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Research Article

Rapid assessment of marine non-native species in Irish marinas

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Abstract

Marinas are known to act as species reservoirs, facilitating the persistence and spread of non-native species. Monitoring of marinas for non-native species is therefore an essential element of any integrated monitoring programme. Baseline and updated survey data are needed to understand the abundance, distribution and rate of spread of non-native species within marinas over time in order to support management strategies, develop applicable biosecurity measures and inform rapid response plans. Rapid assessment surveys (RAS) were employed to survey floating pontoons, as well as opportunistically survey intertidal structures, for non-native biofouling species in 22 marinas across the Republic of Ireland. A total of 25 non-native species were recorded: 19 invertebrates and 6 macroalgae, with ascidians being the most represented group. Four non-native species not previously reported in published literature from the Republic of Ireland were recorded: devil's tongue weed, *Grateloupia turuturu* Yamada, 1941, the compass sea squirt, *Asterocarpa humilis* (Heller, 1878), the San Diego sea squirt, *Botrylloides diegensis* Ritter & Forsyth, 1917 and the branching bryozoan, *Bugulina simplex* (Hincks, 1886). Other species identified as part of this study and previously reported from Ireland were shown to have increased their range.

Key words: rapid assessment survey (RAS), invasive alien species (IAS), non-indigenous species, floating pontoons, Ireland

Introduction

The impact of invasive species is recognised as a major driver of biodiversity loss globally (MEA 2005; Hooper et al. 2012), resulting in species extinctions (Clavero and García-Berthou 2005) and the homogenisation of biological communities (McKinney and Lockwood 1999), with subsequent declines in valuable ecosystem services (Charles and Dukes 2008; Pejchar and Mooney 2009). Invasive species can have economic and social impacts (Pimentel et al. 2005; Williams et al. 2010), reducing crop and aquaculture production (Cook et al. 2007; Pejchar and Mooney 2009), damaging infrastructure (Thomas 2010; ISAC 2016) and affecting recreation and tourism (Mayfield et al. 2021). The global cost associated with invasive species has been estimated at 1.288 trillion USD over the past 50 years (Diagne et al. 2021), with an estimated €117 billion between 1960 and 2020 in Europe alone (Haubrock et al. 2021).

The spread of marine and coastal non-native species has long been attributed to the rise in global shipping and trade (Carlton and Geller 1993; Ruiz et al. 1997; Katsanevakis et al. 2013), with ballast water exchange (Gollasch 2008; Bailey 2015), hull fouling (Gollasch 2002; Coutts and Dodgshun 2007) and aquaculture (Naylor et al. 2001) identified as major pathways of introduction and spread. Consequently, international ports tend to have greater numbers of non-native species compared to areas with less shipping pressure (Wonham and Carlton 2005; Keller et al. 2011; Costello et al. 2022). Secondary pathways, such as recreational boating, facilitate the spread of species away from these international hotspots to natural areas, smaller harbours and recreational marinas (Acosta and Forrest 2009; Clarke Murray et al. 2011). There are growing concerns that marinas provide "stepping stones" (*sensu* Floerl et al. 2009) for the dispersal and spread of non-native species, and that "spillover" from marinas to natural areas—particularly areas containing protected habitats and species—is becoming more common (Epstein and Smale 2018; Afonso et al. 2020).

Recreational marinas contribute to the proliferation of non-native species because they act as species reservoirs (Bax et al. 2002; Neves et al. 2007), facilitating the persistence of fouling species through provision of hard substrate (i.e. floating pontoons and other marina infrastructure) among otherwise uninhabitable habitats (e.g. "soft bottom" sediment habitat; Airoldi et al. 2015). Marinas receive substantially more propagule pressure than other parts of the coast (e.g. natural coastline) as they are often located in or near large harbours that support international commercial shipping. Moreover, marinas support recreational boating traffic, with watercraft remaining berthed in marinas for several days, weeks or months (Wasson et al. 2001; Wonham and Carlton 2005), allowing the dispersal of propagules and larvae from hull-fouling organisms that may have originated from ports and harbours further afield (Ruiz and Smith 2005). Marina pontoons may therefore act as sentinels for the appearance of new species (e.g. Arenas et al. 2006; Collin et al. 2015; Nall et al. 2015), providing opportunities to detect new arrivals before they become established and spread.

Invasive fouling species within marinas can be problematic because they attach to infrastructure, boat hulls and submerged equipment (Coutts and Forrest 2007; Dafforn et al. 2009), resulting in vessels experiencing increased drag in the water (with subsequent increase in frequency of hull cleaning) (Baital and Utama 2017), thus affecting travel time and fuel consumption (Champ 2000; Hakim et al. 2017). Non-native species may reduce local biodiversity (Stachowicz et al. 1999) through overgrowing, smothering and outcompeting native biota (e.g. Griffith et al. 2009; Lengyel et al. 2009). Furthermore, marine non-natives have been shown to be more prevalent on artificial structures, such as floating pontoons and seawalls, compared to natural hard substrate (Glasby et al. 2007; Airoldi et al. 2015). This may be due to several factors at play in marinas, including sheltered conditions, no tidal effect on pontoon assemblages and multilayer fouling creating complex biotic structure facilitating recruitment. Additionally, human disturbances, such as docking and maintenance, which can physically dislodge organisms and create space for new colonisers to exploit, thereby artificially influencing succession of the community can play a significant role in colonisation by non-native species (Byers 2002; Airoldi and Bulleri 2011).

Understanding the distribution of existing invasive species along with the detection of novel introductions is an essential component of any invasive species management programme (Whomersley et al. 2015). Early detection of new harmful species and rapidly responding to an incursion is vital in the marine environment, as invasive species are extremely difficult or impossible to eradicate once established (Katsanevakis et al. 2013). As such, monitoring and surveillance programmes to document existing invasive species populations and detect new arrivals in order to understand abundance and distribution and trigger rapid response plans for harmful invasive species are a key requirement to inform invasive species management (Keefer et al. 2010; Trebitz et al. 2017). The effectiveness of monitoring programmes can be enhanced by carrying out risk assessments of species likely to arrive, establish and cause ecological, social and/or economic harm (i.e. horizon species) (Roy et al. 2014, 2019; Copp et al. 2016), which then can facilitate a targeted monitoring approach. For example, *Caulerpa taxifolia* (M.Vahl) C.Agardh 1817 was discovered during monitoring of a California lagoon in 2000, but because it was already listed on the US Federal Noxious Weed list, rapid response protocols were quickly activated, facilitating containment, treatment and eradication (Anderson 2005).

The rapid assessment survey (RAS) approach is an established method for surveying large geographic areas in a reasonable amount of time by employing a qualitative approach in which target species are recorded on an abundance scale during timed searches of specific habitats (e.g. Cohen et al. 1998, 2005; Arenas et al. 2006). In the context of recreational marinas, RAS facilitate the collection of species abundance data over large areas of submerged and intertidal hard structure (e.g. pontoons, harbour walls), and has been used in marinas around the world to record and map presence, abundance and distribution of non-native species, as well as document new arrivals (e.g. Pederson et al. 2005; Arenas et al. 2006; Minchin 2007a; Bishop et al. 2015a; Collin et al. 2015).

For this study, RAS of 22 marinas across the Republic of Ireland (RoI) were carried out to understand presence and abundance of non-native species within these marinas and to detect any new arrivals.

Table 1. Details of marinas in the Republic of Ireland where rapid assessment surveys (RAS) were conducted in August–September 2021 with assocaited water temperature, salinity, number of scrape samples collected and count of non-native species detected ("NNS richness").

Marina no.	Latitude	Longitude	County	Date of survey	Water temperature (°C)	Water salinity (ppt)	No. scrape samples	NNS richness
1	55.094829	−7.52931	Donegal	02/09/2021	17.2	35	2	4
2	55.088181	−7.480902	Donegal	02/09/2021	16.1	35	4	9
3	54.052407	−6.19122	Louth	01/09/2021	16.4	35	4	12
4	53.297161	−6.134166	Dublin	30/08/2021	16.6	35	6	14
5	53.389075	−6.067498	Dublin	31/08/2021	15.8	35	5	8
6	53.455691	−6.151411	Dublin	31/08/2021	16	35	5	7
7	52.795744	−6.145389	Wicklow	19/08/2021	17.2	1	0	0
8	52.172052	−6.58804	Wexford	20/08/2021	16.1	35	3	5
9	52.260789	−7.10425	Waterford	20/08/2021	17.6	9	3	1
10	51.866761	−8.213499	Cork	12/08/2021	16.4	35	3	6
11	51.805324	−8.307795	Cork	17/08/2021	14.8	35	4	2
12	51.805348	−8.300875	Cork	17/08/2021	14.4	35	3	3
13	51.805256	−8.295731	Cork	17/08/2021	15.1	35	3	4
14	51.697249	−8.516043	Cork	13/08/2021	16.7	34	3	1
15	51.702899	−8.51874	Cork	13/08/2021	14.6	35	5	2
16	51.63592	−8.710927	Cork	18/08/2021	14.0	35	2	1
17	51.482874	−9.375318	Cork	18/08/2021	14.2	35	1	6
18	51.947877	−10.233773	Kerry	23/08/2021	16.8	32	3	2
19	52.137608	−10.277836	Kerry	24/08/2021	17.4	34	3	4
20	52.271191	−9.862923	Kerry	25/08/2021	18.4	35	4	7
21	52.614646	−9.113242	Limerick	26/08/2021	18.4	28	2	3
22	52.634631	−9.495434	Clare	26/08/2021	19.3	30	4	5

Materials and methods

Twenty-two marinas were surveyed around the coast of RoI (Table 1) in August and September 2021, covering the north, east, south and southwest coasts. Several marinas were located in or adjacent to waterways that support international commercial vessel traffic (e.g. Dublin Bay, Cork Harbour, Shannon Estuary), while other marinas were located in less trafficked waterways where the dominant vessels were recreational boats. When reporting on marinas surveyed, marinas were identified using numbers for the actual localities because not all marina operators agreed to their establishments being explicitly named.

RAS were carried out from the surface of floating pontoons. Time spent at each marina varied based on size of marina, but was typically approximately 1 h total, with time at smaller marinas truncated (no less than 45 min total) and larger marinas requiring additional time. Non-native and cryptogenic (i.e. species that cannot be demonstrably classified as native or introduced in a particular region; Carlton 1996) species were searched for and recorded in 15-minute intervals. For all marinas, if the surveyors got to the last 15-minute interval and were still recording new species, an additional 15 minutes was added to the search, and so on until no new species were recorded. Biota on floating pontoons were visually examined, and where biological communities were out of reach, a scraper and collection net were utilised to collect organisms for closer examination. Other submerged artificial material within marinas, such as hanging ropes and fenders, were also examined in order to ensure as many non-native species as possible were accounted for. Although RAS of floating pontoons are able to be done at any state of the tide given the design of pontoons, these surveys were able to be conducted during ebb tide to allow for intertidal structures (e.g. wave breaker walls, marina pilings) to be opportunistically examined in order to capture presence and abundance of intertidal non-native species. Photographs and short videos of species assemblages on floating pontoons were taken underwater using a GoPro HERO 8 camera for confirmation of presence of species where possible.

As some non-native species are inconspicuous because they inhabit the interstitial spaces created by other organisms or can only be positively identified using microscopic examination, samples were taken in the field and transported to the laboratory for further examination. A sample consisted of all biota (native and non-native) within an area of approximately 30 × 30 cm removed from the side of a pontoon using a scraper and sweep net. Intertidal structures were not sampled. The number of scrape samples collected depended on the size of each marina, with number of samples ranging from one in very small marinas (i.e. < 50 berths) to six in very large marinas (i.e. > 350 berths) (see Table 1 for number of samples collected per marina). At all marinas, locations of scrape samples were evenly distributed around the marina, ensuring samples were taken from both sheltered and exposed areas of marina if applicable. The location of each scrape sample was indicated using a handheld GPS unit (Garmin eTrex® 10). Samples were stored in 2.5 Litre collection buckets, preserved in 4% formaldehyde solution following standard practice and health and safety guidelines and sent to the laboratory for taxonomic identification and quantification.

Given the difficulty in distinguishing between *Botrylloides* species (i.e. *B. diegensis*, *B. leachii*, *B. violaceus*) (Bishop et al. 2015b), organisms were only identified as *Botrylloides violaceus* Oka, 1927 when brooding or tadpole larvae were observed. Similarly, organisms were only identified as *Botrylloides diegensis* Ritter & Forsyth, 1917 when the distinctive two-colour morphology was observed. Thus, where neither of these characteristics were observed, these organisms were labelled as *Botrylloides* sp. This method was followed in the field and laboratory. Similarly, to confirm *Didemnum vexillum* Kott, 2002 identified in the field and to distinguish between *D. vexillum* and *D. pseudovexillum* (see Turon et al. 2020), samples of *Didemnum* spp. were taken for examination in the laboratory, and where necessary, molecular analysis was carried out using metabarcoding techniques.

The semi-quantitative SACFOR scale (JNCC 1990) was utilised to estimate overall abundance of non-native and cryptogenic species on a 6-point SACFOR scale: 6 = Super abundant, 5 = Abundant, 4 = Common, 3 = Frequent, 2 =

Table 2. Abundance of non-native species detected from rapid assessment surveys (RAS) in August and September 2021 presented by site number (marina) with number of occurrences of each species included. "S" = Super abundance, "A" = Abundant, "C" = Common, "F" = Frequent, "O" = Occasional, "R" = Rare. Refer to Table 1 for marina details.

marina no.:	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	no. occurrances
Amphibalanus improvisus (Darwin, 1854)																						A	1
Asterocarpa humilis (Heller, 1878)				R	R																		2
Austrominius modestus (Darwin, 1854)	O	O	R	O	F	O		R	A	O	F	O	O	O	C	O	C	O	F	F	F	A	21
Bonnemaisonia hamifera (Hariot, 1891)				R																			1
Botrylloides diegensis Ritter & Forsyth, 1917					R																		1
Botrylloides sp.†			R	R																			2
Botrylloides violaceus Oka, 1927			O		O	O		O					O						F	O			7
Bugula neritina (Linnaeus, 1758)		O	F	O	C					R										C	R		7
Bugulina simplex (Hincks, 1886)			R					R											F				3
Bugulina stolonifera (Ryland, 1960)		R				R											R						3
Caprella mutica Schurin, 1935	A	R	O	A	R	F				S	R				R		R						10
Codium fragile subsp. *fragile* (Suringar) Hariot, 1889		C	O																				2
Colpomenia peregrina (Sauvageau, 1927)		R	R	R																			3
Corella eumyota Traustedt, 1882			R	R				R									R						4
Didemnum sp.†	R	R	R	R	R									R						O			7
Didemnum vexillum Kott, 2002*	R		R	R		F				C													5
Ficopomatus enigmaticus (Fauvel, 1923)				R																		C	2
Grateloupia turuturu Yamada, 1941				O																			1
Magallana gigas (Thunberg, 1793)*	O	O																		R	F		4
Mya arenaria (Linnaeus, 1758)																						R	1
Perophora japonica Oka, 1927			F			O																	2
Potamopyrgus antipodarum (Gray, 1843)																						O	1
Sargassum muticum (Yendo) Fensholt, 1955*		O	R							C		O	R					R					6
Styela clava Herdman, 1881*				R								R	O				R			R			5
Tricellaria inopinata (d'Hondt & Occhipinti Ambrogi, 1985)		O	F	F	F	F				O							A		C	O			9
Undaria pinnatifida (Harvey) Suringar, 1873*				O				C												C			3
Watersipora subatra (Ortmann, 1890)				R	R																		2
Total no. NNS:	**4**	**9**	**12**	**14**	**8**	**7**	**0**	**5**	**1**	**6**	**2**	**3**	**4**	**1**	**2**	**1**	**6**	**2**	**4**	**7**	**3**	**5**	

†Species not included in total non-native species count due to uncertainty in identity.
*Species on Ireland's High Impact invasive species list.

Occasional, 1 = Rare. Abundance scores for each species across marina sections both in the field and laboratory were averaged and presented for each marina.

Results and discussion

General results

Overall, 25 non-native and 20 cryptogenic species were recorded across the 22 marinas visited, with 7 ascidians, 6 macroalgae, 5 bryozoans, 2 bivalves, 2 cirripede, 1 amphipod, 1 gastropod and 1 polychaete recorded (Table 2; Figures 1, 2; Supplementary material Tables S1, S2). The most frequently recorded non-native species was Darwin's barnacle, *Austrominius modestus* (Darwin, 1854) (n = 21 sites), with this species also having the greatest abundance, scoring on average, between "occasional" – "frequent" on the SACFOR scale. The marina with the greatest number of non-native species was located in Co. Dublin (n = 14 species), while the marina with the fewest number was in Co. Wicklow (n = 0 species). Of these 25 species, five are on Ireland's High Impact invasive species list (Kelly et al. 2013): the carpet sea squirt, *Didemnum vexillum*, the Pacific oyster, *Magallana gigas* (Thunberg, 1793), wireweed, *Sargassum muticum* (Yendo) Fensholt, 1955, the leathery sea squirt, *Styela clava* Herdman, 1881 and wakame, *Undaria pinnatifida* (Harvey) Suringar, 1873. Four species not previously recorded in published reports in RoI were recorded: devil's tongue weed, *Grateloupia turuturu* Yamada, 1941, the compass sea squirt, *Asterocarpa humilis* (Heller, 1878), the San Diego sea squirt, *Botrylloides diegensis* Ritter & Forsyth, 1917 and the branching bryozoan, *Bugulina simplex* (Hincks, 1886) (Figure 3).

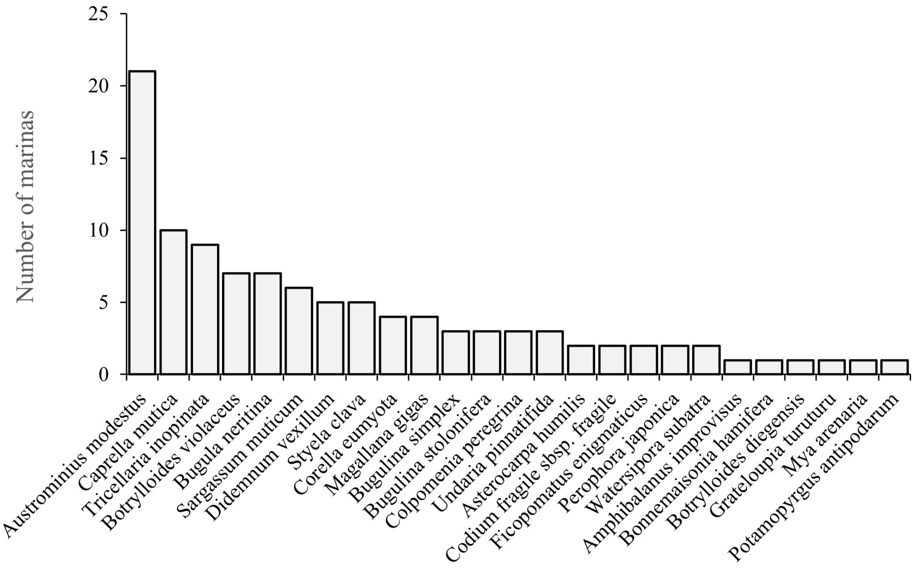

Figure 1. Frequency of non-native species recorded using rapid assesssment surveys in marinas in the Republic of Ireland. A total of 22 marinas were surveyed between August–September 2021.

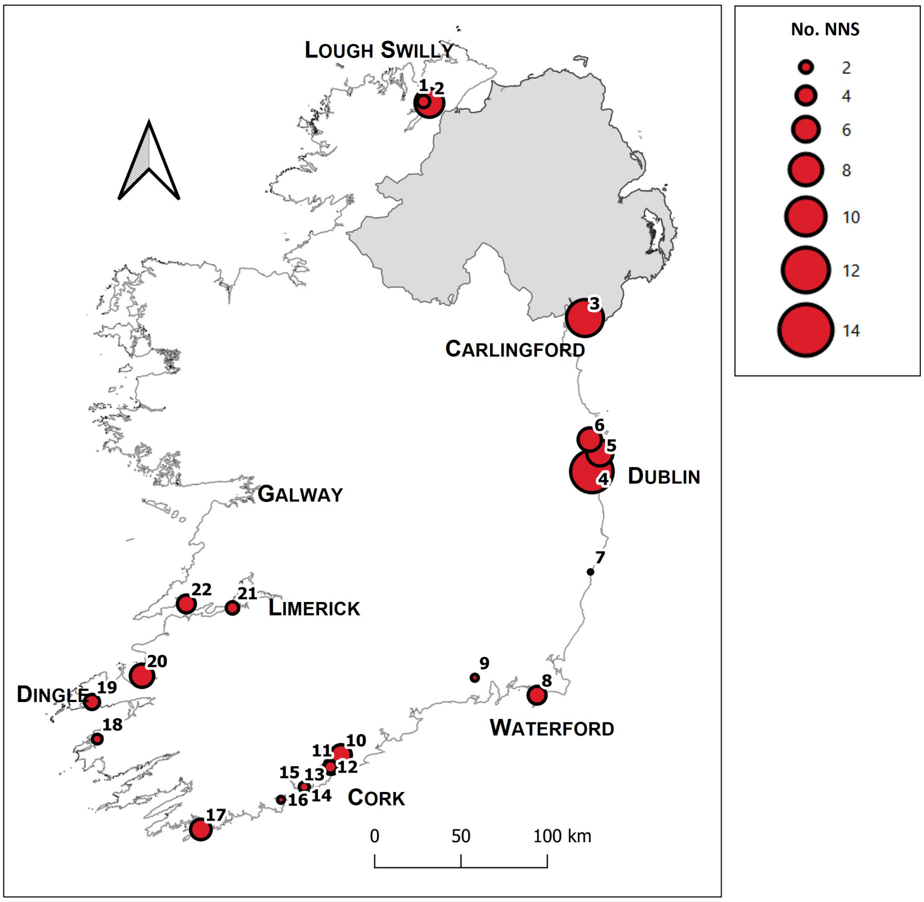

Figure 2. Total number of non-native species ("NNS") recorded by marina from rapid assessment surveys (RAS) carried out in August-September 2021 in the Republic of Ireland. Numbers refer to marina codes (see Table 1). Size of red circles indicate number of non-native species recorded, while the black circle (Marina 7) denotes a survey but no non-natives found.

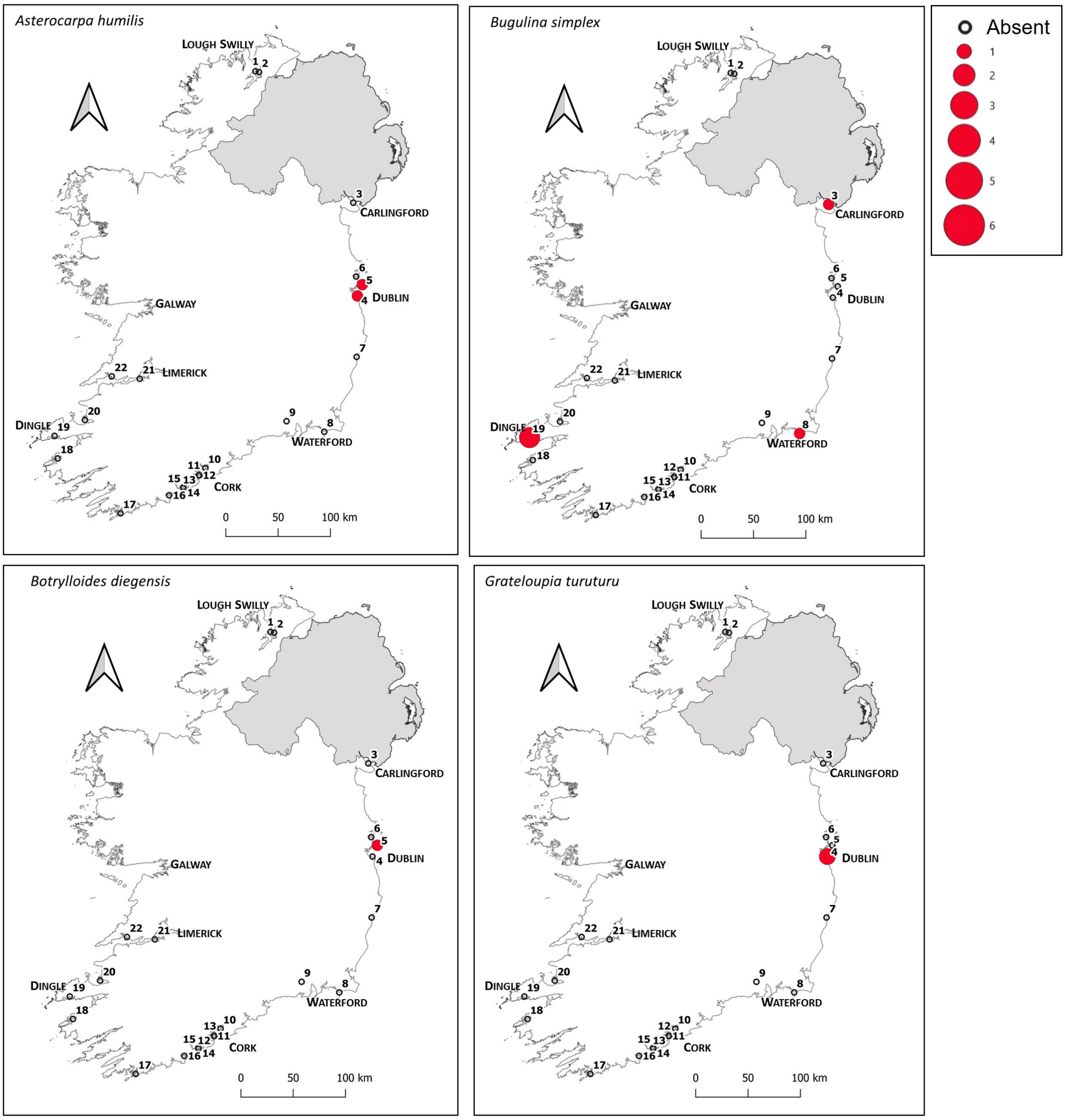

Figure 3. Distribution of new non-native species (species not previously recorded in published literature) in RoI recorded from rapid assessment surveys (RAS) carried out in August-September 2021. Red circles denote presence with the size of the circle representing abundance (key: 1 = rare, 2 = occasional, 3 = frequent, 4 = common, 5 = abundant, 6 = super abundant). Empty circles represent recorded absence of the species (species was searched for but not found). See Table 1 for marina codes (numbers).

Species accounts

Algae

***Codium fragile* subsp. *fragile* (Suringar) Hariot, 1889**

The green alga, *Codium fragile* subsp. *fragile*, was found within marinas in Lough Swilly and Carlingford Lough during the current surveys. Previous reports of this species from the island of Ireland include unpublished records in 2014 and 2018 from Counties Mayo and Galway (Biodiversity Maps 2022; NBN 2022a) as well as several records from Northern Ireland

(NBN 2022a). Bunker et al. (2017) reported *C. fragile* subsp. *fragile* as widespread on all Irish coasts. There is a lack of information on *C. fragile* subsp. *fragile* by this name in the published literature (but see Trowbridge 2001), although this may be due to identity confusions between this species and *C. fragile* subsp. *atlanticum* (native), with Bunker et al. (2017) suggesting that the geographic distribution of these species undergo reassessment. Our records represent the northernmost reports of *C. fragile* subsp. *fragile* in RoI.

***Grateloupia turuturu* Yamada, 1941**

This study recorded the red alga, *Grateloupia turuturu*, commonly known as devil's tongue weed, from a marina in Dublin (Figure 3), which is the first published record of this species on the island of Ireland and the first record in RoI. This species was locally abundant on a small number of floating pontoons which were located near the entrance of the marina and was not found from any other location in the marina. *Grateloupia turuturu* is known to originate from the Pacific, but nearest records of this species include several records from Belfast Lough in Northern Ireland in 2017 and every year since, as well as Mill Bay Larne in 2020 (NBN 2022b; J Nunn *pers. comm.*). It was first recorded in England in the late 1960s (Farnham and Irvine 1979), and more recently was observed from eight sites from Brighton to Exmouth in southern England (Arenas et al. 2006). In a review of non-native species within Scottish marinas, *G. turuturu* was not found in any published reports (Foster et al. 2016). This species has been previously misidentified as *G. doryphora* (Montagne) Howe (Gavio and Fredericq 2002), but no records under that name were found in RoI. The presence of this species in one marina in which it was only locally abundant along a small number of pontoons suggests it may be a relatively recent arrival.

***Undaria pinnatifida* (Harvey) Suringar, 1873**

The first record of *U. pinnatifida* on the island of Ireland was in 2012 from Belfast Lough (Minchin and Nunn 2014), and later from Glenarm in Northern Ireland (J Nunn *pers. comm.*) and in 2016 from Kilmore Quay in Co. Wexford (RoI) (Kraan 2017). This species has subsequently been recorded from Belfast Lough nearly every year since 2017 (J Nunn *pers. comm.*). Chan et al. (2021) recently reviewed the known records of *U. pinnatifida* from Ireland and provided a new record from Co. Wicklow. Thus, up to 2020, six records of *U. pinnatifida* were known from the island of Ireland, including two records in Northern Ireland. The current study also found *U. pinnatifida* attached to floating pontoons in a marina in Kilmore Quay, confirming its continued presence here, and provides new records from Dublin and Tralee Bay (west coast), representing the first published records of this species on Ireland's west coast. This species is on Ireland's

High Alert species list (Kelly et al. 2013), as it is known to spread rapidly in introduced areas. For example, *U. pinnatifida* was first described from England in 1994 attached to floating pontoons in the Hamble Estuary within the Solent (Fletcher and Manfredi 1995) and has subsequently become widespread and common along the south coast of England. Here, it is the dominant kelp within most marinas, outcompeting other habitat forming species (Farrell and Fletcher 2006; Heiser et al. 2014) with there being growing concern of "spillover" of this species into natural habitats (Epstein and Smale 2018) where its impact on native ecosystems is largely unknown.

Ascidians

***Asterocarpa humilis* (Heller, 1878)**

The unitary ascidian, *A. humilis*, commonly known as the compass sea squirt, was recorded from two marinas in the Dublin area (Figure 3), representing the first records of this species in RoI and the first published records of this species for the island of Ireland. The nearest records of this species come from Northern Ireland in Strangford Lough in 2020, Belfast Lough every year since 2017 and Rathlin Island in 2017, 2018 and 2021 (J Nunn *pers. comm.*, NBN 2022c). The first report of this species in the Northern Hemisphere was from Brittany, France, in 2005, with subsequent observations in south and southwest England in 2010 and 2011 and in north Wales in 2011 (Bishop et al. 2013). It was later observed in Oban (Scotland) in 2013 (Nall et al. 2015) and the Orkney Islands in 2014 (Kakkonen et al. 2019). Bishop et al. (2015a) documented this species at a single marina in southwest England in 2010, and when surveying the same marinas in 2013, recorded it at ten marinas, demonstrating that *A. humilis* can spread quickly once introduced and established. In the current study, abundances of *A. humilis* at both marinas were "rare", and in combination with its presence within only two marinas within close proximity to each other (approximately 10 km), possibly indicates a relatively recent arrival; although lack of targeted monitoring for this species in combination with its cryptic form means it could have been present but overlooked for some time (Bishop et al. 2013).

***Botrylloides diegensis* Ritter & Forsyth, 1917**

Botrylloides diegensis, commonly known as the San Diego sea squirt, has only relatively recently been recognised as a non-native species in the northeast Atlantic but has been present since at least 2004 (Bishop et al. 2015b) and is established along southern and south-eastern coasts of England. There are no previously published records of *B. diegensis* on the island of Ireland. In the current study, small colonies of *B. diegensis* were found attached to floating pontoons at a single marina in Dublin (Figure 3). It is not clear whether the species is newly arrived in Ireland or whether it has previously been overlooked due to confusion with other species. For instance, the single colour morphs of *B. diegensis*, which resemble *B. violaceus* in external appearance, would require genetic analysis for positive identification (Bishop et al. 2015b). This species' apparent absence from other marinas visited in the current survey, however, suggests that it is currently limited in distribution in Ireland.

***Botrylloides violaceus* Oka, 1927**

As with *Botrylloides diegensis*, it is possible that the colonial ascidian, *Botrylloides violaceus*, has been overlooked in the past due to confusion with other species and that its current distribution in Ireland is underestimated. Minchin (2007a) provided the first published Irish records of *B. violaceus* within marinas in Carlingford Lough and Malahide Estuary in 2005 and 2006. Additional records reported on National Biodiversity Network Atlas include reports from Northern Ireland in Strangford Lough in 2012 and 2017 (J Nunn *pers. comm.*), Belfast Lough in 2011, 2017, 2020, 2021 and 2022 and Carlingford Lough in 2009 (NBN 2022e). In the present survey, this species was confirmed as still present within the same two marinas reported by Minchin (2007a), and new records were made from marinas in Dublin, Cork Harbour and Kilmore Quay (east coast) as well as on the west coast in Dingle Bay and Tralee Bay, representing the first reports of this species on the west coast of Ireland. Colonies observed in the marina in Dingle Bay, in particular, were concerning as they were the dominant fauna on subtidal marina pilings and were also noticeably dominant on some floating pontoon sections.

***Corella eumyota* Traustedt, 1882**

The first records of the orange-tipped sea squirt, *Corella eumyota*, were from marinas in Carlingford Lough, Dublin, Cork Harbour and Tralee Bay (west coast) in 2005 and 2006, representing the first records of this species in RoI (Minchin 2007a). During the same surveys, it was discovered in Belfast Lough (Minchin 2007a), with subsequent surveys in 2012 determining it had spread rapidly and had become widely distributed across marinas in Northern Ireland (Minchin and Nunn 2013). It has also been found widely distributed around Mullet in Co. Mayo in 2018 (Nunn 2021). The current survey found *C. eumyota* in the same RoI locations with the exception of Cork Harbour, but also on floating pontoons in Kilmore Quay (east coast) and Baltimore (south coast), with the Baltimore record representing the southernmost published report of *C. eumyota* on the island of Ireland (although unpublished records indicate its presence in Lough Hyne and intertidally at Baltimore Quay, Clear Island and Sherkin Island (J Nunn *pers. comm.*)). Abundance of all records of *C. eumyota* from the current study were "rare". This, along with the apparent disappearance of this species from the marina in Cork Harbour in 2021 does not fully align with the reports of abundance from 2005–2006, as Minchin (2007a) reported *C. eumyota* individuals as forming extensive clusters and layers on floating pontoons and hulls of boats. Possible explanations for this reduction in abundance could be the cleaning of pontoons and subsequent inability of *C. eumyota* to recolonise space it had previously dominated. The marina in Cork Harbour where it was not recorded during the current surveys supported large colonies of *D. vexillum* (scored as "common" on the abundance scale), which was the dominant fauna on several sections of floating pontoons, perhaps reducing the ability for *C. eumyota* to recolonise.

Didemnum vexillum **Kott, 2002**

In surveys conducted in 2005 and 2006 across the island of Ireland (Minchin 2007a), colonies initially identified as *Didemnum* sp. were found in a marina in Malahide Estuary (2005) and a marina in Carlingford Lough (2006), both located on the east coast of the island (Minchin and Sides 2006; Minchin 2007a). Due to its sudden appearance in areas that it had not been previously found and association with marinas, it was suspected that the species was introduced, and its unusual growth form matched with specimens recently reported from France and the Netherlands (Minchin and Sides 2006). These records were later identified by Lambert (2009) as *Didemnum vexillum*, commonly known as the carpet sea squirt. Although clearly a recent introduction, the colonies were noted to already be established and highly abundant at both Malahide Estuary and Carlingford Lough. *Didemnum vexillum* was subsequently found attached to oyster trestles in south Galway Bay on the west coast of RoI (Minchin and Nunn 2007). The first record of this species in the north was from Strangford Lough in 2012 (Minchin and Nunn 2013) where it is still present to this day (J Nunn *pers. comm.*). In the current surveys, colonies of *Didemnum* were found at the same two marinas from (Minchin 2007a) and were confirmed as *D. vexillum* via examination of mature larvae as well as the morphology of the spicules. In addition to Malahide Estuary and Carlingford Lough, confirmed colonies of *D. vexillum* were also found within marinas in Dublin, Cork Harbour and Lough Swilly during the current surveys. Past reports of this species from the island of Ireland were from Galway Bay and Clew Bay on the west coast (McKenzie et al. 2017; Prentice et al. 2021), Dunmanus Bay (Prentice et al. 2021) and Strangford Lough in Northern Ireland (Minchin and Nunn 2013). Records of *D. vexillum* from the current study provide the first accounts of this species in Lough Swilly and Cork Harbour. Recently, Prentice et al. (2021) provided evidence of multiple introductions of *D. vexillum* to Britain and Ireland, and it is not clear if the Lough Swilly and Cork Harbour records represent spread within Ireland or discrete introductions from another source population.

Perophora japonica **(Oka, 1927)**

The stoloniferous ascidian, *P. japonica*, commonly known as the creeping sea squirt, was recorded from marinas in Carlingford Lough and Malahide Estuary during the current survey but was not found in these locations despite being searched for in surveys conducted in 2005–2006 (Minchin 2007a). The first records of this species on the island of Ireland were from Strangford Lough in Northern Ireland and Carlingford Lough in RoI in 2012 (Minchin and Nunn 2013), with subsequent records from those locations reported relatively consistently with the most recent records in 2021 (NBN 2022f; J Nunn *pers. comm.*). Subsequent surveys in 2015 found this species in Mulroy Bay on the north coast of RoI in 2013 and Clew Bay on the west coast of RoI (Minchin et al. 2016). Our record from Malahide Estuary represents the southernmost record on the island and indicates the species is expanding its range south along the east coast of Ireland. Previous to records of *P. japonica* in Ireland, the first record of this species in the Atlantic was from northwest France in 1982 where it spread rapidly and was observed in high abundance within marinas and coastal areas (Monniot and Monniot 1985). This species was later recorded in Britain, first from Plymouth in 1999 (Nishikawa et al. 2000) and soon after from the Fleet Lagoon along the south coast of England (Baldock and Bishop 2001).

Bryozoans

Bugula neritina **(Linnaeus, 1758)**

The warm-temperate and subtropical bryozoan, *Bugula neritina*, was found at seven sites in the current study, which included marinas in Lough Swilly in the north, Carlingford Lough and Dublin along the east coast, Cork Harbour in the southeast and Tralee Bay and the Shannon Estuary in the west. Abundances were "rare" to "frequent" except for populations in Dublin and Tralee Bay, which were higher ("common"). Previous reports of this species in RoI include archived material from a marina in Malahide Estuary taken during surveys in 2006 (Minchin 2007a) and later confirmed as *B. neritina* (Ryland et al. 2011; Porter et al. 2017), with subsequent observations in RoI from Carlingford Lough in 2008 (Ryland et al. 2011). In Northern Ireland, the species was first described from Belfast Lough in 2012 (Minchin and Nunn 2013; Porter et al. 2017) and has been recorded here regularly since with the most recent record being in 2022 (J Nunn *pers. comm.*). In 2013, Kelso-Maguire (2020) reported *B. neritina* for the first time from a marina in Dublin. The results from the current study concur with past reports in RoI with the exception of the report from Malahide Estuary (Ryland et al. 2011; Porter et al. 2017). It is possible the species was in low enough abundances to go undetected in the current study in Malahide; however, Ryland et al. (2011) reported the disappearance of *B. neritina* from England from the 1970s to the early 2000s, suggesting the cause to be a reduction in water temperatures. This demonstrates that perhaps the species is susceptible to fluxes in temperatures which may affect the population to the point of local extinction. Our survey reports the first published record of this species on the west coast of RoI.

Bugulina simplex **(Hincks, 1886)**

The current study recorded the branching bryozoan, *Bugulina simplex*, from marinas in Carlingford Lough, Kilmore Quay (east coast) and Tralee Bay (west coast), with abundances ranging from "rare" to "frequent". Little published information of this species exists in RoI. A summary of results from surveys in Britain and Ireland, in which surveys were conducted in Malahide Estuary and Carlingford Lough in 2006 and 2008, did not report *B. simplex* from either location (Ryland et al. 2011). Additionally, it was not detected in a survey of bryozoans from Dun Laoghaire Harbour in Dublin in 2011 (Kelso and Wyse-Jackson 2012). In Northern Ireland, it was first described from Bangor Marina in 2011 and later was reported from five sites in 2012 (Porter et al. 2017). In Britain, it was first described from Wales by Ryland (1958) and has since spread along most coasts in Britain (Ryland et al. 2011; Bishop et al. 2015b; Nall et al. 2015). To our knowledge, *B. simplex* has not been previously recorded in RoI, and our records provide the first published report of the species here.

Bugulina stolonifera **(Ryland, 1960)**

The branching bryozoan, *Bugulina stolonifera,* was found within marinas in Lough Swilly, Dublin and Baltimore (south coast). The distribution of this species is poorly understood, with a lack of published information available from the island of Ireland, although Ryland (1960) included Cobh (Co. Cork) within its known geographic distribution. Records in the National Biodiversity Network Atlas (NBN 2022g) include one accepted report from Kenmare Bay in southwest Ireland in 2007, as well as two verified records in Northern Ireland in 2000 and 2014. The paucity of information available for *B. stolonifera* may be attributed to the absence of this species on target lists of non-native species surveys as well as challenges in identification and confusion with similar species such as *B. avicularia* (Ryland et al. 2011). To our knowledge, this is the first published report of *B. stolonifera* from the island of Ireland since Ryland (1960) and indicates a large range expansion since the initial report from Cork.

Tricellaria inopinata **d'Hondt & Occhipinti Ambrogi, 1985**

The current study found *T. inopinata* at nine locations across Ireland, including marinas in Lough Swilly, Carlingford Lough, Dublin, Cork Harbour, Baltimore, Dingle Bay and Tralee Bay. Previous records from the island of Ireland include published reports from Northern Ireland in 2011 and 2012 (Nunn et al. 2012; Minchin and Nunn 2013) and Dublin in 2011 (Kelso and Wyse-Jackson 2012). In a review of information to determine current European distribution of *T. inopinata*, Cook et al. (2013) provided records of this species from Malahide Estuary in 2006, Cork Harbour in 2009, Carlingford Lough in 2009 and Tralee Bay in 2011. Thus, our records, in combination with previous reports, suggest the species is becoming widespread along Irish coasts.

***Watersipora subatra* (Ortmann, 1890)**

This study found *W. subatra*, commonly known as the red ripple bryozoan, at two marinas in the Dublin area. These records concur with a bryozoan survey of one of these marinas in 2011 in which *W. subatra* was recorded in low abundance (Kelso and Wyse-Jackson 2012), and in follow-up surveys, was reported in noticeably higher abundance (Kelso-Maguire 2020). The first record for Northern Ireland was from in a marina in Ardglass Harbour (east coast) in 2012 (Minchin and Nunn 2013; Porter et al. 2017). In RoI, there are no known records of this species outside of Dublin. *Watersipora subatra* was found in the Channel Islands in 2007 (Ryland et al. 2009b) and was first described from England in 2008 from Plymouth and Poole, and later recorded from Cornwall to Sussex (Bishop et al. 2015b; Wood et al. 2015; O'Shaughnessy et al. 2020). Given its current invasion history in England, Bishop et al. (2015b) predicted that *W. subatra* would expand noticeably in terms of prevalence and geographic range and this trend may also be expected in Ireland.

Molluscs

***Magallana gigas* (Thunberg, 1793)**

In the early 2000s, only occasional settlements of *M. gigas* had been reported in the wild in Ireland (Boelens et al. 2005), with no known established populations despite a substantial oyster fishery in Ireland (Miossec et al. 2009). The current surveys found individuals of *M. gigas* attached to floating pontoons and intertidal marina pilings in low abundance ("occasional") in the north of Ireland (Lough Swilly), very low abundance ("rare") in Tralee Bay on the west coast and moderate abundance ("frequent") in the Shannon Estuary (west coast). The north of Ireland is known to support 25% of the country's oyster fishery (BIM 2019), perhaps explaining the settlement of individuals in Lough Swilly. However, the west coast supports only 4% of the fishery (BIM 2019), suggesting that low propagule pressure from the small number of farms in this area can facilitate settlement and/or these individuals were transported from distant areas as larvae.

Polychaetes

***Ficopomatus enigmaticus* (Fauvel, 1923)**

This study found the trumpet tube worm, *F. enigmaticus*, within marinas in Dublin and Kilrush (west coast). Abundance was "rare" in Dublin, with only a few live individuals observed on a floating pontoon and the hull of a boat, both located in the inner part of the marina. In Kilrush, however, abundance was much greater, with *F. enigmaticus* observed commonly on floating pontoons and found completely fouling many submerged materials such as boat lines and buoys. This species was detected in Kilrush in a survey of this marina in 2005–2006, and additionally, at a site in the Avoca River where, in the current survey, pontoons were observed to support no macroflora/fauna. The first record of this species on the island of Ireland is from Cork in 1971 (Kilty and Guiry 1973) and has since been recorded from Northern Ireland in 2015 (NBN 2022h).

Pathways and vectors of introduction and spread

Several pathways and vectors have been identified that facilitate the introduction and spread of marine invasive species. Global shipping is recognised as a major pathway of introduction and spread of species via hull and sea chest fouling (Gollasch 2002; Coutts and Dodgshun 2007) and ballast water exchange (Gollasch 2008; Bailey 2015). Additionally, aquaculture and recreational boating are also key pathways as a result of deliberate introductions, contaminations and hitchhikers (Critchley et al. 1984; Minchin 2007b; Darbyson et al. 2009; Clarke Murray et al. 2011). An assessment of key introduction vectors in Ireland showed Dublin as a hotspot for species introduction due to commercial shipping and recreational boating (Tidbury et al. 2016). Costello et al. (2022) carried out an analysis of cargo shipping records to determine connectivity of shipping routes between ports in Ireland and international ports in order to facilitate horizon scanning for marine species likely to invade the island. Their analysis showed high connectivity between Ireland and the UK, Western Europe and Scandinavia, and found that Dublin was the primary destination port, with approximately 83% of incoming vessels using the Port of Dublin, followed by Cork Harbour (13%), New Ross (2.6%) and Rosslare (1.5%). In the current study, the Dublin area supported the greatest number of non-native species (n = 18), while Cork Harbour supported fewer species (n = 8) (Figure 2). Furthermore, three of the four new records from the current surveys were reported from marinas in the Dublin area (*A. humilis*, *B. diegensis* and *G. turuturu*).

Secondary spread of marine non-native species is often attributed to recreational vessels, such as yachts and sail boats, distributing species away from primary ports of introduction through local or regional voyages (Clarke Murray et al. 2011). As such, recreational watercraft in popular recreational harbours such as Dublin, which supports international watercraft, may facilitate the spread of species to other harbours around Ireland that do not support regular international traffic. This is a potential explanation for the relatively high number of non-native species recorded on the west and north coasts in Tralee Bay and Lough Swilly, respectively.

Several of the marinas surveyed in this study are located within water bodies that support oyster and mussel aquaculture facilities (DAFM 2022). As aquaculture activities are known to facilitate introduction and spread of associated species (Naylor et al. 2001), movement of aquaculture equipment from one water body to another within Ireland might also facilitate the secondary spread of non-native species around Ireland. In this study, we found that sites with the greatest number of non-native species outside of the Dublin area support aquaculture activities. For example, 12 and nine non-native species were recorded in Carlingford Lough and Lough Swilly, respectively, both of which support multiple oyster and mussel aquaculture facilities (DAFM 2022). Under the Invasive Alien Species Regulations (1142/2014) Member States are required to undertake a pathway analysis followed by the development of pathway action plans for priority pathways. Evidence presented here indicates that the movement of aquaculture stock and equipment may constitute a significant pathway for which a pathway action plan could be developed.

Concluding remarks

Although the RAS approach is often reliable in the detection of fouling species such as sessile invertebrates, macroalgae and associated mobile fauna on hard structures such as marina pilings and floating pontoons, it is not able to capture soft-bottom habitat, benthic and planktonic species. Thus, the list of non-native and cryptogenic species provided within this study may not be a complete list of these species within marinas surveyed. Diverse sampling techniques including plankton tows and soft-bottom habitat sampling such as intertidal core sampling, crab/lobster traps and/or fykes nets would be needed to capture the full suite of species of concern (Kakkonen et al. 2019). As such, it is likely that some non-native species in these under-sampled habitats are under-reported or have gone undetected completely.

In this study, the new species and new localities suggest that consistent and regular monitoring of marinas is needed to understand abundance, distribution and rates of spread of non-native species along Irish coasts. Marinas will likely form a component in future studies of hotspots—geographic areas identified as supporting activities known to facilitate the movement of species—of introduction and spread of marine invasive species (Tidbury et al. 2016). As such, monitoring these hotpots and proximate natural habitats is needed to provide key information on the current baseline and distribution of invasive species. This information can be used by several stakeholder groups such as government agencies, port authorities and marina operators for early detection of horizon species in order to maximise success in preventing establishment of and containing newly arrived species via rapid response processes as well as evidence the need for biosecurity and track the effectiveness of implemented measures.

Regular surveying of marinas for invasive species, using easily applied, repeatable and effective approaches such as the RAS method should be a key component of any integrated management programme for marine invasive species.

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Authors' contribution

KAO contributed to sample design and methodology, investigation and data collection, data analysis and interpretation and writing original draft and reviewing and editing the manuscript; DL contributed to research conceptualization, funding provision and reviewing and editing the manuscript; CWA contributed to sample design and methodology, investigation and data collection and writing original draft and reviewing and editing the manuscript; RC contributed to investigation and data collection and reviewing and editing the manuscript; SP contributed to investigation and data collection, data analysis and interpretation; ET contributed to reviewing and editing the manuscript; RD contributed to investigation and data collection and reviewing and editing the manuscript; PDS contributed to sample design and methodology and writing original draft and reviewing and editing the manuscript. All authors contributed to the revision of the manuscript.

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Supplementary material

The following supplementary material is available for this article:

Table S1. SACFOR abundance of each non-native species ('NNS') recorded by marina.
Table S2. SACFOR abundance of each cryptogenic species recorded by marina.

This material is available as part of online article from:
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