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1 **Unexpected Biotic Homogenization Masks the Effect of a Pollution Gradient on**
2 **Local Variability of Community Structure in a Marine Urban Environment.**

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3

4 **Abstract.** Urbanization of coastal habitats, often exemplified by harbors and marinas, has led to various
5 ecological paradigms, questioning the functioning of these new ecosystems. In the present study we
6 investigated, in a large Mediterranean harbor, whether spatial variation of pollution is present and if
7 this variability drives the structure of the sessile community. We hypothesized locations to have
8 significantly different communities, based on the assumption on the occurrence of environmental
9 gradients of pollution that would constitute selective filters. Three distinct community types were
10 identified in June, coinciding with spatially variable contaminants in sediments. We observed then an
11 unexpected shift of the community between June and August associated to a sharp decrease in
12 biodiversity and a decline of most species, masking the effects of local variation and thus leading to
13 the homogenization of the biodiversity within the harbor. This shift coincided with successive
14 heatwaves (the longest lasting 13 days over 25°C, with a thermal peak at 28°C) which might potentially
15 be associated with a die-off in the harbor communities, regardless of location.

16 Keywords: fouling communities, harbor, Marina, local variation, pollution, heat-wave, introduced
17 species

18

19

1 **Introduction**

2 Urbanization of coastal habitats has led to various ecological paradigms, questioning the functioning
3 of these new ecosystems (Bulleri, 2006; Bulleri and Chapman, 2010). At the forefront, harbors and
4 marinas have been used as models, representative of marine urbanization. A particular focus has been
5 accorded to the role of introduced species, established and abundant in these environments (Mineur
6 et al., 2012). The number of species introductions rose exponentially since the beginning of marine
7 trade traffic, a trend which will continue in the coming decades (Carrasco et al., 2017; Levine and
8 D’Antonio, 2003; Sardain et al., 2019; Seebens et al., 2016). Non Indigenous Species (NIS), if they
9 become invasive, can have high ecological impacts, completely restructuring the ecosystems which can
10 lead to the loss of biodiversity and ecosystem services (Johnston et al., 2015; Pejchar and Mooney,
11 2009; Walsh et al., 2016). This loss of services and the investment to combat these species can also
12 cause severe economic impacts, ranging from hundreds of millions to tens of billions dollars for
13 individual countries (Diagne et al., 2021; Jardine and Sanchirico, 2018).

14 Artificial structures constitute a new substratum to be colonized by sessile fauna, but diverging from
15 natural habitats with distinct communities and altered diversity (Chan and Briski, 2017; Glasby et al.,
16 2007; Mineur et al., 2012; Oricchio et al., 2016). Diversity at all scales is generally regarded as a
17 contributor to biotic resistance against introduced species (Arenas et al., 2006; Elton, 1958; Gestoso
18 et al., 2018; Megina et al., 2016). Associated with the proximity to the primary vector of species
19 introductions (ships’ hull and ballast waters) it makes these substrates highly susceptible to marine
20 introduced species (Ferrario et al., 2017; Glasby et al., 2007; Megina et al., 2016; Oricchio et al., 2016).
21 NIS, especially in urban habitats, are regarded as a major actor of global homogenization of biodiversity
22 (Alberti, 2015; Mckinney and Lockwood, 1999; Mineur et al., 2012; Tsirintanis et al., 2022) and are
23 among the major causes of species extinctions (Blackburn et al., 2019; Clavero and García-Berthou,
24 2005; Mckinney and Lockwood, 1999).

25 High ambient pollution in Persistent Organic Pollutants (POPs), Polycyclic Aromatic Hydrocarbons
26 (PAHs) and Elemental Trace Metals (ETMs) constitutes strong selective filters which profoundly affect
27 local communities, potentially favoring introduced species and leading to strong environmental
28 patchiness (Gauff et al., 2022a; Je et al., 2004; Kenworthy et al., 2018b; Oricchio et al., 2016). Copper,
29 which is used as active component in many antifouling coatings of ships, has been shown to be a major
30 selective agent in harbor environments and seems to favor introduced species, due to their higher
31 tolerance (Canning-Clode et al., 2011; Osborne and Poynton, 2019; Piola et al., 2009). This selective
32 pressure can lead to differential resistance among populations of the same introduced species (Piola
33 and Johnston, 2006) and has been observed to have an effect on populations separated by short

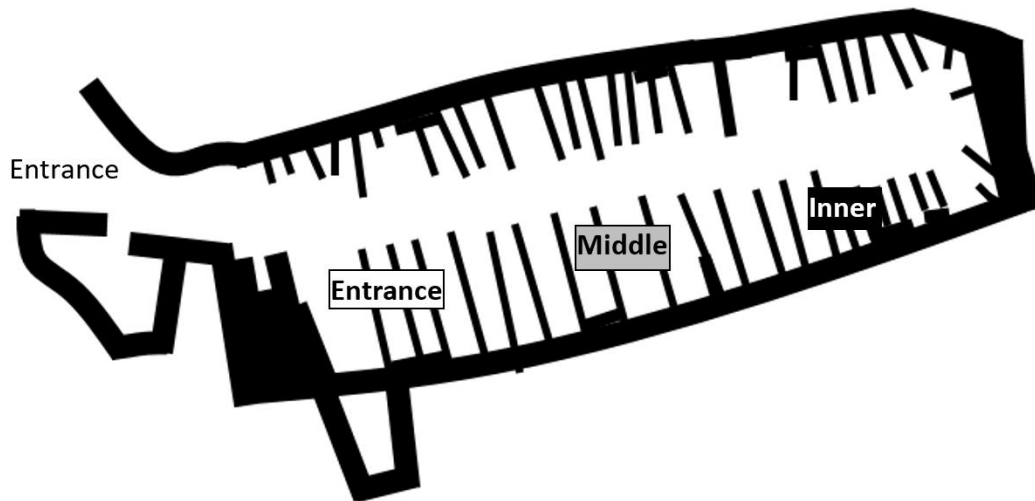
1 distances (~ 100 m; Gauff et al., 2022a). Local adaptation as a response to such selective pressure, can
2 however be impaired by larger scale pulse disturbances exerting identical pressure on the studied
3 environments, favoring genotypes not linked to the original pressure which can homogenize the local
4 populations (Colautti and Lau, 2015; Kawecki and Ebert, 2004). Anthropogenic pulse-disturbances in urban
5 habitats like pulse-pollution (Johnston et al., 2002; Johnston and Keough, 2002, 2000) or marine heat
6 waves (Castro et al., 2021; Smale et al., 2019, 2015) have been demonstrated to have drastic impacts
7 on settled communities and together with introduced species are regarded as contributors to global
8 homogenization of biodiversity (Firth et al., 2016; Mckinney and Lockwood, 1999; Waldock et al.,
9 2018).

10 Previous studies showed that, in some harbors, pollution may be organized as gradient with maximal
11 pollution in the inner parts of the harbor and less strong pollution at the entrance (Gauff et al., 2022a;
12 Je et al., 2004; Kenworthy et al., 2018b; Ryu et al., 2011). Communities are organized along these
13 gradients and display thus high intra-harbor variability, that can be greater than inter-harbor variability
14 when comparing the same functional areas (Je et al., 2004; Kenworthy et al., 2018b; Ryu et al., 2011).
15 In the present study we investigated, in a large Mediterranean harbor, how spatial variation of
16 pollutants may drive the structure of the sessile community. In accordance with previous studies, we
17 expected 1) a spatial variability of pollution with higher pollution at the inner part of the harbor
18 associated to 2) a variability of community structure with 3) lower biodiversity at the inner location.
19 Our experiment however coincided with unexpected extreme heat events, which could, as larger scale
20 disturbance, reduce this observed variability.

21 **Material and Methods**

22 Study site

23 We focused on the Vieux Port de Marseille, a recreational marina with approximately 3200 boat
24 moorings and zones dedicated to careening activities, artisanal fisher boats and short distance ferry
25 transports. The high number of boats and the multiplicity of maritime activities renders this site a key
26 example of human disturbance in anthropized habitats. Within this harbor three locations were chosen
27 to show intra-harbor variability of the environment and the associated community. The locations were
28 oriented on a hypothetic gradient of anthropic disturbance from the entrance to the inner part of the
29 harbor (Fig. 1; Je et al. 2004; Kenworthy et al. 2018b).



1

2 Fig. 1: Site of the Vieux Port de Marseille. The three study locations are indicated.

3

4

5 Environmental variables

6 Per location, three surface sediment samples (depth ranged from 3 - 7 m) of approximately 0.4 kg were
7 taken for Persistent Organic Pollutant analysis, including PolyChlorinated Biphenyls (PCBs), Polycyclic
8 Aromatic Hydrocarbons (PAHs) and the most frequent pesticides (see Table 1). Five more samples were
9 sampled for quantifying Metallic Trace Elements (MTEs). Samples were frozen and lyophilized before
10 analysis.

11 The PCBs and pesticides assessment method has been described in detail in the supplementary
12 material of Gauff et al. (2022b) and in Wafo et al. (2006). We quantified 33 individual PCB congeners
13 which include molecules targeted by the International Council for the Exploration of the Sea (ICES).
14 The targeted pesticides can be seen in Table 1. PAH quantification of all 16 PAH targeted by the US
15 Environmental Protection Agency (USEPA) priority list (Table 1, US EPA 2014), was conducted following
16 Sarrazin et al. (2006), Ratier et al. (2018) and Dron et al. (2019) and is described in detail in the
17 supplementary material of Gauff et al. (2022b).

18 Samples for Metallic Trace Elements (MTEs) quantification were dissolved in a three-acid solution (HCl,
19 HNO₃, HF; all Suprapur grade). The analysis was conducted with a Sector Field Inductively Coupled
20 Plasma Mass Spectrometry (SF-ICP-MS; ELEMENT XR, ThermoElement) as described in Jacquet et al.
21 (2021). The spectrometer was calibrated with standard solutions and an external calibration method
22 using Indium as internal standard. Dissolution procedure efficiency and SF-ICP-MS measurement

Loss of community variability

1 quality were verified by the determination of concentrations of target elements in Certified Reference
2 Materials (MESS-4, SLRS-5, SLEW-3 and CASS-5).

3 Water temperature at 1 m depth was logged at each location with a HOBO® (Onset®) TidbiT v2 Water
4 Temperature Data Logger. We used a 15-minute measure interval for the whole duration of the
5 experiment (135 days). We used the temperature data from the SOMLIT (Service d'Observation en
6 Milieu Littoral; www.somlit.fr; consulted 30th Nov 2021) surveillance station at Marseille: Frioul (11) at
7 surface level from mid-June to end of August for the years 1990 to 2021 as reference to compare our
8 measurements. An extreme heat event would constitute any situation where temperatures exceed the
9 90th percentile of these 30 years of historic data ($T > 24.7^{\circ}\text{C}$), potentially constituting a heat wave if the
10 event lasts for at least 5 days (Hobday et al., 2016). We allowed for a slightly higher temperature in
11 the marina and fixed the threshold at 25°C.

12

13 Experimental design

14 At each location, 10 Correx® Polyethylene panels (0.2 x 0.2 m) were placed horizontally at a constant
15 depth of 1 m on the entirely shaded underside of floating pontoons in April 2019. The distance of the
16 panels to the seafloor varied among locations, between 3 and 7 m. After two months (65 days, June
17 2019), allowing sessile communities to settle, 5 panels per location were randomly sampled. The stress
18 on communities during handling was minimized as much as possible and all manipulations were done
19 underwater and in shaded conditions. After two additional months (70 days more, August) the
20 remaining 5 panels per location were sampled.

21 For both dates, the destructively sampled panels were analyzed in the laboratory. A 144-point grid was
22 superimposed on the panels (stratified point design) and at each point all species were counted. This
23 allowed for an optimal resolution for species contributing to at least 5% cover (~ 0.4 points cm^{-2} ;
24 Taormina et al., 2020). If one species overlapped with another, both were noted to account for
25 epibionts as well as the different strata of the community. Species were identified to the lowest
26 taxonomic level possible (Brunetti and Mastrototaro, 2017; Hayward and Ryland, 1995; Riedl, 1983;
27 Zabala and Maluquer, 1988).

28

29 Statistical analyses

30 Contaminant data were compiled in a table with the individual contaminant name, its family, and its
31 mean concentration for all three locations along with standard deviation. We used uniformly a Kruskal-

Loss of community variability

1 Wallis test (R version 4.0.3; R Core Team 2020) to test for significant differences between locations as
2 conditions for parametric tests were not respected for all contaminants. As numerous contaminants
3 were tested for differences between sites we adjusted the p-value for each contaminant category
4 (MTEs, PAHs, PCBs, Pesticides) with a Benjaminji-Hochberg correction to avoid false positives.
5 Reference values given by the Canadian sediment quality guidelines and by the US Environmental
6 Protection Agency are indicated for all contaminants (CCME, 1999; US EPA, 2005). For MTEs an
7 additional sediment quality category by Guerra-García et al. 2021 is indicated. To avoid problems
8 arising from high multicollinearity, further analysis were conducted using latent variables (Dormann et
9 al., 2013). The contaminant matrix was submitted to a CCA from the 'vegan' R package (version 2.4-6;
10 Oksanen et al. 2018) and centroid coordinates were extracted. The centroid CCA1 value (Axis 1
11 explained 78% of variability) was computed into a 2-degree orthogonal polynomial for use as
12 contamination weight (Cont.wt1, Cont.wt2).

13 All community data were analyzed in 'vegan'. The community matrix derived from the destructive
14 sampling containing the multi strata cover of all present species for each panel was transformed into
15 a Bray-Curtis dissimilarity matrix used in a nMDS comparing panels from the 3 locations in June and
16 August. Multivariate homogeneity of groups (Date : Location) was verified via the 'betadisper' function
17 of the R package 'vegan' (Oksanen et al., 2018). A PERMANOVA (10^5 permutations) tested for the effect
18 of date, location as well as their interaction on community structure. A second PERMANOVA (10^5
19 permutations) was conducted by considering the previously computed contamination weight
20 (Cont.wt) as quantitative variable replacing the categorical location. The model tested the effect of
21 date, contamination weight, as well as their interaction for both polynomial degrees (see Table 3). A
22 subsequent pairwise PERMANOVA (10^5 permutations) from the 'pairwiseAdonis' R package (version
23 0.3; Martinez Arbizu 2019) including a Benjamini-Hochberg correction was conducted to spatially and
24 temporally compare individual locations (Benjamini and Hochberg, 1995). For each panel, species
25 richness, the Shannon-Weaver index as well as the Pielou evenness were calculated. They were
26 compared among locations at each date using a Dunn test from the 'FSA' R package (version 0.8.32;
27 Ogle et al. 2021) accounting for the Benjamini-Hochberg procedure and they were compared between
28 dates using a Kruskal-Wallis test. SIMPER analysis was conducted to determine the species'
29 contribution to the observed temporal difference (Clarke, 1993). The percent cover of the species
30 contributing more than 5% to contrasts between treatments in the SIMPER analysis was compared
31 between June and August with a Kruskal-Wallis test.

32 Results

33 Contaminants

Loss of community variability

1 All contaminants, for which reference values were present, largely exceeded the Canadian sediment
2 quality guidelines as well as concentrations at which 20% and 50% of sediments tested by the US
3 Environmental Protection Agency would become toxic to model amphipods (Table 1; CCME 1999; US
4 EPA 2005). PAHs in the sediments of the Vieux Port de Marseille were highly concentrated. Most
5 notably, Benzo[*a*]pyrene, Benzo[*k*]fluoranthene and Phenanthrene were exceeding three to four times
6 the concentration at which a 50% probability of toxicity is reached (Table 1, US EPA 2005). However,
7 the only POP varying significantly between locations were total PCBs (K-W, $p < 0.05$), which exceeded
8 quality thresholds by orders of magnitude.

9 Most tested MTEs had significant higher concentrations at the inner location compared to the entrance
10 location (K-W, $p < 0.05$), however concentrations in the middle location were close to the inner
11 concentrations. MTE concentrations were all above the Canadian sediment quality guideline and above
12 the concentration of 20% probability of toxicity and fell within a sediment quality category of moderate
13 for As and Cr and bad for Cu and Pb and Zn (Table 2; CCME 1999; US EPA 2005; Guerra-García et al.
14 2021). The Cu concentration in sediments even exceeded a 80% probability of toxicity at all locations
15 (US EPA, 2005).

16

Loss of community variability

1 Table 1: Concentration of POP contaminants in the sediments of the three study locations. Mean values ($\mu\text{g}\cdot\text{kg}^{-1}$
 2 = ppb) with standard deviation. Contaminants with significant differences between locations (Kruskal-Wallis test,
 3 $p < 0.05$) indicated by *. Reference values given by the Canadian Council of Ministers of the Environment (CCME
 4 1999; ISQG: Interim Sediment Quality Guideline; PEL: Probable Effect Level); and the United States
 5 Environmental Protection Agency (US EPA 2005, Table 11; T20: probability of 20% of toxicity among samples;
 6 T50: probability of 50% of toxicity among samples). Bold font for reference values: threshold exceeded.

7

Polycyclic Aromatic Hydrocarbon (PAH) $\mu\text{g}\cdot\text{kg}^{-1}$	Site mean \pm SD			K-W test	CCME		US EPA	
	Inner	Middle	Entrace		ISQG	PEL	T20	T50
Acenaphthene	192 \pm 64.3	230 \pm 33.5	70 \pm 21.4	ns	6.71	88.9	19	120
Acenaphthylene	96.8 \pm 14.4	84.3 \pm 13.2	37.1 \pm 18.4	ns	5.87	128	14	140
Anthracene	660 \pm 75.7	677 \pm 21.3	389 \pm 21.4	ns	46.9	245	34	290
Benzo[a]pyrene	1185 \pm 170	1687 \pm 230	590 \pm 106	ns	88.8	763	69	520
Benzo[a]anthracene	1210 \pm 115	1571 \pm 221	838 \pm 127	ns	74.8	693	61	470
Benzo[g,h,i]perylene	665 \pm 35.3	509 \pm 160	369 \pm 48.8	ns			67	500
Benzo[b]fluoranthene	1184 \pm 180	1184 \pm 120	702 \pm 4.6	ns			130	1110
Benzo[k]fluoranthene	1523 \pm 275	1971 \pm 242	953 \pm 106	ns			70	540
Chrysene	103 \pm 16	174 \pm 77.1	62.9 \pm 14.6	ns	108	846	82	650
Dibenzo[a,h]anthracene	1652 \pm 272	2112 \pm 313	1112 \pm 98.6	ns	6.22	135	19	110
Fluorene	580 \pm 31.5	1099 \pm 716	290 \pm 30.3	ns	21.2	144	19	110
Fluoranthene	486 \pm 117	575 \pm 128	207 \pm 23.2	ns	113	1494	120	1030
Indeno[1,2,3-c,d]pyrene	570 \pm 20.6	581 \pm 142	494 \pm 80.1	ns			68	490
Naphthalene	57.4 \pm 15.3	78 \pm 13	41.2 \pm 12.4	ns	34.6	391	30	220
Phenanthrene	1233 \pm 169	1691 \pm 293	621 \pm 42.9	ns	86.7	544	68	460
Pyrene	767 \pm 46.3	962 \pm 144	389 \pm 65.2	ns	153	1398	120	930
Polychlorinated Biphenyl (tPCB) $\mu\text{g}\cdot\text{kg}^{-1}$								
tPCBs	1252 \pm 6.27	1395 \pm 15.9	455 \pm 7.61	*	21.5	189	35	370
Pesticides $\mu\text{g}\cdot\text{kg}^{-1}$								
aldrin	2.74 \pm 0.22	2.53 \pm 0.14	5.32 \pm 0.36	ns				
Trans_chlordane	1.57 \pm 0.28	1.18 \pm 0.05	2.42 \pm 0.03	ns				
cis_chlordane	0.73 \pm 0.22	0.11 \pm 0.04	0.38 \pm 0.21	ns				
tChlordane	2.3 \pm 0.2	1.29 \pm 0.08	2.81 \pm 0.19	ns	2.26	4.79		
diazinon	4.3 \pm 0.38	2.92 \pm 0.65	2.09 \pm 0.55	ns				
dieldrin	7.11 \pm 0.09	11.3 \pm 1.44	2.4 \pm 0.26	ns	0.71	4.3	0.83	2.9
pp'-DDD	5.33 \pm 0.55	12.3 \pm 1.59	13.1 \pm 1.46	ns	1.22	7.81	2.2	19
pp'-DDE	8.58 \pm 1.39	15 \pm 1.61	17.8 \pm 1.96	ns	2.07	374	3.1	100
pp'-DDT	0.66 \pm 0.19	1.82 \pm 0.1	2.55 \pm 0.16	ns	1.19	4.77	1.7	11
endosulfan-1	1.83 \pm 0.44	3.74 \pm 0.53	2.26 \pm 0.1	ns				
endosulfan- 2	21.2 \pm 0.94	13.8 \pm 1.26	2.57 \pm 0.17	ns				
heptachlor	6.09 \pm 0.54	6.02 \pm 0.43	3.89 \pm 0.28	ns	0.6	2.74		
heptachlor_epoxide_a	23.9 \pm 0.95	15.3 \pm 2.67	5.8 \pm 0.57	ns				
heptachlor_epoxide_b	1.6 \pm 0.4	0.84 \pm 0.14	0.47 \pm 0.11	ns				
isodrin	5.68 \pm 0.38	5.52 \pm 0.43	8.45 \pm 0.14	ns				
lindane	3.89 \pm 0.34	3.83 \pm 0.28	2.07 \pm 0.4	ns	0.32	0.99		
methoxychlor	5.95 \pm 0.48	15.7 \pm 0.83	16 \pm 1.03	ns				

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Loss of community variability

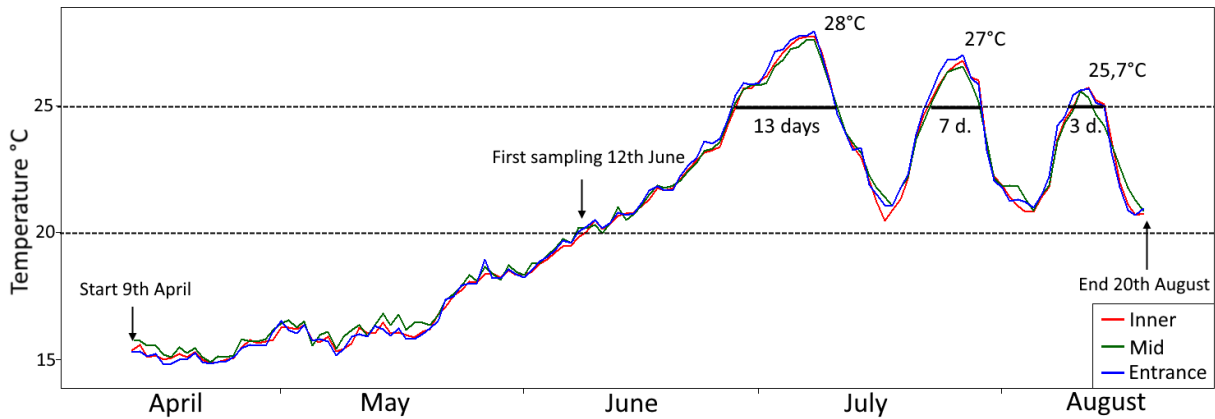
1 Table 2: Concentrations of MTE contaminants in the sediments of the three study locations. Mean values (mg.kg⁻¹ = ppm) with standard deviation. Contaminants with
 2 significant differences between locations (Kruskal-Wallis test) indicated by * p < 0.05. Reference values given by the Canadian Council of Ministers of the Environment (CCME
 3 1999; ISQG: Interim Sediment Quality Guideline; PEL: Probable Effect Level); and the United States Environmental Protection Agency (US EPA 2005, Table 11; T20: probability
 4 of 20% of toxicity among samples; T50: probability of 50% of toxicity among samples). Quality category after Guerra-García et al. (2021) indicated. Bold font for reference
 5 values: threshold exceeded.

Metallic Trace Elements (MTE) mg.kg ⁻¹	Location mean ± SD			K-W test	CCME		US EPA		Quality category ¹
	Inner	Middle	Entrance		ISQG	PEL	T20	T50	
Al27(MR)	12927 ± 3151	8400 ± 3829	5501 ± 1881	*					
As75(HR)	29.2 ± 5.95	31.5 ± 3.62	18.6 ± 2.78	*	7.24	41.6	7.4	20	Moderate
Co59(MR)	4.55 ± 1.65	4.82 ± 2.13	2.07 ± 0.2	*					Good
Cr52(MR)	68.6 ± 14.6	66.4 ± 9.42	27.8 ± 4.54	*	52.3	160	49	140	Moderate
Cu63(MR)	611 ± 111	545 ± 73.4	279 ± 103	*	18.7	108	32	94	Bad
Fe56(MR)	12881 ± 3158	12867 ± 1357	6859 ± 2341	*					
Mn55(MR)	164 ± 33.3	158.2 ± 13.2	126 ± 9.19	*					
Ni60(MR)	17.7 ± 2.76	17.3 ± 2.36	7.89 ± 1.49	*			15	47	Good
Pb208(LR)	299 ± 63.7	262 ± 26.2	136 ± 27.5	*	30.2	112	30	94	Bad
Ti47(MR)	1373 ± 301	1414 ± 251	723 ± 129	*					
V51(MR)	68.5 ± 16.7	70.8 ± 5.28	28.7 ± 1.65	*					
Zn66(MR)	665 ± 151	531 ± 85.8	198 ± 18.5	*	124	271	94	240	Bad

Loss of community variability

1 Temperature

2 The temperature curve of all three locations revealed a similar trend among locations with no striking
3 temperature differences among locations. Taking into account the framework proposed by Hobday et
4 al. (2016), between June and August two major heatwaves can be noted with temperatures exceeding
5 25°C during 13 and 7 days respectively, and a maximum temperature of 28°C, followed by a smaller
6 heat spike (Fig. 2).



7

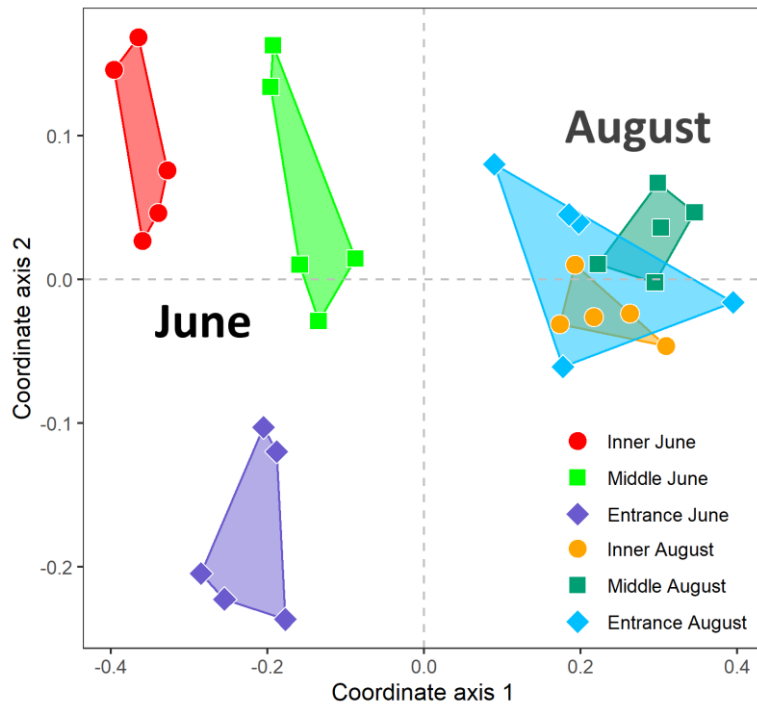
8 Fig. 2: Temperature curve for each location. Start and sampling dates as well as notable values are indicated.

9

10 Community

11 The analysis of community data derived from the laboratory analysis of panels in June and August
12 showed a significant variability both in space and time (Fig. 3). Homogeneity of group dispersion was
13 verified (Betadisper, $F = 1.97$, $p = 0.12$) allowing the application of a PERMANOVA showing a significant
14 effect of location, date as well as their interaction (Table 3). This interaction indicated that the effect
15 of location on community structure was unequal between both dates and may be attributed to the
16 overall more similar communities in August compared to June, when higher differences between
17 locations can be observed (Fig. 3). This was also observable when replacing location with
18 contamination weight derived from the CCA centroids of the contaminant matrix. The PERMANOVA
19 identified a significant effect of date, contamination weight and their interaction, for both polynomial
20 degrees of contamination weight (Table 3). In both cases, interpretation of interaction terms required
21 a subsequent pairwise comparison. The pairwise-PERMANOVA revealed a systematic difference for
22 each of the three locations between June and August ($P_{adj.} < 0.05$, $R^2 > 0.75$; Table 3). All locations
23 differed significantly from each other in June ($P_{adj.} < 0.05$; $R^2 > 0.54$; Table 3), while they were only
24 significantly different between the inner and the middle location in August ($P_{adj.} = 0.02$, $R^2 = 0.44$; Table
25 3).

Loss of community variability



1

2 Fig. 3: Nonmetric Multidimensional Scaling (nMDS) of analyzed communities. Communities from a location are
3 systematically different between June and August (PERMANOVA, p adj. < 0.05 , $R^2 > 0.75$). Communities differ
4 between locations in June (PERMANOVA, p adj. < 0.05 , $R^2 > 0.54$). Inner and Middle different in August
5 (PERMANOVA p adj. $= 0.02$, $R^2 = 0.44$)

6

7

8

9

PERMANOVA results					
Overall Test with Location (Community structure ~ Date * Location)					
	Df	F.Model	R ²	Pr (>F)	
Date	1	117.55	0.65	< 0.001	***
Location	2	8.34	0.09	< 0.001	***
Date : Location	2	10.85	0.12	< 0.001	***
Residuals	24		0.13		
Overall Test with Contamination (Community structure ~ Date * Cont.wt1 + Date * Cont.wt2)					
	Df	F.Model	R ²	Pr (>F)	
Date	1	117.55	0.65	< 0.001	***
Cont.wt1	1	7.74	0.04	0.003	**
Cont.wt2	1	8.95	0.05	0.001	**
Date : Cont.wt1	1	7.38	0.04	0.003	**
Date : Cont.wt2	1	14.31	0.08	< 0.001	***
Residuals	24		0.13		
Pairwise PERMANOVA (Community structure ~ Date : Location)					
Within date June					
	Df	F.Model	R ²	Pr (>F)	
J E vs J M	1	9.22	0.54	0.012	*
J E vs J I	1	19.01	0.7	0.012	*
J M vs J I	1	16.7	0.68	0.012	*
Within date August					
	Df	F.Model	R ²	Pr (>F)	
A E vs A I	1	1.56	0.16	0.26	
A E vs A M	1	2.04	0.2	0.16	
A M vs A I	1	6.36	0.44	0.018	*
Between June and August					
	Df	F.Model	R ²	Pr (>F)	
J E vs A E	1	24.06	0.75	0.012	*
J M vs A M	1	45.74	0.85	0.012	*
J I vs A I	1	118.88	0.94	0.012	*

Table 3: Results from the PERMANOVA and the pairwise PERMANOVA on community structure. Cont.wt1: Contamination weight polynomial degree 1; Cont.wt2: Contamination weight polynomial degree 2; J: June; A: August; E: Entrance location; M: Middle location; I: Inner location. * p < 0.05; ** p < 0.01; *** p < 0.001

1 While a total of 28 species were identified during this study (Table 4), species richness per panel varied
2 from 5 to 12 species. The average number of species per panel was 9.1 and remained constant
3 between both studied periods (8.8 in June, 9.4 in August). Shannon index ranged from 1.09 to 2.58,
4 and Pielou evenness ranged between 0.39 and 0.85. Significantly higher values of indices were
5 detected in June (Dunn test, p < 0.001, Fig. 4), but not for species richness. For each date, no significant
6 differences of diversity indices were found among locations (Dunn test). The polychaete annelids
7 Serpulinae (Rioja, 1923), the serpulid worm *Simplaria pseudomilitaris* (Thiriot-Quievreux, 1965), the

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1 bryozoan *Bugula neritina* (Linnaeus, 1758) and the colonial ascidians *Botryllus schlosseri* (Pallas, 1766)
2 and *Diplosoma listerianum* (Milne Edwards, 1841) accounted to over 85% of the contrast between June
3 and August (Fig. 5). These species are also the five most abundant species with a mean cover of more
4 than 5% at both dates. Most of these species suffered a significant reduction of their percent cover
5 between June and August ($p < 0.01$, Fig. 6A) except the sub-family of the Serpulinae which significantly
6 increased ($p < 0.001$, Fig. 6A). The latter is linked to a significantly increased total cover from June to
7 August ($p = 0.002$, Fig. 6B). Significant differences of total cover between locations could only be
8 identified in June between the middle and the entrance location ($P_{adj.} = 0.04$).

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12 Table 4: Species list for the Marina. Non-Indigenous Species (NIS) and cryptogenic species (crypto.) are indicated
13 with a respective source.

14

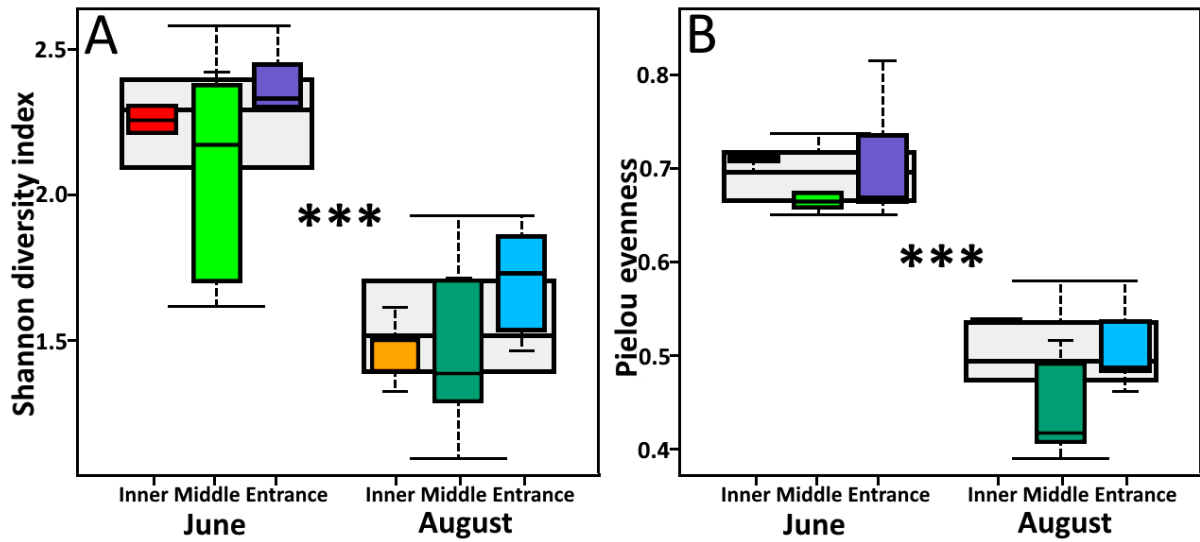
Loss of community variability

Annelida		Status	Source
<i>Hydroides elegans</i>	(Haswell, 1883)	NIS	(Langeneck et al., 2020)
<i>Hydroides norvegica</i>	Gunnerus, 1768		
<i>Janua heterostropha</i>	(Montagu, 1803)		
<i>Jugaria granulata</i>	(Linnaeus, 1767)		
<i>Neodexiospira brasiliensis</i>	(Grube, 1872)	NIS	(Hayward and Ryland, 1995)
<i>Pileolaria berkeleyana</i>	(Rioja, 1942)	NIS	(Langeneck et al., 2020)
<i>Pileolaria militaris</i>	(Claparède, 1870)		
<i>Simplaria pseudomilitaris</i>	(Thiriot-Quievreux, 1965)		
<i>Serpula concharum</i>	(Langerhans, 1880)		
<i>Serpula vermicularis</i>	(Linnaeus, 1767)		
<i>Spirorbis marioni</i>	(Caullery & Mesnil, 1897)	NIS	(Langeneck et al., 2020)
<i>Spirorbis sp.</i>	(Daudin, 1800)		
Arthropoda			
<i>Perforatus perforatus</i>	(Bruguière, 1789)		
Urochordata			
<i>Asciadiella aspersa</i>	(Müller, 1776)	NIS	(Zenetos et al., 2017)
<i>Bortylloides sp.</i>	(Milne-Edwards, 1841)		
<i>Botryllus schlosseri</i>	(Pallas, 1766)	Crypto.	(Zenetos et al., 2017)
<i>Ciona intestinalis</i>	(Linnaeus, 1767)		
<i>Diplosoma listerianum</i>	(Milne Edwards, 1841)	NIS	(Ulman et al., 2017)
Bryozoa			
<i>Bugula neritina</i>	(Linnaeus, 1758)	NIS	(Harmelin et al., 2016)
<i>Bugulina simplex</i>	(Hincks, 1886)		
<i>Bugulina stolonifera</i>	(Ryland, 1960)		
<i>Cryptosula pallasiana</i>	(Moll, 1803)		
<i>Watersipora subatra</i>	(d'Orbigny, 1852)	NIS	Gauff et al. in prep.
Cnidaria			
<i>Laomedea flexuosa</i>	(Alder, 1857)		
<i>Obelia geniculata</i>	(Linnaeus, 1758)		
Mollusca			
<i>Crassostrea gigas</i>	(Thunberg, 1793)	NIS	(Hayward and Ryland, 1995)
<i>Parvicardium pinnulatum</i>	(Conrad, 1831)		
Porifera			
<i>Sycon ciliatum</i>	(Fabricius, 1780)		

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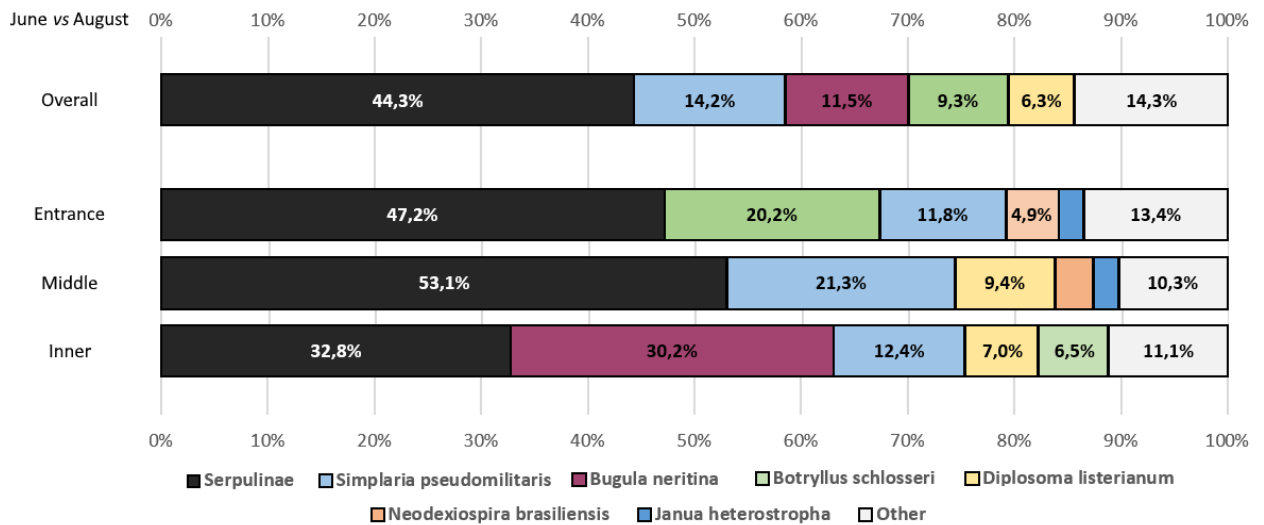
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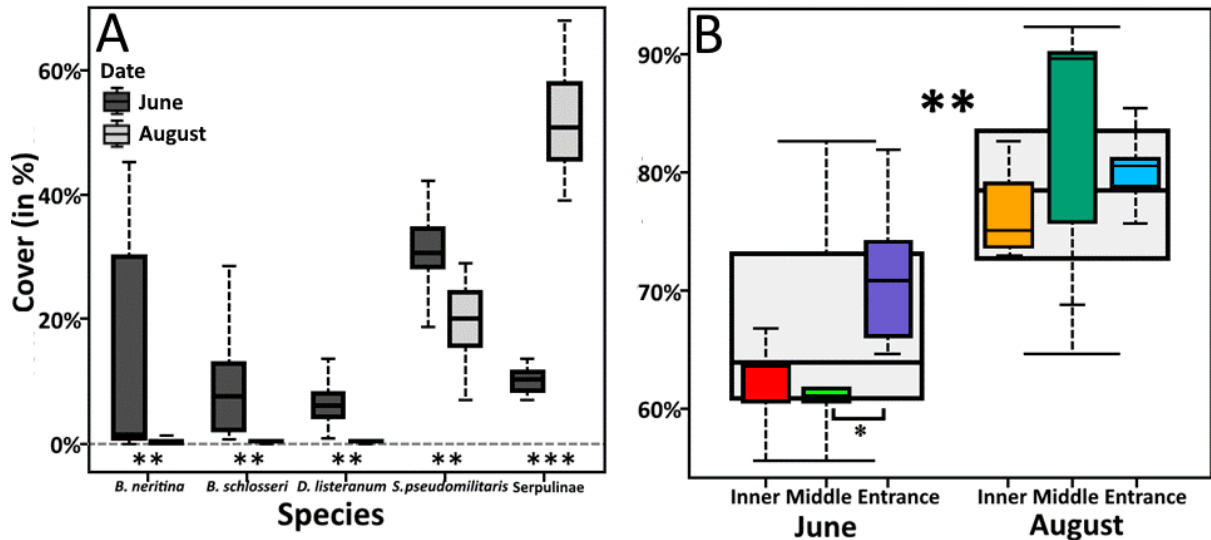
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Fig. 4: Spatial and temporal variation of the Shannon-Weaver diversity index (A) and Pielou evenness (B). Significant differences between dates are indicated with big stars (Kruskal-Wallis test). No significant differences between locations within dates. *** $p < 0.001$



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Fig. 5: SIMPER analysis showing the species that contribute the most to the temporal contrast among communities between June and August (percentage, SIMPER analysis)



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Fig. 6: Percent cover exerted by the A) five most abundant species (which are also the species, most contributing to the contrasts between June and August) between June (dark grey) and August (light grey) and the B) total community for each location and period. Significant differences between dates are indicated with stars (Kruskal-Wallis test). Significant differences between locations are indicated by brackets (Dunn test). * $p < 0.05$; ** $p < 0.01$; *** $p < 0.001$.

9 **Discussion**

10 The present study aimed to describe how the sessile sciaphilic communities of the Vieux Port de
 11 Marseille are composed and how they vary in space. We hypothesized that different locations in the
 12 harbor have significantly different communities, associated to a local variation of pollution, as
 13 described in previous studies (Je et al., 2004; Kenworthy et al., 2018b; Rondeau et al., 2022; Ryu et al.,
 14 2011). In accordance with these studies, most contaminant concentrations and especially Metallic
 15 Trace Elements (MTEs) were minimal at the entrance of the harbor, where more exchange with the
 16 outer water masses occurs (Schiff et al., 2007). Conversely, we did not observe a clear pollution
 17 gradient, as concentrations in the middle and the inner parts of the harbor were similar which might
 18 be explained by local hydrodynamic conditions. Community structure was still significantly affected by
 19 location and by overall contamination.

20 While an expected community heterogeneity was observed among locations in June, it was in contrast
 21 strongly reduced in August with overall similar communities. In addition to this loss of spatial variability
 22 between locations in August, it was possible to observe a strong temporal shift of the communities
 23 between June and August. *Simplaria pseudomilitaris*, *Bugula neritina*, *Botryllus schlosseri* and
 24 *Diplosoma listerianum* had their cover significantly reduced. Conversely, calcareous worms of the

Loss of community variability

1 Serpulinae sub-family are the only group increasing their cover between both periods. We were not
2 able to identify most Serpulinae species on the settlement plates, due to their small size making
3 dissection impossible. Some species of this sub-family could however be identified: *Salmacina*
4 *incrustans* Claparède, 1870, *Hydroides elegans* (Haswell, 1883), followed by *Serpula vermicularis*
5 Linnaeus, 1767 were the most abundant identified Serpulinae while we noted the anecdotal presence
6 of *Hydroides norvegica* Gunnerus, 1768 and *Serpula concharum* Langerhans, 1880.

7 The observed temporal decrease of the cover of most species (including the dominant ascidians and
8 bryozoan) might be explained by the species' life cycles, biotic interactions among the organisms of
9 the fouling community or by an abiotic factor leading to homogenization among the studied
10 communities. Focusing on biotic interactions, ecological succession could explain our observations
11 (Gardner and Wear, 2006; Loke et al., 2016; Raja, 1959) as it highly impacts community structure and
12 biodiversity on settlement plates (Marraffini et al., 2017). The present case might reflect competitive
13 exclusion due to the strong increase of the cover of Serpulinae (Hardin, 1960; Mayfield and Levine,
14 2010). However, while the total species cover increased between June and August, it reached 65%-
15 90% in August with Serpulinae covering 39-68%, which would still leave enough space for other species
16 to grow and thus limit competition for space. Furthermore, Serpulinae are mostly prone species
17 forming calcareous tubes which should vastly limit spatial competition they could exert towards *B.*
18 *neritina* an erect bryozoan. *B. neritina* could also simply overgrow the calcareous tubes. Equally,
19 encrusting colonial ascidians *B. schlosseri* and *D. listerianum* are unlikely to be competitively excluded
20 by Serpulinae. Both colonial ascidians can overgrow solitary animals such as serpulid worms due to
21 their high competitive ability (Grosberg, 1981). The Spirorbinae serpulid *Simplaria pseudomilitaris* is
22 the only major species that occupies a very similar ecological niche as Serpulinae and would be more
23 likely to be competitively excluded by them. Yet, despite the reduction of its cover between June and
24 August, this species seems to be the least affected one since it still covers 7-44% in August. Similar
25 experiments with settlement plates conducted in another marina have shown to have constant
26 community differences between the entrance and the inner part of a marina over a span of more than
27 4 years (Gauff et al., 2022a; Rondeau et al., 2022). These differences remained constant regardless of
28 timing of panel deployment, or length between individual assessments (Rondeau et al., 2022), *i.e.*,
29 regardless of successional state of the community. Thus, the observed differences between June and
30 August might not be attributed to biotic interaction among fouling species or to succession, but rather
31 to external abiotic factors impacting the communities at all three locations.

32 A possible explanation could be a larger scale environmental disturbance, leading to environmental
33 homogenization of all locations of the harbor between June and August. The present Serpulinae could

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1 have a quicker recolonization rate due to their opportunistic nature and/or be more resistant to this
2 disturbance than the other species (Diaz-Castaneda, 2000; Nishi and Nishihira, 1994; Pernet, 2001).
3 Large scale climatic variation and extreme climatic events can affect local communities (Astudillo et
4 al., 2016; Chang et al., 2018; Cascade J. B. Sorte et al., 2010). Water temperature has been subject to
5 high variations between June and August with three successive phases of increased heat, the longest
6 lasting 13 days over 25°C, and a thermal maximum of over 28°C. Two of this phases can be considered
7 as heat waves as the 90th percentile threshold of historic temperature data at surface level in summer
8 (24.7°C; SOMLIT) was exceeded for more than 5 days (Hobday et al., 2016). Research on the potential
9 impacts of climate change shows a drastic impact of rising temperature on communities (Lejeusne et
10 al., 2010; Smale et al., 2017, 2011; Sorte et al., 2010). Heat waves have particularly been shown to
11 have severe consequences on marine communities, their physiology, and their ecosystem services
12 (Britton et al., 2020; Garrabou et al., 2009; Lonhart et al., 2019; Pansch et al., 2018; Smale et al., 2019),
13 with impacts potentially lasting long beyond the punctual time-scale (Gómez-Gras et al., 2021). High
14 temperature events with surface temperatures reaching over 26.5°C in 1997, 1999 and 2003 have led
15 to mass mortalities of multiple sessile species in the studied region (Garrabou et al., 2009; Lejeusne et
16 al., 2010; Perez et al., 2000) and the concerned communities did not recover from this event (Gómez-
17 Gras et al., 2021). Heat waves, as short term events, highly impact community composition in natural
18 and artificial environments (Lonhart et al., 2019; Pansch et al., 2018; Sorte et al., 2010). Climate change
19 and heat waves may also favor introduced species which might be more resistant to heat (Kelley, 2014;
20 Kenworthy et al., 2018a; Lord, 2017). *Hydroides elegans*, one of the most abundant Serpulinae in our
21 samples, has been shown to be highly resistant to temperatures up to 30°C and seems even to profit
22 from higher temperatures due to faster development if this is not negated by other associated
23 variables (Qiu and Qian, 1998, 1997).

24 Given the magnitude of the observed temperature anomaly, it constitutes a candidate for explaining
25 the observed community shift and homogenization. We are here, however, not able to exclude all
26 potential other factors and stochastic variation, or a simple correlational effect could still exist. It is
27 important to note that contaminant concentrations measured in sediments integrate long-term
28 tendencies of pollution and do not account for temporal variation which has been shown to strongly
29 influence fouling community structure (Johnston et al., 2002; Johnston and Keough, 2002, 2000). A
30 pollution spike could thus constitute an alternate explanation for our observations. However, it seems
31 likely that such pollution spike may also be organized as a gradient of concentration. Increased vessel
32 and tourist activity in the summer months might also constitute a potential explanation, it remains
33 however difficult to quantify.

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1 Here we showed a spatially organized community structure in the Vieux port de Marseille similar to
2 previous studies (Gauff et al., 2022a; Kenworthy et al., 2018b; Rondeau et al., 2022), followed by an
3 unexpected change in the community associated to a sharp decrease in biodiversity and a decline of
4 most species except Serpulinae, eclipsing the effects of local variation of environmental factors and
5 thus leading to the homogenization of the biodiversity within the harbor. Urbanization and climate
6 change have led to the worldwide homogenization of ecosystems (Clavel et al., 2011; Magurran et al.,
7 2015; Mckinney and Lockwood, 1999; Qian and Ricklefs, 2006). Heat waves are predicted to increase
8 with climate change, both in frequency and intensity (Holl, 2009; Meehl et al., 2000; Stott, 2016), a
9 trend which is particularly intense in Europe (Rousi et al., 2022). Regular heat waves might favor
10 altered communities with highly opportunistic and/or species with high thermal tolerance,
11 characteristics often described in introduced species (Kelley, 2014; Kenworthy et al., 2018a; Lejeusne
12 et al., 2014; Sorte et al., 2010) and heat-stressed communities have been shown to have an increased
13 invasibility (Castro et al., 2021). Here heat waves might constitute a potential explanation for the
14 observed biodiversity homogenization. However, our experiment was originally not designed to study
15 the effect of heat waves on marina communities and clear conclusions on the reason for the present
16 homogenization are not possible. The temporal concomitance of the homogenization with the heat
17 events remains, however, intriguing and illustrates that more research in this subject is needed.
18 Further experiments with in situ heating structures that could simulate heat waves directly on the field
19 (Gauff et al., 2022; Smale et al., 2017, 2011) would help to investigate the effect such events have on
20 local communities and biodiversity to effectively understand them.

21

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