Journal of Marine Systems 163 (2016) 23–30

Contents lists available at ScienceDirect

Journal of Marine Systems

journal homepage: www.elsevier.com/locate/jmarsys

Introduction of non-native marine fish species to the Canary Islands waters through oil platforms as vectors

José G. Pajuelo, José A. González, Raül Triay-Portella *, José A. Martín, Raquel Ruiz-Díaz, José M. Lorenzo, Ángel Luque

Instituto Universitario de Investigación en Estudios Ambientales y Recursos Naturales i-UNAT, University of Las Palmas de Gran Canaria, Campus de Tafira, Las Palmas de Gran Canaria, 35017, Las Palmas, Spain

ARTICLE INFO

Article history:
Received 9 February 2016
Received in revised form 20 June 2016
Accepted 20 June 2016
Available online 21 June 2016

Keywords:
Non-indigenous species
Oil rigs
Input vector
Translocation
Canary Islands

ABSTRACT

This work documents the introduction of non-native fish species to the Canary Islands (central-eastern Atlantic) through oil rigs. Methodological approaches have included surveys by underwater visual censuses around and under oil platforms and along the docking area of rigs at the Port of Las Palmas. Eleven non-native fish species were registered. *Paranthias furcifer*, *Abudefduf hoefleri*, *Acanthurus bahianus*, *Acanthurus chirurgus*, and *Acanthurus coeruleus* are first recorded from the Canaries herein. Other three species could not be identified, although they have never been observed in the Canaries. *Cephalopholis taeniops*, *Abudefduf saxatilis*, and *Acanthurus monroviae* had been previously recorded. Native areas of these species coincide with the areas of origin and the scale of oil rigs with destination the Port of Las Palmas. The absence of native species in the censuses at rigs and their presence at rigs docking area, together with the observation of non-native species after the departure of platforms, reject the possibility that these non-native species were already present in the area introduced by another vector. *C. taeniops*, *A. hoefleri*, *A. saxatilis*, *A. chirurgus*, *A. coeruleus* and *A. monroviae* are clearly seafarer species. *A. bahianus* seems to be a potential seafarer species. *P. furcifer* is a castaway species. For the moment, the number of individuals of the non-native species in marine ecosystems of the Canaries seems to be low, and more investigation is needed for controlling these translocations.

1. Introduction

The presence of non-native marine fish species in ecosystems of biogeographical regions well separated from the giver region has been related with the natural population extensions, in many cases associated with climate change, habitat disturbance (Côté and Green, 2012; Mineur et al., 2012; Webber and Scott, 2012), and anthropological factors (Carlton, 1987, 1989). In response to warmer conditions, marine fishes tend to shift their distributions to higher latitudes and fish are predicted to keep shifting under climate change scenarios (Cheung et al., 2009; Nye et al., 2009). In the case of anthropological factors, the new distribution areas are linked to the transport vector of non-native fish species. In this sense, some vectors of introduction of non-native fish species have been described including, among others, ballast water transport (Carlton and Geller, 1993; Wonham et al., 2000), marine aquaculture (Liao et al., 2010; Grosholz et al., 2015), aquarium trade (Semmens et al., 2004), oil platforms (Foster and Willan, 1979; Friedlander et al., 2014), and movements through channels (Carlton, 1989; Mavruk and Avsar, 2008). Introduction and spread of non-native species are considered one of the main threats to biodiversity in the marine context (Molnar et al., 2008) and, in recent years, the level of interest in introduced species has increased worldwide (Lockett and Gomon, 2001; Molnar et al., 2008).

Many small benthic marine fishes introduced as eggs, larvae or juveniles are first recorded from regions with important commercial ports (Wonham et al., 2000; Lockett and Gomon, 2001), and the method of transport associated is via the large amounts of ballast water carried by international shipping. In this regard, the ships' ballast water has been considered to date as the principal vector for the transfer of non-native species among marine ecosystems. However, in recent years numerous occurrences of adult and subadult individuals of non-native species have been recorded in waters near commercial ports of the Canary Islands (Brito et al., 2011; Triay-Portella et al., 2015). In the case of one of them, *Cephalopholis taeniops*, Brito et al. (2011) hypothesized about the possibility of the arrival of large adult specimens through ballast water from oil platforms. However, results of research indicate that ships' ballast water is potentially an important vector for transferring just marine non-native species of algae, plankton, small-sized invertebrates and chordate species, but not adult fish individuals (Lockett and Gomon, 2001; Molnar et al., 2008).

---

* Corresponding author.
E-mail address: emapulpgc@gmail.com (R. Triay-Portella).

http://dx.doi.org/10.1016/j.jmarsys.2016.06.008
0924-7963/

In recent years, special attention has been paid to oil rigs as input vector of non-native fish species (Ferreira et al., 2006; Wanless et al., 2010; Yeo et al., 2010; Friedlander et al., 2014; Triay-Portella et al., 2015). Oil platforms are among the largest artificial structures in the ocean where they act as artificial reefs and serve as important habitat for a number of species of fish populations (Atchison et al., 2008). These artificial habitats are colonized by diverse ecological communities that have been shown, in some instances, to enhance fisheries production and biodiversity, increasing the growth and survival of individuals, affording shelter for protection from predation and spawning substrate (Jørgensen et al., 2002; Macreadie et al., 2011; Friedlander et al., 2014). However, spread of non-native species has been described as some of the negative consequences associated with oil rigs (Page et al., 2006; Wanless et al., 2010; Friedlander et al., 2014).

The role of oil rigs as aggregating places and as a mechanism of dispersal of fish and other marine organisms is well known (Foster and Willan, 1979; Carlton, 1987, 1989; Bax et al., 2003; Wanless et al., 2010; Yeo et al., 2010; Friedlander et al., 2014). These platforms provide artificial hard substrates in places where do not exist naturally, exhibiting an extraordinary diversity and biomass of fishes and rocky invertebrates (Foster and Willan, 1979; Yeo et al., 2010; Friedlander et al., 2014). The majority of old oil platforms lack self-propulsion to navigate and are towed at low speed (1–6 knots) with one or two tugs (Foster and Willan, 1979; Yeo et al., 2010). Even modern oil platforms, which are equipped with engines to navigate, also move at very low speeds (<8 knots). The low speeds at which platforms are moved between areas reduce the loss of the fouling, which favours the maintenance of the community, making oil rig an artificial habitat moving slowly through the ocean (Yeo et al., 2010). In addition to the enormous hull columns, oil platforms have multiple submerged or semi-submerged structural elements (pontoon and tubular members as horizontal girders) whose main function is to favour buoyancy and stability. This submerged surface area of the oil rig is entirely covered with algae and sessile or sedentary organisms such as sponges, corals and bivalves (Wanless et al., 2010; Yeo et al., 2010). Under these favourable conditions, numerous and various fish species are attracted by these huge floating rigs for food or shelter, and many of them are able to travel to through hundreds or thousands of nautical miles (Foster and Willan, 1979; Carlton, 1987).

The aims of the present study are to identify arrivals of the non-native fish species associated to oil platforms, and to verify whether these oil rigs represent a primary input vector of adult and subadult individuals of non-native fish species in waters of the Canary Islands.

2. Material and methods

The Port of Las Palmas is a sheltered seaport in the northeastern coast of the island of Gran Canaria, Canary Islands (Fig. 1). It is the first port in the mid-Atlantic and offers excellent berthing piers for larger oil platforms, being able to berth up to six oil rigs at the time at the end of Pier Reina Sofía, where the depth is between 18 and 30 m (Fig. 1).

Traffic information of oil platforms at the Port and Bay of Las Palmas between 2010 and 2015 was obtained from various sources, including local newspapers, port logs of the Port of Las Palmas, and annual reports of activities of the Port Authority of Las Palmas. The information gathered allowed to build a map with the main routes of oil rigs with arrival to and departure from Gran Canaria.

Biogeographical information on detailed distribution tracking of those non-native fishes was added to specific maps and associated to present and past platforms travelling routes. A figure with maps combining this kind of information was used to discuss in light of biogeographic patterns and tropicalization event in the area.

To determine the occurrence of adult and subadult individuals of non-native fish species and indirectly verify whether oil platforms represent a primary input vector of introduction, underwater visual censuses by video-transects were carried out in both oil rigs and pier. Two semi-submersible drilling rigs (rig 1, December 2014; rig 2, April 2015) were sampled, around and under, a few hours after being docked at Pier Reina Sofía. The pontoons of the oil rigs surveyed were 90 m length, 12 m breadth and 10 m draught each one. The bottom area of the Pier Reina Sofía was also surveyed twice in the absence of oil platforms (four weeks after their departure), in order to detect the presence

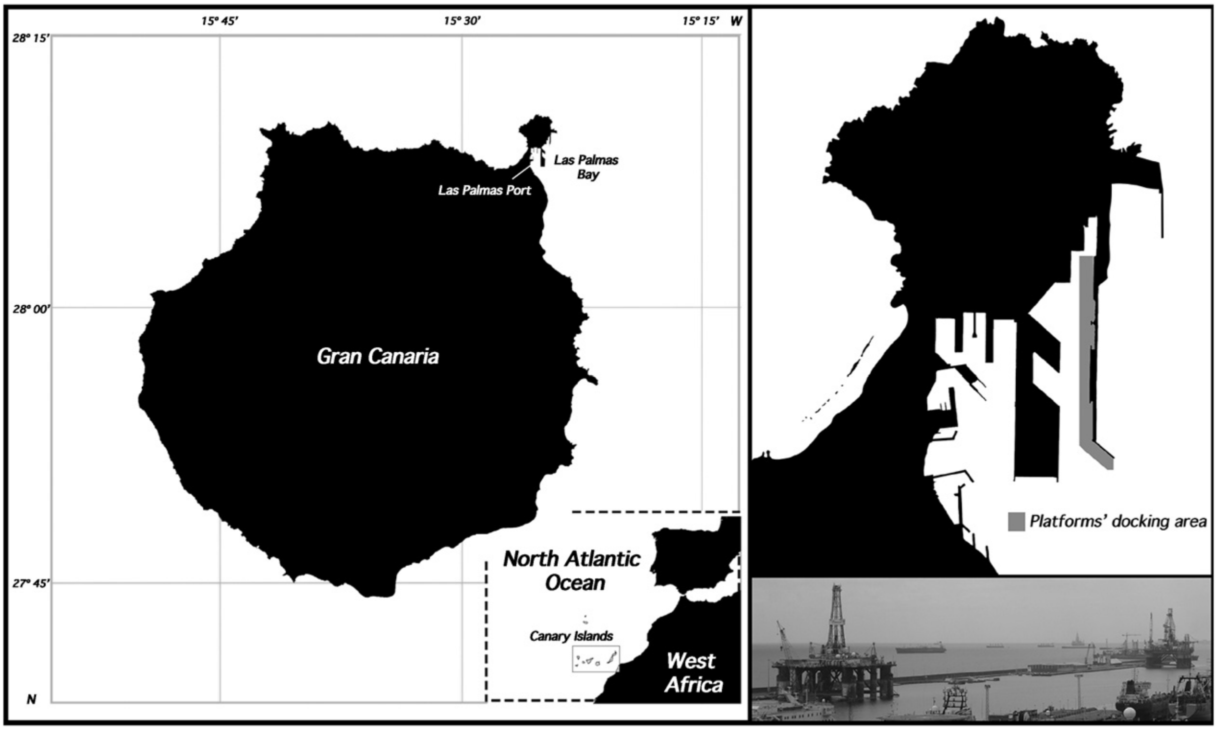

Fig. 1. Situation of the island of Gran Canaria (Canary Islands), showing the study area. Detail of the Port of Las Palmas, with the oil platforms' docking area. In the photograph, real picture of Pier Reina Sofía with the presence of semisubmersible oil rigs in April 2015.

of non-native species escaped from them (pier 1, February 2015; pier 2, May 2015). The surveyed area corresponded to the docking area for oil rigs. For each oil rig four 100 m long replicates (i.e. a surface of 400 m^2) were developed following Lincoln-Smith (1988) and Friedlander et al. (2014), surveying the top and bottom areas of both pontoons simultaneously to avoid non-native fish to be counted twice. For the pier, three replicated similar video-transects were performed simultaneously 2 m over the seabed.

To discard the presence of non-native species at the pier prior to the increase of oil rig traffic in Las Palmas Port, data on nine replicates obtained in 2003 (pier 0) from the entire Pier Reina Sofía (not only from the rigs docked area) were used to compare the species' occurrence at the pier, before (pier 0) and after (pier 1–2) of the massive docking of oil rigs in the studied area.

The abundance (total number of individuals) of all species was recorded on waterproof paper and by underwater pictures of live animals. Estimation of fish abundance in schools larger than 25 individuals, their numbers were calculated by using image counter software. When possible, one or few individuals of each non-native species were captured to be examined at the laboratory for taxonomic identification. When not, the zoological family of the species observed was recorded.

For each census, the species composition and the abundance (expressed as the number of individuals/400 m^2) were estimated. The abundance data were analyzed by means of a cluster analysis applied to the censuses using the Bray-Curtis index of similarity and the group-average linkage for the dendrogram (Clarke and Warwick, 2001; Clarke and Gorley, 2006) with PRIMER v6.1.13 software (Clarke and Warwick, 1994). The significance of the cluster branches was tested using the SIMPROF subroutine within the Primer software (Clarke and Warwick, 2001; Clarke and Gorley, 2006). The similarity percentage analysis procedure (SIMPER) was used to identify the species that characterize each structure (Clarke and Warwick, 1994). Measurements of species diversity at each rig and for the pier census were assessed by means of species richness (S), Shannon-Wiener diversity index (H′) and Pielou evenness index (J’), using the DIVERSE subroutine within the Primer software (Clarke and Warwick, 1994, 2001). These measurements were tested for differences between rigs and pier censuses, and between pier censuses along the time (pier 0 vs pier 1–2) through an ANOVA test, considering each underwater census at oil rig and pier as one observation. This analysis was used to evaluate the null hypothesis of equality in S, H′ and J’ among structures (rigs and pier) and among time periods (pier 0 vs pier 1–2), with a significance level of 5% ($\alpha = 0.05$) and a critical value of $F_{0.05,3,13} = 3.41$. The Student *t*-test was used to evaluate the null hypothesis of equality in mean S, H′, N and J′ between oil rigs, between pier samplings, and between pier samplings along time, with a significance level of 5% ($\alpha = 0.05$) and a critical value of $t_{0.05,>6} = 2.245$.

This study follows the best practices approach to overcoming unverified and unverifiable “first records”, as proposed by Bello et al. (2014). Voucher specimens were labelled, curated, data-based and deposited, available for verification, in the collections of the Tenerife Museum of Natural History (TFMC, Spain) and the Funchal Natural History Museum (MMF, Portugal). Further details on the identification process and fish data can be found in Triay-Portella et al. (2015).

3. Results

The introduction of 11 non-native fish species was found as associated to oil platforms (Table 1). Of these, five non-native fish species, *Paranthias furcifer* (Valenciennes, 1828), *Abudefduf hoefleri* (Steindachner, 1881), *Acanthurus bahianus* Castelnau, 1855, *Acanthurus chirurgus* (Bloch, 1787), and *Acanthurus coeruleus* Bloch & Schneider, 1801, are recorded for the first time from the Canaries herein. Other three non-native species belonging to families Lutjanidae, Pomacentridae and Blenniidae have not been identified, although they have never been observed in the Canaries (Table 1). Another three non-native fish species recorded, *Cephalopholis taeniops* (Valenciennes, 1828), *Abudefduf saxatilis* (Linnaeus, 1758), and *Acanthurus monroviae* Steindachner, 1876, had been previously found in the Canaries.

From January 2010 to May 2015, a total of 23 different oil platforms arrived at Gran Canaria Island, and then docked at the Port of Las Palmas. In total, 23 transoceanic or interoceanic voyages with destination/scale Gran Canaria were carried out by these oil platforms in this 5-year period (Fig. 2). These voyages started/arrived from/to ports in the eastern-central Atlantic (Nigeria, Cameroon, Ivory Coast and Cape Verde Islands), southeaster Atlantic (Namibia and Congo Brazzaville), West Atlantic (Brazil, sometimes with stop in Cape Verde Islands), South Africa, and the Mediterranean (Fig. 2). In addition, three transoceanic routes in the South Atlantic were verified using official and unofficial websites and incorporated to the route maps (Fig. 2). Departure areas of the sampled rigs were Angola-Gulf of Guinea (rig 1) and Brazil-Cape Verde (rig 2). These departure areas are coincident with the supposedly native distribution range of the non-indigenous species found at the rigs. In this regard, *A. hoefleri*, *A. saxatilis*, *P. furcifer*, *A. monroviae* and *C. taeniops* found at oil rig 1 have their native range in waters of the Gulf of Guinea and even southwards in Angola, where *A. bahianus* is a vagrant species. *A. bahianus*, *A. chirurgus* and *A. coeruleus* found at oil rig 2 abundantly occur in Brazilian waters. At last, *A. hoefleri* found at oil rig 2 would join the platform during its technical stop in the Cape Verde Islands (see Supplementary material) (Fig. 2).

Analyzing each oil rig and pier, the highest number of non-native fish species was observed in the pier census 2 in absence of oil rigs ($n = 10$). The whole number of non-native species recorded at the oil platforms were oil rig 1 ($n = 8$) and oil rig 2 ($n = 7$). The mean fish density observed ranged from 40.50 individual/400 m^2 at oil rig 2 to 413.25 individuals/400 m^2 at oil rig 1 (Table 1). In oil rig 1 the numerically dominant species was *A. hoefleri* (64.92%) with a mean density of 268.25 individuals/400 m^2. This species was followed in abundance by *A. saxatilis* (31.45%) with a mean density of 130.00 individuals/400 m^2. In oil rig 2 the numerically dominant species was a non-identified *Blenniidae* (61.72%) with a mean density of 25.00 individuals/400 m^2. This species was followed in abundance by *A. hoefleri* (26.54%, 10.75 individuals/400 m^2) and *A. bahianus* (6.79%) with a mean density of 2.75 individuals/400 m^2. The remaining species associated to both oil platforms had percentages below 2%. No native fish species were observed at the rigs (Table 1). At Pier Reina Sofia, numerically dominant species were *Thalassoma pavo* (Linnaeus, 1758) (10.50–15.92%) with a mean density of 12.00–12.33 individuals/400 m^2, *Chromis limbata* (Valenciennes, 1833) (11.00–17.00%, 13.67–14.00 individuals/400 m^2), *Diplodus sargus cadenati* de la Paz, Bauchot & Daget, 1974 (8.42–11.43%, 9.00–9.33 individuals/400 m^2), and *Boops boops* (Linnaeus, 1758) (10.22–13.00%, 10.67–11.33 individuals/400 m^2). Dominant non-native species at pier were *A. hoefleri* (4.76–8.42%, 6.77–7.33 individuals/400 m^2) and *A. saxatilis* (9.00–12.38%, 10.00–10.67 individuals/400 m^2). The remaining native species had percentages ranging between 1.00 and 6.81% and the non-native between 0.01 and 4.21%.

The dendrogram of similarities among the censuses of the cluster analysis revealed three main distinct groups (confirmed by SIMPROF at $p < 0.05$) (Fig. 3): a first group at the pier in 2003 comprising 9 censuses, a second group corresponding to the rigs censuses, and a third group corresponding with the censuses at pier in 2015. The results of the SIMPER, with the contribution in percentage of the main contributor species, are shown in Table 2 for the two oil rigs and the Pier Reina Sofía. Two species of *Abudefduf*, *A. hoefleri* and *A. saxatilis*, showed the major contribution for defining transects in the oil rig 1. In the oil rig 2, the most typifying species for defining transects were the Blennidae not identified and *A. hoefleri*. The main contributors of the average similarity at Pier Reina Sofía transects were *C. limbata*, *T. pavo*, *B. boops*, and *D. sargus* and the non-native species *A. hoefleri* and *A. saxatilis*. The main contributors of the average dissimilarity between the two oil rigs sampled were *A. hoefleri* and *A. saxatilis*. The main contributors of the

Table 1
Estimates of mean density (±s.d.) of individuals (individuals/400 m^2) recorded by visual census of oil rigs and at pier Reina Sofía. *, non-native species. n.i., non-identified species.

Species	3–6/2003 Pier 0 Mean density	12/2014 Oil rig 1 Mean density	1/2015 Pier 1 Mean density	3/2015 Oil rig 2 Mean density	4/2015 Pier 2 Mean density
**Abudefduf hoefleri*		268.25 ± 136.99	6.67 ± 1.40	10.75 ± 2.87	7.33 ± 1.15
Abudefduf luridus	8.13 ± 0.83		4.67 ± 1.15		5.00 ± 1.00
**Abudefduf saxatilis*		130.00 ± 35.56	10.67 ± 2.08		10.00 ± 0.58
**Abudefduf* n.i.		6.25 ± 2.50	0.33 ± 0.71	0.50 ± 0.35	0.67 ± 0.58
**Acanthurus bahianus*		2.00 ± 0.58	1.00 ± 0.50	2.75 ± 0.96	1.33 ± 0.58
**Acanthurus chirurgus*				0.75 ± 0.50	1.67 ± 0.71
**Acanthurus coeruleus*				0.25 ± 0.00	0.33 ± 0.25
**Acanthurus monroviae*		1.50 ± 1.00	3.00 ± 0.82		2.67 ± 0.58
Aulostomus strigosus	2.38 ± 1.51		2.33 ± 0.58		2.67 ± 0.57
Spondyliosoma cantharus	0.50 ± 0.53		0.33 ± 0.25		
**Blenniidae* n.i.		2.50 ± 1.29		25.00 ± 7.07	
Bodianus scrofa	0.38 ± 0.52				
Boops boops	9.50 ± 1.69		10.67 ± 1.53		11.33 ± 1.53
Canthigaster capistrata	6.25 ± 2.55		5.33 ± 0.58		4.67 ± 1.15
**Cephalopholis taeniops*		0.25 ± 0.48	0.33 ± 0.60		0.67 ± 0.25
Chromis limbata	9.75 ± 1.16		14.00 ± 2.00		13.67 ± 2.52
Diplodus sargus	4.75 ± 1.16		9.33 ± 2.31		9.00 ± 2.00
Diplodus vulgaris	1.63 ± 0.92				
Epinephelus marginatus	0.38 ± 0.52		0.67 ± 0.01		1.33 ± 0.58
Kyphosus sectatrix	2.50 ± 1.41		1.67 ± 0.58		2.33 ± 1.15
**Lutjanidae* n.i.				0.50 ± 0.00	0.33 ± 0.00
Mullus surmuletus	0.38 ± 0.52				
Oblada melanura	1.38 ± 1.06		4.00 ± 1.00		3.67 ± 0.58
Pagrus pagrus	0.50 ± 0.53				
**Paranthias furcifer*		2.50 ± 0.58	1.00 ± 0.71		0.67 ± 0.25
Serranus scriba	0.38 ± 0.52				
Sparisoma cretense	4.13 ± 1.64		5.67 ± 1.53		5.00 ± 2.00
Synodus synodus	1.63 ± 0.74		2.33 ± 1.15		1.67 ± 0.58
Thalassoma pavo	11.00 ± 1.85		12.00 ± 2.00		12.33 ± 1.53
N	65.50 ± 8.02	413.25 ± 137.01	102.67 ± 12.66	40.50 ± 7.85	103.33 ± 5.77
S	13.87 ± 1.36	6.50 ± 0.58	17.33 ± 0.58	5.00 ± 0.82	19.33 ± 0.58
J	0.89 ± 0.03	0.43 ± 0.07	0.88 ± 0.01	0.63 ± 0.02	0.88 ± 0.01
H′	2.32 ± 0.91	0.81 ± 0.12	2.52 ± 0.02	1.013 ± 0.14	2.61 ± 0.03

average dissimilarity between each oil rig and pier Reina Sofía were the Blennidae not identified and *A. saxatilis* (oil rig 2) and *A. hoefleri* and *A. saxatilis* (oil rig 1). The dissimilarity results of the SIMPER analysis between each oil rig and pier Reina Sofía showed that the major contributor species were the Blennidae not identified, *C. limbata*, *T. pavo* and *B. boops* (oil rig 2), and *A. hoefleri*, *A. saxatilis*, *C. limbata* and *T. pavo* (oil rig 1). When comparing pier censuses data in 2003 (pier 0) with 2015 (piers 1–2), non-native species were not recorded in 2003. Dissimilarity between both periods was 32.15%, being the non-native *A. saxatilis*, *A. hoefleri*, *A. monroviae* and *A. bahianus* the major contributor species (Table 1). There were significant differences in species richness and diversity between both periods ($F > 23.09 > F_{0.05,2,13} = 3.41$, $p < 0.0001$), the greatest species richness and highest diversity being observed in 2015.

For the structures surveyed in 2014–2015, two samplings at oil rigs and two censuses at pier, the greatest mean species richness (S) was reached at pier (17.33–19.33) and the lowest at oil rig 2 (5.00) (Table 1). There were significant differences in the species richness among structures ($F = 425.6795 > F_{0.05,3,13} = 3.41$, $p < 0.0001$). The Student *t*-test indicated that null hypothesis for equality in mean species richness (S) between oil rigs was rejected ($t = 3.00 > t_{0.05,6} = 2.245$, $p = 0.027$) and between pier censuses was also rejected ($t = 4.24 > t_{0.05,4} = 3.306, p = 0.0.13$). The highest diversity (H′) was obtained at pier (2.52–2.61) and the lowest at oil rig 1 (0.81). The H′ values showed differences among sampled structures ($F = 311.34 > F_{0.05,3,13} = 3.41$, $p < 0.0001$). The Student *t*-test indicated that null hypothesis for equality in mean diversity (H′) between oil rigs ($t = 2.208 < t_{0.05,6} = 2.245$, $p = 0.069$) or between pier censuses ($t = 2.621 < t_{0.05,4} = 3.306$, $p = 0.590$) was retained. Pielou J′ indices showed the highest value at pier (0.88) and the lowest at two oil rigs (0.43–0.63). J′ values showed significant differences among the sampled structures ($F = 109.51 > F_{0.05,3,13} = 3.41$, $p < 0.0001$). The Student *t*-test indicated that null hypothesis for equality in mean J' values between oil rigs ($t = 5.510 > t_{0.05,6} = 2.245$, $p = 0.002$) was retained and between pier censuses ($t = 0.689 < t_{0.05,4} = 3.306$, $p = 0.529$) was rejected. Mean number of individuals for the four structures reached the highest value at oil rig 1 (413.24) and the lowest at oil rig 2 (40.50). Number of individuals showed significant differences among the sampled structures ($F = 19.74 > F_{0.05,3,13} = 3.41$, $p < 0.0001$). The mean number of individuals between oil rigs showed significant differences ($t = 5.433 > t_{0.05,6} = 2.245$, $p = 0.002$) and between pier censuses retained ($t = 0.445 < t_{0.05,4} = 3.306$, $p = 0.679$).

4. Discussion

Species dispersing naturally and propagules from the current range to a new area is a process fundamental to the development of biogeographic patterns. However, the human-mediated dispersal of species to regions beyond their normal range of dispersal, as in this case, has been a major force in shaping global biodiversity, especially in the last few centuries (Wilson et al., 2010). Luiz et al. (2015) have recently proposed to assess the species-level ecological traits associated with the rafting behaviour, to better understand and predict the species' capacity for long-distance oceanic dispersal in isolated areas of benthic habitats. These authors proposed a dual classification of seafarer or castaway species. Rafter fish are like seafarers because they possess special traits that facilitate long oceanic journeys, whereas the castaway fish represent species marooned by chance in the flotsam (Luiz et al., 2015). In this regard, this work evaluates the non-native fish species recently found in waters of the Canary Islands as follows (for further details on species

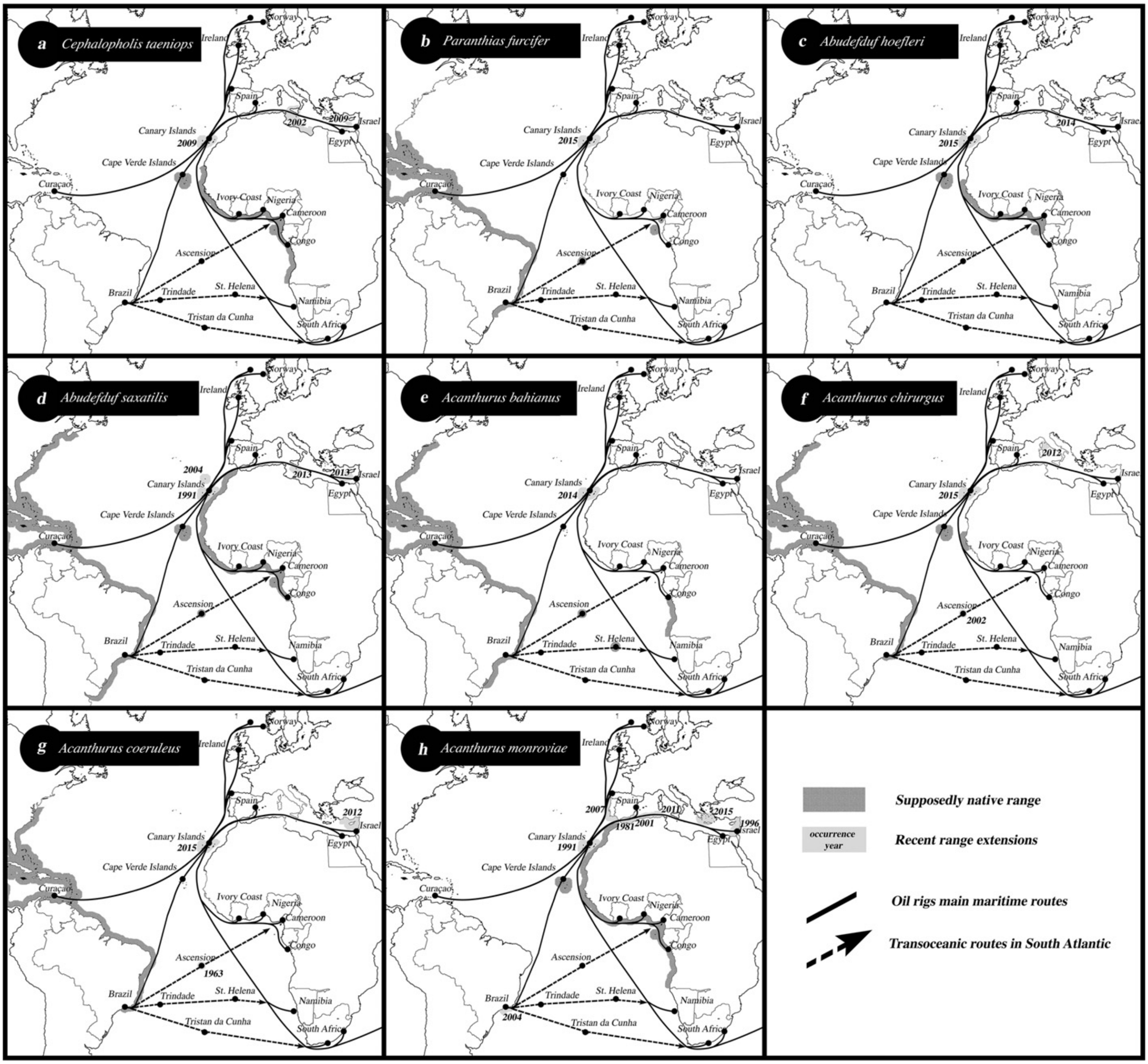

Fig. 2. Main maritime routes used by oil platforms with arrival to and departure from Gran Canaria (Canary Islands) between 2010 and 2015 (solid line) including three transoceanic routes in South Atlantic (discontinuous line), and maps with biogeographical information on distribution tracking of each non-native species found at Gran Canaria. Dark grey surface, supposedly native distribution range. Light grey surface, recent range extensions with year of first occurrence.

distribution and related bibliography, see Supplementary material). *C. taeniops*, *A. hoefleri*, *A. saxatilis*, *A. chirurgus*, *A. coeruleus* and *A. monroviae* are clearly seafarer species. *A. bahianus* seems to be a potential seafarer species. At last, *P. furcifer* is a castaway species. *A. monroviae* has been capable to perform a double long-distance oceanic dispersal. In the East Atlantic, from the early 1980′s to these days it repeatedly has reached the Canary Islands and the entire Mediterranean Sea, also extending to Israel, the easternmost Mediterranean. It also spread northward to the Atlantic European waters off Portugal in 2007. For these reasons, this species was included in the CIESM Atlas of Exotic Species in the Mediterranean, and therefore considered as an alien/invasive species in FishBase. Moreover, it crossed to Brazil, western Atlantic in 2004, so being included in the IABIN Brazil catalogue of invasive species. Several hypotheses should be considered to explain the reasons of dispersal, including both natural (current system, flotsam) and human-mediated causes particularly the increasing of oil rigs traffic among regions (Triay-Portella et al., 2015). *A. saxatilis* has progressively reached the Canary and Madeira Islands, as well as the western and the easternmost Mediterranean Sea favoured by both natural and human-mediated factors, and it has been considered as an alien/invasive species in FishBase. *C. taeniops* has sequentially reached, at least since 1990′s, the Canary and Madeira Islands, as well as the western and the easternmost Mediterranean Sea, likely favoured by the global warming and the tropicalization process described for these eastern Atlantic and Mediterranean areas (Craig et al., 2011), together with the harbour given by the oil rigs. Both *A. chirurgus* (in 2012–2015) and *A. hoefleri* (in 2014–2015) have simultaneously extended to the Canary Islands and Mediterranean Sea, likely favoured by global warming, tropicalization process, and rigs traffic. *A. coeruleus* has reached the mid-Atlantic island of Ascension (Cadenat and Marchal, 1963) as a natural range extension (marine current system, drafting behaviour), and extending fifty years after simultaneously (in

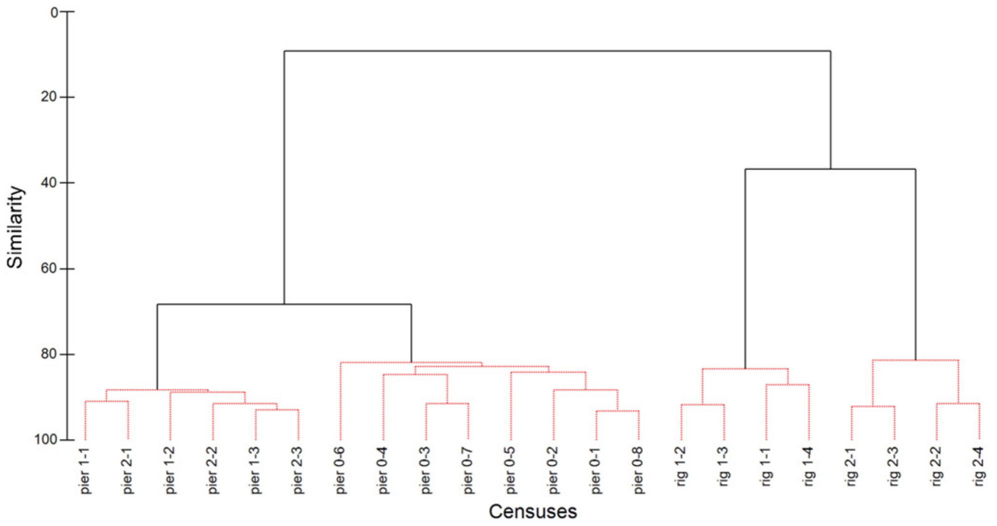

Fig. 3. Dendrogram of similarities among censuses of the cluster analysis confirmed by SIMPROF at $p < 0.05$. Oil rigs, censuses at semi-submersible drilling rigs (rig 1, December 2014; rig 2, April 2015). Pier, censuses at Pier Reina Sofia (pier 0, March–June 2003; pier 1, February 2015; pier 2, May 2015).

2012–2015) to the Canary Islands and Cyprus waters, the easternmost Mediterranean likely favoured by global warming, tropicalization process, and heavy rigs traffic. *A. bahianus* would be capable, if conditions remain favourable, to naturally extend its range, with the additional help of human-caused activities, to the Mediterranean and southwestern coast of Europe until 43° N as in the western Atlantic basin. *P. furcifer* has reached in 2015 similar latitude in the eastern (Canary Islands) than in the western Atlantic basin (Bermuda and Florida), and it can be predicted that will probably ascend to the Madeira Archipelago. Its capacity for long-distance oceanic dispersal has been favoured by the heavy overseas traffic of oil platforms from South America with destination to the Canaries mainly in the last decade (Fig. 2).

Up to date, spread records and histological examination of the gonads of individuals of different non-native species introduced by platforms (unpublished data) suggest that all these species can survival under the new biotic and abiotic conditions, but they do not show capacity of spawning in the area. However, *A. monroviae* apparently is, year after year, increasingly abundant in waters of the Canaries, as proven by some specimens in the spawning stage caught by local fishermen. In this regard, global warming change and the widely accepted tropicalization in the area could be favouring the existence of abiotic conditions more suitable for these species, at least for their survival. Therefore, one or some of these abiotic conditions are currently limiting the reproductive capacity of these species in the Canaries. Moyle and Light (1996) suggested that abiotic factors, such as sea temperature, could be more important in determining the success of introduced non-native fishes than biological factors. Also, non-native species that are ecological generalists in terms of either food or habitat and have morphological specializations of aggregation behaviour and aggressiveness, such as *A. saxatilis*, have often more successful in the survival (Moyle and Marchetti, 2006; Côté et al., 2013). Survival and spread of individuals of non-native species can occur in part due to the intrinsic characteristics of the species, such as fast life history and competitive ability (especially against the congeneric indigenous species), but it can also be facilitated by some characteristics of the recipient community, such as the scarcity of effective native competitors, predators and parasites and resource availability, among other factors (Côté et al., 2013). In this respect, *A. saxatilis* seem to have occupied vacant niches that have been opened from the effects of overfishing, whose consequences have been a change in the fish community with the decrease of native fish species and their biomasses (Bustos et al., 2010).

Considering that a transport vector is the form in which species are carried along a pathway, and a pathway is the route between the source region and the region of release (Lockwood et al., 2007), our data clearly indicate that oil platforms are very important vectors for transferring (translocation and introduction) non-indigenous fish species and other large-sized organisms. This vector has already been described by Foster and Willan (1979) based on the monitored observation of a couple of *A. saxatilis* adopted by a semisubmersible exploratory drilling platform in the coastal waters of the Solomon Islands, as well as by Friedlander et al. (2014) who stated that the demersal fish fauna from São Tomé and Príncipe, consisting of both western and eastern Atlantic species, are being extended onto the West African shelf by oil platforms. Also, the oil rigs have been described as an important pathway for translocation and introduction of vagile non-native species such as brachyurans and possibly other decapods (Page et al., 2006; Yeo et al., 2010). Of the six general types of extra range dispersal pathways proposed by Puth and Post (2005), oil rigs can be considered as mass dispersal type. This type is likely to capture a large proportion of the total genetic diversity of one species because of the large number of individuals translocated, so that the species introduced in a new area has levels of genetic diversity comparable to that of native populations, favouring the propagation over large areas (Puth and Post, 2005). These structures provide artificial substrates exhibiting an extraordinary abundance and diversity of fishes and, in some cases, the fish biomass is larger than for most tropical reef fishes, and even higher than many pristine reefs of the Pacific (Foster and Willan, 1979; Yeo et al., 2010; Friedlander et al., 2014). These oil platforms not only provide greater opportunity for vagile organisms to enter new environments as adults (Ferreira et al., 2006), but also raises the high probability that species will be reproductive in situ, as has been described by Yeo et al. (2010) and Friedlander et al. (2014).

Results obtained herein support that adult or subadult individuals of non-native fish species arrived associated to oil platforms. However, Brito et al. (2011) pointed the possibility of arrival to the Canaries of one adult of African hind, *C. taeniops*, and other large adult specimens through ballast waters. In this regard, the present results do not support Brito et al. (2011) hypothesis and provide a more suitable vector. However, the transport of small fish in ships' ballast water cannot be totally rejected because live fish have been observed in ships' ballast water (Williams et al., 1988; Wonham et al., 2000).

The absence of native species in the censuses at the rigs and their presence at the pier, together with the observation of non-native species after departure of platforms, reject the possibility that these non-native species were already present in the area introduced by another vector and then observed at the rigs, as a consequence that these non-native species approach to the platform once docked. Also, the results of the cluster analysis verified that the dissimilarity along the time in

Table 2
Results of the SIMPER routine to analyze the contribution (C, in %) of typifying species.

12/2014		3/2015		3–6/2003		2/2015		4/2015	
Oil rig 1, Average similarity: 83.85		Oil rig 2 Average similarity: 82.97		Pier 0 Average similarity: 83.92		Pier 1 Average similarity: 88.73		Pier 2 Average similarity: 88.01	
Species	C %	Species	C %	Species	C %	Species	C %	Species	C %
A. hoefleri	45.35	Blennidae n.i.	46.40	*T. pavo*	13.53	*C. limbata*	10.66	*C. limbata*	9.85
A. saxatilis	35.31	*A. hoefleri*	30.18	*C. limbata*	12.99	*T. pavo*	9.78	*T. pavo*	9.58
Abudefduf n.i.	7.62	*A. bahianus*	14.91	*B. boops*	12.52	*B. boops*	9.31	*B. boops*	9.15
Blenniidae n.i.	4.33			*A. luridus*	11.92	*A. saxatilis*	9.15	*A. saxatilis*	8.70
				C. capistrata	9.06	*D. sargus*	8.48	*D. sargus*	7.87
				D. sargus	8.65	*A. hoefleri*	7.13	*A. hoefleri*	7.34
				S. cretense	7.56	*C. capistrata*	6.71	*A. luridus*	5.92
				S. synodus	4.69	*S. cretense*	6.44	*C. capistrata*	5.70
				K. sectatrix	4.63	*A. luridus*	6.00	*S. cretense*	5.41
				A. strigosus	4.28	*O. melanura*	5.46	*O. melanura*	5.18
						A. monroviae	4.55	*A. monroviae*	4.33
						A. strigosus	4.24	*A. strigosus*	4.33
								K. sectatrix	3.54
Oil rigs 1 & 2 Average dissimilarity: 72.60		Oil rig 1 & Pier 1 Average dissimilarity: 77.69		Oil rig 2 & Pier 1 Average dissimilarity: 84.61		Oil rig 2 & Pier 2 Average dissimilarity: 81.09		Pier 0 & Pier 1–2 Average dissimilarity: 32.15	
Species	C %	Species	C %	Species	C %	Species	C %	Species	C %
A. hoefleri	36.66	*A. hoefleri*	23.43	*Blenniidae* n.i.	11.88	*Blenniidae* n.i.	11.83	*A. saxatilis*	15.03
A. saxatilis	33.36	*A. saxatilis*	14.31	*C. limbata*	8.92	*C. limbata*	8.79	*A. hoefleri*	12.37
Blenniidae n.i.	10.03	*C. limbata*	6.62	*T. pavo*	8.28	*T. pavo*	8.36	*A. monroviae*	7.83
Abudefduf n.i	5.77	*T. pavo*	6.14	*B. boops*	7.79	*B. boops*	8.01	*A. bahianus*	5.44
P. furcifer	4.14	*B. boops*	5.78	*A. saxatilis*	7.77	*A. saxatilis*	7.53	*D. vulgaris*	5.01
		D. sargus	5.39	*D. sargus*	7.25	*D. sargus*	7.12	*O. melanura*	4.45
		S. cretense	4.19	*S. cretense*	5.64	*A. luridus*	5.31	*D. sargus*	3.97
		C. capistrata	4.10	*C. capistrata*	5.52	*S. cretense*	5.24	*P. furcifer*	3.43
		A. luridus	3.82	*A. luridus*	5.16	*C. capistrata*	5.12	*A. luridus*	3.08
		Abudefduf n.i.	3.78	*O. melanura*	4.75	*O. melanura*	4.55	*E. marginatus*	3.06
		O. melanura	3.53	*A. monroviae*	4.10	*A. monroviae*	3.88	*C. limbata*	2.77
		Blennidae n.i.	2.76	*A. strigosus*	3.63	*A. strigosus*	3.87	*K. sectatrix*	2.55
		A. strigosus	2.69	*S. synodus*	3.56	*K. sectatrix*	3.54	*C. capistrata*	2.55
		S. synodus	2.64	*K. sectatrix*	3.04	*S. synodus*	3.05	*A. strigosus*	2.43
				P. furcifer	1.89	*E. marginatus*	2.72	*A. chirurgus*	2.41

the pier are due to the introduction of non-native species, and therefore, that rigs must be considered, for their species composition, a structure without previous relation with natural fauna of the pier.

The market for oil rigs operating in West Africa and South America has increasingly consolidated the Port of Las Palmas. It is very probable that the heavy overseas traffic from West Africa and South America arriving to the Canary Islands was responsible for many of the first records that were attributed to other sources in the past ten years. According to our data, this also could be applied to southern Brazil and the Cape Verde Islands. In this regard, in waters of the Cape Verde Islands, Freitas et al. (2014) attributed the presence of 13 reef-associated species endemic or native to the biogeographical Brazilian Province to the North Equatorial Countercurrent as a potential dispersal route for marine organisms crossing the Atlantic Ocean in a west–east direction. These authors also indicated that Brazilian species have been recently recorded as vagrants in the southern Caribbean and West African offshore islands. Previously, however, Rocha (2003) and Wirtz et al. (2007) indicated a widespread extension of the Brazilian fish distribution beyond the border of the Brazilian Province, mainly toward the southern Caribbean and African offshore islands. The origin of recorded non-native species (from these areas and from the Canaries) and places of occurrence are consistent with the routes of these rigs, which would explain the recent rise in the observations of non-native species via the two main oil platforms' Atlantic routes, coming from the Gulf of Guinea and/or southern Brazil (sometimes with one stop in the Cape Verde Islands) and arriving at the Canary Islands. Oliveira and Creed (2008); Loebmann et al. (2010) and Wanless et al. (2010) suggested that some non-indigenous benthic species could have been introduced into Brazil on oil platforms from the Caribbean coast. In this regard, Ferreira et al. (2006) recorded 22 species exotic to Brazil, with 10 species as first records, on oil platforms in the Brazilian coast.

The greater the degree of international trade, the higher the number of non-native species. Thus, future management decisions on non-native species transport should consider the role of international overseas traffic in promoting species introduction. For the moment, the number of individuals of the non-native species in marine ecosystems of the Canaries seems to be low. However, this requires more investigation with emphasis on controlling the spread and growth of their populations (Gardner et al., 2015).

Acknowledgements

The authors are very grateful to two anonymous reviewers for their critical and useful comments on the manuscript that enabled the contents to be improved. Many thanks are due to Óliver González (MET Engineering and Consulting, Las Palmas de Gran Canaria), with plenty of experience on oil platforms, for enlightening us on the operation thereof.

Appendix A. Supplementary data

Supplementary data to this article can be found online at http://dx.doi.org/10.1016/j.jmarsys.2016.06.008.

References

Atchison, A.D., Sammarco, P.W., Brazeau, D.A., 2008. Genetic connectivity in corals on the Flower Garden Banks and surrounding oil/gas platforms, Gulf of Mexico. J. Exp. Mar. Biol. Ecol. 365, 1–12.

Bax, N., Williamson, A., Aguero, M., González, E., Geeves, W., 2003. Marine invasive alien species: a threat to global biodiversity. Mar. Policy 27, 313–323.

Bello, G., Causse, R., Lipej, L., Dulčić, J., 2014. A proposed best practice approach to overcome unverified and unverifiable “first records” in ichthyology. Cybium 38, 9–14.

Brito, A., Clemente, S., Herrera, R., 2011. On the occurrence of the African hind, *Cephalopholis taeniops*, in the Canary Islands (eastern subtropical Atlantic): introduction of large-sized demersal littoral fishes in ballast water of oil platforms? Biol. Invasions 13, 2185–2189.

Bustos, R., Luque, Á., Pajuelo, J.G., 2010. Reproductive biology of the island grouper (*Mycteroperca fusca*) in the Canary Islands, northwest coast of Africa. Sci. Mar. 74, 613–619.

Cadenat, J., Marchal, E., 1963. Poissons, in: Résultats des campagnes océanographiques de la Reine-Pokou aux îles Sainte-Hélène et Ascension. Bull. IFAN (A) Sci. Nat. 25, 1235–1315 (51 unnumbered pls).

Carlton, J.T., 1987. Patterns of transoceanic marine biological invasions in the Pacific Ocean. Bull. Mar. Sci. 41, 452–465.

Carlton, J.T., 1989. Man's role in changing the face of the ocean: biological invasions and implications for conservation of near-shore environments. Conserv. Biol. 3, 265–273.

Carlton, J.T., Geller, J.B., 1993. Ecological roulette: the global transport of nonindigenous marine organisms. Science 261, 78–82.

Cheung, W.W.L., Lam, V.W.Y., Sarmiento, J.L., Kearney, K., Watson, R., Pauly, D., 2009. Projecting global marine biodiversity impacts under climate change scenarios. Fish Fish. 10, 235–251.

Clarke, K.R., Gorley, R.N., 2006. PRIMER v6: User Manual/Tutorial. PRIMER-E Ltd, Plymouth, p. 190.

Clarke, K.R., Warwick, R.M., 1994. Change in Marine Communities: An Approach to Statistical Analysis and Interpretation. Natural Environmental Research Council, Plymouth Marine Laboratory, Plymouth.

Clarke, K.R., Warwick, R.M., 2001. Change in Marine Communities: An Approach to Statistical Analysis and Interpretation. PRIMER-E, Plymouth.

Côté, I.M., Green, S.J., 2012. Potential effects of climate change on a marine invasion: the importance of current context. Current Zool. 58, 1–8.

Côté, I.M., Green, S.J., Hixon, M.A., 2013. Predatory fish invaders: insights from Indo-Pacific lionfish in the western Atlantic and Caribbean. Biol. Conserv. 164, 50–61.

Craig, M.T., Sadovy, Y., Heemstra, P.C., 2011. Groupers of the World: A Field and Market Guide. NISC (Pty), Grahamstown.

Ferreira, C.E.L., Gonçalves, J.E.A., Coutinho, R., 2006. Ship hulls and oil platforms as potential vectors to marine species introduction. J. Coast. Res. 39, 1340–1345.

Foster, B.A., Willan, R.C., 1979. Foreign barnacles transported to New Zealand on an oil platform. N.Z. J. Mar. Fresh. 13, 143–149.

Freitas, R., Luiz, O.J., Silva, P.N., Floeter, S.R., Bernardi, G., Ferreira, C.E.L., 2014. The occurrence of *Sparisoma frondosum* (Teleostei: Labridae) in the Cape Verde Archipelago, with a summary of expatriated Brazilian endemic reef fishes. Mar. Biodivers. 44, 173–179.

Friedlander, A.M., Ballesteros, E., Fay, M., Sala, E., 2014. Marine communities on oil platforms in Gabon, West Africa: high biodiversity oases in a low biodiversity environment. PLoS One 9, e103709.

Gardner, P.G., Frazer, T.K., Jacoby, C.A., Yanong, R.P.E., 2015. Reproductive biology of invasive lionfish (*Pterois* spp.). Front. Mar. Sci. 2, 1–10.

Grosholz, E.D., Crafton, R.E., Fontana, R.E., Pasari, J.R., Williams, S.L., Zabin, C.J., 2015. Aquaculture as a vector for marine invasions in California. Biol. Invasions 17, 1471–1484.

Jørgensen, T., Løkkeborg, S., Soldal, A.V., 2002. Residence of fish in the vicinity of a decommissioned oil platform in the North Sea. ICES J. Mar. Sci. 59, 288–293.

Liao, Y.C., Chen, L.S., Shao, K.T., 2010. The predatory Atlantic red drum, *Sciaenops ocellatus*, has invaded the western Taiwanese coast in the Indo-West Pacific. Biol. Invasions 12, 1961–1965.

Lincoln-Smith, M.P., 1988. Effects of observer swimming speed on sample counts of temperate rocky reef fish assemblages. Mar. Ecol. Prog. Ser. 43, 223–231.

Lockett, M.M., Gomon, M.F., 2001. Ship mediated fish invasions in Australia: two new introductions and a consideration of two previous invasions. Biol. Invasions 3, 187–192.

Lockwood, J.L., Hoopes, M.F., Marchetti, M.P., 2007. Invasion Ecology. Blackwell, Oxford, UK.

Loebmann, D., Cecilia, A., Mai, G., Lee, J.T., 2010. The invasion of five alien species in the Delta do Parnaíba Environmental Area, North-eastern Brazil. Rev. Biol. Trop. 58, 909–923.

Luiz, O.J., Allen, A.P., Robertson, D.R., Floeter, S.R., Madin, J.S., 2015. Seafarers or castaways: ecological traits associated with rafting dispersal in tropical reef fishes. J. Biogeogr. 42, 2323–2333.

Macreadie, P.I., Fowler, A.M., Booth, D.J., 2011. Rigs-to-reefs: will the deep sea benefit from artificial habitat? Front. Ecol. Environ. 9, 455–461.

Mavruk, S., Avsar, D., 2008. Non-native fishes in the Mediterranean from the Red Sea, by way of the Suez Canal. Rev. Fish Biol. Fish. 18, 251–262.

Mineur, F., Cook, E.J., Minchin, D., Bohn, K., MacLeod, A., Maggs, C.A., 2012. Changing coasts: marine aliens and artificial structures. Oceanogr. Mar. Biol. 50, 189–234.

Molnar, J.L., Gamboa, R.L., Revenga, C., Spalding, M.D., 2008. Assessing the global threat of invasive species to marine biodiversity. Front. Ecol. Environ. 6, 485–492.

Moyle, P.B., Light, T., 1996. Fish invasions in California: do abiotic factors determine success? Ecology 77, 1666–1670.

Moyle, P.B., Marchetti, M.P., 2006. Predicting invasion success: freshwater fishes in California as a model. Bioscience 56, 515–524.

Nye, J.A., Link, J.S., Hare, J.A., Overholtz, W.J., 2009. Changing spatial distribution of fish stocks in relation to climate and population size on the Northeast United States continental shelf. Mar. Ecol. Prog. Ser. 393, 111–129.

Oliveira, A.E.S., Creed, J.C., 2008. Mollusca, Bivalvia, *Isognomon bicolor* (C. B. Adams 1845): Distribution Extension. Check List. 4, pp. 386–388.

Page, H.M., Dugan, J.E., Culver, C.S., Hoesterey, J.C., 2006. Exotic invertebrate species on offshore oil platforms. Mar. Ecol. Prog. Ser. 325, 101–107.

Puth, L.M., Post, D.M., 2005. Studying invasion: have we missed the boat? Ecol. Lett. 8, 715–721.

Rocha, L.A., 2003. Patterns of distribution and processes of speciation in Brazilian reef fishes. J. Biogeogr. 30, 1161–1171.

Semmens, B.X., Buhle, E.R., Salomon, A.K., Pattengill-Semmens, C.V., 2004. A hotspot of non-native marine fishes: evidence for the aquarium trade as an invasion pathway. Mar. Ecol. Prog. Ser. 266, 239–244.

Triay-Portella, R., Pajuelo, J.G., Manent, P., Espino, F., Ruiz-Díaz, R., Lorenzo, J.M., González, J.A., 2015. New records of non-indigenous fishes (Perciformes and Tetraodontiformes) from the Canary Islands (north-eastern Atlantic). Cybium 39, 163–174.

Wanless, R.M., Scott, S., Sauer, W.H.H., Andrew, T.G., Glass, J.P., Godfrey, B., Griffiths, C., Yeld, E., 2010. Semi-submersible rigs: transporting marine ecosystems around the world. Biol. Invasions 12, 2573–2583.

Webber, B.L., Scott, J.K., 2012. Rapid global change: implications for defining natives and aliens. Glob. Ecol. Biogeogr. 21, 305–311.

Williams, R.J., Griffiths, F.B., van der Waal, E.J., Kelly, J., 1988. Cargo vessel ballast water as a vector for the transport of non-indigenous marine species. Estuar. Coast. Shelf Sci. 26, 409–420.

Wilson, J.R.U., Dormontt, E.E., Prentis, P.J., Lowe, A.J., Richardson, D.M., 2010. Something in the way you move: dispersal pathways affect invasion success. Trends Ecol. Evol. 24, 136–144.

Wirtz, P., Ferreira, C.E.L., Floeter, S.R., Fricke, R., Gasparini, J.L., Iwamoto, T., Rocha, L.A., Sampaio, C.L., Schliewen, U., 2007. Coastal fishes of São Tomé and Príncipe — an update. Zootaxa 1523, 1–48.

Wonham, M.J., Carlton, J.T., Ruiz, G.M., Smith, L.D., 2000. Fish and ships: relating dispersal frequency to success in biological invasions. Mar. Biol. 136, 1111–1121.

Yeo, D.C., Ahyong, S.T., Lodge, D.M., Ng, P.K., Naruse, T., Lane, D.J., 2010. Semisubmersible oil platforms: understudied and potentially major vectors of biofouling-mediated invasions. Biofouling 26, 179–186.