


# The amphipod fauna assemblage along the Mediterranean Israeli coast, a spatiotemporal overview


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## Abstract

In recent decades, intense human activities and increased maritime transportation triggered heavy pressure on the Israeli coastal ecosystem and, meanwhile, the eastern Mediterranean experienced the highest temperature values. With the assumption that both kinds of anthropogenic drivers could alter the benthic taxocenosis, a monitoring survey covering a period ranging from 2010 to 2017 was addressed to assess changes in the macrozoobenthos on spatial and temporal scales. The present study focused on the crustacean amphipod fauna collected on the soft bottom littoral area. Twenty-five species and taxa (genus level) were detected under the temporal survey in the same stations, located along a north-south axis, from the northernmost Haifa Bay to the southern area. The dataset showed a temporally stable assemblage of the most common species. Seven species showed the greatest abundance and a constant presence: the Levantine endemic *Cheiriphotis mediterranea*; the Mediterranean endemic *Megaluropus massiliensis*; the NE-Atlantic–Mediterranean *Bathyporeia guilliamsoniana* and *Perioculodes longimanus*; and the widely distributed *Photis longicaudata* and *Urothoe grimaldii*. In addition, an alien species was recorded, the circumtropical aorid *Grandidierella bonnieroides* which resulted naturalized. Differently, the spatial dataset discriminated two geographical regions, the northernmost Haifa Bay and the remaining Southern Israeli Coast, based on the different substratum granulometry. On the whole, the Israeli assemblage showed a low diversity composed of dominant species strictly associated with the features of the sediment and characterised by local abundance fluctuations. It is a pattern that provides baseline knowledge for successive monitoring years and contributes to implementing the knowledge of the ecological traits of amphipod fauna.

**Key words** Amphipoda, Israel, Levantine Sea, Mediterranean Biodiversity.

## Introduction

The Mediterranean Basin can be classified as a highly biodiverse sea where numerous marine species, particularly cryptic, rare and recently introduced species, remain understudied, rendering the

biodiversity assessment challenging (Hochkirch et al., 2021; Lo Brutto et al., 2024). Such limits could be overcome by monitoring marine ecosystems and their taxocenosis profile. This is particularly relevant in the context of global warming, where a comprehensive understanding of how changes in species assemblages are driven by anthropogenic pressures is essential for effective management and mitigation of the impacts of climate change (Coll et al., 2010; Gerovasileiou et al., 2020). Further, although significant progress has been made in monitoring bio-invasions, improving knowledge of specific areas and specific taxa is an urgent task. For these reasons, it is essential to address the establishment of regional taxonomic assemblages, taxocenosis, particularly in the view of a biogeographical approach which can help to detect global and regional changes.

Within the Mediterranean, the Levantine basin is distinguished by a negative freshwater balance, which gives rise to specific abiotic conditions characterised by warm water and high salinity. The presence of the Suez Canal and the effects of climate change facilitate the entry and establishment of a significant number of non-indigenous species (Galil, 2023). Besides, the Levantine Basin has been subjected to increasing anthropogenic pressures in recent decades (Lo Brutto et al., 2019; Lubinevsky et al. 2019). Locally, the prevalence of intense human activities and augmented maritime traffic exerted considerable pressure on the coastal ecosystem (Schneider et al., 2010; Marriner et al., 2014; Galil, 2023; Polinov, 2023) that is characterised by a sandy shallow shelf due to the Nile delta deposition and a continental clay transport from streams.

The present study focused on the Amphipoda order, one of the most abundant marine taxa that plays a key role in aquatic ecosystems and significantly influences trophic webs (Ritter and Bourne, 2024). In recent years, there has been a notable interest in these peracarid crustaceans as indicators of the Mediterranean biodiversity (Servello et al., 2019; Navarro-Barranco et al., 2020; Bakalem et al., 2024). It is therefore of great importance to update knowledge to maintain an accurate overview of the basin and regional patterns (Iaciofano & Lo Brutto 2017; Martínez-Laiz et al., 2019; Ragkousis et al., 2023).

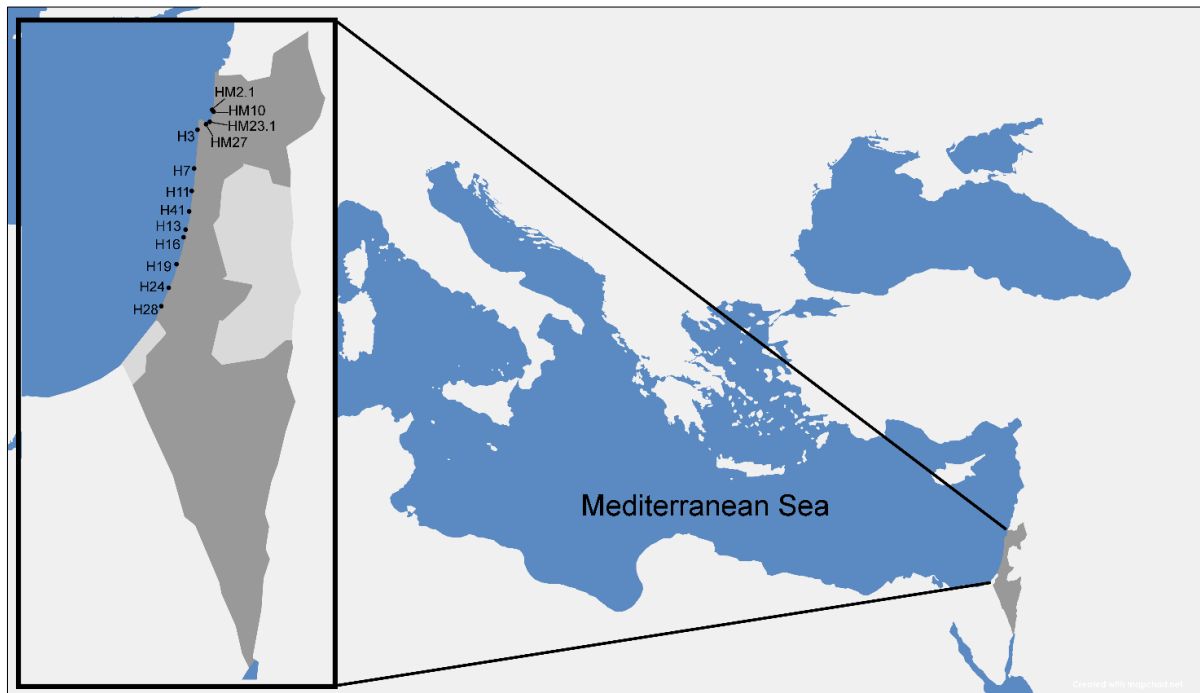
A monitoring taxonomic survey covering a period ranging from 2010 to 2017 was addressed to identify macrozoobenthos including the crustacean amphipods along the Israeli coast. It was a period that preceded a government decision (December 2018) to expand the National Monitoring Program in line with the United Nations Environmental Program's (UNEP) Integrated Monitoring and Assessment Program (IMAP) (Herut & IOLR Scientists, 2022). The study aimed to assess the Israeli taxocenosis to obtain data on a timeframe useful for implementing warning models focused on detecting changes in the Mediterranean fauna and to contribute to understanding the ecology and distribution of amphipod species.

## Materials and methods

The present dataset is given for thirteen National Monitoring stations (Table 1) located along the Mediterranean coast of Israel. The thirteen sites were sampled once a year, from 2010 to 2016, while in 2017 the sampling was limited to six stations (i.e. only the sites listed at the numbers 1, 4, 5, 7, 10, and 13 in Table 1). Thus, the sampling effort corresponded to 97 site replicates. Three replicate sediment samples per site were taken, counting a total of 291 (97×3) sample replicates. The geographical position of the sites followed a north-south axis distribution and a criterion that included the most anthropised area (i.e. the localities in Haifa Bay, HB) and the localities in the remaining Southern Israeli Coast (SIC) (Figure 1). Haifa Bay, identified by the HM station code, is a locally impacted region by heavy metals pollution, maritime traffic development, rapid urbanization and industrialization. The sites were characterised by mobile substrate; the SIC zone was characterised by a homogeneous fine sandy sediment with a granulometry mode around the 125–250- $\mu\text{m}$  size; whereas the HB zone showed a heterogeneous sediment, from a fraction of gravel (> 1000  $\mu\text{m}$ ), the latter probably originating from rocky debris of the nearby sandstone (calcareous) ridges, to coarse, smaller-sized sand and the largest silt–clay fraction (< 63  $\mu\text{m}$ ), probably contributed by the Na'aman (close to HM2.1 station) and Qishon (close to HM27 station) rivers outflow (Figure 2).

Samples were collected for each site with a 32 × 35 cm Van-Veen grab (KAHLSICO, model WA265/SS214), volume 20 L, which penetrated the substrate by up to 20 cm (see Lubinevsky et al. 2019 for other details). The fauna was segregated on board by sieving and preserved temporarily in 4%

formaldehyde. Thereafter, the sediment samples were preserved in 70% alcohol and sieved onto a 250 mm mesh within a few days from collection. The samples were sorted, and amphipod fauna was identified to species level, using the most recent literature to highlight both new records and non-indigenous species; in case of uncertainty, the identification stopped up the genus rank.



**Figure 1.** Map of the geographical position of the sites following a north-south axis and a criterion that included the most anthropised area (i.e. the localities in Haifa Bay, HB, in the northern Israeli coast) and the localities in the remaining Southern Israeli Coast (SIC).

It was not feasible to identify all individuals to the species level, given that this work is part of a much larger investigation encompassing over 53,473 individuals (data not published yet) all of which were observed and identified almost to the genus level. The identification to species level was conducted when possible. Consequently, when species identification was unfeasible due to various impediments, such as juvenile stage, female sex, damaged specimens, or particularly high abundance, it was performed at the genus level. In the case of a genus that was monospecific, i.e. represented by one species in the area, all the individuals were ascribed to that species. Regarding the genera that were polyspecific, i.e. represented here by more than one species, the taxon considered for statistical analyses was the genus. The identified species belonging to genera represented by more than one species are reported in the Results section as a qualitative assessment; they were identified with certainty, but not for all individuals of the genus to which they belonged.

The World Amphipoda Database (Horton et al., 2023) was used to update nomenclature and data from the literature.

### Statistical analyses

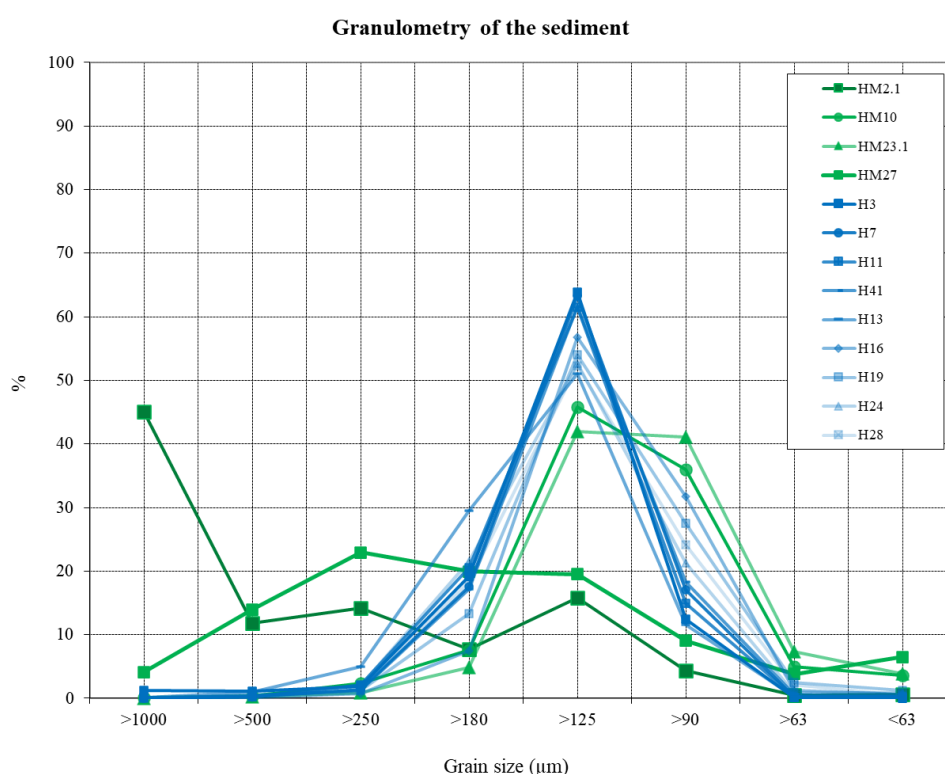
The taxocenosis profile was first defined by the list of taxa and their abundance per year and site. All the sampled individuals were included in the abundance calculations. Temporal variation in the richness of taxa and their total abundance was assessed for each station.

Following previous studies (see Lubinevsky et al. 2019), the sites were grouped into two zones within the Israeli maritime area: the northernmost Haifa Bay (HB) and the Southern Israeli Coast (SIC). A nonmetric multidimensional scaling (nMDS) performed on the taxon matrix using Bray–Curtis distances was used to detect the degree of heterogeneity between these two zones (spatial scale) and among all the years (temporal scale). The Principal Component Analysis (PCA) was performed to detect similarity among the sites, based on the amphipod assemblage and the granulometry. The statistical

analyses were based on the seven most common species plus *Ampelisca* sp. and performed using the PRIMER 6 software package with the PERMANOVA extension (Clarke and Gorley, 2006).

**Table 1.** Information of the sampled stations/sites. The stations were subdivided into two zones within the Israeli maritime area, the Haifa Bay (HB) and the Southern Israeli Coast (SIC).

	Zone	Stations/Sites code	Depth (m)	Lat (N)	Long (E)	Sediment features and corresponding site code in Lubinevsky et al. 2019
1	Haifa Bay	HM2.1	7.8	32.91797	35.07223	Gravel, coarse and smaller-sized sand - A1
2	Haifa Bay	HM10	8.94	32.90101	35.07257	Fine sand and silt – A2
3	Haifa Bay	HM23.1	9.78	32.83244	35.04214	Fine sand and silt – A3
4	Haifa Bay	HM27	10.98	32.82105	35.01958	Coarse and fine sand – A4
5	Southern Israeli Coast	H3	10.16	32.78954	34.94745	Fine sand – A5
6	Southern Israeli Coast	H7	12.73	32.54347	34.8925	Fine sand – A6
7	Southern Israeli Coast	H11	12.81	32.40034	34.85592	Fine sand – A7
8	Southern Israeli Coast	H41	10.23	32.27189	34.82359	Fine sand – A8
9	Southern Israeli Coast	H13	10.26	32.15887	34.78664	Fine sand – A9
10	Southern Israeli Coast	H16	12.82	32.10801	34.76567	Fine sand – A10
11	Southern Israeli Coast	H19	11.7	31.94531	34.69834	Fine sand – A11
12	Southern Israeli Coast	H24	10.86	31.80319	34.62448	Fine sand – A12
13	Southern Israeli Coast	H28	8.41	31.68809	34.55698	Fine sand – A13



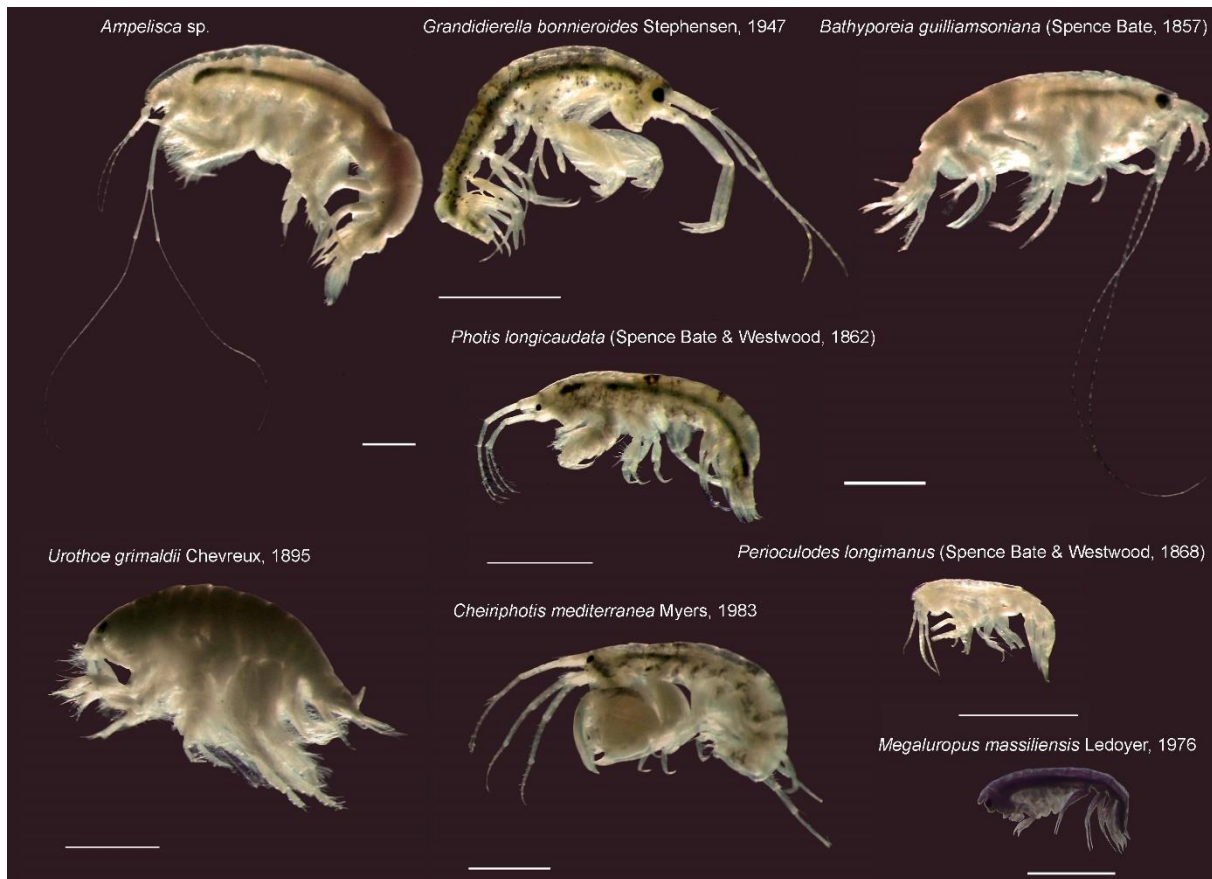
**Figure 2.** Granulometric profile of the shallow sediment samples. The graph shows the percentage of grain size for each station. The green lines indicate the sites in Haifa Bay and the blue lines stations on the Southern Israeli Coast. The site HM2.1 was characterised by a high portion of gravel (>1000  $\mu\text{m}$ ) and coarse sand (500-180  $\mu\text{m}$ ); the site HM27 was characterised by medium sand (500-250  $\mu\text{m}$ ) fine sand (250-125  $\mu\text{m}$ ), very fine sand (125-63  $\mu\text{m}$ ) and silt clay (<63  $\mu\text{m}$ ). The remaining stations were characterised by fine sandy substrate (with mode around 125  $\mu\text{m}$ ).

## Results

A total of 11,838 individuals belonging to 25 species or taxa identified to genus level were counted during the present study. Table 2 shows a detailed list and the number of individuals collected each year.

Some genera were composed of different species, e.g. *Ampelisca* spp., but they were statistically treated as one taxon. Despite the impediments in identifying all the individuals to the species rank within these genera, the samples where the identification could reach the species level are here reported. The genus *Ampelisca* included *A. brevicornis* (A. Costa, 1853) (detected in HB and SIC zones; in 2010 and 2015), *A. dalmatina* Karaman, 1975 (SIC zone, in 2015 and 2016), *A. pseudospinimana* Bellan-Santini & Kaim-Malka, 1977 (SIC zone, in 2015 and 2016), *A. sarsi* Chevreux, 1888 (HB and SIC zones, in 2015, 2016). The *Leptocheirus* individuals (detected in 2013 in HB zone, at HM2.1 site, with 5 individuals; HM27 site, 2 individuals) were never identified at the species level. The genus *Leucothoe* included *L. incisa* Robertson, 1892 (in HB and SIC zones, in 2015 and 2016) and *L. occulta* Krapp-Schickel, 1975 (in HB and SIC zones; in 2010 and 2015). The individuals belonging to the genus *Maera* (detected in HB zone) included specimens ascribable to *Maera* cf. *grossimana* (Montagu, 1808) (identified in HM23.1 and HM27 sites).

The dataset showed an almost stable assemblage of the most common species belonging to the sandy and muddy shallow biocenosis, with sporadic records of species usually associated with a substrate of macroalgae or seagrasses.



**Figure 3.** The soft bottom amphipod taxocenosis from Israel. Here, the most abundant species – *Bathyporeia guilliamsoniana*, *Cheiriphotis mediterranea*, *Grandidierella bonnieroides*, *Megaluropus massiliensis*, *Periculodes longimanus*, *Photis longicaudata* and *Urothoe grimaldii* – plus the taxon *Ampelisca* sp. are shown. Scale bar 1 mm.

### Species highlights

The most abundant species were seven – *Bathyporeia guilliamsoniana*, *Cheiriphotis mediterranea*, *Megaluropus massiliensis*, *Periculodes longimanus*, *Photis longicaudata* and *Urothoe grimaldii* – plus one non-indigenous species – *Grandidierella bonnieroides* (Figure 3). The ecology and distribution

**Table 2.** List of the species and the taxa identified to genus rank collected in the eight-year survey. The abundance for each species/taxon per year, the total abundance of amphipods per year, the total abundance of the species/taxon over the eight years and the number of years of occurrence per species/taxon are shown.

Family	Species	2010	2011	2012	2013	2014	2015	2016	2017	Total abundance	Years of presence
Ampeliscidae	<i>Ampelisca</i> spp.	97	216	30	26	54	309	159	59	950	8
Amphilochoidea	<i>Apolochus neapolitanus</i> (Della Valle, 1893)					5				5	1
Ampithoidae	<i>Ampithoe ramondi</i> Audouin, 1826	1				2				3	2
Aoridae	<i>Grandidierella bonnieroides</i> Stephensen, 1947					227	7	1	8	243	4
Atylidae	<i>Nototropis massiliensis</i> (Bellan-Santini, 1975)						1		1	2	2
Bathyporeiidae	<i>Bathyporeia guilliamsoniana</i> (Spence Bate, 1857)	190	260	8	108	56	118	435	208	1383	8
Caprellidae	<i>Pseudolirius kroyeri</i> (Haller, 1879)	1	5				10	4	1	21	5
Corophiidae	<i>Cheiriphotis mediterranea</i> Myers, 1983	129	303	26	14	1805	246	70	26	2619	8
	<i>Apocorophium acutum</i> (Chevreux, 1908)	8	8	35	5	16	4		11	87	7
	<i>Leptocheirus</i> sp.				7					7	1
Dexaminidae	<i>Dexamine spinosa</i> (Montagu, 1813)		1	2	1	5			1	10	5
Ischyroceridae	<i>Erichthonius punctatus</i> (Spence Bate, 1857)	1	14			7			3	25	4
	<i>Siphonoecetes</i> sp.							20		20	1

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**Table 2.**

Leucothoidae	<i>Leucothoe</i> spp.	19	13	7	8	14	54	25	10	150	8
Liljeborgiidae	<i>Liljeborgia</i> sp.	18						3		21	2
Maeridae	<i>Quadrimaera</i> cf. <i>inaequipes</i> (A. Costa in Hope, 1851)								5	5	1
	<i>Maera</i> sp.			3	14	13			2	32	4
Megaluropidae	<i>Megaluropus massiliensis</i> Ledoyer, 1976	143	54	15	43	35	142	36		468	7
Photidae	<i>Photis longicaudata</i> (Spence Bate & Westwood, 1862)	119	16	75	751	6	3	18		988	7
Phoxocephalidae	<i>Metaphoxus</i> sp.	2								2	1
Oedicerotidae	<i>Periocolodes longimanus</i> (Spence Bate & Westwood, 1868)	132	482	107	75	162	1414	550	205	3127	8
	<i>Synchelidium longidigitatum</i> Ruffo, 1947				1					1	1
	<i>Pontocrates arenarius</i> (Spence Bate, 1858)	10	2		12	6	20	9	16	75	7
Stenothoidae	<i>Stenothoe</i> sp.								1	1	1
Urothoidae	<i>Urothoe grimaldii</i> Chevreux, 1895	426	192	113	119	344	235	113	51	1593	8
<b>Total abundance per year</b>		1032	1760	401	480	3510	2459	1534	662		

information of such species is here presented. They represent the *core* of the taxocenosis on which environmental and anthropogenic drivers can significantly act.

Family Aoridae Stebbing, 1899

*Grandidierella bonnieroides* Stephensen, 1947

*Grandidierella bonnieroides* is a cosmopolitan circumtropical species distributed in tropical and temperate regions (Myers 1970, 2009; LeCroy et al., 2009). This aorid is very abundant in the Caribbean and on the tropical west Atlantic coast and to date is recorded in Arabian Gulf, India, South Africa, Indonesia, Mexico, Puerto Rico, Colombia, Cuba, Bahamas, Brazil, and North America (Myers, 1970; Zimmerman et al., 1979; Stoner, 1980; Heard, 1982; Lalana-Rueda and Gosselck, 1986; Virnstein and Curran 1986; Stearns and Dardeau 1990; Stoner and Acevedo 1990; Ortiz and Lalana, 1996, 2010; Ortiz and Lemaitre 1997; LeCroy et al., 2002; Satheeshkumar, 2011; Paz-Rios and Ardisson, 2013; Hindarti et al., 2015; Sweatman et al., 2017; Manokaran et al., 2021; Winfield et al., 2023). This species was originally described in the Netherlands Antilles, Caribbean Sea (Stephensen, 1933). Regarding the Mediterranean basin, it has recently been reported in Israel as a non-indigenous species (Lo Brutto et al., 2016). How this aorid arrived in Mediterranean water has not yet been determined but the most likely transport vectors could be ballast water and ship fouling (Lo Brutto et al., 2016).

In its natural areas of distribution, *G. bonnieroides* is considered a dominant species in estuarine and euryhaline environments (Stearns and Dardeau, 1990). In addition, this aorid has also been established in intertidal mangroves (Satheeshkumar, 2011), seagrass meadows such as *Stryngodium filiforme*, *Halodule wrightii* and *Thalassia testudinum* (Stoner et al., 1980; Virnstein and Curran, 1986; Sweatman et al., 2017), soft bottoms in shallow bays, tide pools (Heard, 1982) and muddy bottoms with sparse vegetation (Oliva-Rivera, 1998). *Grandidierella bonnieroides* can also exploit muddy substrates characterized by low oxygen tension (Day, 1981).

This species is considered a gregarious “detritus-blanket tube-builder” as it lives inside masses of detritus that are utilised for enrolling themselves in a blanket-like semi-permanent tube (Thomas, 1976; Barnard et al., 1991). The tube is agglutinating with silk and sediments particles, micro and macroalgae debris and foraminifera (Barnard et al., 1991; Ortiz and Lalana, 2010). *Grandidierella bonnieroides* is a subsurface interstitial omnivore deposit feeder (Manokaran et al., 2021) and is classified as a microphage that feeds mainly on epiphytic diatoms and debris accumulated on vegetation (Zimmerman et al., 1979). Moreover, this species can change its feeding modes encompassing filter-feeders, grazer, detritivore, and deposit feeders according to food availability (Thomas, 1976; LeCroy et al., 2002; Reis Filho et al., 2018) and promotes nutrient recycling by reintegrating carbon into trophic networks (Reis Filho et al., 2018).

*Grandidierella bonnieroides* is cryptic in colour and behaviour and hides in sediments to reduce its vulnerability to visual predators (Stoner, 1980). In fact, in estuarine environments, this species is an important food resource for benthic and nektonic species, especially fish, crabs and penaeid shrimps (Lalana-Rueda and Gosselck, 1986; Paiva and Da Silva, 1998).

This species is regarded as an opportunistic pollution indicator (Grizzle, 1984; Barnard et al., 1991) and has been employed in ecotoxicological studies to assess sediment toxicity and the presence of chemical compounds in environments (Hindarti et al., 2015; Reis Filho et al., 2018). Furthermore, *G. bonnieroides* is also known to colonize substrates that undergo recurrent defaunation, sometimes forming dense populations (Santos and Simon, 1980).

The resistance characteristics of this species may have contributed to its successful colonization of Haifa Bay (Israel), which is characterized by anthropogenic pollution and eutrophication (Herut & IOLR Scientists, 2022; Lo Brutto et al., 2016). Our analyses revealed the presence of 243 individuals of *G. bonnieroides* distributed over soft bottoms located at depths between 7.8 and 11.7 m. The species was found with high abundance close to the Haifa harbour, in the only site characterized by the presence of coarse sediment (HM2.1 station, in 2014, 227 individuals); subsequently, a few individuals were found in a site within Haifa Bay, characterized by finer sediments (HM27 station, in 2015, 7 individuals). A single individual was sampled along the Israeli coast in 2016 (H19 station), and, finally, the species was again recorded in 2017 at its first settlement site (HM2.1 station, 8 individuals).



Family Bathyporeiidae d'Udekem d'Acoz, 2011  
*Bathyporeia guilliamsoniana* (Spence Bate, 1857)

This species is present in the northeastern Atlantic, North Sea, Marmara Sea and Black Sea. It is recorded in the Mediterranean and European waters: France, Scotland, England, Italy, Egypt, Israel Spain, Belgium, Netherlands, Algeria, Norway, Portugal, Tunisia and Turkey (Sars, 1890-95; Gottlieb, 1960; Atta, 1988; N'Da, 1992; Eleftheriou and Robertson, 1992; Marques and Bellan-Santini, 1993; Bakalem, 1998; d'Udekem d'Acoz and Vader, 2005; Kirkim et al., 2006; Colosio et al., 2007; Pérez-Domingo et al., 2008; Kröncke, 2011; Passarelli et al., 2012; Curatolo et al., 2013; Navarro-Barranco et al., 2013; Bakir and Katağan, 2014; Mülayim et al., 2015a,b; Coates et al., 2016; Belatoui et al., 2017; Mayer et al., 2018; Wijnhoven et al., 2018; Ballesteros et al., 2020).

All the species of the genus *Bathyporeia* are considered burrowers and confined to sandy bottoms (Toulmond, 1964); in fact, *B. guilliamsoniana* is generally associated with shallow water well sorted fine sand sensu Pérès & Picard (1964) and other sand sublittoral biocenosis (Jones, 1950; Diaviacco and Bianchi, 1987; Occhipinti Ambrogi et al., 1988; Robertson et al., 1989; Eleftheriou and Robertson, 1992; Heip and Craeymeersch, 1995; de-la-Ossa-Carretero et al., 2010) and often dominates in the communities of the shallow-water sand habitat (Elmhirst, 1932). In detail, *B. guilliamsoniana* has been found on several types of soft substrates such as fine depositional sands with *Tellina* spp. (Warwick and Davies, 1977), *Spisula subtruncata* sands, sand with shells fragments, intertidal sand (Robertson et al., 1989; De Grave and Casey, 2000; Pérez-Domingo et al., 2008), lagoon sands (Reid, 1941; Diaviacco and Bianchi, 1987), muddy sands and mud (Kirkim et al., 2006). The optimal sediment type for this bathyporeid is sand with median grain size with high percentage of carbonate particles but this species can live on slightly coarser sediments (Toulmond, 1964; Degraer et al., 2006). Moreover, *B. guilliamsoniana* can be observed rarely on *Mytilus galloprovincialis* facies and oyster beds, in photophilic algae communities (Millar, 1961; Mülayim et al., 2015a,b) and in areas subject to anthropogenic impact (Bakalem, 1998; Colosio et al., 2007).

Regarding its bathymetric distribution range this species is generally observed in shallow waters between 0.5 and 20 m (Bossanyi, 1957; Occhipinti-Ambrogi et al., 1988; Kirkim et al., 2006; de-la-Ossa-Carretero et al., 2010; Curatolo et al., 2013; Mülayim et al., 2015a,b; Belatoui et al., 2017) but some specimens have also been sampled at about 75 m depth (d'Udekem d'Acoz, 2004).

*Bathyporeia guilliamsoniana* feeds on detritus (Navarro-Barranco et al., 2013) and eats organic particles adhered to sand grains (Nicolaisen and Kannevorff, 1969), for these reasons can be considered as selective deposit-feeders (Wolff, 1973). This species actively burrows into the sand by the action of the head, used to penetrate the sediment, and use the anterior appendages for generate an intense “ventral groove” current (Watkin, 1939a, b). This amphipod species shows a pronounced sexual dimorphism (d'Udekem d'Acoz, 2004) and is capable of nocturnal vertical movement (d'Udekem d'Acoz and Vader, 2005); particularly, the adult males swim in the water column at night and these vertical displacements are probably regulated by lunar and tidal cycles (Watkin, 1939b; d'Udekem d'Acoz, 2004; d'Udekem d'Acoz and Vader, 2005). Some authors have pointed out the presence of parasites on the body of *B. guilliamsoniana*; this species can be infested both by epizoan ciliates (Wijnhoven et al., 2018) and copepods such as *Sphaeronella paradoxa* Hansen, 1897 (d'Udekem d'Acoz, 2004).

A total of 1383 individuals of *B. guilliamsoniana* exclusively on soft substrate localized between 7.8 and 12.82m depth was collected. The species *Bathyporeia guilliamsoniana* was observed to have high abundances in the central section of the Israeli coastline (H3-H41 stations) and lower abundances in the southernmost areas. In contrast, Haifa Bay exhibited the occasional presence of a few individuals supporting that the species was consistently absent in the stations characterized by high levels of anthropogenic activities.

Family Corophiidae Leach, 1814  
*Cheiriphotis mediterranea* Myers, 1983

This corophiid is endemic to the Levantine basin; it had been reported only in Israel, but recent observations in Turkey extended the northernmost area of distribution (Myers 1983; Sorbe et al. 2002; Bakir and Katağan, 2014; Çinar et al., 2015; Lo Brutto and Iacofano, 2020; Lo Brutto et al., 2022). It is important to point out that though its collection has been limited to the eastern Mediterranean coast, some individuals were also observed in semi-enclosed bays in China (Ren, 2006; Shi et al., 2022), a geographical occurrence that needs to be confirmed. Since its limited range, this amphipod is considered

a relict species that survived the paleogeographic and paleoclimatic events that affected the Mediterranean Sea (Lo Brutto et al., 2022).

*Cheiriphotis mediterranea* is an infralittoral filter-feeder species and was generally observed on fine sand, coarse sand, detritic rubble and sandy-muddy bottoms also in association with the photid *Photis longicaudata* (Sorbe et al., 2002; Bakir and Katağan, 2014; Lo Brutto et al., 2022). Its bathymetric distribution ranges between 3 and 38 m depth, with maximum abundances recorded between 10 and 25 m (Lo Brutto et al., 2022).

High abundances of this species have been observed in the anthropised Haifa Bay and, for this reason, *C. mediterranea* seems to be a good bioindicator of environmental contamination such as pollutants, organic matter, and high carbon levels. The presence of 2619 specimens of *C. mediterranea*, with a significant peak (1805 individuals) in 2014 in Haifa Bay, was distributed at depths between 7.8 and 12.81 m.

#### Family Megalurotidae Thomas & Barnard, 1986

##### *Megaluropus massiliensis* Ledoyer, 1976

*Megaluropus massiliensis* is an endemic Mediterranean species that has also colonized the Black Sea (Sorbe et al., 2002; Karaçuha et al., 2009; Mülayim, 2021; Grinstov, 2022). It has been recorded in Algeria, Corsica, France, Italy, Israel, Greece, Morocco, Spain, Tunisia and Turkey (León and Corrales, 1995; Conradi & López-González, 1999; Sánchez-Jerez et al., 1999; San Vicente and Sorbe, 1999; Sorbe et al., 2002; Scipione et al., 2005; Kirkim et al., 2006; Sezgin and Katağan, 2007; Luís, 2007; Karaçuha et al., 2009; Vázquez-Luis et al., 2009; de-la-Ossa-Carretero et al., 2010, 2012; Lorenti et al., 2011; Bakir, 2012; Lattanzi et al., 2013; Bakir & Katağan, 2014; Guerra-García et al., 2014; Çinar et al., 2015; Mülayim et al., 2015a,b; Moidanou et al., 2017, 2021; Zakhama-Sraieb et al., 2017; Targusi et al., 2019; Navarro-Barranco et al., 2020; Bakalem et al., 2024; Saenz-Arias et al., 2024).

This amphipod is a fossorial filter feeder species (de-la-Ossa-Carretero et al., 2012), and it is also an important component of the hyperbenthic fauna of surface sandy substrates (León & Corrales, 1995; Galparsolo, 1999). Indeed, *M. massiliensis* commonly colonizes fine sand, very fine sand, muddy sand and, less commonly, mud (Conradi and López-González, 1999; de-La-Ossa-Carretero et al., 2010; Targusi et al., 2019; Mülayim, 2021). In some studies, this species has been observed in association with *Spisula subtruncata* sand (Mülayim et al., 2015a) and *Phaseolina* sludge zoocenosis (Sezgin and Katağan, 2007). *Megaluropus massiliensis* has also been observed in relation to shallow-water phanerogam meadows, such as *Posidonia oceanica*, *Cymodocea nodosa*, *Zostera marina* and *Zostera noltii* (Sánchez-Jerez et al., 1999; San Vicente and Sorbe, 1999; Karaçuha et al., 2009; Vázquez-Luis et al., 2009) and with rocky substrates colonized by photophilic macroalgae (Navarro-Barranco et al., 2020). Some studies have shown that this species also inhabits areas characterized by the presence of the caulerpaces *Caulerpa racemosa* and *C. prolifera* (Luís, 2007; Vázquez-Luis et al., 2009; Lorenti et al., 2011; Moidanou et al., 2017, 2021).

This megaluropid is most abundant in shallow coastal waters between 1 and 10 m (León & Corrales, 1995; Mülayim et al., 2015a; Moidanou et al., 2017; Saenz-Arias et al., 2024) but some authors have reported it at greater depths up to about 50 m (Scipione et al., 2005; Conradi and López-González, 1999; Lattanzi et al., 2013).

Ecologically, *M. massiliensis* was considered particularly sensitive to pollution and organic enrichment (de-la-Ossa-Carretero et al., 2012; Çinar et al., 2015). However, Lattanzi et al. (2013) observed that this amphipod is tolerant to the increase of fine sediment in suspension/deposition. *Megaluropus massiliensis* feeds on suspended detritus and planktonic crustaceans (Guerra-García et al., 2014); it is most active at night and can be sampled using light traps (Saenz-Arias et al., 2024), indicating that it probably moves vertically along the water column.

Our analyses revealed the presence of 468 individuals exclusively on soft sandy bottoms, distributed at depths between 7.8 and 12.81 m. *Megaluropus massiliensis* was found particularly numerous within Haifa Bay. This species demonstrated low abundances along the entire southern coast of Israel, with two notable peaks in abundance at sites H13 and H28.

## Family Oedicerotidae Lilljeborg, 1865

*Periocolodes longimanus* (Spence Bate & Westwood, 1868)

This species is widely distributed in the Mediterranean Basin, Black Sea and European waters and is reported in France, Scotland, England, Spain, Italy, Anatolia, Germany, Norway, Algeria, Ireland, Portugal, Spain, Cyprus, Sweden, Turkey, Romania, Greece, Holland, Tunisia, and Israel (Falciai and Spadini, 1985; Buhl-Jensen, 1986; Occhipinti-Ambrogi et al., 1988; Dauvin and Gentil, 1990; Buhl-Jensen and Fosså, 1991; Faasse and Stikvoort, 2002; Blanchet et al., 2005; Kirkim et al., 2006; Sezgin and Katağan, 2007; Nickell et al., 2009; Zakhama-Sraieb et al., 2009, 2017; de-la-Ossa-Carretero et al., 2010; Sezgin et al., 2010; Schückel et al., 2011; Carvalho et al., 2012; Navarro-Barranco et al., 2013; Nikitik and Robinson, 2003; Kudrenko et al., 2016; Misic et al., 2016; Belatoui et al., 2017; Maidanou et al., 2017; Ballesteros et al., 2020; Rousou et al., 2020; Mülayim, 2021; Tănase et al., 2022).

*Periocolodes longimanus* is exclusively associated with soft infralittoral substrates and sand sublittoral biocenosis (Buhl-Jensen and Fosså, 1991, Sezgin and Katağan, 2007; Ballesteros et al., 2020) although it has also been found on deep substrates and bathyal bottom (Cartes et al., 2007). Due to the close association between this species and sandy substrates, *P. longimanus* is considered an exclusive species of the well calibrate fine sands biocenosis sensu Pérès & Picard (1964) (Falciai and Spadini, 1985; Niccolai et al., 1993). The bottom types and sands communities in which this species is observed are intertidal sands (Viéitez and Baz, 1988; Robertson et al., 1989), fine sands (Lourido et al., 2010), coarser sands (Parker, 1984), mud, muddy sands and clay (Kirkim et al., 2006), slightly gravelly sandy mud (Cruz et al., 2003), sand and mud biodetrital bottom (Dumitrache et al., 2013), very fine sands with *Abra alba-Tellina fabula* communities (Warwick and Davies, 1977; Dauvin and Gentil, 1990), *Venus striatula* sand communities (Klein et al., 1975). Furthermore, this oedicerotid is also present in substrates colonized by marine phanerogams and algae such as *Posidonia oceanica*, *Cymodocea nodosa*, *Zostera marina*, *Z. noltii* and *Caulerpa prolifera* (Sánchez-Jerez et al., 1999; Karaçuha et al., 2009; Bakir and Katağan, 2014; Bellisario et al., 2016; Maidanou et al., 2017), and is observed also in association with the communities of shallow hydrothermal vents (Dando et al., 1995).

*Periocolodes longimanus* can be considered resistant to pollution (Belatoui et al., 2017) and in fact it was reported in estuarine areas and in other localities characterised by the presence of anthropogenic discharges such as fish farms discharges, oil spill and waste pollution (Bakalem, 1998; Pearson and Black, 2000; Faasse and Stikvoort, 2002; Nickell et al., 2009; Sánchez-Moyano and García-Asencio, 2010; Dumitrache et al., 2013; Nikitik and Robinson, 2003; de-la Ossa-Carretero et al., 2016); but in this context, it is important to emphasize that de-la-Ossa-Carretero et al. (2012) have pointed out that this species showed high sensitivity to sewage pollution.

This species is considered eurybathic and associated with hyperbenthic communities (Greze, 1968; Cartes et al., 2009; Koulouri et al., 2013; Navarro-Baranco et al., 2013; Guerra-García et al., 2013). Also, even if *P. longimanus* is common and abundant in shallow waters between 1 and 30 m (Klein et al., 1975; Dauvin and Gentil, 1990; Buhl-Jensen & Fosså, 1991; Dumitrache et al., 2013; Ballesteros et al., 2020), its bathymetric distribution can be deeper up to about 100 m (Viéitez and Baz, 1988; Cartes et al., 2009; Koulouri et al., 2013). Probably the high variability of its bathyal distribution range is related to its regular vertical migrations in the water column; in fact, *P. longimanus* performs vertical nocturnal migrations of variable amplitude (Watkin, 1939b; Sorbe, 1982) and, in some areas, the abundance of this species varies on a daily scale resulting in an increase in its abundance during the night hours (Bossanyi, 1957).

Although this oedicerotid is not characterised by sexual dimorphism generally the female individuals are larger in size. Beare and Moore (1998) analyzed its sex ratio and observed that there is a dominance of females in coastal water populations. In addition, these authors pointed out that female specimens can be parasitized by the copepod *Sphaeronella minuta* Scott T., 1904.

*Periocolodes longimanus* is considered a common prey of some flatfish such as *Buglossidium luteum* (Risso, 1810), *Arnoglossus laterna* (Walbaum, 1792), *Limanda limanda* (Linnaeus, 1758) and *Pleuronectes platessa* Linnaeus, 1758 (Beare and Moore, 1997; Schückel et al., 2011).

In this work, 3127 specimens of *P. longimanus* were observed on soft substratum distributed over a bathymetric range of 7.8 - 12.81 m. *Periocolodes longimanus* exhibited a uniform distribution along the entire coastline, except for the anthropised site of Haifa port (HM27), where the abundance was relatively low.

## Family Photidae Boeck, 1871

*Photis longicaudata* (Spence Bate & Westwood, 1862)

*Photis longicaudata* is considered Atlantic-Indo-Mediterranean species and its distribution extends to the Indian Ocean, Barents Sea, South Africa, India, China, Korean Peninsula, Caribbean Sea, Costa Rica, Venezuela, Gulf of Mexico, Indonesia, Philippine, and New Zealand (Bellan-Santini, 1990). In the Mediterranean Basin and European waters this species is recorded in France, Ireland, Scotland, Spain, Israel, Italy, Portugal, England and Algeria (Falconetti, 1970; Bellan-Santini and Ledoyer, 1973; Parker, 1984; Falciai and Spadini, 1985; Dauvin, 1987; Dauvin, 1999; Moore and Cameron, 1999; Sorbe et al., 2002; Miloslavich et al., 2010; de-la-Ossa-Carretero et al., 2012; Lo Brutto et al., 2016; Plicanti et al., 2017; Navarro-Barranco et al., 2017).

This photid can be observed on several types of substrates such as fine sands (Dauvin, 1987), muddy sands and mud (Falciai and Spadini, 1985), detritic substrates and coralline algae, hard and artificial substrates (Flynn and Valério-Berardo, 2009, 2012; de-la-Ossa-Carretero et al., 2012) and *Sabellaria alveolata* biocostructions (Plicanti et al., 2017). *Photis longicaudata* is also abundant in the photophilic and intertidal algae epifaunal community and is founded in association with some species of *Sargassum* (e.g. *Sargassum stenophyllum*) (Tanaka and Leite, 2003; Ortiz et al., 2005). Moore and Cameron (1999) observed some specimens of *P. longicaudata* among the tentacles of the anthozoan *Cerianthus lloydii* Gosse, 1859 and hypothesized that this amphipod exploits this cnidarian to gain protection and defence from predators. Moreover, *P. longicaudata* is also considered resistant to pollution and has been found in areas affected by anthropogenic impact and sewage pollution (Flynn and Valério-Berardo, 2009, 2012; de-la-Ossa-Carretero et al., 2012).

This species is most abundant between 0 and 50 m but it can also exploit deep substrates located at depths of about 200 m (Ortiz et al., 2005).

*Photis longicaudata* is considered a tube dweller opportunistic suspension feeder as it builds small muddy tubes that can form dense aggregations in fouling communities (Moore and Cameron, 1999; Flynn and Valério-Berardo, 2009).

Our analyses revealed the presence of 988 individuals of this species which was distributed at depths between 7.8 and 11.7 m. *Photis longicaudata* was recorded mainly at the site HM2.1 where an increase in abundance was exhibited in 2014 (751 individuals). A single observation was recorded in 2011 at the Haifa Port site (HM27 station; 118 individuals), while the species was present with occasional occurrences along the southernmost portion of the Israel coasts (H19-H28).

## Family Urothoidae Bousfield, 1979

*Urothoe grimaldii* Chevreux, 1895

This urotoid is reported in Atlantic Ocean, Indian Ocean, and Mediterranean Sea (Sorbe et al., 2002) and in particular for the European waters and Mediterranean Basin is observed in France, England, Germany, Spain, Portugal, Italy, Israel, Tunisia and Turkey (Gottlieb, 1960; Toulmond, 1964; Ladle, 1975; Kingston and Rachor, 1982; Grémare et al., 1998; Martínez and Adarraga, 2001; Sorbe et al., 2002; Marín-Guirao et al., 2005; Covazzi Harriague et al., 2008; Moreira et al., 2008; Bakalem et al., 2009; Zakhama-Sraieb et al., 2009; Bakir and Katağan, 2014; Sampaio et al., 2016).

*Urothoe grimaldii* is considered a characteristic species of the well sorted fine sand sensu Pérès & Picard (1964) (Dauvin et al., 2017) and is closely associated with surface and intertidal soft substrates (Grémare et al., 1998; Momtazi and Maghsoudlou, 2022). This species has been reported on different types of seabeds and intertidal zone of sandy beach (Dahl, 1952; Daief et al., 2014), sandflats (Wynberg and Branch, 1997; Siebert and Branch, 2007), coarse sands (Penas and Gonzalez, 1983; Moreira et al., 2008) fine sands (Bakalem et al., 2009), *Tellina fabula* sand communities (Kingston and Rachor, 1982), *Spisula subtruncata* sand community (Grémare et al., 1998), *Ampelisca brevicornis* fine sand community (Toulmond, 1964), *Amphioxus* sands (Chen et al., 2013), mud, mud and sand mixture (Bakir and Katağan, 2014). Moreover, it was observed in lagoon environments (Wynberg and Branch, 1997; du Plessis and Pillay, 2022), in soft substrates vegetated by *Fucus* spp. and *Caulerpa cylindracea* (Toulmond, 1964; Lorenti et al., 2011) and as epibiont of loggerhead sea turtle *Caretta caretta* (Zakhama-Sraieb et al., 2009). *Urothoe grimaldii* is also adapted to live in polluted coastal areas (Ibanez et al., 1993) and can survive near urban discharges (Avramidi et al., 2022).

This species lives exclusively in shallow coastal waters between 0.5 and 20 m (Penas and Gonzalez, 1983; Moreira et al., 2008; Bakir and Katağan, 2014) and performs seasonal depth migrations.

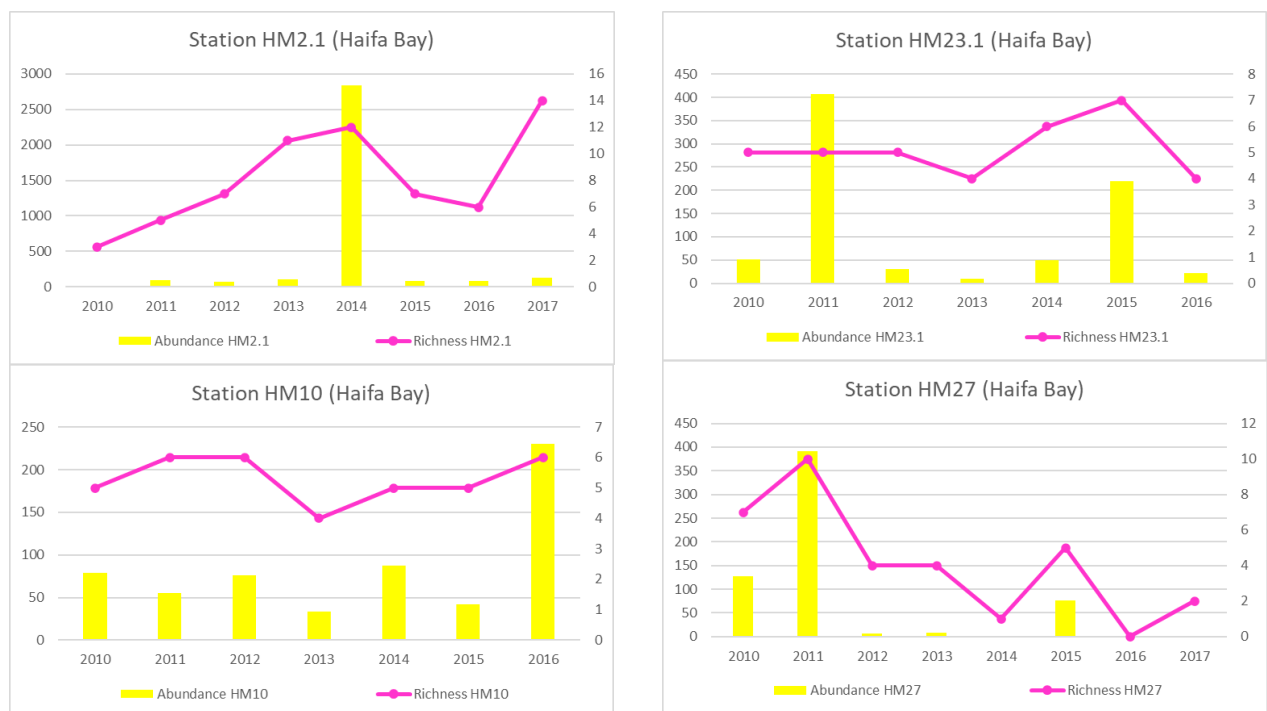
Indeed, it descends to greater depths in summer until it reaches a maximum depth of around 20-30 m (Amouroux, 1974).

Like all species of the genus *Urothoe*, *U. grimaldii* is considered a predator carnivorous species that preys on benthic meiofauna (Macdonald et al., 2010). Females of this species generally live for two years, while the less long-lived males mature within a year (Ladle, 1975). This urothoid can also exploit the burrows of some benthic fossorial species as a source of protection and prey seeking, in fact Goulliart (1952) observed that *U. grimaldii* can live in the tunnels burrowed by the polychaete *Arenicola marina* (Linnaeus, 1758). This amphipod is preyed by coastal fish species such as gobies (Villiers, 1982) and is part of the diet of the flamingo *Phoenicopterus roseus* Pallas, 1811 (du Plessis and Pillay, 2022). In this work, 1593 individuals of this species were sampled and identified. *Urothoe grimaldii* was observed only on mobile substrates distributed between 8.41 and 12.81 m. *Urothoe grimaldii* has been consistently observed along the entire coast of Israel, with occasional records in Haifa Bay, but never in the anthropised sites (HM2.1 and HM27).

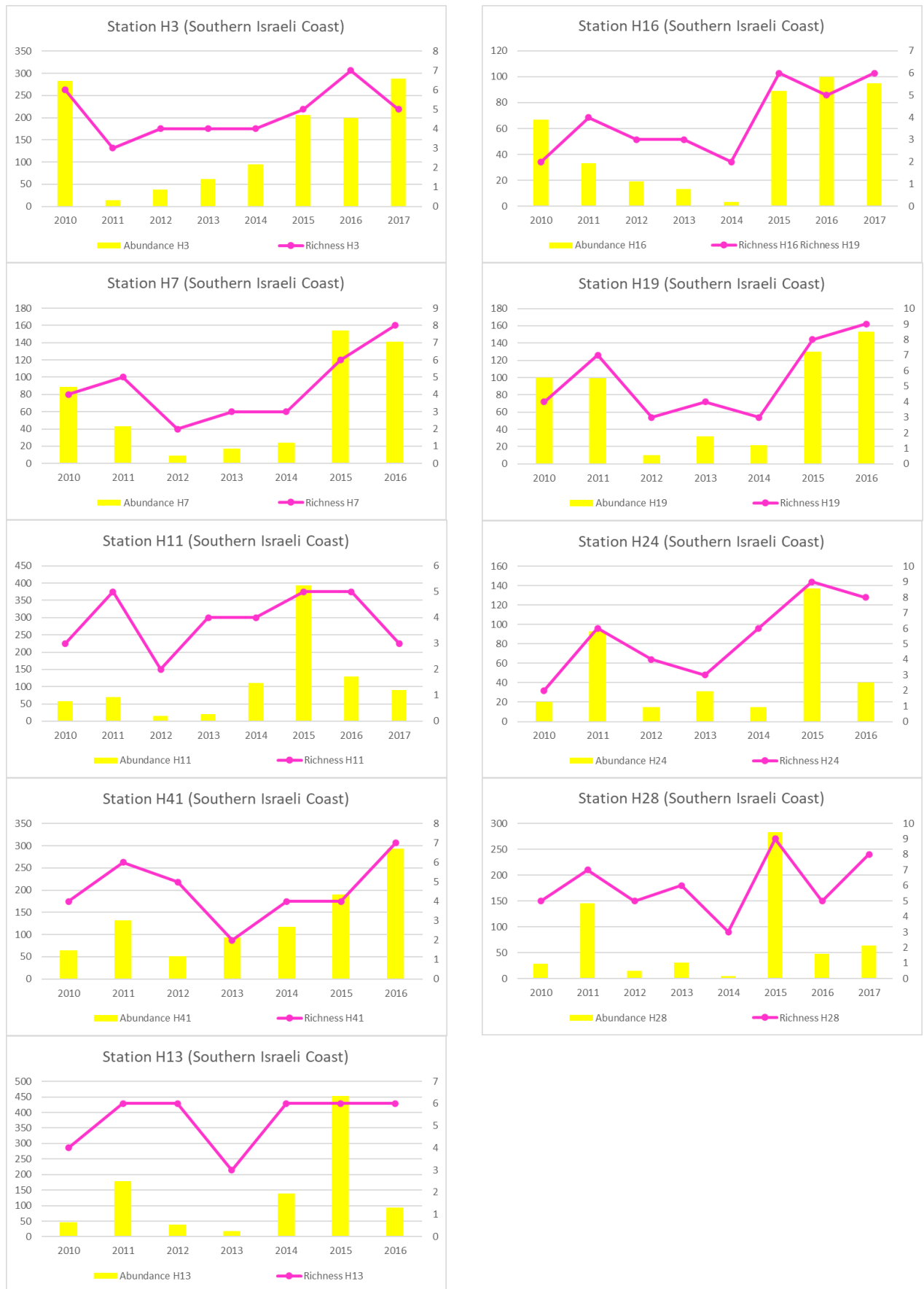
**Spatiotemporal variation**

The temporal variation in the richness of all the taxa, i.e. species and genus taxa, and their total abundance was shown for each station during the consecutive years (Figure 4). In general, the highest abundances were not linked to the highest richness, probably due to the low number of species and the features of sandy amphipod assemblages where local explosions of a few species often occur. An example is the occurrence of 1805 *Cheiriphotis mediterranea* individuals in station HM2.1 in the year 2014, not comparable to a proportional increase in species richness (Figure 4).

The absence of correlation between the abundance and the species richness was caused by a differential contribution of different species. The fluctuations in abundance were due to an increase in the dominant species not the rare and sporadic species; in contrast, the richness was determined by the total number of species and taxa, and influenced by the occasional taxa (Figure 4).

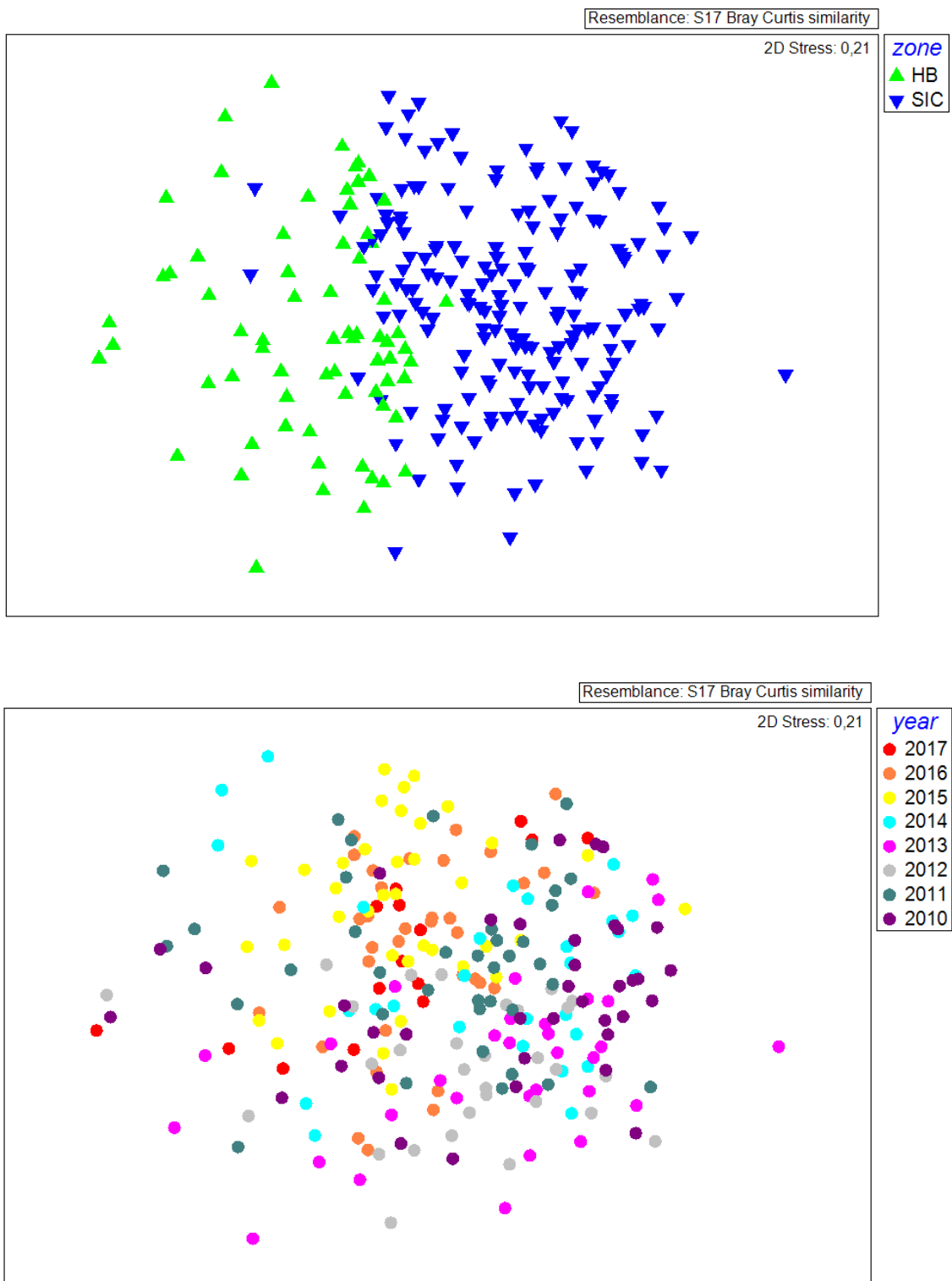


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**Figure 4.** The temporal variation in the richness of all the taxa, including species and genera as listed in Table 2, and their total abundance were shown for each station during the consecutive years.

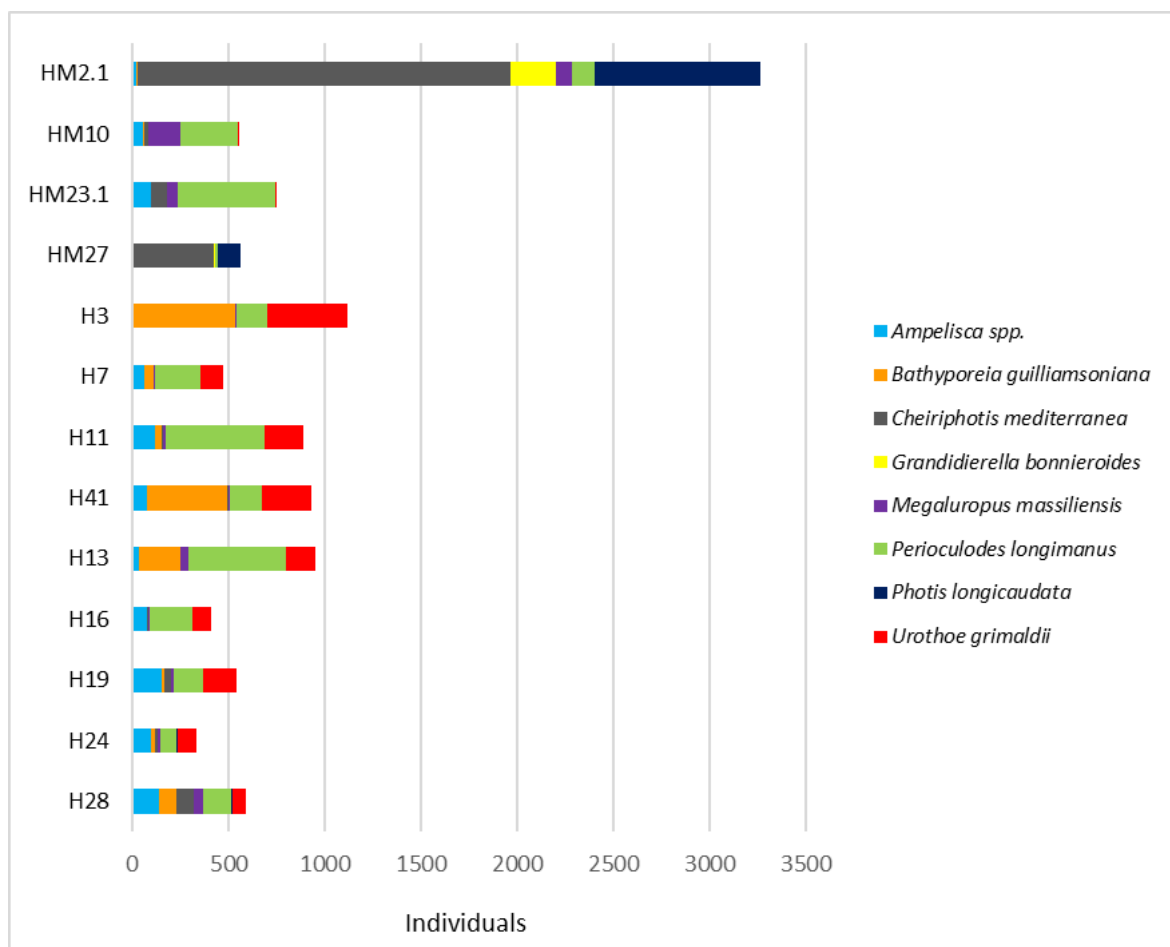
The nMDS did not reveal any temporal variation which could have been associated with changes over the eight years (Figure 5). The contribution of the most abundant species to the taxocenosis profile (SIMPER analysis in Supplement) supported such result, as *Periocolodes longimanus* and *Urothoe grimaldii* were the species with the maximum weight in similarity analyses, showing a stable presence.



**Figure 5.** Non-metric multidimensional scaling (nMDS) plot for the amphipod taxocenosis variation on a spatial scale, between the Haifa Bay – HB – and the Southern Israeli Coast – SIC – zone, (above) and on a temporal scale, across the years (below).

To explore the spatial variability, the long-term monitoring supported the division into two principal zones. The nMDS analysis (Figure 5) showed a discrepancy in amphipod assemblage between the area corresponding to Haifa Bay (HB) and the zone corresponding to the Southern Israel Coast (SIC).

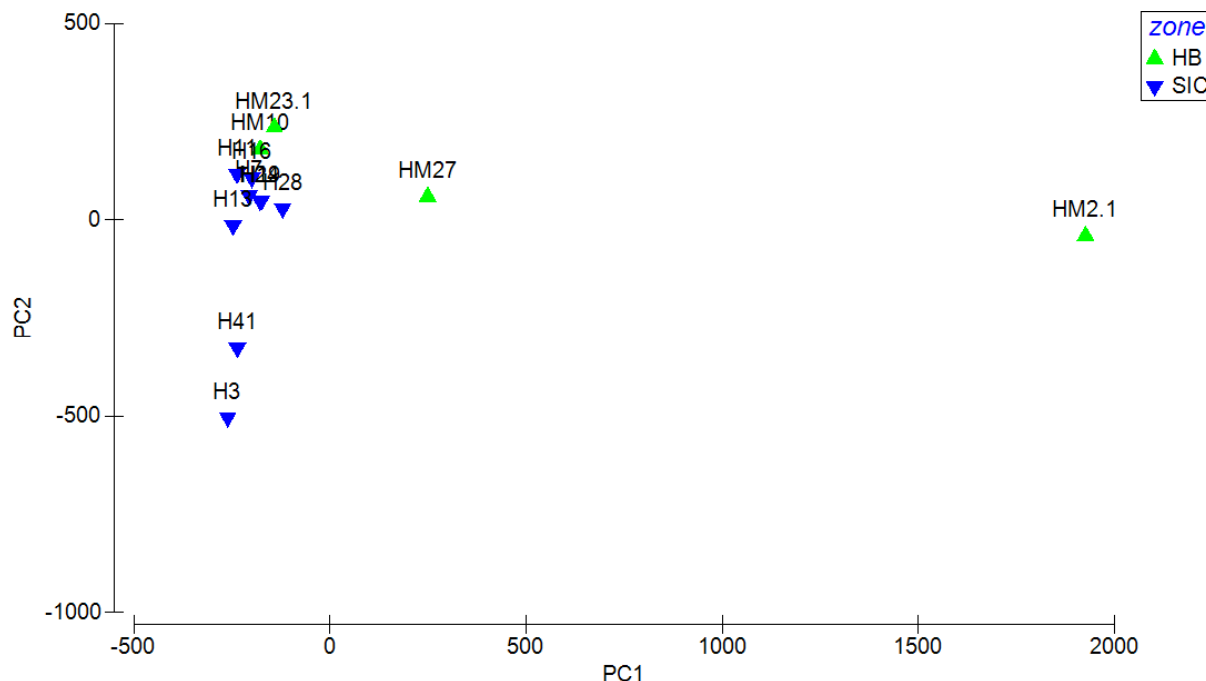
The spatial variation of the most abundant taxa, i.e. the taxa with more than 150 individuals in the period 2010-2017, was observed to understand how the species were distributed among the stations (Figure 6). Figure 6 shows the total number of individuals detected per site. Two different assemblages between the two areas (stations “HM” in Haifa Bay vs. stations “H” in Southern Israeli Coast) are evidenced. *Ampelisca* spp. was present along the coast of Israel and occasionally close to the Haifa promontory (sites HM27 and H3). *Bathyporeia guilliamsoniana* was found along the southern coast of Israel and only sporadically in Haifa Bay in 2014 and 2016-2017; it was very abundant in the southern coastal area, in the region between the Haifa promontory and Tel Aviv (H3-H13), with lower abundances southernmost close to the Israeli desalination plants (sites H19-H24 and H28). *Cheiriphotis mediterranea* was found abundant in Haifa Bay and only occasionally along the southern coast of Israel, with a localised discrete amount in the southern coast close to the Israeli desalination plants (sites H19-H24 and H28). *Grandidierella bonnieroides*, a species recorded for the first time in 2014 in Haifa Bay, showed sporadic occurrence in other sites. *Megaluropus massiliensis* showed a homogeneous distribution along the coast, and higher abundances around the Haifa port sites (HM10 and HM23.1), Tel Aviv (H13), and Ashkelon (H28). *Periculodes longimanus* was consistently present in high abundance along the entire coast of Israel, confirming its tolerance to different conditions. *Photis longicaudata* was collected only in the Bay of Haifa, close to the ports, (H2.1 and HM27), and sporadically close to the desalination plants, particularly starting from 2015 (sites H19-H24 and H28). *Urothoe grimaldii* was a dominant species along the southern Israeli coast and occasionally in Haifa Bay.



**Figure 6.** Spatial variation of the most abundant taxa, i.e. more than 150 individuals in the period 2010-2017. Total number of individuals detected per site.



The spatial distribution of the most abundant taxa and granulometry dataset was analysed through a principal component analysis (PCA). A substantial diversity was scored between Haifa Bay Port (HM27) and Haifa Bay harbour (HM2.1) sites and the other stations (Figure 7). The PCA plot indicates significant site discrepancy (Figure 7). The first two PCA axes explained respectively the 83.1 and 9.6 % of the total variation. The first principal component separated the sites based on the relevant presence of *Cheiriphotis mediterranea* + *Photis longicaudata* species in Haifa Bay (HM2.1 and HM27) (see also Figure 6) and the gravel/coarse sediment. The second principal component distinguished the sites due to the presence of *Bathyporeia guilliamsoniana* + *Perioculodes longimanus* + *Urothoe grimaldii* assemblage (see also Figure 6) and the fine sand sediment.



**Figure 7.** PCA scatterplot depicting the spatial variation along the Israeli coast. The stations are clustered by the taxonomic composition and the substrate granulometry. HB, Haifa Bay; SIC, Southern Israeli Coast.

## Discussion

Marine biodiversity changes across spatial and temporal scales and the extent of such changes can depend on the context and the taxon investigated (Steger et al. 2024). In this paper, a monitoring survey along the Israeli coast provided an example of what a multiscale approach can reveal.

A study of the Israeli amphipod fauna – a dominant taxon of the marine ecosystems – was conducted along the coast on the soft littoral bottom area for eight years. This was the first temporal quantitative study performed on the benthic amphipod fauna in the country. Twenty-five taxa (species or genera) were recorded from a sampling effort in the same stations, located in the northernmost Haifa Bay and along the southern coast, at the same depth range.

The dataset showed an overall stable assemblage of the most common species, with sporadic records of occasional species usually associated with macroalgae or seagrasses reaching very low abundances, generally, less than 150 individuals or detected once over the eight years.

Seven species showed the highest abundances and a temporally constant presence: the Levantine endemic *Cheiriphotis mediterranea*; the Mediterranean endemic *Megaluropus massiliensis*; the NE Atlantic–Mediterranean *Bathyporeia guilliamsoniana* and *Perioculodes longimanus*; and *Photis longicaudata* and *Urothoe grimaldii* presumably widely distributed in the Atlantic Ocean, the Mediterranean Sea and the Indian Ocean. The most significant change was the detection of an alien species in 2014, the circumtropical aorid *Grandidierella bonnieroides* which resulted naturalized (Lo Brutto et al. 2016).

This is not the only Non-Indigenous (NIS) amphipod species detected in Israeli waters; *Bemblos leptochirus*, and *Paracaprella pusilla* were documented in the region at different sites not included in the present paper (Lo Brutto et al., 2019; Lo Brutto and Iaciofano, 2020).

The abundance of these seven species was observed to be unstable at the local level, as fluctuations occurred in the different stations. The range of abundance fluctuations was considerable, encompassing peaks of high numbers of individuals concentrated in specific years which differed among the species; for instance, *Cheiriphotis mediterranea* reached 1805 individuals in station HM2.1 in 2014. No correlation was observed between the total abundance per site and year and the species richness, as the fluctuating abundances were attributable to the few dominant species and the assemblages showed a low  $\alpha$ -diversity.

The random fluctuations mirrored the ecological traits of the species (Navarro-Barranco et al., 2017). The taxa identified in the present study can be classified as hyperbenthos, representing the predominant benthic boundary layer faunal component. The inhabitants of the water layer adjacent to the seabed feed on organic particles on the bottom and, at the same time, are capable of vertical migrations, playing a significant trophic role in the benthic communities and within water column food webs (Buhl-Jensen and Fosså, 1991; Koulouri et al., 2013). These features made the species influenceable by anthropogenic drivers impacting the littoral communities such as nutrient enrichment.

The taxocenosis observed was characterised by deposit feeders on the surface of the bottom, such as ampeliscids, *Bathyporeia* and *Urothoe* genus, and species able to perform vertical migrations, such as *B. guilliamsoniana*, *M. massiliensis* and *P. longimanus*. However, a long-temporal variation in the faunal structure which was expected due to the increase of anthropogenic and environmental stressors was not observed.

The analyses detected only a significant spatial variation that discriminated Haifa Bay from the Southern Israeli Coast.

The physical features of the sediment frequently play a crucial role in shaping amphipod assemblage structure (Buhl-Jensen & Fosså, 1991; Fanelli et al., 2011; de-la-Ossa-Carretero et al., 2012; Scipione, 2013). Along Israeli coast, the bottom, in terms of sediment grain size and chemical composition (Lubinevsky et al., 2019), displayed two areas of different substratum, with which amphipods were associated. Haifa Bay area was more polluted and eutrophic and with a higher portion of gravel, and coarse and medium sand than the Southern Israeli Coast (Lubinevsky et al., 2019). As a consequence, the seven most abundant species were spatially distributed according to the type of sediment that favoured their feeding habit. The different species compositions between the two areas reflected the local environmental features.

The information about the sensitivity of the dominant species to disturbances is worthy of remarks. According to de-la-Ossa-Carretero et al. (2012) shallow soft-bottom amphipods can show different sensitivity levels due to their burrowing behaviour; fossorial can show higher sensitivity than domicolous species. This prediction is confirmed in the present paper. Fossorial species, such as *Bathyporeia guilliamsoniana*, *Perioculodes longimanus*, and particularly *Urothoe grimaldii* (Scipione, 2013), showed a negative response to polluted stations, where they reduced their abundance; other fossorial species showed an unclear pattern or, indeed, a certain tolerance such as *Megaluropus massiliensis*, previously indicated as sensitive to polluted areas (Çinar et al., 2015) and present here with the high abundances around the Haifa harbour localities. The domicolous filter feeders *Cheiriphotis mediterranea* and *Photis longicaudata* (Scipione, 2013) characterized the gravel/coarse sediment stations in Haifa Bay, an area which receives pollutants from rivers effluent of chemical and petrochemical industries, urban and agricultural runoff, and from the Haifa municipality domestic sewage treatment plants.

The dataset also provided biogeographical information on certain species that, to date, appear to exhibit a wide geographic distribution, including *U. grimaldii* and *P. longimanus*, the latter of which was unexpectedly documented from the Barents Sea to New Zealand. These geographical ranges must be subjected to rigorous examination and verification, as it is highly improbable that some of these species can be found in areas that are geographically distant or in habitats with markedly dissimilar environmental characteristics. *Urothoe grimaldii*, a species typically found in sandy habitats, was documented as an epibiont of loggerhead sea turtles (Zakhama-Sraieb et al., 2009). Similarly, *P. longimanus* was reported in the literature as both an infralittoral and bathyal species (Cartes et al., 2007).

The present paper shows that the benthic fauna in Israeli coastal marine environments has not changed over time, showing a pattern congruent with long-term analyses of other taxa (molluscs in Steger et al., 2024). Considering the low species richness, changes in amphipod assemblages are expected to be particularly evident if they occur in the future. In this respect, this comprehensive dataset contributes to our knowledge of the Levantine area and its fauna. Data extrapolated from a long time series provide an accurate baseline for detecting putative changes in biodiversity, and a precise understanding of species distributions is essential for monitoring the impact of climate change on marine ecosystems.

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## Supplementary material

### SIMPER analyses

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