other kinds of anthropogenic disturbance. This could be the discharge of radioactive waste, thermal pollution or physical disturbance (e.g. dredging disposal), which may lead to partial or complete defaunation. Physical disturbance, i.e. processes that lead to the disruption of sediments (see Boyd et al., 2000 and references therein), is a key factor in controlling the spatial and temporal composition of benthic communities. Anthropogenic activities, including coastal development, dredged material disposal and bottom trawling, may cause widespread physical disturbance of the seabed and changes in sedimentation patterns in shelf seas.

At the present time, only a very few studies have addressed the effect of these disturbances on benthic foraminiferal assemblages (e.g. Arieli et al., 2011; Bartlett, 1972; Hechtel et al., 1970; Reish, 1983); (Fig. 2e). A decrease in foraminiferal diversity and a rapid increase of density were documented in response to heated effluent from a power plant in Long Island Sound (northeastern US) (Hechtel et al., 1970). Although it is difficult to separate the effect of thermal pollution from other kinds of stresses, Bartlett (1972) suggested that there was impoverished diversity and lower density in areas along the Atlantic Provinces (eastern Canada). Morphological abnormalities and low density were among the effects on benthic foraminifera in the eurythermal environment of Guayanilla Bay (Puerto Rico).

The most comprehensive investigation of the impact of thermal pollution from a power plant on benthic foraminifera has been carried out in Hadera (Israel) (Arieli et al., 2011). This *in situ* monthly monitoring has revealed that thermal pollution has a detrimental effect on benthic foraminiferal assemblages. In particular, decreased density and diversity values were documented as the sea surface temperature (SST) increased. It was also suggested that 30°C is the critical threshold above which foraminifera' growth and reproduction are severely retarded, and some species are better adapted to tolerating a high SST than others.

The effects of dredged material on the structure of nematode assemblages have been investigated by several authors (e.g. Boyd et al., 2000; Schratzberger et al., 2000, 2006; Somerfield et al., 1995) (Fig. 2e). Boyd et al. (2000) reported a decrease in all of the diversity indices, but not in the abundance values. In accordance with the experiments carried out by Schratzberger et al. (2000, 2009), the most remarkable effect of dredging disposal on the structure of nematode assemblages was the proliferation within the disposal sites of the non-selective deposit feeders *S. pulchra* group (both *breviseta* and *punctata*) and *D. tenuispiculum*. Somerfield et al. (1995) also found *D. tenuispiculum* and *S. punctata* (part of the *pulchra* group) to be numerically abundant in the same dredged material disposal site of Liverpool Bay. As a consequence, the authors suggested that these species were indicators of dredging disturbance. Schratzberger & Jennings (2002) documented the impact of trawling on nematodes. Their data revealed that trawling had a significant impact on the composition of nematode assemblages: diversity and species' richness were significantly lower at high levels of trawling disturbance than at low or medium levels thereof. Recently, Schratzberger et al. (2009) have reviewed the effects of several types of physical disturbance on nematodes. They noted that epigrowth feeding genera such as *Spirinia* and *Desmodora* were highly susceptible to seabed disturbance, thus leading to genus diversity reduction at the most affected stations. A decreased trophic diversity at these stations was primarily due to the increased dominance of non-selective deposit feeders. In contrast, a decrease in genus diversity as a result of bottom trawling did not lead to any changes in trophic diversity. *Sabatieria*, meanwhile, proliferated at high levels of anthropogenic disturbance as a result of coastal development, dredged material disposal and bottom trawling.

5. Conclusions

According to the European Marine Strategy Framework Directive (2008/56/EC), seafloor integrity should be at a level ensuring the safeguarding of the structure and function of ecosystems. Consequently, monitoring the quality of the environment appears to be essential for devising effective protection strategies and appropriate forms of management of marine systems. In particular, the Water Framework Directive (WFD, Directive 2000/60/EC) highlights the importance of biological descriptors when it comes to evaluating and monitoring environmental conditions. Among these, benthic foraminifera and nematodes are highly suitable and sensitive biological organisms through which our comprehension of marine and transitional marine environments can be further explored. In particular, benthic foraminifera have been demonstrated to be particularly sensitive microorganisms and they have been successfully utilized for their value as bioindicators of environmental change in a wide range of marine environments. Since the complex interplay of different biological, chemical, ecological and physical element, and the synergic and antagonistic relationships among them operating at different times and scale the identification of foraminiferal response to specific environmental factor(s) is sometimes hampered. The complex behavior is reflected in shifting patterns of parameters like density, diversity, assemblages composition and percentages of abnormality that are far to be completely understood. The covariance among the nature of the substrate, and the oxygen and organic matter contents within it, which affect the benthic foraminiferal density, clearly, represents a case. The lower foraminiferal diversification of transitional marine environments (i.e., lagoon, coastal lake, salt marsh), when compared to more open water, cannot be readily used as pollution indicator if not associate with similar environments. In the same environmental settings, the percentages of foraminiferal abnormalities would be used with care when additional disturbance factors (i.e., rapid salinity changes) are involved. Moreover, the comparison of different studies is further complicated by the adoption of different techniques and methods. In order to compare these studies, the same set of techniques must be used from the initial sampling to the final treatment of data at least for the same environmental setting. This can only be guaranteed if there is an agreement among scientists and a flexible protocol(s) is developed. Although a definitive scientific agreement has not yet been reached, the development of a protocol represents a milestone for foraminiferal application within governmental and international programs which regulate environmental surveys on marine and transitional marine environments. Notwithstanding nematodes are particularly suitable as bio-indicators in all the types of environments (freshwater, marine and terrestrial ones), a general limitation of ecological investigations of nematodes is related to their difficult taxonomic identification. In this respect, a new and important challenge for the future is the implementation of molecular technique applications, which could be an important tool for making nematode identification easier and faster. The taxonomic approach is certainly the best and more sensitive way for the evaluation of the nematode species response to the pollution effects. However, even if the functional roles of nematodes may be highly species-specific, the application of the taxonomic sufficiency (at genus level) may overall give excellent results. In particular, a multivariate analysis of the assemblage structure and the detection of specific bio-indicators give the best results in all the types of pollution discussed. Also taxonomic diversity may be a good tool for detecting the responses of nematodes, especially to sewage, aquaculture and trace elements pollutions. However, the results from the diversity index should be treated with caution because of the influence of some natural environmental parameters (e.g. sediment grain size). At the present time, the functional trait approach (e.g. trophic groups, life history strategies) seems to be no more powerful than the traditional taxonomic methods, but it can provide additional ecological information on the ecosystem functioning. In particular, Maturity Index and colonizer-persister class percentages are successfully applied in literature on the effects of aquaculture and trace elements. Furthermore, the c-p percentage has been recently suggested as the best descriptor of the quality ecological status in marine ecosystems, together with the taxonomic approach. Although significant variations of single trophic groups are usually observed in several types of pollutions, the Index of Trophic Diversity shows controversial responses, especially in aquaculture and hydrocarbon pollutions, and in physical disturbance. This might suggest a non-selective impact of these stressors on the global trophic structure of the nematode assemblage or it may be related to the limits of the Wieser classification, which does not perfectly reflect the trophic position of all the nematode species. A good alternative to feeding types appears to be based on the life history of nematodes or on the integration of both of these functional traits. However, the ecological and practical advantages of using Foraminifera and Nematoda in benthic ecological studies are basic elements for choosing these taxa as descriptor groups in the assessment of the quality status of marine sediments.

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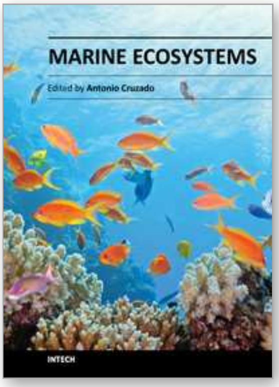

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Marine ecosystems, a very wide topic, includes many different processes, groups of organisms and geographical peculiarities. The objective of this book is to present various topics of great importance for understanding the marine ecosystems, what they are, how they work and how we can model them in order to forecast their behaviour under changing conditions. They have been thoroughly reviewed and accepted for publication. The chapters cover aspects such as: Threats to ultraoligotrophic marine ecosystems (Ch. 1); Modelling the pelagic ecosystem dynamics: the NW Mediterranean (Ch. 2); The marine ecosystem of the Sub-antarctic, Prince Edward Islands (Ch. 3); Meiofauna as a tool for marine ecosystem biomonitoring (Ch. 4); Chemical interactions in Antarctic marine benthic ecosystems (Ch. 5); An Interdisciplinary Approach on Erosion Mitigation for Coral Reef Protection- A Case Study from the Eastern Caribbean (Ch. 6); A revisit to the evolution and ecophysiology of the Labyrinthulomycetes (Ch. 7); Seabed mapping and marine spatial planning: a case-study from a Swedish marine protected area (Ch. 8); Management strategies to limit the impact of bottom trawling on VMEs in the High Seas of the SW Atlantic (Ch. 9); Hydrocarbon contamination and the swimming behavior of the estuarine copepod Eurytemora affinis (Ch. 10), and Interactions between marine ecosystems and tourism on the Adriatic and Mediterranean (Ch. 11).

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