

ORIGINAL ARTICLE

Distribution patterns of the invasive herbivore *Siganus luridus* (Rüppell, 1829) and its relation to native benthic communities in the central Aegean Sea, Northeastern Mediterranean

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Abstract

The present study explores the distribution of the invasive herbivore fish *Siganus luridus* (Rüppell 1829) and its relationship to native herbivores and macroalgal assemblages in the shallow sublittoral of the Cyclades Archipelago, Aegean Sea, Greece. *In situ* underwater surveys of herbivore abundance (fishes and sea urchins) and algal coverage were carried out at 180 sampling sites in 18 islands. *Siganus luridus* accounted for 17% of the total herbivore abundance, with a decrease in relative abundance from the southeastern to the northwestern islands. In Santorini Island (in the southeast of the study area) *S. luridus* abundance accounted for 90% of the total herbivore fish abundance, while in Kea Island (at the northwestern limit of the study area), *S. luridus* was absent. The spatial variation of minimum sea surface temperature is possibly the reason for its distributional pattern. *Siganus luridus* abundance was found to be positively correlated to the native herbivore *Sparisoma cretense* (Linnaeus 1758). A significant negative relationship was found between the abundance of the invasive species and the sum of erect and canopy algae cover (Dictyotales and *Cystoseira* spp.), which are the main components of its diet in the region. On the other hand, its occurrence was particularly high in barren sites. The results arising from this study reinforce evidence from studies in the Eastern Mediterranean Basin that the expansion of the invasive species *S. luridus* may have profound impacts on native communities in the Mediterranean infralittoral zone.

Introduction

The biodiversity of the Mediterranean Sea has changed significantly over the past two centuries due to the continuous and accelerating introduction of non-indigenous species via human activities (Golani 2010; Zenetos *et al.* 2012). The main vector for introductions into the Mediterranean Sea is direct transport of species from the Red Sea through the Suez Canal, followed by shipping through ballast water and fouling (Gofas & Zenetos 2003; Rilov & Galil 2009). Currently, with more than 900 introduced species, the Mediterranean Sea is considered a hot-

spot of marine bioinvasions (Rilov & Galil 2009; Zenetos *et al.* 2012).

The dusky spinefoot *Siganus luridus* (Rüppell 1829), belongs to the tropical/subtropical species that immigrated into the Mediterranean Sea through the Suez Canal, also known as 'Lessepsian' migrants (Por 1978). *Siganus luridus* was first recorded in the Mediterranean in 1931 off Syria (Gruvel 1931). Since then, its geographical distribution has expanded greatly from the Levantine Sea, where it is considered invasive (Zenetos *et al.* 2010), to the northeast of Tunisia (Ktari-Cha Kroun & Bouhlal 1971), the island of Linosa in the Sicily Strait (Azzurro &

Andaloro 2004), and Cape d'Orlando, Northern Sicily (Castriota & Andaloro 2005). There is evidence that *S. luridus* distribution is continuing to expand westwards and northwards, as recent records have reported its occurrence off the coasts of Marseille (Daniel *et al.* 2009) and Croatia in the South Adriatic Sea (Dulcic *et al.* 2011).

Hassan *et al.* (2003) suggest that the colonisation of the herbivore *S. luridus* and its congeneric *Siganus rivulatus* has been successful because of the large eco-physiological plasticity of the species. Genetic tests of siganid populations in both the Mediterranean and Red Seas indicate that a great number of individuals migrated into the Mediterranean Sea without experiencing any bottleneck event. Furthermore, colonisation of the Mediterranean Sea by the siganids may have been facilitated by the presence of only two native herbivorous fish, *Sarpa salpa* (Linnaeus 1758) and *Sparisoma cretense* (Linnaeus 1758), and hence low competition pressure (Bariche *et al.* 2004; Golani 2010). Along the Levantine coasts, siganids account for over 80% of the total herbivore abundance (Diamant *et al.* 1986; Goren & Galil 2001; Bariche *et al.* 2004), whereas in the Gulf of Aqaba in the Red Sea, where herbivore species are abundant and competition is fierce, they account for only 2% of total herbivore abundance (Bouchon-Navaro & Harmelin-Vivien 1981).

Although biotic factors explaining siganid distribution patterns are considered complicated and need further study, it is clear that the most important abiotic factor influencing the expansion of the siganids westwards is temperature (Golani 2010 and references therein). Golani (1998) suggested a correlation between the early commencement of the spawning season in the Mediterranean (March or April) and western expansion of Lessepsian fish species. The increase of the sea temperature in the Mediterranean, mainly due to human-induced global warming, is expected to reinforce this trend (Lejeusne *et al.* 2010).

Globally, marine herbivores have profound impacts on primary producers, reducing their abundance significantly (Poore *et al.* 2012). In the eastern Mediterranean Sea, *S. luridus* and *S. rivulatus* compete with the native herbivores *S. cretense* and *S. salpa* for food resources and habitat (Bariche *et al.* 2004; Bariche 2006; Azzurro *et al.* 2007) and there is evidence that they have altered the community structure and the native foodweb along the Levantine rocky coasts (Galil 2007). The siganid feeding preferences for coarse brown algae (Stergiou 1988; Lundberg & Golani 1995; Lundberg *et al.* 2004; Azzurro *et al.* 2007), together with their high fecundity, may be responsible for the elimination of canopy-forming algae of the genus *Cystoseira* and hence the creation and maintenance of barren areas (Sala *et al.* 2011). *Cystoseira* spp. account for most of the biomass and production of the shallow rocky algal assemblages (Ballesteros 1989) and harbour a

high biodiversity (Ballesteros *et al.* 2009). Therefore, the expansion of siganid populations may lead to the impoverishment of the biodiversity of sublittoral benthic communities.

This study presents the composition and distribution of invasive and native herbivore fishes in the Cyclades Archipelagos (Greece), Northeastern Mediterranean Sea. Moreover, the relationships between the abundance of the invasive *S. luridus* and native herbivores (fishes and sea urchins) and between the abundance of herbivores and the percentage coverage of macroalgal assemblages were studied.

Study area

The study area was located in 18 islands of the Cyclades Archipelago, Central Aegean, Northeastern Mediterranean Sea. A total of 180 sites with rocky bottoms were sampled between June and July 2008 (Fig. 1). The Aegean Sea is characterised by warm and low nutrient waters (SoHelME 2005). Currently, 76 non-indigenous species have been recorded in the region and this list is expanding continuously (Zenetos *et al.* 2011; Nicolaidou *et al.* 2012). *Siganus luridus* first appeared in the Dodecanese in the late 1950s and since then its population has increased immensely (Stergiou 1988). This invasive species has successfully established populations in the South Aegean, Crete, and recently in the Ionian Sea (Corsini-Foka & Economidis 2007; Bardamaskos *et al.* 2008; Kalogirou *et al.* 2010; Katsanevakis 2011). At present, Chios Island (Northeastern Aegean Sea) has been reported as the northernmost limit of its geographical distribution in Greek waters (Katsanevakis & Tsiamis 2009).

Material and methods

At each site, standard underwater surveys of the abundance of the main herbivore fishes were conducted along visual transects (La Messa & Vacchi 1999). At each sampling site, three replicate 25 × 5 m belt transects were conducted. Transects were placed parallel to the shore at approximately 3 m depth. The observer snorkelled at a constant speed, identifying the herbivore species and counting their abundance within 2.5 m on either side of the transect line.

At each site, algal percentage coverage was estimated using 10 replicates of photoquadrats randomly located along the transect lines. A Canon G7 camera was mounted on a 50 × 50 cm frame with a custom-built camera set-up. Within each photoquadrat, the percentage coverage of algae belonging to four functional groups (defined in Giakoumi *et al.* 2012) – crustose coralline algae (CCA), small filamentous algae (turf algae), *Cystoseira* spp.

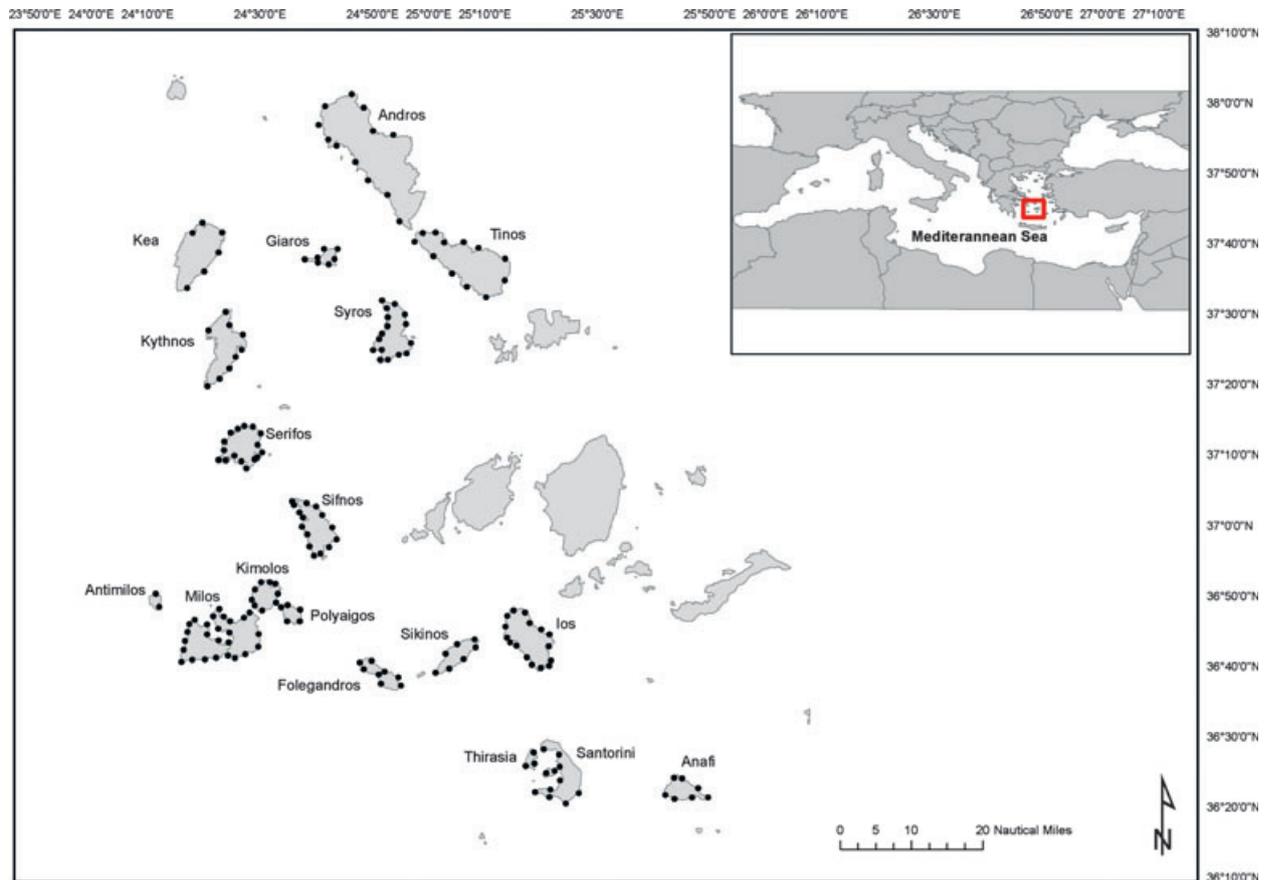


Fig. 1. Study area in 18 islands of the Cyclades Archipelago, Northeastern Mediterranean. Points represent the 180 sampling stations.

(canopy algae), and Dictyotales (erect algae) – along with the percentage coverage of bare rock, was estimated by analysing all photographs with software PHOTOGRID 1.0 beta (C. Bird, Department of Botany, University of Hawaii, Hilo, HI, USA). PHOTOGRID allows the user to estimate percentage coverage of species through point counts; points are superimposed on quadrats in a random manner and the percentage of the total number of points assigned to a species is determined (Perskitt *et al.* 2004). Furthermore, sea urchin abundance was estimated at each site by counting individuals of sea urchins (*Paracentrotus lividus*, *Arbacia lixula* and *Sphaerechinus granularis*) found inside the photoquadrats.

Differences in the mean abundance of herbivore fishes and sea urchins, and mean algal percentage coverage among islands were tested using the non-parametric Kruskal–Wallis test as data did not follow normality and homogeneity assumptions. For pairwise comparisons, the non-parametric Mann–Whitney *U*-test was used. Multivariate patterns in algal and herbivorous communities were visualised using non-metric multidimensional scaling (MDS) ordination (Kruskal & Wish 1978) based on the Bray–Curtis similarity matrix calculated from percent

algae coverage and herbivore abundance data. Square-root transformation was used to homogenize and normalise the data. A hierarchical group average agglomerative clustering method accompanied by a SIMPROF test (Clarke *et al.* 2008) (9999 permutations, 0.1% significance level; Potter *et al.* 2001) was used to explore potential grouping structures among samples. SIMPER analysis was used to determine which species are responsible for dissimilarity between groups of islands. Moreover, we determined correlations among herbivore abundance, and between herbivore abundance and the percentage coverage of algal functional groups, using Spearman's rank order test. This test does not assume bivariate normality and is robust to outliers. The statistical package STATISTIKA v.7 (StatSoft Inc. 2004) was used for the univariate analyses and PRIMER v.6 (Clarke & Gorley 2006) was used for multivariate analyses.

The software ArcGIS 10 was used to map the spatial distribution and relative abundance of the herbivore fishes throughout the Cyclades Archipelago. Satellite data on sea surface temperature (SST) were obtained from Germany's National Research Centre for Aeronautics and Space (DLR-EOWEB 2012). SST was derived from an

advanced very high resolution radiometer (AVHRR) dataset with a grid resolution of 1.5 km. March is considered the month of the year with the lowest SST in the study area (SoHelME 2005). Therefore, data on SST of March 2008 were used to illustrate the overlap between the distribution of the invasive species and the variation in minimum sea temperature. Pearson's correlation coefficient was used to determine the correlation between *S. luridus* abundance and islands' mean SST across the sampling sites. Square-root transformation of abundance data was used to normalise the data.

Results

In all, 1473 individuals of *Sarpa salpa* (76% of the total herbivore abundance), 136 of *Sparisoma cretense* (7%)

and 337 of *Siganus luridus* (17%) were recorded. In most islands the dominant benthic group was turf algae (Fig. 2a). However, in the western islands (Antimilos, Milos, Sifnos, Serifos, Kimolos, Kea and Kythnos) the percentage coverage of *Cystoseira* spp. was significantly higher than in other islands (Fig. 2a; Table 1; Mann-Whitney *U*-test $P < 0.05$). The mean abundance of sea urchins per m² varied between 0 individuals in Antimilos and 6.8 in Kimolos, where significantly higher values were observed (Fig. 2b; Table 1; Mann-Whitney *U*-test $P < 0.05$). However, sea urchin distribution did not reveal latitudinal or longitudinal patterns. On the other hand, the relative abundance of *S. luridus* was higher in the south-eastern part of the study area, especially in Santorini Island, where *S. luridus* accounted for 90% of total herbivore abundance (Figs 2c and 3; Table 1; Mann-Whitney

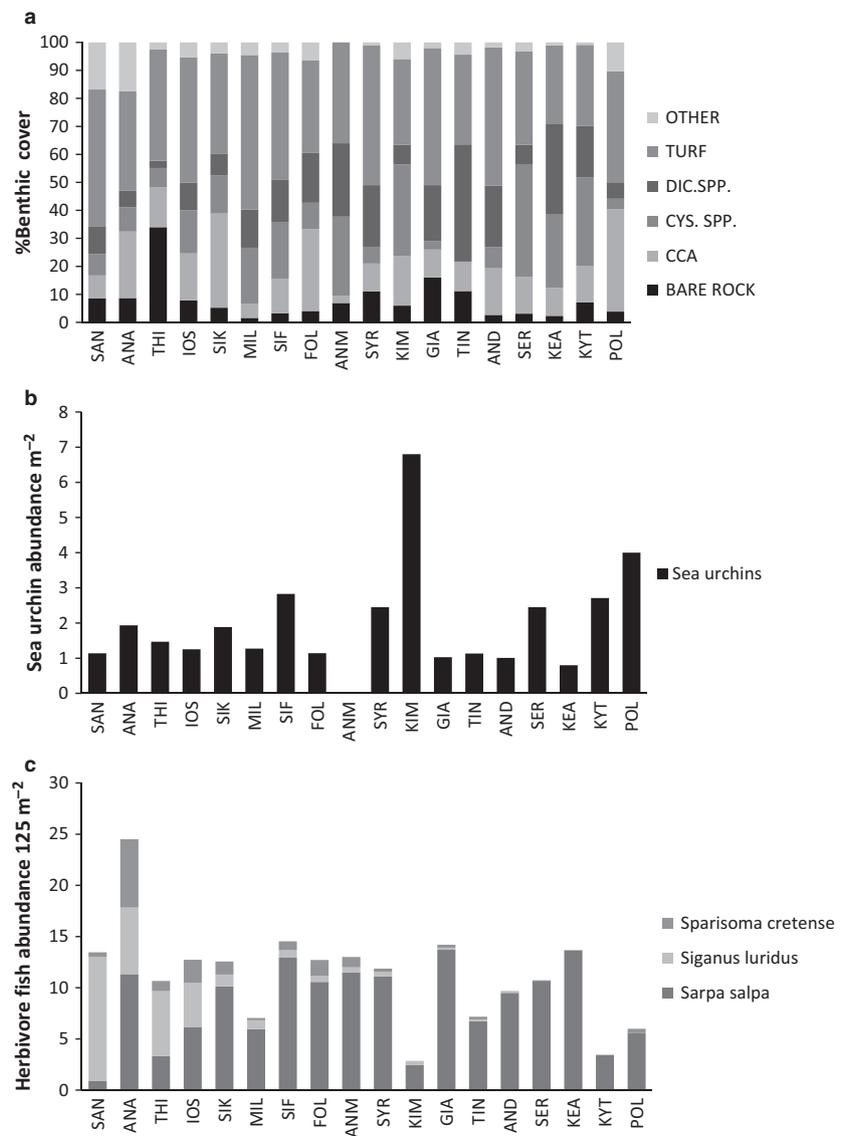


Fig. 2. Distribution of (a) mean percentage coverage of benthos categories: BARE ROCK; CCA: crustose coralline algae; CYS.SPP.: *Cystoseira* spp.; DIC.SPP.: Dictyotales; TURF: turf algae; and OTHER: benthic categories, defined in Giakoumi *et al.* (2012) (e.g. sand, fauna, *Caulerpa prolifera*) with small mean percentage coverage not used in the present study; (b) mean sea urchin abundance; and (c) mean herbivorous fish abundance (*Sarpa salpa*, *Siganus luridus*, *Sparisoma cretense*). Values were estimated per island: ANA: Anafi; AND: Andros; ANM: Antimilos; FOL: Folegandros; GIA: Giaros; IOS: Ios; KEA: Kea; KIM: Kimolos; KYT: Kythnos; MIL: Milos; POL: Polyaigos; SAN: Santorini; SER: Serifos; SIF: Sifnos; SIK: Sikinos; SYR: Syros; THI: Thirasia; TIN: Tinos.

Table 1. Results of Kruskal–Wallis tests on differences in the mean abundance of herbivore fishes and sea urchins, and mean algal percentage coverage among islands.

	<i>Siganus luridus</i>	<i>Sarpa salpa</i>	<i>Sparisoma cretense</i>	Sea urchins	Bare rock	CCA	<i>Cystoseira</i> spp.	Dictyotales	Turf
Chi-squared	101.6721	35.3047	53.5174	28.1909	38.6172	38.4085	73.6297	59.8079	35.9851
df	17	17	17	17	17	17	17	17	17
P-value	<0.05	<0.001	<0.001	<0.05	<0.05	<0.05	<0.001	<0.001	<0.05

U-test $P < 0.05$). The relative abundance of *Siganus luridus* decreased from the southeast to the northwest part of the study area (*i.e.* Kythnos and Kea Islands), where *S. luridus* was absent. Areas of higher *S. luridus* abundance also showed a significant overlap with areas of higher minimum winter SST (Fig. 3). A significant correlation between *S. luridus* abundance and SST was detected ($r^2 = 0.585$, $P < 0.05$; Fig. 4). During the study period, June and July 2008, SST values were more homogeneous throughout the study area (average SST approxi-

mately 22.2 and 23.3 °C, respectively) and no correlation to *S. luridus* abundance was found.

MDS ordination, based on groups identified by SIMPROF test, showed two major groups: the western islands (Antimilos, Milos, Sifnos, Serifos, Kimolos, Kea and Kythnos) and the central and eastern islands (Andros, Tinos, Giaros, Syros, Ios, Folegandros, Polyaiagos, Anafi, Thirasia, Santorini) (Fig. 5). The islands were separated into two distinct groups in the cluster analysis at 80% similarity level. SIMPER analysis revealed that the groups

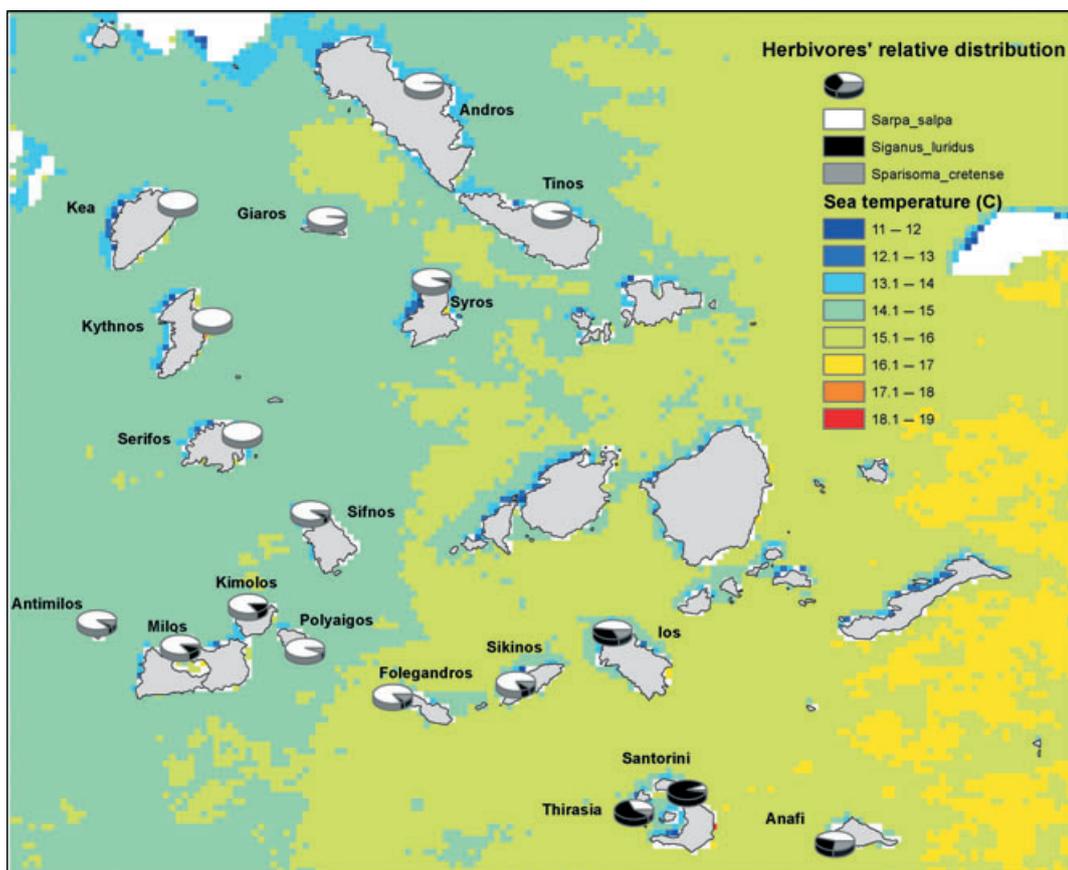


Fig. 3. Spatial distribution of relative abundance of herbivores and spatial variation of minimum sea surface temperature. Pie charts illustrate the relative abundance of herbivores. The white proportion corresponds to *Sarpa salpa*, the grey to *Sparisoma cretense* and the black to *Siganus luridus*. Higher sea surface temperatures for the month of March (minimum sea temperature of the year) is displayed with warm colours, and lower temperatures with cold colours.

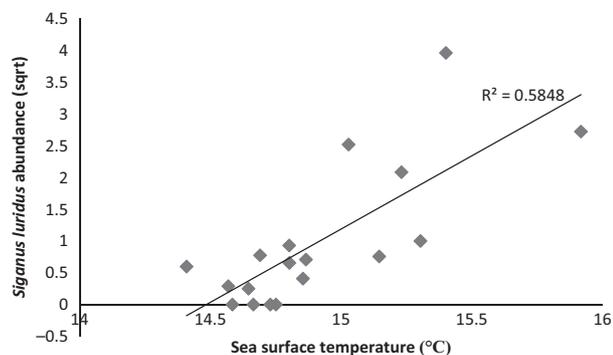


Fig. 4. Correlation between *Siganus luridus* abundance (square-root transformation) and minimum winter sea surface temperature (values in March 2008) in the study area. Each point corresponds to the mean abundance of the species and the mean temperature for each island across the sampling sites.

determined by SIMPROF test differed from each other predominantly in the percentage coverage of *Cystoseira* spp., the percentage coverage of barrens (bare rock with crustose coralline algae) and relative abundance of *S. luridus*. The western islands had higher *Cystoseira* spp. coverage, and the second group of islands were characterised by larger areas of barrens and increased abundance of *S. luridus*.

The abundance of *S. luridus* was positively correlated with the abundance of the thermophilic *Sparisoma cretense* ($r = 0.716$, $P < 0.05$) but no statistically significant relation was found with *S. salpa* ($r = -0.252$, $P > 0.05$) or sea urchins ($r = 0.082$, $P > 0.05$). A significant negative relationship was revealed between the percentage coverage of Dictyotales and *S. luridus* abundance ($r = -0.427$, $P < 0.05$), whereas no significant relationship was detected between *Cystoseira* spp. and *S. luridus* ($r = -0.074$,

$P > 0.05$). However, when the percentage coverage of Dictyotales and *Cystoseira* spp. were combined, a negative relationship with the abundance of the invasive species was revealed ($r = -0.625$, $P < 0.05$). Other statistically significant relations between herbivores and macroalgal assemblages detected include a positive relationship of sea urchin abundance to the percentage coverage of CCA ($r = 0.522$, $P < 0.05$), and a negative relationship between the abundance of *S. cretense* and the percentage coverage of Dictyotales ($r = -0.431$, $P < 0.05$).

Discussion

Herbivores exert strong control over benthic primary producers throughout the world's oceans, regardless of latitude, mean annual temperature, nutrient availability or major herbivore taxon (Poore *et al.* 2012). In the Mediterranean Sea, fish herbivory, although significant (Sala & Boudouresque 1997; Verges *et al.* 2009), has been attributed a secondary role in shaping sublittoral benthic communities. Sea urchins have been considered to play the predominant role in modifying macroalgal communities through intense grazing (e.g. Sala *et al.* 1998; Micheli *et al.* 2005). However, recent studies show that the introduction of non-indigenous herbivorous fishes (*i.e.* the siganids) in the Eastern Mediterranean has substantially changed the foodweb and the structure of algal communities in the rocky sublittoral (Goren & Galil 2005). Moreover, Sala *et al.* (2011) showed, through experimental treatment, that *Siganus luridus* and *Siganus rivulatus* may create and maintain barren areas in the shallow sublittoral of Turkey. The authors suggest that siganids may play a major role in determining the existence and composition of benthic assemblages and hence the biodiversity in the Eastern Mediterranean Sea.

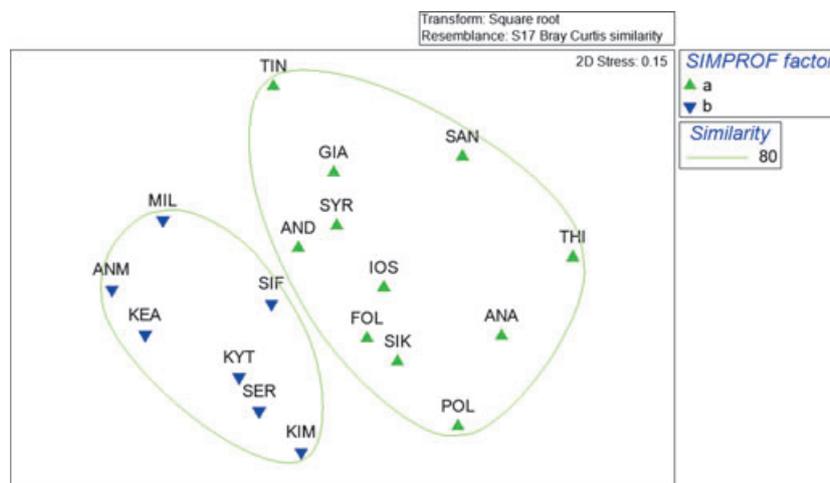


Fig. 5. MDS on herbivore (fishes and sea urchins) abundance and algae cover for all islands. Two different groups were identified by the SIMPROF analysis. Lines represent the groupings at an 80% similarity level.

In the Cyclades Archipelago, the two native species, *Sarpa salpa* and *Sparisoma cretense*, and the invasive *S. luridus* were encountered. Within the study area, *S. luridus* displayed a gradient of decreased abundance from the southeastern islands to the northwestern islands, following a typical spatial distribution pattern of a Lessepsian species (Zenetos *et al.* 2010). In the southeastern parts of the study area, the overlap in higher *S. luridus* abundances with higher SST during March (month of minimum sea temperature) may possibly explain the current distribution of *S. luridus* in the Cyclades Archipelago. The high correlation of its abundance to the abundance of the thermophilic *S. cretense* reinforces such an argument. According to Por (1978), the expansion of Lessepsian migrants into the Mediterranean Sea would be limited by temperature, defining a minimum threshold SST of 16 °C. However, as Bardamaskos *et al.* (2008) observed, the dusky spinefoot may be less sensitive to low temperature than expected. In the Cyclades Archipelago, *S. luridus* was present in areas with minimum winter SST below 14.5 °C, while no clear minimum temperature threshold was revealed. Similarly, the Lessepsian seagrass *Halophila stipulacea* was found to cross the barrier of 16 °C SST and reached areas where SST reaches 15 °C (Gambi *et al.* 2009).

The contribution of the invasive herbivorous fish to the total herbivore density and biomass was much lower in the Cyclades Archipelago than in southeastern areas of the Aegean Sea and the Levantine basin. In the southern Cyclades, where *S. luridus* was more abundant, the invasive species accounted for 37% of the herbivore biomass (Giakoumi *et al.* 2012). East and southwards, off the Turkish Mediterranean coast, siganids were found to account for between 83 and 95% of the biomass of herbivorous fish (Sala *et al.* 2011), and in *Posidonia oceanica* beds in Rhodes Island, they contribute 99% to the total herbivore density and biomass (Kalogirou *et al.* 2012). In the shallow rocky coasts of Israel and Lebanon, the siganid proportion of the total herbivore abundance is approximately three times higher than in the Cyclades (Diamant *et al.* 1986; Bariche *et al.* 2004; Harmelin-Vivien *et al.* 2005) and siganids make up one-third of the total fish biomass (Goren & Galil 2005).

Siganus luridus was frequent in barren areas, especially in islands situated in the southeast (*i.e.* Santorini, Thirasia and Anafi). At the same time, a significant negative relationship was found between *S. luridus* and the sum of erect (Dictyotales) and canopy algae (*Cystoseira* spp.), which are the main components of the siganid diet in the Southern Aegean (Stergiou 1988) as well as in other parts of the Mediterranean Sea (Bariche 2006; Azzurro *et al.* 2007). Although the abundance of *S. cretense* was also found to be negatively correlated to the percentage coverage of Dictyo-

tales, these algae have not been found to be important items of *S. cretense* diet (Azzurro *et al.* 2007). Therefore, the low percentage cover of canopy and erect algae, particularly in the southeastern region of the Cyclades, could be mainly or partly attributed to *S. luridus* grazing, reinforcing sea-urchin activity (Giakoumi *et al.* 2012). Experimental work should be conducted to test this hypothesis.

Grazing by multiple herbivores, fishes and invertebrates, may have synergistic or antagonistic impacts on macroalgal assemblages. Pages *et al.* (2012) showed that intense grazing of the sea grass *P. oceanica* by *S. salpa* indirectly increased the size-dependent predation risk on the competing herbivore sea urchin *Paracentrotus lividus* by reducing canopy height. Similar studies, exploring the interactions among herbivores in Mediterranean rocky reefs would be valuable in understanding direct and indirect impacts of invasive herbivorous fish on benthic communities and more broadly on the Mediterranean marine food web.

Conclusions

Currently, the most abundant herbivorous fish in the Cyclades Archipelago is the native species *Sarpa salpa*, although this situation might change in the near future. The absence of commercial value for *Siganus luridus* in the Cyclades Islands together with the expected rise of SST (and salinity) due to global climate change may act additively to the high competitiveness and fecundity of this invasive fish (Bariche *et al.* 2004; Raitso *et al.* 2010). Another species of *Siganus* (*Siganus rivulatus*) which occurs in the southeastern areas of the Mediterranean (Ben-Tuvia 1964), was absent in the Cyclades during the study period. Yet, three years later (in July 2011) large schools of *S. rivulatus* thrived along the rocky shores of Santorini Island (S. Giakoumi, personal observation). Quantitative studies are essential in understanding future shifts in ecosystem functioning due to climate change (Lejeune *et al.* 2010). The future increase of these invasive species and their interaction with native herbivores may have profound community-wide consequences at the Mediterranean infralittoral zone.

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References

- Azzurro E., Andaloro F. (2004) A new settled population of the Lessepsian migrant *Siganus luridus* (Pisces: Siganidae) in Linosa Island, Sicily Strait. *Journal of the Marine Biological Association of the United Kingdom*, **84**, 819–821.
- Azzurro E., Fanelli E., Mostarda E., Catra M., Andaloro F. (2007) Resource partitioning among early colonizing *Siganus luridus* and native herbivorous fish in the Mediterranean: an integrated study based on gut-content analysis and stable isotope signatures. *Journal of the Marine Biological Association of the United Kingdom*, **87**, 991–998.
- Ballesteros E. (1989) Production of seaweeds in Northwestern Mediterranean marine communities: its relation with environmental factors. *Scientia Marina*, **53**, 357–364.
- Ballesteros E., Garrabou J., Hereu B., Zabala M., Cebrian E., Sala E. (2009) Deep-water stands of *Cystoseira zosteroides* C. Agardh (Fucales, Ochrophyta) in the Northwestern Mediterranean: insights into assemblage structure and population dynamics. *Estuarine Coastal and Shelf Science*, **82**, 477–484.
- Bardamaskos G., Tsiamis K., Panayotidis P., Megalofonou P. (2008) New records and range expansion of alien fishes and macroalgae in Greek waters (southeast Ionian Sea). *Journal of Marine Biological Association – Biodiversity Records* **2**, www.mba.ac.uk/jmba/pdf/6361.
- Bariche M. (2006) Diet of the Lessepsian fishes, *Siganus rivulatus* and *S. luridus* (Siganidae) in the eastern Mediterranean: a bibliographic analysis. *Cybium*, **30**, 41–49.
- Bariche M., Letourneur Y., Harmelin-Vivien M. (2004) Temporal fluctuations and settlement patterns of native and Lessepsian herbivorous fishes on the Lebanese coast (Eastern Mediterranean). *Environmental Biology of Fishes*, **70**, 81–90.
- Ben-Tuvia A. (1964) Two siganid fishes of Red Sea Origin in the Eastern Mediterranean. *Bulletin of the Sea Fisheries Research Station, Haifa*, **37**, 3–9.
- Bouchon-Navaro Y., Harmelin-Vivien M. (1981) Quantitative distribution of herbivorous reef fishes in the Gulf of Aqaba (Red Sea). *Marine Biology*, **63**, 79–86.
- Castriota L., Andaloro F. (2005) First record of the Lessepsian fish *Siganus luridus* (Osteichthyes: Siganidae) in the Tyrrhenian Sea. *Journal of Marine Biological Association – Biodiversity Records*, **2**, 5122.
- Clarke R.K., Gorley R.N. (2006) *PRIMER v6: User Manual/ Tutorial*. PRIMER-E, Plymouth: 10 pp.
- Clarke R.K., Somerfield P.J., Gorley R.N. (2008) Testing of null hypotheses in exploratory community analyses: similarity profiles and biota-environment linkage. *Journal of Experimental Marine Biology and Ecology*, **366**, 57–69.
- Corsini-Foka M., Economidis P.S. (2007) Allochthonous and vagrant ichthyofauna in Hellenic marine and estuarine waters. *Mediterranean Marine Science*, **8**, 79–101.
- Daniel B., Piro S., Charbonnel E., Francour P., Letourneur Y. (2009) Lessepsian rabbitfish *Siganus luridus* reached the French Mediterranean coasts. *Cybium*, **33**, 163–164.
- Diamant A., Ben-Tuvia A., Baranes A., Golani D. (1986) An analysis of rocky coastal Eastern Mediterranean fish assemblages and a comparison with an adjacent small artificial reef. *Journal of Experimental Marine Biology and Ecology*, **97**, 269–285.
- DLR-EOWEB 2012 <http://eoweb.dlr.de:8080/servlets/template/> (accessed March 2012).
- Dulcic J., Dragicevic B., Crgicevic R., Lipej L. (2011) First substantiated record of a Lessepsian migrant – The dusky spinefoot, *Siganus luridus* (Actinopterygii: Perciformes: Siganidae) in the Adriatic Sea. *Acta Ichthyologica et Piscatoria*, **41**, 141–143.
- Galil B.S. (2007) Loss or gain? Invasive aliens and biodiversity in the Mediterranean Sea *Marine Pollution Bulletin*, **55**, 314–322.
- Gambi M.C., Barbieri F., Bianchi C.N. (2009) New record of the alien seagrass *Halophila stipulacea* (Hydrocharitaceae) in the western Mediterranean: a further clue to changing Mediterranean Sea biogeography. *Journal of the Marine Biological Association 2 – Marine Biodiversity Records*, **e84**, doi:10.1017/S175526720900058X
- Giakoumi S., Cebrian E., Kokkoris G.D., Ballesteros E., Sala E. (2012) Relationships between fish, sea urchins and macroalgae: the structure of shallow sublittoral communities in the Cyclades, Eastern Mediterranean. *Estuarine, Coastal and Shelf Science*, **109**, 1–10.
- Gofas S., Zenetos A. (2003) Exotic molluscs in the Mediterranean basin: current status and perspectives. *Oceanography and Marine Biology*, **41**, 237–277.
- Golani D. (1998) Distribution of Lessepsian migrant fish in the Mediterranean. *Italian Journal of Zoology*, **65**, 95–99.
- Golani D. (2010) Colonization of the Mediterranean by Red Sea fishes via the Suez Canal – Lessepsian migration. In: Golani D., Appelbaum-Golani B. (Eds), *Fish Invasions of the Mediterranean Sea: Change and Renewal*. Pensoft Publishers, Sofia-Moscow: 145–188
- Goren M., Galil B.S. (2001) Fish biodiversity in the Vermetid reef of Shiqmona (Israel). *Marine Ecology*, **22**, 369–378.
- Goren M., Galil B.S. (2005) A review of changes in the fish assemblages of Levantine Inland and marine ecosystems following the introduction of non-native fishes. *Journal of Applied Ichthyology*, **21**, 364–370.
- Gruvel A. (1931) *Les Etats De Syrie. Richness Marines Et Fluviales. Exploitation Actuelle, Avenir*. Societe d'editions Geographiques, Marines et Coloniales, Paris: pp. 72–134.
- Harmelin-Vivien M.L., Bitar G., Harmelin J.-G., Monestiez P. (2005) The littoral fish community of the Lebanese rocky coast (eastern Mediterranean Sea) with emphasis on Red Sea migrants. *Biological Invasions*, **7**, 625–637.

- Hassan M., Harmelin-Vivien M., Bonhomme F. (2003) Lessepsian invasion without bottleneck: example of two rabbitfish species (*Siganus rivulatus* and *Siganus luridus*). *Journal of Experimental Marine Biology and Ecology*, **291**, 219–232.
- Kalogirou S., Corsini Foka M., Sioulas A., Wennhage H., Pihl L. (2010) Diversity, structure and function of fish assemblages associated with *Posidonia oceanica* beds in an area of the eastern Mediterranean Sea and the role of non-indigenous species. *Journal of Fish Biology*, **77**, 2338–2357.
- Kalogirou S., Wennhage H., Pihl L. (2012) Non-indigenous species in Mediterranean fish assemblages: contrasting feeding guilds of *Posidonia oceanica* meadows and sandy habitats. *Estuarine, Coastal and Shelf Science*, **96**, 209–218.
- Katsanevakis S. (2011) Rapid assessment of the marine alien megabiota in the shallow coastal waters of the Greek islands, Paros and Antiparos, Aegean Sea. *Aquatic Invasions*, **6** (Suppl. 1), S133–S137.
- Katsanevakis S., Tsiamis K. (2009) Records of alien marine species in the shallow coastal waters of Chios Island (2009). *Mediterranean Marine Science*, **10**, 99–107.
- Kruskal J.B., Wish M. (1978) *Multidimensional scaling*. Sage University paper series on Quantitative Applications in the Social Sciences No11. Beverly Hills, Sage: 95 pp.
- Ktari-Cha Kroun F., Bouhhal M. (1971) Record of *Siganus luridus* (Rüppel) in the Gulf of Tunisia. *Bulletin de l'Institut National Scientifique et Technique d'Océanographie et de Pêche de Salammbô*, **2**, 49–52.
- La Messa G., Vacchi M. (1999) An analysis of the coastal fish assemblage of the Ustica Island Marine Reserve (Mediterranean Sea). *Marine Ecology*, **20**, 147–165.
- Lejeune C., Chevaldonne P., Pergent-Martini C., Boudouresque C.F., Perez T. (2010) Climate change effects on a miniature ocean: the highly diverse, highly impacted Mediterranean Sea. *Trends in Ecology & Evolution*, **25**, 250–260.
- Lundberg B., Golani D. (1995) Diet adaptations of Lessepsian migrant rabbitfishes, *Siganus luridus* and *S. rivulatus*, to the algal resources of the Mediterranean coast of Israel. *Marine Ecology*, **16**, 73–89.
- Lundberg B., Ogorek R., Galil B.S., Goren M. (2004) Dietary choices of siganid fish in Shiqmona reef, Israel. *Israel Journal of Zoology*, **50**, 39–53.
- Micheli F., Benedetti-Cecchi L., Gambaccini S., Bertocci I., Borsini C., Osio G. C., Roman F. (2005) Cascading human impacts, marine protected areas, and the structure of Mediterranean reef assemblages. *Ecological Monographs*, **75**, 81–102.
- Nicolaidou A., Alongi G., Aydogan O., Catra M., Cavas L., Cevik A., Dosi A., Cirosta V., Giakoumi S., Gimenez-Casaldueiro F., Filiz H., Izquierdo-Munoz A., Kalogirou S., Konstantinidis E., Kousteni V., Kout J., Legaki A., Megalofonou P., Ovalis P., Paolillo G., Paschos I., Perdikaris C., Poursanidis D., Ramos-Espla A.A., Reizopoulou S., Sperone E., Taskin E., Triperi S., Vazquez-Luis M. (2012) New Mediterranean Biodiversity Records (June 2012). *Mediterranean Marine Science*, **13**, 36–48.
- Pages J.F., Farina S., Gera A., Arthur R., Romero J., Alcoverro T. (2012) Indirect interactions in seagrasses: fish herbivores increase predation risk to sea urchins by modifying plant traits. *Functional Ecology*, **26**, 1015–1023.
- Preskitt L.B., Vroom P.S., Smith C.M. (2004) A rapid ecological assessment (REA) quantitative survey method for benthic algae using photoquadrats with Scuba. *Pacific Science*, **58**, 201–209.
- Poore A.G.B., Campbell A.H., Coleman R.A., Edgar G.J., Jormalainen V., Reynolds P.L., Sotka E.E., Stachowicz J.J., Taylor R.B., Vanderklift M.A., Duffy E.J. (2012) Global patterns in the impact of marine herbivores on benthic primary producers. *Ecology Letters*, **15**, 912–922.
- Por F.D. (1978) Lessepsian migration. The influx of Red Sea biota into Mediterranean by way of the Suez Canal. In: W.D. Billings et al. (Eds), *Ecological Studies, Analysis and Synthesis* 23. Springer-Verlag, Berlin: 1–228.
- Potter I.C., Bird D.J., Claridge P.N., Clarke K.R., Hyndes G.A., Newton L.C. (2001) Fish fauna of the Severn Estuary. Are there long-term changes in abundance and species composition and the recruitment patterns of the main marine species correlated? *Journal of Experimental Marine Biology and Ecology*, **258**, 15–37.
- Raitsos D.E., Beaugrand G., Georgopoulos D., Zenetos A., Pancucci-Papadopoulou A.M., Theocharis A., Papatthanassiou E. (2010) Global climate change amplifies the entry of tropical species into the Eastern Mediterranean Sea. *Limnology and Oceanography*, **55**, 1478–1484.
- Rilov G., Galil B. (2009) Marine bioinvasions in the Mediterranean Sea – History, distribution and ecology. In: Rilov G., Crooks J.A. (Eds), *Biological Invasions in Marine Ecosystems*. Springer, Berlin: pp 549–575
- Sala E., Boudouresque C.F. (1997) The role of fishes in the organization of a Mediterranean sublittoral community I: algal communities. *Journal of Experimental Marine Biology and Ecology*, **212**, 25–44.
- Sala E., Boudouresque C.F., Harmelin-Vivien M. (1998) Fishing, trophic cascades, and the structure of algal assemblages: evaluation of an old but untested paradigm. *Oikos*, **82**, 425–439.
- Sala E., Kizilkaya Z., Yildirim D., Ballesteros E (2011) Alien marine fishes deplete algal biomass in the eastern Mediterranean. *PLoS ONE*, **6**, e17356.
- SoHelME (2005) *State of the Hellenic Marine Environment*. Papatthanassiou E., Zenetos A. (Eds). HCMR Publ., Athens, Greece: 360 pp.
- StatSoft Inc. (2004) *STATISTICA (data analysis software system), Version 7*. www.statsoft.com
- Stergiou K. (1988) Feeding habits of the Lessepsian migrant *Siganus luridus* in the eastern Mediterranean, its new environment. *Journal of Fish Biology*, **33**, 531–543.

- [Verges A., Alcoverro T., Ballesteros E. \(2009\) Role of fish herbivory in structuring the vertical distribution of canopy algae *Cystoseira* spp. in the Mediterranean Sea. *Marine Ecology Progress Series*, **375**, 1–11.](#)
- Zenetos A., Gofas S., Verlaque M., Cinar M.E., Garcia Raso J.E., *et al.* (2010) Alien species in the Mediterranean Sea by 2010 A contribution to the application of European Union's marine strategy framework directive (MSFD). Part I. Spatial distribution. *Mediterranean Marine Science*, **11**, 381–493.
- [Zenetos A., Katsanevakis S., Poursanidis D., Crocetta F., Damalas D., Apostolopoulos G., Gravili C., Vardalla-Theodorou E., Malaquias M. \(2011\) Marine alien species in Greek Seas: additions and amendments by 2010. *Mediterranean Marine Science*, **12**, 95–120.](#)
- [Zenetos A., Gofas S., Morri C., Rosso A., Violanti D., Garcia Raso J.E., Cinar M.E., Almogi-Labin A., Ates A.S., Azzurro E., Ballesteros E., Bianchi C.N., Bilecenoglu M., Gambi M.C., Giangrande A., Gravili C., Hyams-Kaphzan O., Karachle P.K., Katsanevakis S., Lipej L., Mastrototaro F., Mineur F., Pancucci-Papadopoulou M.A., Ramos Espla A., Salas C., San Martin G., Sfrizo A., Streftaris N., Verlaque M. \(2012\) A contribution to the application of European Union's marine strategy framework directive \(MSFD\). Part 2. Introduction trends and pathways. *Mediterranean Marine Science*, **13/2**, 328–352.](#)