



Six decades of change in a Mediterranean port fouling assemblage: winners and losers in the era of global change

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ABSTRACT

Fouling assemblages in ports are typically dominated by euryoecious species capable of tolerating wide environmental fluctuations, suggesting that they may be little affected by seawater warming and related global factors. Some sixty years after the pioneering studies of the 1960s, the fouling of the Port of Genoa was re-investigated in the 2020s using similar methods to assess any long-term changes. During this period, annual average sea water temperature in the port increased from 17.7 ± 0.8 °C to 19.3 ± 0.7 °C, while pH decreased from 8.17 ± 0.01 to 8.03 ± 0.002 . A dramatic decline in species richness (52 to 28) and average monthly biomass (8.4 ± 4.4 to 2.1 ± 0.8) was detected. Out of the total of 64 species inventoried, only 16 were common to both periods, while 36 (the 'losers') disappeared (including once abundant species, such as the calcareous sponges *Sycon ciliatum* and *Leucosolenia variabilis*, the hydrozoan *Ectopleura crocea*, and the ascidian *Ciona intestinalis*). The 12 species newly appeared (the 'winners') included 4 non-indigenous species with warm-water affinity, namely the hydrozoan *Calypsothrix cerulea*, the sabellid *Branchiomma luctuosum*, and the serpulids *Hydroides dirampha* and *Spirobranchus arabicus*. The recruitment and abundance of the three most common species, the serpulid *Hydroides elegans*, the barnacle *Amphibalanus amphitrite*, and the encrusting bryozoan *Schizoporella errata* decreased, remained stable or increased, respectively. Contrary to expectations, the modifications observed in the fouling assemblage were remarkable, and consistent with the overall environmental change in the Mediterranean Sea biota.

1. Introduction

Global change is affecting marine ecosystems worldwide (Hodapp et al., 2023), causing biological re-shuffling of communities (Padial et al., 2020). In the Ligurian Sea (NW Mediterranean), for instance, the rapid temperature rise in the 1980-90s was accompanied by dramatic and unprecedented changes in the biota of various habitats, such as the pelagic environment, seagrass meadows, rocky reefs, and marine caves (Bianchi et al., 2019).

However, no information is available on ports, whose biota is typically dominated by euryoecious species capable of tolerating the multiple stressors that characterise these environments, including eutrophication, chemical pollution, and wide fluctuations in physico-chemical parameters (Relini, 1980; Ferrario et al., 2020; Tempesti et al., 2020a).

It can therefore be assumed that organisms living in ports are little sensitive to changes in environmental conditions caused by global change. Since ports are hubs of global maritime traffic and therefore the

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main receptacles of non-indigenous species (Tempesti et al., 2020b, 2025), it can also be assumed that global change is leading to an increase in the occurrence of such species (Occhipinti-Ambrogi, 2021; Zarcero et al., 2026).

To verify both assumptions, time series data would be required (Sukhotin and Berger, 2013). However, time series are rarely available, for ports as well as for other habitats (Gatti et al., 2017). In the absence of continuous long-term data, repeating past studies using comparable methods can provide a successful alternative for assessing stability or change in biotic assemblages (Bianchi et al., 2014).

The city of Genoa, located in the central-northern part of the Ligurian Sea, has one of the largest ports of the Mediterranean Sea, covering an area of 7 km² along 20 km of coastline. Its waters are polluted by urban waste, river mouths, oil terminals, and industrial effluents, resulting in a high concentration of nutrients and organic matter (Ruggieri et al., 2011; Betti et al., 2025). In the 1960s, the fouling assemblage of the Port of Genoa was the subject of extensive research, which used experimental panels submerged for different periods of time (Relini, 1964, 1966, 1968).

In this study, new data collected in 2023–2024 (hereinafter: “the 2020s”) at one of the same stations and using the same methods as in the past (Abate et al., 2025), were compared with the corresponding data from the 1960s in order to assess potential shifts in the composition and biomass of the fouling assemblage, under the hypothesis that no significant changes had occurred (other than some potential additions of non-indigenous species) due the high ecological resistance of the species living in the Port of Genoa.

2. Materials and methods

2.1. Study area

The experiments of the 2020s were conducted at the floating pier of the CNR-IAS Experimental Marine Station, in the ship repair area of the Port of Genoa (44°23'43.8" N; 8°55'52.8" E). This station is located near the eastern entrance to the port and roughly corresponds to the historic station “Zattera” used in the 1960s (Abate et al., 2025), but the port expansion work carried out in recent decades have resulted in it now being situated in a comparatively more confined sub-basin with respect to the past (Fig. 1). Field observations suggest, however, that water exchange with the open sea remains highly efficient, to the point that the new sub-basin is visited by bottlenose dolphins (*Tursiops truncatus*). The depth of the experimental area in particular is 16 m.

Sea Surface Temperature (SST) and water pH data collected during the 1960s (Mor et al., 1970; Relini et al., 1976) were compared with those available from IAS-CNR weather station collected between January 2022 and July 2024.

2.2. Experimental setting

In the 2020s, the fouling assemblage settled on experimental panels was investigated as had been done in the 1960s (details can be found in Relini, 1964, 1977). The sessile invertebrate species were examined under a stereomicroscope and identified to the lowest possible taxonomic level, adopting the nomenclature reported in the World Register of Marine Species (<https://www.marinespecies.org/>), which was also applied to update the species names used in the 1960s.

Two complementary sets of panels were deployed at a depth of 1 m for different purposes: (i) to establish an exhaustive inventory of species, and (ii) to analyse the monthly recruitment cycle of the most abundant species and the total biomass accumulation.

The first set consisted of six panels vertically left submerged for long periods (one–three years) by IAS-CNR. The list of species that settled on these panels was compared with that obtained in the past from similar panels. The species turnover between the 1960s and the 2020s was computed according to the following formula, which has the advantage to be independent of species richness (Bianchi et al., 2014, and references therein):

$$\beta = \min(N_{1960s}, N_{2020s}) \times [\min(N_{1960s}, N_{2020s}) + N_c]^{-1},$$

where β is a turnover index, N_{1960s} is the number of species recorded only in the 1960s, N_{2020s} is the number of species recorded only in the 2020s, and N_c is the number of species recorded in both periods. β , which can range from 0 (complete overlap, no change in the species list) to 1 (no overlap, completely different species composition), was calculated for the entire species list and separately for each phylum.

The second set consisted of five replicated panels vertically deployed and replaced monthly from June 2023 to May 2024. Data were obtained for the three most abundant species: the serpulid *Hydroides elegans*, the barnacle *Amphibalanus amphitrite*, and the encrusting bryozoan *Schizoporella errata* (Canessa et al., 2025); the abundance of the former two species was estimated in terms of number of individuals, that of the latter in terms of number of colonies, both referred to a square decimetre. After identifying and counting the species, the total non-decalcified wet biomass (g•dm⁻²) was quantified as in the 1960s (Relini, 1968), by measuring the difference in weight between the panel colonised by fouling and a clean panel soaked in seawater prior to submersion. A two-tailed paired *t*-test ($p < 0.05$) was used to compare the abundances of the three species and the total biomass of the 1960s and 2020s using the free software PAST (PALEontological STATistics, version 4.03; Hammer et al., 2001).

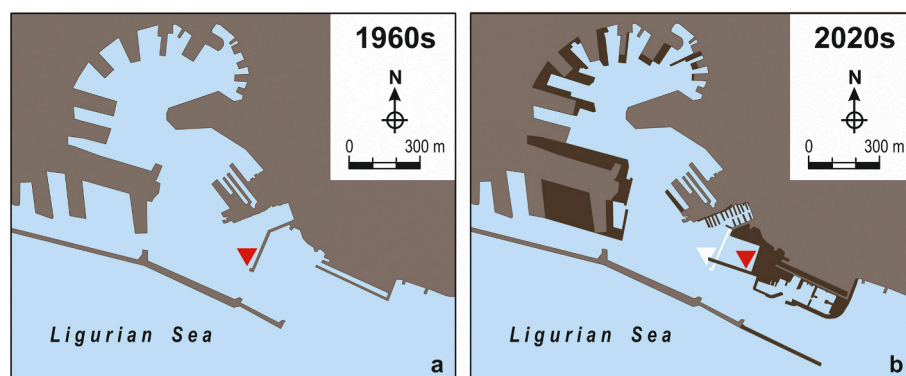


Fig. 1. Maps of the eastern part of the Port of Genoa in the 1960s (a) and in the 2020s (b). The red triangles indicate the location of the station “Zattera”, where the fouling panels were submerged. The darker areas in **b** represent new port structures, whilst the white ones represent the pier that has been demolished and the Zattera that has been moved. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

3. Results

3.1. Sea water temperature and pH variations

The sea surface temperature (SST) in the 2020s remained distinctly higher than in the 1960s throughout the year (19.3 ± 0.7 °C vs 17.7 ± 0.8 °C), with average differences ranging from 0.8 °C in October to 2.6 °C in May (Fig. 2).

The pH of the water decreased from 8.17 ± 0.01 in the 1960s to 8.03 ± 0.002 in the 2020s, with the highest differences recorded in December-January and the lowest in September-October (Fig. 2).

3.2. Species richness

In the two periods analysed, a total of 64 species were recorded, belonging to 8 phyla (Table 1): Porifera (with 15 species), Cnidaria (represented exclusively by Hydrozoa, with 8 species), Mollusca (with 2 species of Bivalvia), Annelida (represented exclusively by Polychaeta, with 16 species, 12 of which belonging to the Serpulidae), Arthropoda (with 4 species of Cirripedia), Bryozoa (with 10 species), Ectoprocta (with 1 species), and Chordata (with 8 species of Ascidiacea). Out of these 64 species, only 28 were found in the 2020s (Fig. 3, upper graph). Annelida was the richest phylum, with 12 species, followed by Cnidaria, with 7 species. Bryozoa had 4 species, while Arthropoda and Chordata had 2 species each. Porifera were represented by a single species. In the 1960s, the total number of species was 52 (Fig. 3, upper graph). The richest phylum was Porifera, with 14 species, followed by Bryozoa, with 10 species. Chordata had 8 species, Annelida 7, Cnidaria 6, Arthropoda 4, Mollusca 2, and Ectoprocta 1.

The comparison between the two periods revealed a dramatic decline in diversity: only 16 species (the 'constants') were common to both periods (Fig. 3, central graph), resulting in a turnover index $\beta = 0.64$ (Fig. 3, lower graph). In the 2020s, 36 species (the 'losers') were no longer observed, and 12 (the 'winners') were recorded for the first time (Fig. 3, central graph). Mollusca and Ectoprocta completely disappeared. Porifera exhibited an extremely high proportional loss (14 out of 15 species), followed by Chordata (6 out of 8) and Bryozoa (6 out of 10). The number of species of Arthropoda halved (from 4 to 2), while Annelida and Cnidaria experienced comparatively less severe losses (4 out of 16 and 1 out of 8 species, respectively). Annelida and Cnidaria were also the only two phyla that gained in richness (from 7 to 12 and from 6 to 7 species, respectively), the number of winners (9 and 2, respectively) being higher than that of losers. Accordingly, the turnover index ranged from $\beta = 1$ for Porifera, Mollusca and Ectoprocta, to $\beta = 0.70$ for Annelida, to $\beta = 0.55$ for Cnidaria and to $\beta = 0.50$ for

Arthropoda, Bryozoa and Chordata (Fig. 3, lower graph).

3.3. Monthly abundance of the main species

The quantitative data obtained from the monthly panels illustrated different trends between the 1960s and the 2020s for the three most abundant species in the fouling assemblage (Fig. 4).

The largest difference concerned *Hydroides elegans*, the species that exhibited the highest abundance in absolute terms in both periods. However, its density underwent a significant ($p = 0.03$) decline of over 90%. While recruitment in the 1960s only occurred between April and December, with a peak in July, in the 2020s it took place throughout the year (including the winter months), reaching a peak in August.

In the case of the barnacle *Amphibalanus amphitrite*, the difference between the two periods was not significant ($p = 0.80$), notwithstanding a slight increase in total abundance in the 2020s. Its recruitment reached a major peak in July and a minor one in September in both periods, while a third, lower peak was observed in February in the 1960s but not in the 2020s.

Finally, the difference was again significant ($p = 0.04$), but in the opposite direction compared to *H. elegans*, for the encrusting bryozoan *Schizoporella errata*, which increased its abundance by an order of magnitude and prolonged its recruitment period: restricted to August in the 1960s, it extended from June to September, with a peak in July, in the 2020s.

3.4. Biomass

Despite the high monthly variability, the average biomass in the two periods was significantly different ($p = 0.01$), being that of the 2020s four times lower than that of the 1960s. Its trend exhibited a polymodal pattern in the 1960s, with major peaks in June and August. In contrast, its trend was flatter and unimodal in the 2020s, with only a moderate peak in August (Fig. 5).

4. Discussion

The comparison between historical (1960s) and recent (2020s) data on the fouling assemblage in the Port of Genoa revealed a substantial reduction in species richness. This comparison warrants caution: unlike Relini's three-year baseline (1964, 1966, 1968), our data represent a single-year snapshot, which may not fully account for interannual variability. Nevertheless, the magnitude of the observed change is so substantial as to indicate that the reduction in species number is fully credible.

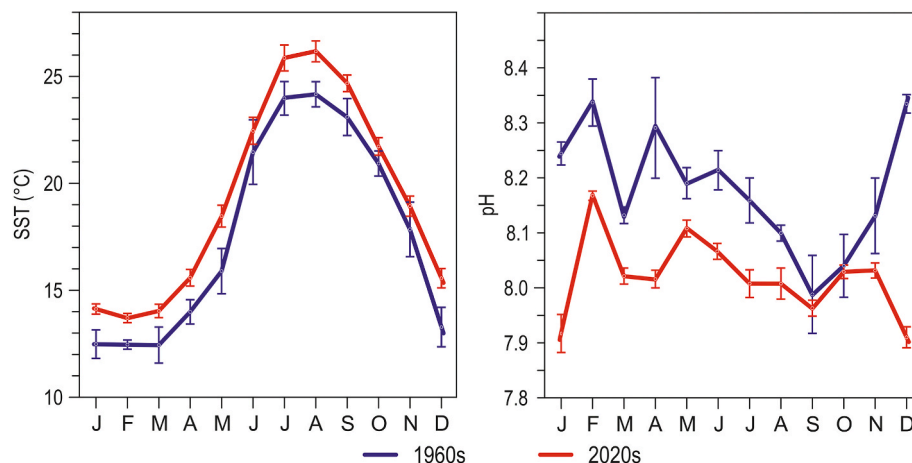


Fig. 2. Monthly variations in mean ($\pm$ SE) Sea Surface Temperature (SST) and pH in the 1960s (blue line) and in the 2020s (red line) at the station "Zattera". (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

Table 1

List of the sessile invertebrate species in the fouling settled on panels submerged in the Port of Genoa (station "Zattera") in the 1960s and in the 2020s.

	1960s	2020s
PORIFERA		
<i>Cacospongia mollior</i> Schmidt, 1862	+	
<i>Cliona viridis</i> (Schmidt, 1862)	+	
<i>Halichondria panicea</i> (Pallas, 1766)	+	
<i>Haliclona cinerea</i> (Grant, 1826)	+	
<i>Haliclona</i> sp.	+	
<i>Hymeniacidon perlevis</i> (Montagu, 1814)	+	
<i>Leucandra crambessa</i> Haeckel, 1872	+	
<i>Leucandra</i> sp.		+
<i>Leuconia</i> sp.	+	
<i>Leucosolenia variabilis</i> Haeckel, 1870	+	
<i>Mycale macilenta</i> (Bowerbank, 1866)	+	
<i>Mycale rotalis</i> (Bowerbank, 1874)	+	
<i>Suberites carnosus</i> (Johnston, 1842)	+	
<i>Sycon ciliatum</i> (Fabricius, 1780)	+	
<i>Sycon raphanus</i> Schmidt, 1862	+	
CNIDARIA		
<i>Calyptospadix cerulea</i> Clarke, 1882		+
<i>Clytia hemisphaerica</i> (Linnaeus, 1767)	+	+
<i>Ectopleura crocea</i> (Agassiz, 1862)	+	
<i>Eudendrium racemosum</i> (Cavolini, 1785)	+	+
<i>Kirchenpaueria halecioides</i> (Alder, 1859)	+	+
<i>Obelia dichotoma</i> (Linnaeus, 1758)	+	+
<i>Obelia geniculata</i> (Linnaeus, 1758)	+	+
<i>Sertularella ellisii</i> (Deshayes and Milne Edwards, 1836)		+
MOLLUSCA		
<i>Mytilus galloprovincialis</i> Lamarck, 1819	+	
<i>Ostrea edulis</i> Linnaeus, 1758	+	
ANNELIDA		
<i>Branchiomma luctuosum</i> (Grube, 1870)		+
<i>Hydroides dirampha</i> Mörch, 1863		+
<i>Hydroides elegans</i> (Haswell, 1883)	+	+
<i>Janua heterostrophia</i> (Montagu, 1803)		+
<i>Josephella marenzelleri</i> Caullery and Mesnil, 1896		+
<i>Neodexiospira pseudocorrugata</i> (Bush, 1905)		+
<i>Pileolaria militaris</i> Claparède, 1870		+
<i>Polydora ciliata</i> (Johnston, 1838)	+	
<i>Protula tubularia</i> (Montagu, 1803)		+
<i>Sabella pavonina</i> Savigny, 1822	+	
<i>Sabella spallanzanii</i> (Gmelin, 1791)	+	
<i>Serpula concharum</i> Langerhans, 1880	+	+
<i>Simplaria pseudomilitaris</i> (Thiriot-Quievreux, 1965)	+	+
<i>Spirobranchus arabicus</i> Mörch, 1863		+
<i>Spirobranchus polytrema</i> (Philippi, 1844)		+
<i>Spirobranchus triquetra</i> (Linnaeus, 1758)	+	
ARTHROPODA		
<i>Amphibalanus amphitrite</i> (Darwin, 1854)	+	+
<i>Amphibalanus eburneus</i> (Gould, 1841)	+	
<i>Balanus trigonus</i> Darwin, 1854	+	
<i>Perforatus perforatus</i> (Bruguère, 1789)	+	+
ENTOPROCTA		
<i>Pedicellina cernua</i> (Pallas, 1774)	+	
BRYOZOA		
<i>Amathia imbricata</i> (Adams, 1800)	+	
<i>Amathia verticillata</i> (delle Chiaje, 1822)	+	
<i>Bugula neritina</i> (Linnaeus, 1758)		+
<i>Bugulina avicularia</i> (Linnaeus, 1758)	+	
<i>Bugulina flabellata</i> (Thompson in Gray, 1848)	+	
<i>Bugulina stolonifera</i> (Ryland, 1960)	+	+
<i>Cryptosula pallasiana</i> (Moll, 1803)	+	
<i>Electra monostachys</i> (Busk, 1854)	+	
<i>Schizoporella errata</i> (Waters, 1878)	+	+
<i>Watersipora cucullata</i> (Busk, 1854)	+	+
CHORDATA		
<i>Botrylloides leachii</i> (Savigny, 1816)	+	+
<i>Botryllus schlosseri</i> (Pallas, 1766)	+	
<i>Ciona intestinalis</i> (Linnaeus, 1767)	+	
<i>Didemnum maculosum</i> (Milne Edwards, 1841)	+	+
<i>Diplosoma gelatinosa</i> (Milne Edwards, 1841)	+	
<i>Diplosoma listerianum</i> (Milne Edwards, 1841)	+	
<i>Styela canopus</i> (Savigny, 1816)	+	
<i>Styela plicata</i> (Lesueur, 1823)	+	

Based on presence/absence data, all phyla—except Cnidaria and Annelida—experienced a loss exceeding 50%. Losers included typical harbour species that were once abundant, such as the calcispongiae *Sycon ciliatum* and *Leucosolenia variabilis* (Pronzato, 1972), the solitary ascidian *Ciona intestinalis* (Montanari and Relini, 1970), and the athecate hydrozoan *Ectopleura crocea* (Morri and Bianchi, 1983). According to Morri et al. (2017), the latter species shifted to deeper waters to escape heatwaves and greater temperature variability in shallow water.

Winners belonged only to Cnidaria and Annelida and were mainly of three types: i) species that already existed elsewhere in the Port of Genoa (but not at Zattera station) in the 1960s; ii) species that typically thrived (and still thrive) outside ports and were only occasionally recorded at Zattera in the 1960s; iii) non-indigenous species found for the first time at Zattera in the 2020s.

The best examples of the former were provided by the two Spirorbinae *Janua heterostrophia* and *Neodexiospira pseudocorrugata*, which were once limited to the innermost parts of the port (Rossi et al., 1970). Their occurrence at Zattera in 2020s may have been favoured by the more confined situation of this station caused by the recent expansion of the basin.

The hydrozoan *Eudendrium racemosum* is commonly found on rocky Mediterranean coasts in summer (Boero and Fresi, 1986; Di Camillo et al., 2012) but was rare in the 1960s at Zattera (Morri and Bianchi, 1983). However, it was present throughout the year in the 2020s, as also seen on natural rocks outside the port (Moglia et al., 2025), suggesting a positive response to the temperature increase. In 2001-02 it was also observed throughout the year in the harbour of Santa Margherita Ligure, about 30 km east of Genoa (Azzini et al., 2003).

Four non-indigenous species were found at Zattera in the 2020s, which added to those already known to exist in the Port of Genoa (Relini, 1964). The hydrozoan *Calyptospadix cerulea* had been first recorded in the Mediterranean Sea in 1978-79 in the Lagoon of Venice (Morri, 1982; Garveia franciscana), while in the Ligurian Sea was only found in 1988-89, on an offshore buoy (Relini et al., 2000). The sabellid *Branchiomma luctuosum*, first reported in the Mediterranean Sea in 1978 near Naples (Knight-Jones et al., 1991) and in the Ligurian Sea in 1989 (Bianchi et al., 2018), is now abundant throughout the Port of Genoa (Moreni et al., 2022). The serpulid *Hydroides dirampha*, known from southern Mediterranean ports since the late 19th century and currently expanding its range throughout the Mediterranean Sea (Taybi et al., 2025), had already been found in the Port of Genoa in the late 1970s only in the vicinity of the hot-water discharge of a thermoelectric power plant (Bianchi). Finally, a second serpulid species, *Spirobranchus arabicus*, is a recent invader of the Mediterranean Sea, hitherto known only from the ports of Valencia and Crete (Kupriyanova et al., 2025). All these non-indigenous species are of tropical origin, and their settlement and survival at Zattera may have been facilitated by the increase in sea water temperature, especially during winter. It is foreseeable that, as the trend of seawater warming continues, boat traffic could further spread these species to other ports and marinas across the region (Ferrario et al., 2024).

The substantial modification in the composition of the fouling assemblage between the 1960s and the 2020s implied a temporal species turnover of more than 60% ($\beta = 0.64$): high turnover values are considered indicative of the impact of seawater warming by Hillebrand et al. (2010). Three times as many species were lost (36) as were gained (12), which conforms to the paradigm that a few winners will replace many losers in the modern biodiversity crisis, with the substitution of non-indigenous species, usually introduced by humans, for local biotas (McKinney and Lockwood, 1999).

Notwithstanding such high species turnover, the organisms dominating the fouling assemblage at Zattera remained the same: *Hydroides elegans*, *Amphibalanus amphitrite*, *Schizoporella errata*. However, they reacted differently to the changed environmental conditions. *H. elegans* showed a dramatic decline in abundance with respect to the 1960s (Rossi et al., 1970); its reproduction is known to be little affected by

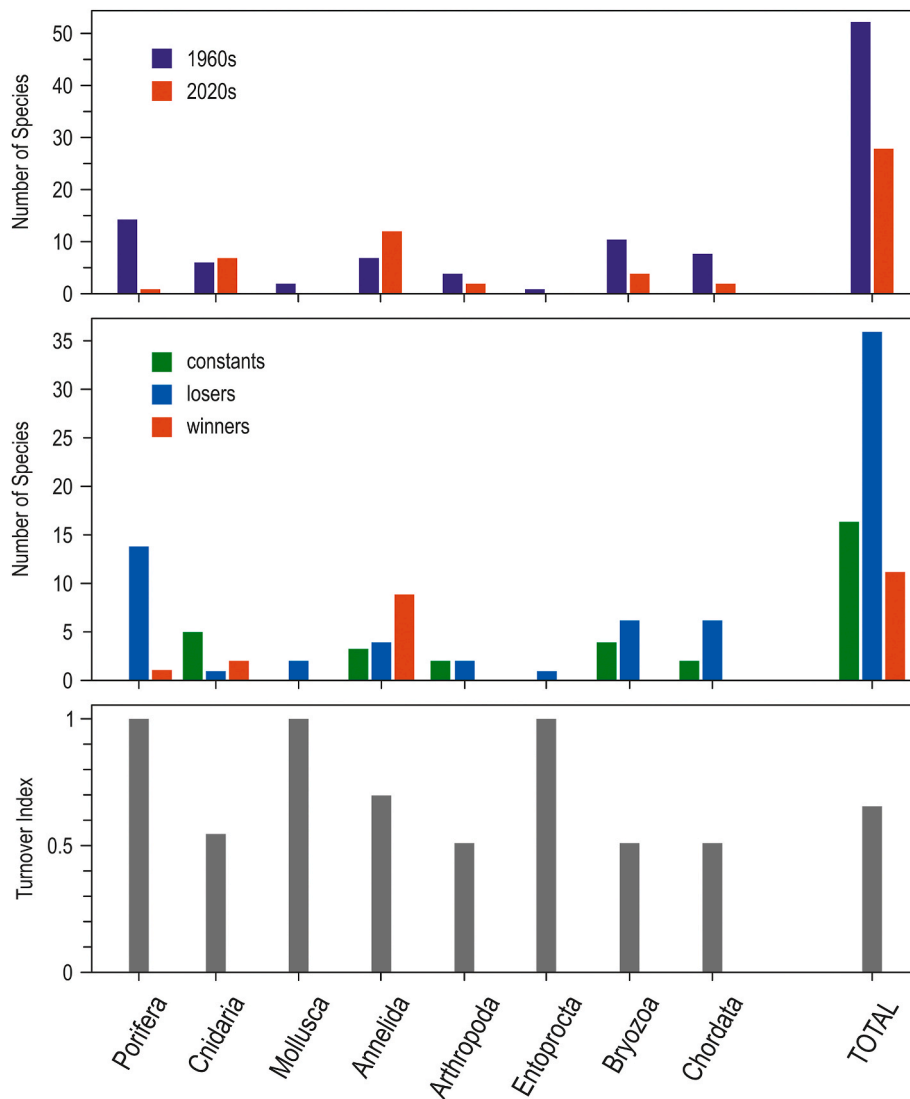


Fig. 3. Overall number of sessile invertebrate species (upper graph), number of species common to both periods (constants), disappeared (losers) or newly appeared (winners) (central graph), and species turnover index (lower graph), by phylum and in total, collected on the fouling panels of the station "Zattera" in the 1960s (blue bars) and in the 2020s (red bars). (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

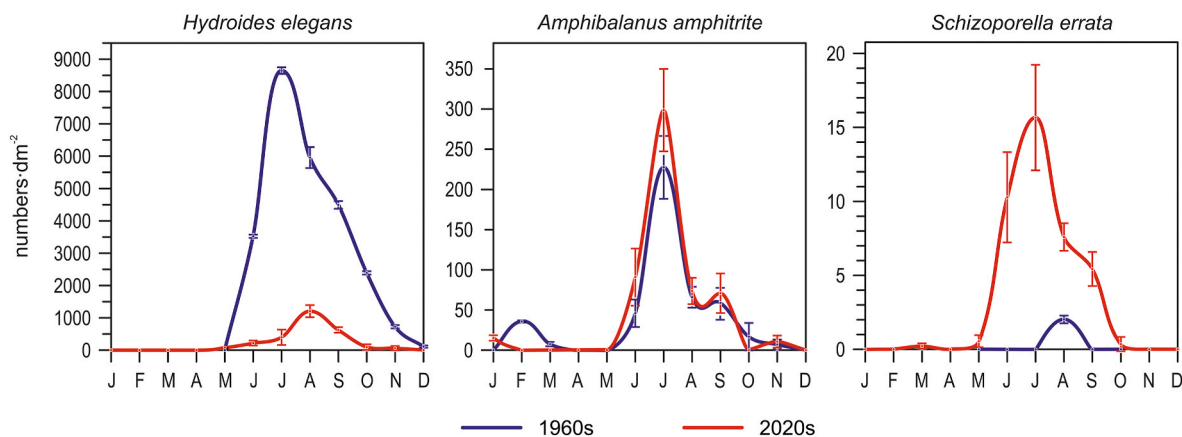


Fig. 4. Mean ($\pm$ SE) monthly recruitment of the three main species on the fouling panels of the station "Zattera" in the 1960s (blue line) and in the 2020s (red line): the serpulid *Hydroides elegans*, the barnacle *Amphibalanus amphitrite* and the encrusting bryozoan *Schizoporella errata*. Note the different scales on the y axes. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

temperature (Qiu and Qian, 1998) but increased water acidification hinders tube production (Li et al., 2014). Recruitment and abundance of

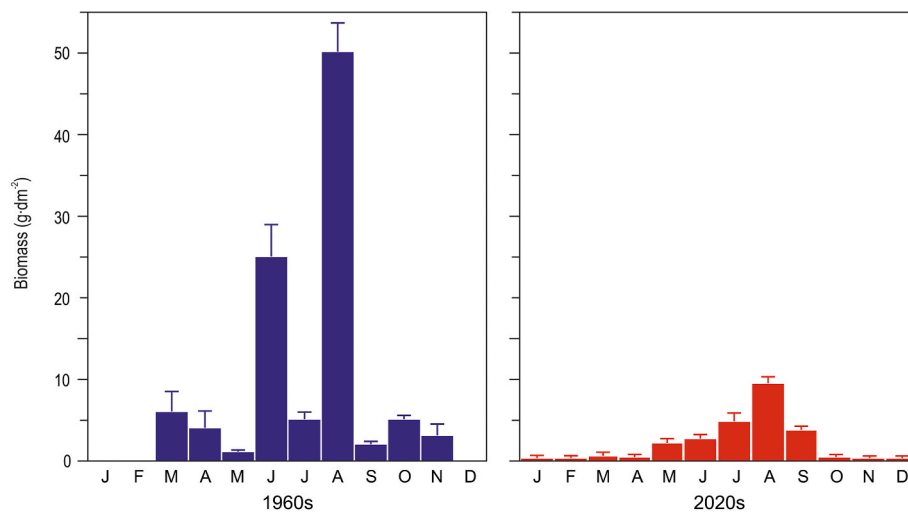


Fig. 5. Mean (+SE) monthly biomass (non-decalcified wet weight) accumulated on the fouling panels of the station "Zattera" in the 1960s (blue bars) and in the 2020s (red bars). (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

A. amphitrite remained similar to those of the 1960s (Relini and Relini Orsi, 1970), consistently with its tolerance to variations of water temperature (Qiu and Qian, 1999) and pH (Nardone et al., 2018). Scarce and occasional at Zattera on panels at 1 m depth in the 1960s (Geraci and Relini, 1970a, 1970b), *S. errata* greatly increased its abundance and prolonged its recruitment period; being a thermophilic species that thrives under various hydrodynamic regimes (Cocito et al., 2000) and tolerates pH reduction (Lombardi et al., 2011), it may have taken advantage of relieved competition with other species (e.g., *H. elegans*).

The significant reduction in the biomass accumulated on monthly panels in the 2020s, compared to the 1960s (Relini, 1968), aligns with predictions of a decline in the biomass of Mediterranean marine ecosystems driven by warming (Artana et al., 2026); however, seawater acidification may have played a non-secondary role, as the most abundant fouling species (serpulids, barnacles, bryozoans) at Zattera possess skeletal structures that are negatively affected by reduced pH conditions (Foo et al., 2018).

5. Conclusions

The dramatic decline in diversity and biomass of the fouling assemblage living in the Port of Genoa between the 1960s and the 2020s has clearly disproved the hypothesis that the organisms living in ports are capable of withstanding environmental change. The high species turnover caused a substantial reshuffling of the fouling assemblage, with a few winners replacing many losers, especially among sponges, bivalves and ascidians, with the only exception of hydrozoans and serpulids. From a quantitative point of view, the dominant species have remained the same, which would appear to provide a partial support to the hypothesis. However, even in this case, the change in their relative importance over time has been considerable.

On the other side, further non-indigenous species, considered by Occhipinti-Ambrogi and Galil (2010) as an aspect of global change, appeared in the Port of Genoa, thus corroborating our second hypothesis. Thriving especially in tropical seas, all these species are to be considered as strictly thermophilic.

Thus, the biological shift observed agrees with the known effect of the factors of global change, such as seawater warming (Kubin et al., 2023) and acidification (Hassoun et al., 2022), which are transforming the biota of the Ligurian Sea (Azzola et al., 2024) and the entire Mediterranean (Azzola et al., 2026). Obviously, other factors such as pollution levels, eutrophication and topographical modifications, may have played a role. Unfortunately, no quantitative data are available to allow for a thorough analysis of these factors, but they appear to be of

secondary importance compared to the strong climate signal.

The amount of information presented here, which includes both qualitative data (species composition) and quantitative data (abundance of the main species and total biomass), provided the first account of the effect of global change on the fouling assemblage of a major port. The fouling assemblages of several Mediterranean ports (particularly Italian ones) have been the subject of extensive research in the past (Relini, 1980): repeating those experiences to allow comparisons with the past will help to consolidate or challenge the results of the present study.

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CRediT authorship contribution statement

Alessandro Abate: Formal analysis, Writing – original draft, Writing – review & editing. **Giorgio Bavestrello:** Conceptualization, Data curation, Funding acquisition, Project administration, Supervision, Validation, Writing – original draft, Writing – review & editing. **Francesca Garaventa:** Resources. **Francesco Enrichetti:** Data curation, Writing – original draft, Writing – review & editing. **Marco Faimali:** Resources, Supervision. **Carla Morri:** Data curation, Formal analysis, Supervision, Validation, Visualization, Writing – original draft, Writing – review & editing. **Veronica Piazza:** Resources. **Beatrice Ridella:** Formal analysis. **Carlo Nike Bianchi:** Data curation, Formal analysis, Supervision, Validation, Visualization, Writing – original draft, Writing – review & editing. **Martina Canessa:** Formal analysis, Validation, Writing – original draft, Writing – review & editing.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

The data supporting the outcomes of this study are included in the article.

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