



Recent changes on the abundance and distribution of non-indigenous macroalgae along the southwest coast of the Bay of Biscay

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ABSTRACT

Twenty-three rocky shores along approximately 225 km on the southwest coast of the Bay of Biscay were sampled during the springs of 2014 and 2021, to explore changes in the distribution and abundance of four non-indigenous species (NIS) macroalgae (i.e., *Asparagopsis armata*, *Grateloupia turuturu*, *Sargassum muticum*, and *Undaria pinnatifida*) by using a semi-quantitative scale. Results showed relevant changes in the distribution and abundance of NIS. The kelp *U. pinnatifida* was recorded in 2021 for the first time on two shores. The distribution of *G. turuturu* showed an extension in its range of distribution of 200 km to the east. The other two target species *S. muticum* and *A. armata* were widely distributed along the whole 225 km of the studied area in 2014 and 2021, with higher abundance in 2021. Therefore, we strongly advise the necessity of future monitoring programs for these four NIS species. These monitoring programs will explore the progress of invasion and resilience of native species.

1. Introduction

The introduction of non-indigenous species (NIS) is one of the major threats to biodiversity conservation (Pimentel et al., 2005). Non-indigenous species normally compete with local species which may result in the loss of native diversity and niche contractions (Guerra-García et al., 2012). About 41% of the NIS that concerns the European Commission due to their invasive potential are marine macroalgae (EASIN, 2019). Different types of shores (e.g. rocky shores, sandy shores, marinas...) are favored by NIS macroalgae that are able to modify the complexity of the habitat and cause an impact on the marine communities, species composition, ecosystem services, and ecological interactions (Schaffelke et al., 2006; Tait et al., 2015; Vaz-Pinto et al., 2014).

Marine macroalgae assemblages are key elements for ecosystem functioning because they increase habitat complexity, modify the hydrodynamic conditions and sedimentary processes, and also provide food and shelter for many organisms (Christie et al., 2009; Gee and Warwick, 1994; Venier et al., 2012). Abiotic (e.g. temperature,

hydrodynamic conditions, light, chemical conditions), biotic (e.g. primary consumers' assemblages), and anthropic factors (e.g. distance to aquaculture facilities, commercial ports, population density) determine the distribution of native and NIS macroalgae, thus controlling the community composition (Báez et al., 2010; Blanco et al., 2021). The arrival of NIS macroalgae causes changes in ecosystem functioning, affecting the macroalgal biomass and productivity by reducing primary production (Salvaterra et al., 2013; Sánchez et al., 2005). Invasions by NIS macroalgae are also frequently associated with the replacement of native macroalgae species, increasing filamentous epiphytic algae and changing the composition of native assemblages (Sánchez and Fernández, 2005), affecting the patterns of abundance, distribution, and structure of higher trophic levels (Cacabelos et al., 2010; Rubal et al., 2018; Veiga et al., 2014a).

Our general understanding of the distribution of NIS macroalgae is limited and biased due to the lack of geographically continuous information. The available data is limited to non-continuous geographical areas, resulting in several gaps in their known distribution. Besides, most of the studies do not take into account the same NIS macroalgae,

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making comparisons even more challenging (Araújo et al., 2016). Therefore, to understand changes and trends in the abundance and distribution of NIS macroalgae, a need for mid and long-term monitoring studies arises to provide baseline data to assess NIS impacts, invasion dynamics, and to verify predictive models (Báez et al., 2010; Blanco et al., 2021; Cacabelos et al., 2010).

The management of NIS to enhance the “Good Environmental Status” is supported by European regulations like the European Marine Strategy Framework Directive (MSFD; 2008/56/EC) or Water Framework Directive (WFD; 2000/60/EC) (Airoldi and Beck, 2007; Danovaro et al., 2016; Ramos et al., 2017; Rice et al., 2012). However, such management is limited due to the lack of information in some regions (Blanco et al., 2021). This is the case of the SW Bay of Biscay, where the presence of NIS macroalgae has been poorly studied. In this particularly relevant area, where distribution overlap of cold and warm species occurs (Fernández, 2016; Fischer-Piette, 1955), there are only a few records available that come from broader studies, usually not designed specifically to provide accurate data on the presence, distribution, and abundance of NIS macroalgae (Cacabelos et al., 2013; Cremades et al., 2006; Peteiro, 2008). More than 57% of NIS macroalgae along the Iberian Peninsula coast have been recorded on the Atlantic NW coast (ICES, 2009), and these populations might likely extend their range of distribution to the SW Bay of Biscay. Therefore, in this area, subtle variations in the environmental conditions as a consequence of climate change could modify the limit of distribution of some native and NIS macroalgae species.

This study aims to explore changes after seven years in the abundance and distribution of the most easily in-situ identifiable widespread NIS macroalgae in the SW of the Bay of Biscay: *Asparagopsis armata* Harvey 1855, *Grateloupia turuturu* Y. Yamada 1941, *Sargassum muticum* (Yendo) Fensholt 1955, and *Undaria pinnatifida* (Harvey) Suringar 1873 (Báez et al., 2010; Bárbara and Cremades, 2004; Cacabelos et al., 2013; Cremades et al., 2006).

2. Materials and methods

2.1. Study area

The SW part of the Bay of Biscay is limited West by Cape Ortegal (43°46'17.2"N 7°52'10.1"W) and has its limit in the Ria de Ribadeo (43°32'02.0"N 7°01'59.9"W). SW Bay of Biscay is included in the warm-temperate area of the Atlantic Region (Alvarez et al., 1988). This area is affected by mesoscale phenomena such as the Iberian Poleward Current that modifies hydrographic properties of the seawater (Haynes and Barton, 1990; Isla and Anadón, 2004) and contributes to the summer thermal gradient along the coast of the Bay of Biscay, where the sea

surface temperature (SST) of the inner part of the Bay gets hotter than the western and northern SST because of the continental influence (García et al., 2011).

This study was carried out on 23 rocky shores between the localities of Cariño (43°44'39.07"N, 7°52'0.42"W), the westernmost limit of the Bay of Biscay, and Ribadeo (43°33'13.41"N, 007°02'39.14"W) which represent the NE border of the Upper Galician Rias (Ramos et al., 2012) (Fig. 1). Of the total 225 km of the coast, 145 are rocky shores, whereas 81 are sandy beaches. The wave regime is dominated by swell from the NW, and most storms occur during autumn and winter. This area shows a semidiurnal mesotidal regime, with the largest tidal range during spring tides of 3.5–4.5 m. During high tide, the Upper Rias presents a mean annual water temperature of 15.26 ± 1.58 °C, with a maximum daily oscillation of 2.1 °C (Cacabelos et al., 2013).

2.2. Target species

Asparagopsis armata (Rhodophyta) is native to southern Australia and New Zealand and it has become widely distributed along European shores since its first record in 1925 (Andreakis et al., 2007; Guiry and Guiry, 2023). It was introduced in the Iberian Peninsula through the Strait of Gibraltar in the 80 s (South and Tittley, 1986) but it had been previously recorded in 1933 in the Ria de Pontevedra (NW Spain) (Miranda, 1934). It is widely distributed on the Mediterranean Iberian coast and along the Atlantic coast to the Bay of Biscay (Díez et al., 2012; García et al., 2011; Guerra-García et al., 2012; Ramos et al., 2020). *Asparagopsis armata* shows a three-phase life-cycle (Guiry and Guiry, 2023), with a free-floating tetrasporophyte (= *Falkenbergia rufolanosa* F. Schmitz (Harv.)) that increases its propagation ability by fragmentation (Guiry and Guiry, 2023; Pinteus et al., 2018). Additionally, due to their capability to produce toxic substances (brominated compounds and acetones) and its unpalatability, this species lacks predators, which increases its invasive capacity showing conspicuous increases in biomass (Guiry and Guiry, 2023; Pinteus et al., 2018; Sala and Boudouresque, 1997).

The foliose irregular macroalga *Grateloupia turuturu* (Rhodophyta) is native to Japan and was first recorded in Europe in 1969 (Farnham and Irvine, 1973), associated with the aquaculture of the oyster *Magallana gigas* (Thunberg, 1793) (Eno et al., 1997). Since then, it spread along the European Atlantic coasts (Cabioch et al., 1997; Eno et al., 1997) and the western Mediterranean (Giaccone et al., 1985; Verlaque, 2001). The first report on the Iberian Peninsula dates back to the early 1990 s (Araújo et al., 2011; Bárbara and Cremades, 2004), and it has been frequently reported in recent studies (Blanco et al., 2021; Montes et al., 2017; Rubal et al., 2011). *Grateloupia turuturu* exhibits a trigenetic life cycle, including two independent isomorphic reproductive phases (Araújo

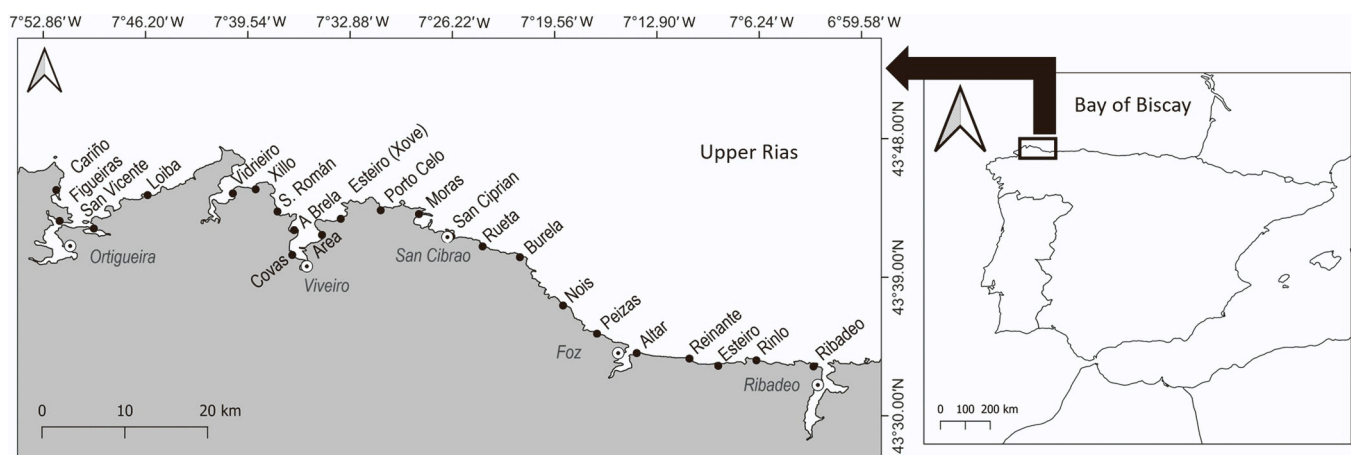


Fig. 1. Map of the Southwest coast of the Bay of Biscay showing the location of the 23 studied rocky shores.

et al., 2011; Bárbara and Cremades, 2004).

The macroalga *Sargassum muticum* (Phaeophyta) is native to Japan and was first detected in Europe in 1973 on the British coast (Farnham et al., 1973) arriving in the western part of the Bay of Biscay in 1989 (Pérez-Cirera et al., 1989). *Sargassum muticum* became one of the most successful NIS macroalgae (Cacabelos et al., 2013) because it can grow in different kinds of substrates, from unstable intertidal rocky boulders where other macrophytes are not able to attach, rockpools where it displaces native seaweeds, to the shallow sublittoral of sheltered areas (Fernández, 2020, 1999). This species is characterized by a pseudoperennial marked seasonal cycle (Fernández, 1999) that consists of one period of slow growth (October–March), another of fast growth (March–May) followed by senescence in the summer (Arenas et al., 1995).

The kelp *Undaria pinnatifida* (Phaeophyta) is native to China, Korea and Japan, and in the past decades, has expanded its distribution worldwide (Cremades et al., 2006; Epstein and Smale, 2017). It was first detected on the NW coast of the Iberian Peninsula in 1988 (Cremades, 1995) and then in 1995 in the inner part of the Bay of Biscay (Salinas et al., 1996). *Undaria pinnatifida* was cultivated around Santander (northern Iberian Peninsula) and the Ria de Ares (northwest Iberian Peninsula) in the early 2000 s (Báez et al., 2010; Cremades et al., 2006; Peteiro, 2008). Nowadays, *U. pinnatifida* has spread all over the NW Atlantic coast of the Iberian Peninsula to the English Channel (Araújo et al., 2016; Blanco et al., 2021; Epstein and Smale, 2017; Veiga et al., 2014b). It is now considered a successful invasive seaweed on the European Atlantic coast because of its great ability to colonize artificial substrates, disturbed habitats (high turbidity or eutrophicated), and oyster or mussel shells (ICES, 2007). *Undaria pinnatifida* has an annual heteromorphic life cycle, with microscopic gametophytes that may also act as seed banks and macroscopic sporophytes (Cremades et al., 2006).

2.3. Sampling strategy

The studied area was divided into consecutive zones of 2 km. When rocky shores were present and accessible within these zones, one rocky shore was selected and sampled. A total of 23 rocky shores were sampled during the spring low tides (April–May) in 2014 and 2021 (Appendix A). Spring was selected because the targeted NIS macroalgae present greater sizes and densities (Arenas et al., 1995; Bárbara and Cremades, 2004; Cremades, 1995; Guerra-García et al., 2012). At each shore, the distribution and abundance of *A. armata*, *G. turuturu*, *S. muticum*, and *U. pinnatifida* were assessed by recording the presence/absence along a 100 m transect parallel to the shore between the low tide to the mid-tide level at each shore. When target species were present, their abundance

was quantified using a modification of the categorical abundance scale MNCR SACFOR (Connor et al., 2004) considering 6 levels: 1) absent (no presence of NIS); 2) rare (NIS individuals solitary or loosely distributed along the transect with a total cover of <1%); 3) occasional (NIS individuals restricted to a minor part of the transect forming patches but total covering not exceeding 19%); 4) frequent (NIS individuals distributed along the transect, with a total cover of 20–39%); 5) abundant (NIS individuals distributed along the transect forming patches, with a total cover of 40–79%); and 6) dominant (NIS individuals distributed along all the transect and clearly dominating the surface, with a total cover of >80%) (Fig. 2). This semi-quantitative approach allowed us to cover a wide area in a short period, avoiding excessive temporal variability among studied shores.

2.4. Data analyses

The variation of the abundance of NIS macroalgae on the SW Bay of Biscay was represented in a map built based on the shape file downloaded from “Xeportal / Infraestructura de Datos Espaciales de Galicia” (<https://mapas.xunta.gal>) representing the study area coastline. The final maps were built under QGIS 3.14 environment (QGIS Development Team, 2022), and an estimated coastline of 225 km was considered (length function applied on the whole shape file).

A Score Variation Change (Sc) between 2014 and 2021 was calculated to explore potential changes in each NIS macroalgae abundance. A value of 0 was assigned to absent, 1 to rare, 2 to occasional, 3 to frequent, 4 to abundant, and 5 to dominant, and a simple score subtraction (Sc2021–Sc2014) was calculated and represented. Moreover, based on the previous scores’ assignments, the frequency, as a percentage of shores that presented each score in 2014 and 2021, was calculated. Finally, to explore significant differences between different score frequencies between the two sampling dates (i.e. 2014 vs. 2021) a two independent sampled Chi-squared test (χ^2) was calculated, using a statistical significance alpha of 0.05. However, the Chi-squared test was not possible to calculate for *G. turuturu* and *U. pinnatifida* due to zero inflation.

3. Results

3.1. *Asparagopsis armata*

Asparagopsis armata showed differences in presence and abundance between 2014 and 2021. It was absent on 12 shores in 2014 but in 2021 it was absent only on seven shores. In 2021, *A. armata* became abundant on seven shores whereas it was frequent on four shores in 2014 (Fig. 3

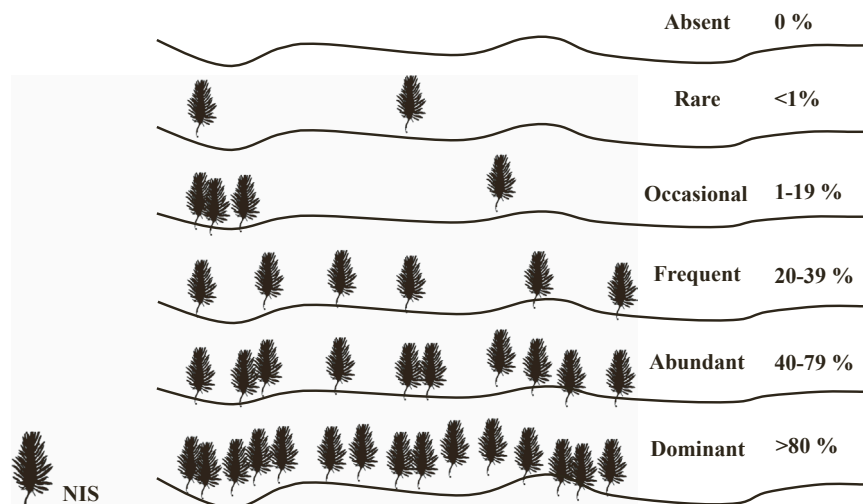


Fig. 2. Presence-Absence categorical scale (% of cover) used to evaluate the abundance of targeted macroalgal NIS. The solid line represents the 100 m-long transect.



A). Regarding the Score Variation Change, the abundance increased in 12 shores, eight kept the same value and abundance decreased in three. The Score Variation Change of *A. armata* was high, reaching four points on one shore and three on two shores (Fig. 3 B). The frequencies of abundance between 2014 and 2021 also showed significant differences ($\chi^2 = 9.802$, d.f. = 4, $p = 0.04$). In 2014, *A. armata* was absent on 50% of the shores and frequent on 20%. However, in 2021 it was absent only on 30% of the shores and it became abundant on 30% (Fig. 3 C).

In 2014, the distribution of *G. turuturu* was limited to the Ria de Ortigueira, appearing as rare on two shores and as frequent only on one. In 2021, this species appeared again in the Ria de Ortigueira, becoming abundant on two shores but disappearing on another two. Furthermore, *G. turuturu* increased its distribution 200 km to the east, becoming rare on three shores where it was not observed in 2014 (Fig. 4 A). The Score Variation Change of *G. turuturu* was very low, only reaching one positive point on five shores and one negative point on one shore (Fig. 4 B). In 2014, *G. turuturu* was absent on 90% of the shores; however, in 2021 was

S. muticum was widespread along the studied area, being present on 20 and 21 rocky shores in 2014 and 2021, respectively (Fig. 5 A). In 2021, *S. muticum* became abundant on seven rocky shores in comparison to only three in 2014, and dominant on four rocky shores (three in 2014). Regarding the Score Variation Change, abundance increased in 11 shores, seven kept the same value and the abundance score decreased in five (Fig. 5 B). The Score Variation Change of *S. muticum* was low, reaching a maximum score of two positive points on four shores, one positive point on seven shores, and three negative points only on one shore (Fig. 5 B). *Sargassum muticum* was frequent in 2014 on 50% of the shores but in 2021 became abundant on 35% and dominant on 20% of the sampled sites (Fig. 5 C). Despite this apparent increase in the abundance of *S. muticum*, the Chi-square formal analysis of the frequencies indicated that there were no significant differences ($\chi^2 = 8.457$, d.f. = 5, $p = 0.13$) between both sampling dates.

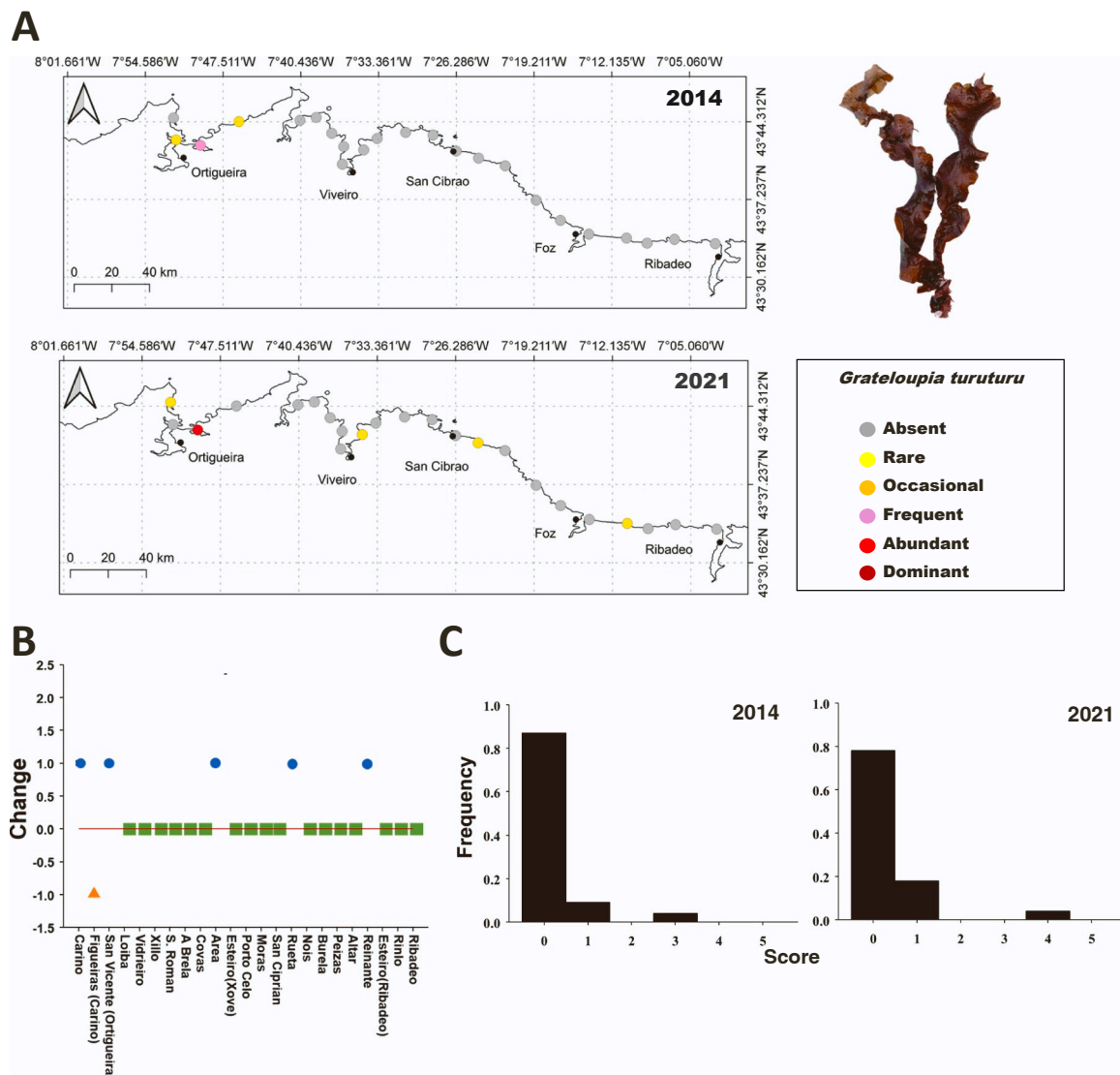


Fig. 4. A) Abundance data of *Grateloupia turuturu* along the SW Bay of Biscay in 2014 and 2021. B) Score Variation Change (Sc2021-Sc2014) of *G. turuturu* in the SW Bay of Biscay. Blue dots: abundance increase; green squares: no changes in abundance; brown triangles: abundance decrease. C) Frequency of shores based on abundance score (0-Absent, 1-Rare, 2-Occasional, 3-Frequent, 4-Abundant, and 5-Dominant) of *G. turuturu* in the SW Bay of Biscay in 2014 and 2021.

3.4. *Undaria pinnatifida*

In 2021, two populations of *U. pinnatifida* were found between Ria do Barqueiro and Ria de Viveiro. However, this species was absent in 2014. This finding of populations of *U. pinnatifida* on the shores of Xilloi (frequent) and San Roman (occasional) represents the first known reported observations of the species in the SW of the Bay of Biscay (Fig. 6 A and B). In 2014, *U. pinnatifida* was absent on all the shores and in 2021 it was present on 9% of the shores (Fig. 6 C).

4. Discussion

Our results showed that *U. pinnatifida* is now present at the SW Bay of Biscay, with its first record in 2021. Moreover, *G. turuturu* has expanded its distribution range to the East, now being present on three additional shores spanning 200 km from the 2014 records. Finally, *S. muticum* and *A. armata* are well established along the 225 km of the study area with stable populations even showing increasing abundances.

Asparagopsis armata is widespread along the northern Iberian Peninsula (Blanco et al., 2021). Negative impacts of *A. armata* both on native assemblages of invertebrates and macroalgae have been

described in the Iberian Peninsula (Rubal et al., 2018; Silva et al., 2021). Our results showed an increase in the abundance of this NIS in the SW Bay of Biscay, which aligns with results by Ramos et al. (2020) who reported an increase in its abundance on the northern shores of the Iberian Peninsula. Ramos et al. (2020) also suggested that in the central Bay of Biscay shores, the increase of *A. armata* abundance could be due to the disappearance of canopy algae that would facilitate the establishment of *A. armata*. Blanco et al. (2021) pointed out a medium-high probability of *A. armata* presence in the SW Bay of Biscay, thus reporting its presence in almost all shores sampled in the area. Therefore, and according to our results, this model fits with field observations. The presence of *A. armata* has been reported on several coasts of the Mediterranean Basin, North Atlantic, Macaronesia, etc (see Guiry and Guiry, 2023). However, in most of these localities, there are no studies of the abundance evolution of the invasion of *A. armata*. Therefore, considering the impacts of *A. armata* in native assemblages and its increased abundance and distribution along the SW Bay of Biscay, monitoring their distribution and abundance dynamics is mandatory to disentangle the invasion trends and explore the local impacts on native assemblages.

Grateloupia turuturu is widely spread from the Ria de Ortigueira to northern Portugal (Araújo et al., 2011; Bárbara and Cremades, 2004;

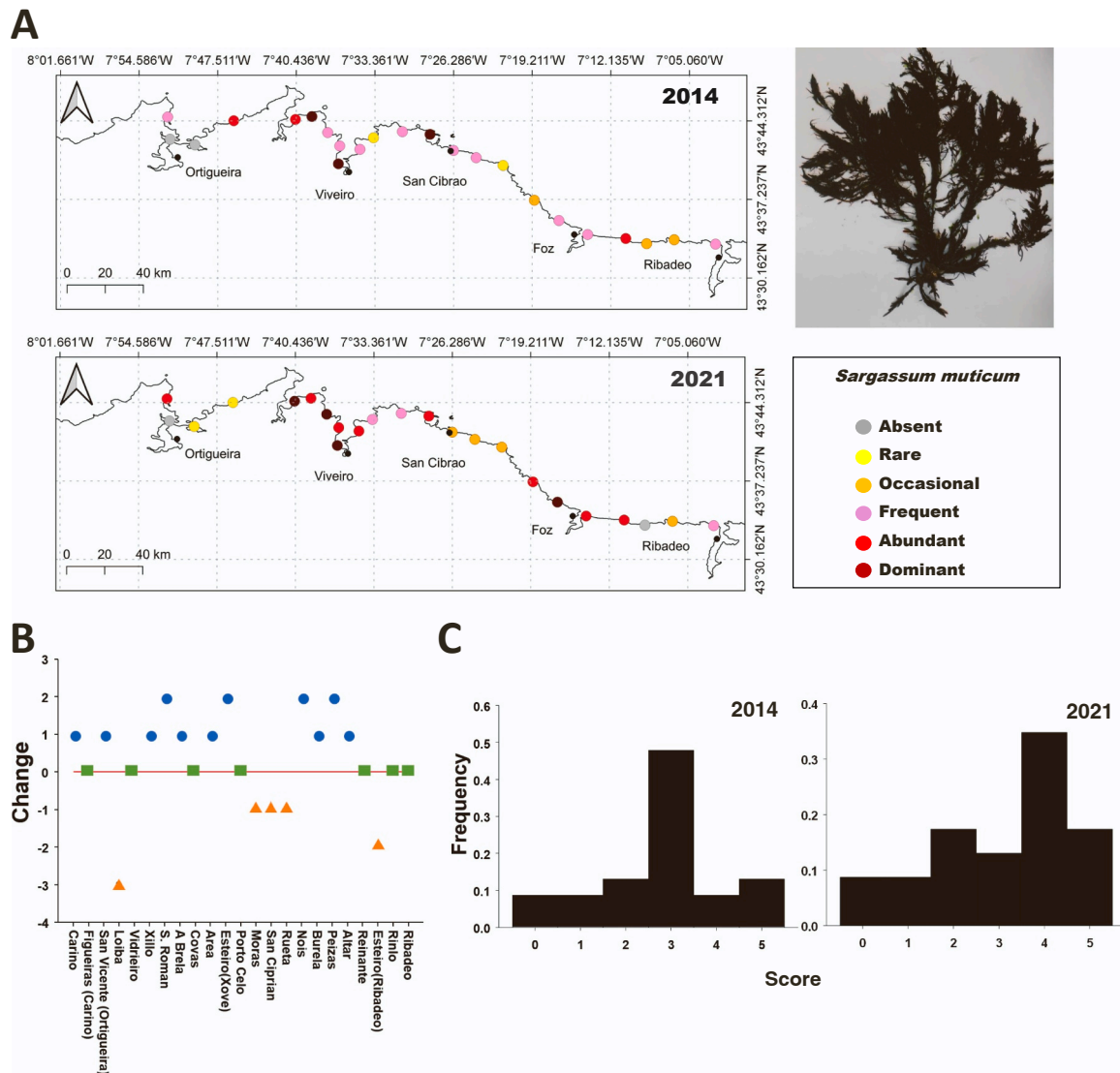


Fig. 5. A) Abundance data of *Sargassum muticum* along the SW Bay of Biscay in 2014 and 2021. B) Score Variation Change (Sc2021-Sc2014) of *S. muticum* in the SW Bay of Biscay. Blue dots: abundance increase; green squares: no changes in abundance; brown triangles: abundance decrease. C) Frequency of shores based on abundance score (0-Absent, 1-Rare, 2-Occasional, 3-Frequent, 4-Abundant, and 5-Dominant) of *S. muticum* in the SW Bay of Biscay in 2014 and 2021.

Cremades et al., 2006; Rubal et al., 2011). Since its first record in the Ria de Ortigueira in the 1990 s (Cremades, 1995), the populations of *G. turuturu* seemed restricted to this area at least until 2014. However, in 2021, the distribution of *G. turuturu* has extended 200 km eastward, aligning with the predictions of Blanco et al. (2021). They indicated that the probability of *G. turuturu* presence in the SW Bay of Biscay was low or medium, which is consistent with our observations. Mulas and Bertocci (2016), demonstrated that in northern Portugal the expansion of *G. turuturu* relies on disturbances that removed native competitors. Therefore, the presence in 2021 of *G. turuturu* on shores where it was previously absent could be explained because of disturbance events. Despite the numerous records of *G. turuturu*, there is a general lack of information about its effects on native assemblages (but see Freitas et al., 2016; Janiak and Whitlatch, 2012). Similar to *A. armata*, the presence of *G. turuturu* has been reported along European Shores, North and South America shores, etc (see Guiry and Guiry, 2023). Once again, there are no studies that assess the evolution of *G. turuturu* invasion in most of those localities. Since the results of our study have shown an increase in the distribution of *G. turuturu*, further monitoring studies are necessary to explore the evolution of the invasion in the SW Bay of Biscay and its impact on native assemblages.

The invasion by *S. muticum* can result in changes in the diversity of native assemblages (Sánchez and Fernández, 2018; Veiga et al., 2014a) and shifts in the ecosystem functioning and trophic webs (Salvaterra et al., 2013). However, despite the lack of significant changes in the abundance of *S. muticum* between 2014 and 2021, their populations seem to slightly increase or at least remain stable. *S. muticum* fits a 'boom-bust' model, having decreased its abundance in the inner part of the Bay of Biscay in the last years (Fernández, 2020), on the contrary, the population at the SW part showed a stable/increasing abundance. Our results agree with, Fernández (2020), who reported that the biggest fertile plants of *S. muticum* appeared in the SW of the Bay of Biscay. Additionally, our findings align with Blanco et al. (2021) and Cacabelos et al. (2013), indicating that *S. muticum* is often found in rockpools in the SW Bay of Biscay. Besides, Blanco et al. (2021) pointed out that *S. muticum* is likely to be present in the nearby Ria de Foz, also agreeing with our observations. On the other hand, in the nearby areas of the Ria de Viveiro, Blanco et al. (2021) highlighted that the probability of the presence of *S. muticum* is low, but our results showed that this area had the highest values of abundance. The impact of *S. muticum* in the northern Iberian Peninsula has diminished in parallel to a decrease in its abundance (Fernández, 2020). Therefore, the future impact of

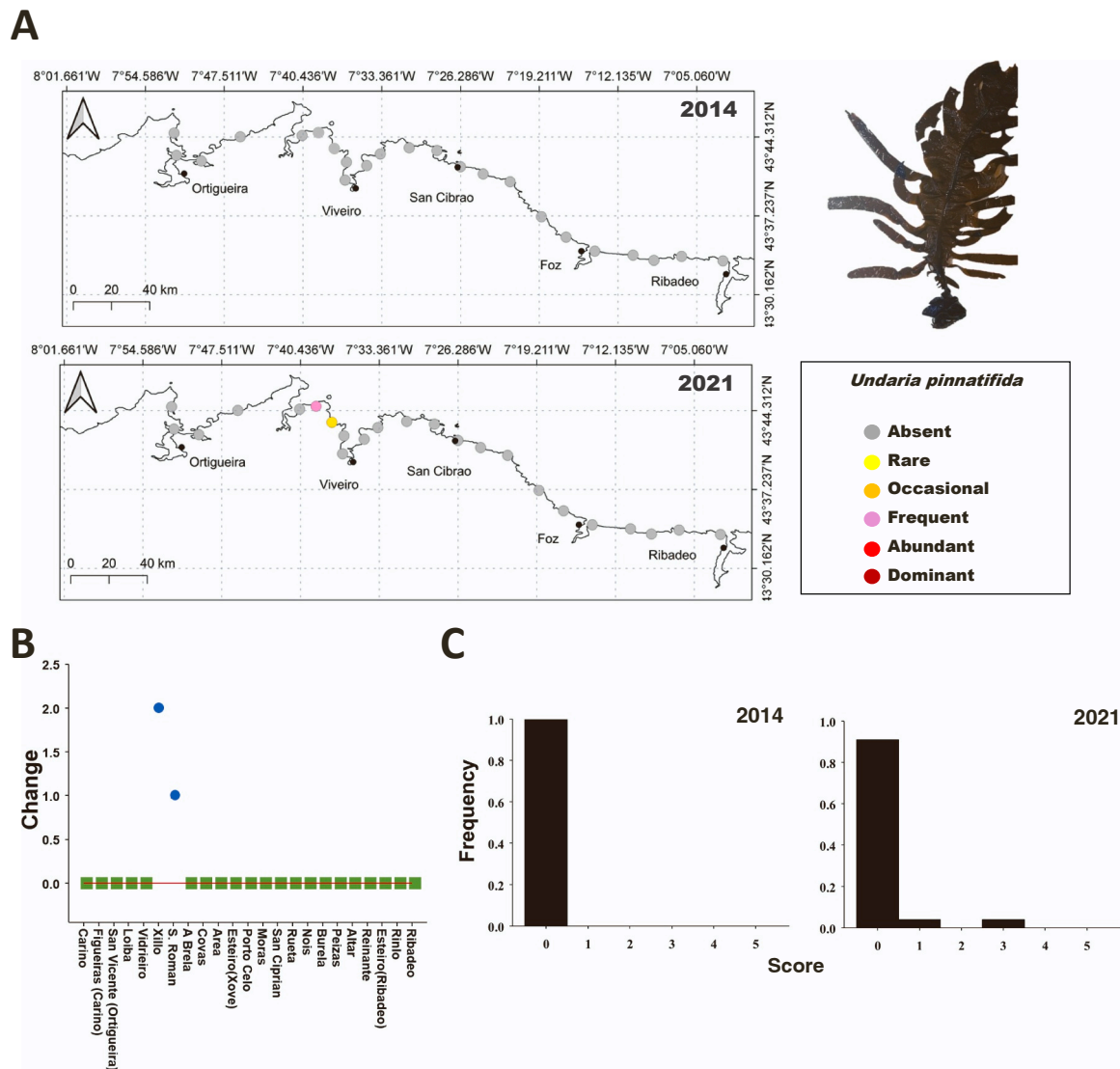


Fig. 6. A) Abundance data of *Undaria pinnatifida* along the SW Bay of Biscay in 2014 and 2021. B) Score Variation Change (Sc2021-Sc2014) of *U. pinnatifida* in the SW Bay of Biscay. Blue dots: abundance increase; green squares: no changes in abundance. C) Frequency of shores based on abundance score (0-Absent, 1-Rare, 2-Occasional, 3-Frequent, 4-Abundant, and 5-Dominant) of *U. pinnatifida* in the SW Bay of Biscay in 2014 and 2021.

S. muticum on native assemblages will be dependent on long-term changes in its abundance. Consequently, long-term monitoring studies are necessary for the SW Bay of Biscay, where *S. muticum* still shows high abundance, to assess the magnitude of its eventual impact on the native benthic assemblages.

The impact of *U. pinnatifida* on native assemblages has not been explored in the Iberian Peninsula; nevertheless, negative effects have been reported for other regions (Arnold et al., 2016; Casas et al., 2004; Epstein and Smale, 2017). Therefore, understanding the current extent of *U. pinnatifida* expansion is crucial to assess its potential for invasiveness and its consequences in the studied area. In this context, the model proposed by Baez et al. (2010) predicted that the SW Bay of Biscay is an unfavorable region for *U. pinnatifida* settlement. Blanco et al. (2021) also predicted that the presence of *U. pinnatifida* in the study area is improbable. However, Cremades et al. (2006) suggested that the area could be suitable for this NIS macroalgae but did not report the presence of *U. pinnatifida* in the SW part of the Bay of Biscay. Cremades et al. (2006) also reported that the expansion rate of *U. pinnatifida* is low (between 0.5 and 1.7 km per year in Galicia), suggesting that any increase in its distribution in the area would likely be slow. Our study reported the first record of *U. pinnatifida* in the SW Bay of Biscay and

confirmed that its expansion rate in this region is relatively slow, probably influenced by the absence of favorable anthropogenic vectors such as large ports or extensive aquaculture facilities. Taking into account the discrepancies between models and the field observations (e.g. Veiga et al., 2014b), it should be considered that predictive models did not take into account relevant biotic factors (Blanco et al., 2021). This highlights the need to complement these predictive models with field surveys calibration. Until now, along the West Iberian Peninsula, there is no evidence of displacement of native algae by *U. pinnatifida* in undisturbed habitats; this species is dominant only in strongly anthropic-disturbed habitats where other NIS macroalgae are also present (Cremades et al., 2006). Accordingly, *U. pinnatifida* is more common in artificial habitats such as ports and marinas (Araújo et al., 2016; Veiga et al., 2014b). In the SW Bay of Biscay, we observed *U. pinnatifida* only on two natural rocky shores, suggesting that its expansion in our study area depends on natural disturbances. Previous studies also suggest that *U. pinnatifida* may compete with other native kelps such as *Saccorhiza polyschides* (Lightfoot) Batters 1902, and that the arrival of *U. pinnatifida* displaces the sporophytes of this native macroalgae (Cremades et al., 2006) causing a reduction in its abundance. In agreement with Cremades et al. (2006) suggestions, the displacement of native

kelps by *U. pinnatifida* in natural reefs was reported in SW United Kingdom (Arnold et al., 2016). The increase in extremely hot days, resulting in higher temperature conditions in the last years (Lima and Wetthey, 2012), may be one of the drivers responsible for the recent retreat of cold temperate kelp and furoid species in the Bay of Biscay (Fernández, 2016, 2011). Loss of native kelps related to heatwaves could facilitate the settlement of *U. pinnatifida* as it could potentially occupy the niche left by the native kelps (*sensu* Leij et al., 2017). Given the capacity of *U. pinnatifida* as a pioneer species in disturbed habitats (Leij et al., 2017), along with the retreat of kelp forests in the Bay of Biscay, it becomes crucial to implement monitoring programs that assess the potential invasiveness of these NIS macroalgae in the mid and long term. Finally, it should be noted that this study has been carried out only in the intertidal realm; further subtidal sampling would help to assess the current distribution of *U. pinnatifida* in the area.

5. Conclusions

As demonstrated by the results of this study, which are in line with previously published records, invasive processes involving NIS macroalgae are dynamic, exhibiting frequent changes in abundance and distribution (Fernández, 2020). However, these changes do not always align with the expectations set by predictive models, reinforcing the need for monitoring programs that provide a real and updated picture of the distribution and abundance of NIS macroalgae on the field.

In summary, in seven years there has been an extension in the range of distribution of *G. turuturu* and *U. pinnatifida*, as well as a consistent trend in the increase of the abundance of *A. armata* and *S. muticum*. These results can be used as a baseline for future monitoring programs aimed to exploring the evolution of the invasion in the SW Bay of Biscay and its potential impact on native assemblages, with a specific focus on these four NIS species.

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

No data was used for the research described in the article.

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Appendix A. Coordinates of the shores and NIS abundances of 2014 and 2021. (Abs= Absent; R= Rare; Oc= Occasional; F = Frequent; Ab= Abundant; D = Dominant)

Shores		S. muticum		A. armata		G. turuturu		U. pinnatifida	
Name	Coordinates	2014	2021	2014	2021	2014	2021	2014	2021
Ribadeo	43°33'13.2"N 7°02'41.3"W	F	F	R	Ab	Abs	Abs	Abs	Abs
Rinlo	43°33'35.06"N 7° 6'23.65"W	Oc	Oc	R	R	Abs	Abs	Abs	Abs
Esteiro(Ribadeo)	43°33'19.1"N 7°08'51.1"W	Oc	Abs	Abs	Ab	Abs	Abs	Abs	Abs
Reinante	43°33'32.54"N 7°10'36.05"W	Ab	Ab	Abs	F	Abs	R	Abs	Abs
Altar	43°34'5.86"N 7°14'13.48"W	F	Ab	Abs	Abs	Abs	Abs	Abs	Abs
Peizas	43°34'56.98"N 7°15'55.66"W	F	D	R	Oc	Abs	Abs	Abs	Abs
Burela	43°40'18.55"N 7°21'51.44"W	R	Oc	R	F	Abs	Abs	Abs	Abs
Nois	43°36'32.52"N 7°18'28.53"W	Oc	Ab	Oc	Abs	Abs	Abs	Abs	Abs
Rueta	43°41'0.13"N 7°24'16.67"W	F	Oc	Abs	Ab	Abs	R	Abs	Abs
San Ciprian	43°41'40.29"N 7°26'19.25"W	F	Oc	F	F	Abs	Abs	Abs	Abs
Moras	43°43'6.43"N 7°28'24.63"W	D	Ab	Abs	R	Abs	Abs	Abs	Abs
Porto Celo	43°43'19.81"N 7°30'52.89"W	F	F	R	Ab	Abs	Abs	Abs	Abs
Esteiro(Xove)	43°42'48.19"N 7°33'29.62"W	R	F	Abs	F	Abs	Abs	Abs	Abs
Area	43°41'45.89"N 7°34'43.79"W	F	Ab	Abs	Oc	Abs	R	Abs	Abs
Covas	43°40'27.95"N 7°36'38.53"W	D	D	Abs	Abs	Abs	Abs	Abs	Abs
A Brela	43°42'3.68"N 7°36'30.58"W	F	Ab	Oc	Abs	Abs	Abs	Abs	Abs
S. Roman	43°43'16.48"N 7°37'38.22"W	F	D	F	Ab	Abs	Abs	Abs	R
Xillo	43°44'45.73"N 7°39'3.51"W	F	Ab	Abs	Oc	Abs	Abs	Abs	F
Vidrieiro	43°44'26.71"N 7°40'30.69"W	D	D	Abs	Abs	Abs	Abs	Abs	Abs
Loiba	43°44'19.09"N 7°46'6.80"W	Ab	R	F	Ab	R	Abs	Abs	Abs
San Vicente (Ortigueira)	43°42'9.84"N 7°49'34.53"W	Abs	R	Abs	Abs	F	Ab	Abs	Abs
Figueiras (Cariño)	43°42'39.22"N 7°51'48.22"W	Abs	Abs	Abs	Abs	R	Abs	Abs	Abs
Cariño	43°44'39.09"N 7°51'59.60"W	F	Ab	F	Ab	Abs	R	Abs	Abs

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