Spatial and temporal monitoring of invasive *Hydroides dianthus* (Verrill, 1873) (Annelida, Serpulidae) in Eel Lake, Argyle, Nova Scotia using a species-specific molecular assay

Stephanie A. Hall¹*, Sarah E. Stewart-Clark¹ and Elena Kupriyanova²,³

¹Department of Animal Science and Aquaculture, Dalhousie Agriculture Campus, Truro, Nova Scotia, Canada, B2N 5E3
²Australian Museum Research Institute, Australian Museum, 1 William Street, Sydney, 2010 NSW Australia
³Department of Biological Sciences, Macquarie University, Sydney, NSW 2109, Australia

*Corresponding author
E-mail: Stephanie.hall@dal.ca

Abstract

In 2012, an invasion by a serpulid tubeworm, unknown to the Atlantic Canada region, caused heavy biofouling on an oyster lease in Eel Lake, Argyle, Nova Scotia. The species was identified as *Hydroides dianthus* (Verrill, 1873), and this morphological identification was confirmed by comparing a newly sequenced COI gene fragment with *H. dianthus* sequences available in GenBank. Heavy biofouling on the oyster lease resulted in the need to develop mitigation strategies. It was hypothesized that by understanding the spawning behaviour and spatial/temporal patterns of *H. dianthus* larvae in Eel Lake, biofouling could be mitigated by determining an optimal depth that oyster cages should be maintained at during its active spawning period. To monitor *H. dianthus* in Eel Lake, species-specific primers associated with the COI gene were developed. Specificity and sensitivity of primers were tested, and the primer set *H. dianthus* COI4F/R was chosen for monitoring. Water samples were collected from Eel Lake from June to August 2013, and eDNA was extracted. Spatial and temporal monitoring of *H. dianthus* using eDNA was completed at four sites (three fouled, one non-fouled), and at three depths (0.3 m, 1.5 m, and 3.0 m) in Eel Lake. Water samples tested positive for *H. dianthus* in most sites and depths on June 20th, indicating a spawning event had occurred. Subsequently, no significant differences between sites and depths were found and *H. dianthus* was detected throughout the summer in all locations and depths monitored. The mean % of positive PCR results significantly increased from 18.7 to 65.0% from June to August. Results suggest that *H. dianthus* spawned frequently during a reproductive season ranging from at least June 20 to August 30 in Eel Lake at temperatures ~18–22 °C. The results of this study indicated that an optimal oyster cage depth for biofouling mitigation could not be determined and highlighted the challenges to aquaculture associated with biofouling of *H. dianthus*.

Key words: biofouling, environmental DNA (eDNA), cytochrome c oxidase gene (COI), invasive species

Introduction

The shellfish aquaculture industry in Atlantic Canada is facing significant challenges with biofouling organisms, especially invasive ascidians (tunicates) such as *Botryllides violaceus* Oka, 1927, *Styela clava* Herdman,
Monitoring of invasive *Hydroides dianthus* in Eel Lake, Argyle, Nova Scotia

Hall et al. (2022), Management of Biological Invasions 13(2): 326–343, https://doi.org/10.3391/mbi.2022.13.2.05

1881, *Botryllus schlosseri* Pallas, 1766, and *Ciona intestinalis* Linnaeus, 1776 (Locke et al. 2007; Ramsay et al. 2009; Sephton et al. 2011, 2016; Simard et al. 2013; McKenzie et al. 2016). Polychaetes in the family Serpulidae Rafinesque, 1815 belong to another important group of biofouling organisms that impact the aquaculture industry (Yan et al. 2008; Ma et al. 2009). Despite 20 species of Serpulidae being documented in Atlantic Canada (Carr 2012), none have been a source of significant biofouling on aquaculture leases in Atlantic Canada. However, in the summer (June to August) of 2012, a serpulid heavily fouled Eel Lake, Argyle, Nova Scotia (NS), Canada. Eel Lake contains brackish water (~ 12–22 ‰), with its saltwater supply in-flowing from strong tides of the Bay of Fundy. The serpulid species responsible for the biofouling was not reported before in the Atlantic Canada region. Taxonomists from Fisheries and Oceans Canada identified the serpulid as a *Hydroides* spp. and consulted Dr. Harry ten Hove (Naturalis, Leiden, the Netherlands), who confirmed the serpulid as *Hydroides dianthus* (Verrill, 1873) (pers. comm. between Dawn Sephton and Sarah Stewart-Clark 2013).

*Hydroides dianthus* belongs to the genus *Hydroides* Gunnerus, 1768 with over 100 species, five of which are confirmed invasive biofoulers, including *H. dianthus*, *Hydroides dirampha* Mörch, 1863, *Hydroides elegans* (Haswell, 1883), *Hydroides ezonesis* Okuda, 1934, and *Hydroides sanctaecrusis* Kroeyer [in] Mörch, 1863 (Capa et al. 2021). The reported range for *H. dianthus* spans as far north as Maine, USA and as far south as the coast of Florida and Texas (Hedgpeth 1950; Bastida-Zavala and Salazar-Vallejo 2000; Bastida-Zavala and ten Hove 2002; Trott 2004). However, *H. dianthus* has been documented in many geographical locations globally (Sun et al. 2017). The introduction of invasive biofouling species globally has been accredited to shellfish transfers (Mckindsey et al. 2007), shipping vectors (ballast water systems and hull fouling) (Ruiz et al. 2011; Sylvester et al. 2011; Briski et al. 2012) and recreational boats with hull fouling (Darbyson et al. 2009; Davidson et al. 2010). These invasions can be facilitated by propagule pressure and anthropogenic changes in environmental conditions (Locke et al. 2017).

Heavy biofouling on shellfish aquaculture leases causes equipment and gear to be weighed down and can negatively affect growth and condition in shellfish (Braithwaite and McEvoy 2005). This ultimately leads to economic losses to the shellfish aquaculture industry through increased cost of maintenance through mechanical/chemical removal, waste removal, and labor of harvesting, transporting, and processing (Fisheries and Oceans 2006). The cost of chemical and physical removal of biofouling is estimated to be between 5–10% of industry labor costs, or between US $1.5–3 billion per year for the industry globally (Fitridge et al. 2012). *Hydroides dianthus* has been found to settle on aquaculture infrastructure and shellfish, and secrete calcium carbonate tubes that the worms reside in for the remainder of its lifecycle (Wisely 1958; Scheltema et al. 1981). The removal of the
calcium carbonate tube from the shellfish and equipment increases the labour costs for shellfish growers (Fitridge et al. 2012). Even if the worms are successfully removed, they may have impacts on the aesthetics of the shell which can result in reduced shellfish value (Watson et al. 2009; Dürr and Watson 2010).

In the summer of 2012, heavy biofouling of *H. dianthus* on oysters and cages in Eel Lake resulted in increased labour costs associated with removal of tubeworms. Therefore, mitigation strategies to prevent biofouling from *H. dianthus* were sought. In the suspended oyster aquaculture industry, floating bags or OysterGro® cages are flipped periodically to expose biofouling organisms to air which results in desiccation (Mallet et al. 2009). This flipping method has been found effective for removing biofouling of barnacles, mussels, and tunicates (Mallet et al. 2009; Government of New Brunswick 2018). However, initial field observations found *H. dianthus* particularly problematic at Eel Lake, as desiccation alone did not remove biofouling from oysters and cages.

Serpulid larvae are known to exhibit photoresponse. For example, trochophores of *H. ezoensis* have been found to exhibit photopositive behaviour, whereas older larval stages exhibit photonegative responses (Miura and Kajihara 1981; Kupriyanova et al. 2001). Other marine invertebrates also exhibit photoresponse. Thorson (1964) found that out of 141 marine bottom invertebrates studied, 82% of early pelagic larvae migrated to light in surface water (photopositive), while later in their pelagic life-stage they migrated away from light (photonegative). Little is known about *H. dianthus’* photoresponse, other than they have been reported to settle in illuminated areas of containers in laboratory experiments (Zeleny 1905; Kupriyanova et al. 2001), therefore, more research is warranted on this subject to understand their behaviour in a natural setting. In addition to flipping, oyster bags/cages have the ability to be floated on surface waters or sunk during storms or when sea ice is present (Doiron 2008). In this study it was hypothesized that it might be possible to hold oyster cages at an optimal depth when *H. dianthus* spawning was occurring, in order to prevent heavy biofouling. Therefore, knowledge of the frequency of spawning and larval depth dispersal of *H. dianthus* in Eel Lake would be required to determine feasibility of such a depth-based management strategy.

The objective of this study was to monitor *H. dianthus* over the summer of 2013 in Eel Lake, NS to better understand spatial and temporal distribution of larvae at different depths and sites. Distribution of *H. dianthus* in Eel Lake was monitored by development of a molecular assay capable of detecting *H. dianthus* in environmental water samples (eDNA). The use of molecular assay to detect larval aquatic organisms can be optimal, as identifying different species of planktrophic larvae using visual morphological taxonomy is difficult (Darling and Mahon 2011).
Molecular assays are being used to track and monitor invasive species globally (e.g., Stewart-Clark et al. 2009, 2013; Willis et al. 2011; Egan et al. 2015; Ardura et al. 2015; Dougherty et al. 2016; Thomas et al. 2020; LeBlanc et al. 2020). In Atlantic Canada, this approach has been successfully used to detect invasive tunicates S. clava, C. intestinalis, B. schlosseri, B. violaceus and Diplosoma listerianum (Milne-Edwards, 1841) in Prince Edward Island (PEI), Canada, in several life history stages such as in eggs, larvae, and adults (Willis et al. 2011; Stewart-Clark et al. 2013; Ma et al. 2016). Target invasive tunicates were also detected in PEI through molecular assay in environmental water samples (Stewart-Clark et al. 2009). This method of water sampling is useful in understanding geographical, spatial, and temporal distribution of invasive species. Therefore, the molecular assay in this study was used to understand H. dianthus larval spatial and temporal patterns, to identify peak spawning periods and preferential depths that could be useful as a method of biofouling mitigation.

Materials and methods

Sample collection and DNA extractions

Samples of adult H. dianthus were collected from cages and oysters from Eel Lake, Argyle, NS in March 2013. The worms were preserved in 100% ethanol and extracted from their tubes using a scalpel blade. DNA was extracted from eight abdominal tissue samples (~10–15 mg) using the Qiagen DNeasy Blood and Tissue Kit (Qiagen, Hilden Germany) according to the manufacturer’s instructions.

Amplification of mitochondrial cytochrome c oxidase subunit I gene

A fragment of the mitochondrial gene cytochrome c oxidase subunit I (COI) was amplified in H. dianthus using Hydro-COIF/Hydro-COIR primer set (Sun et al. 2012). Polymerase Chain Reaction (PCR) was performed in a 50 µl reaction, containing 25 µl of AmphiTaq Gold™ 360 Master Mix (Applied Biosystems™, Massachusetts, USA), 2 µl of the forward primer, 2 µl of the reverse primer, 19 µl of sterile molecular grade water, and 1 µl of template DNA. The COI gene was amplified using the following thermocycler protocol: 94 °C for 5 minutes, 5 cycles of 94 °C for 30 seconds, 47 °C for 30 seconds and 72 °C for 40 seconds. After the initial 5 cycles, 30 additional cycles were completed at 94 °C for 30 seconds, 51 °C for 30 seconds and 72 °C for 40 seconds. Then, the PCR amplicons underwent elongation for 7 minutes at 72 °C. PCR products were separated using agarose gel electrophoresis on a 2% agarose gel containing 0.1 µl/ml of SYBR™ Safe Gel Stain (Invitrogen™, Massachusetts, USA). The gel was run at 135 V for 45 minutes, then visualized using ultraviolet light in the Gel Doc™ XR + Molecular Imager® with BIO-RAD Image lab 5.1 software (California, USA).
Sequencing and species confirmation

The amplified COI gene fragment was sequenced using Guelph Laboratory Services, University of Guelph, Ontario, Canada. The amplicon of one specimen was sequenced in both directions, and a consensus strand was created using ClustalW 2.0 software (Larkin et al. 2007). The consensus sequence was used in a blastn search of the National Center of Biotechnology Information (NCBI) GenBank database to confirm species identity of *H. dianthus*.

Species specific molecular assay

Primer development:

Primers were developed from the COI gene fragment isolated from Eel Lake. COI sequences of 10 *Hydroides* species were downloaded from GenBank (Based on May 2013 sequence availability) (Table 1). Eleven sequences were aligned using MEGA6 (Tamura et al. 2013) to determine unique fragments suitable for primer design. Primers were designed using Integrated DNA Technologies (IDT) (Iowa, USA) primer design tool. Each newly developed primer was examined for possible hairpin structures, homodimers, and heterodimers using the built-in quality check within the primer design tool, to ensure its suitability. Primer sequences were searched on NCBI database using the Primer-BLAST tool to check for unintended amplification of non-target organisms.

Efficacy, sensitivity, and specificity testing

Each primer set developed was tested against a positive Eel Lake *H. dianthus* DNA sample and a negative control to determine their efficacy using PCR. Primer sensitivity was tested with DNA samples of four concentrations (100 ng/l, 10 ng/l, 1 ng/l, and 0.1 ng/l) to determine the lowest limit of detection. To evaluate primer set specificity, we used ethanol-preserved tissues of specimens belonging to 10 species of *Hydroides* from the collection of the Australian Museum (Sydney, Australia) (Table 2). DNA was extracted using the Qiagen DNeasy Blood and Tissue Kit (Qiagen, Hilden Germany),

---

**Table 1.** COI sequences of *Hydroides* spp. used to align against *Hydroides dianthus* Eel Lake sequence to develop species specific primers.

| Species                          | Reg. #     | Accession #        | Collection location       |
|----------------------------------|------------|--------------------|--------------------------|
| *Hydroides brachyacantha* Rioja, 1941 | AM W.40537 | JQ885942.1         | Mazatlán, Mexico         |
| *Hydroides crucigera* Mörch, 1863 | AM W.40538 | JQ885947.1         | Mazatlán, Mexico         |
| *Hydroides dirampha* Mörch, 1863  | AM W.40539 | JQ885946.1         | Hong Kong, China         |
| *Hydroides elegans* (Haswell, 1883) | AM W.40540 | JQ885938.1         | Hong Kong, China         |
| *Hydroides ezoensis* Okuda, 1934  | AM W.40544 | JQ885951.1         | Vladivostok, Russia       |
| *Hydroides fusicola* Mörch, 1863  | AM W.40545 | JQ885950.1         | Manazuru, Japan           |
| *Hydroides operculata* (Treadwell, 1929) | AM W.40550 | JQ885949.1         | Hong Kong, China         |
| *Hydroides operculata* (Treadwell, 1929) | AM W.40551 | JQ885948.1         | Hong Kong, China         |
| *Hydroides sanctaecrucis* (Mörch, 1863) | AM W.40549 | JQ885943.1         | Mazatlán, Mexico         |
| *Hydroides sanctaecrucis* Krayer in Mörch, 1863 | AM W.40548 | JQ885944.1         | Hong Kong, China         |
following the manufacturer’s instructions. Initially, PCR amplification was completed using Hydro-COIF/Hydro-COIR primer set (Sun et al. 2012) to ensure that DNA extracted from *Hydroides* spp. was amplifiable. Then, PCR was completed using the designed *H. dianthus* primers to validate specificity.

PCR was performed with 50 µl reactions, using 25 µl of AmphiTaq Gold® 360 Master Mix (Applied Biosystems™, Massachusetts, USA), 2 µl of the forward primer, 2 µl of the reverse primer, 19 µl of sterile molecular grade water, and 1 µl of template DNA. PCR was completed using the following protocol: 95 °C for 5 minutes; 35 cycles of: 95 °C for 30 seconds, annealing (50–60 °C) for 30 seconds, and 72 °C for 60 seconds; followed by a final extension of 72 °C for 7 minutes. PCR products were separated using agarose gel electrophoresis on a 2% agarose gel containing 0.1 µl/ml of SYBR® Safe Gel Stain (Invitrogen™, Massachusetts, USA). The gels were run at 135 V for 45 minutes and were visualized using ultraviolet light in the Gel Doc™ XR + Molecular Imager® with BIO-RAD Image lab 5.1 software (California, USA).

**Spatial and temporal monitoring**

**Eel Lake sites and sampling procedures**

The study site location was Eel Lake, NS (43°49’39.2″N; 65°54’31.2″W), where the invasion of the serpulid occurred (Figure 1). Eel Lake contains brackish water, with its saltwater supplied through a brook in Ste. Anne du Ruisseau. The lake experiences semi-diurnal tidal influence from the Bay of Fundy. The exact tidal influence and current speeds are unknown in the lake; however, data collected in Salt Bay, NS (43°49’06″N; 65°55’06″W, adjacent to Eel Lake) was found to have an average current speed 10 cm/s (NS Department of Fisheries and Aquaculture data 2017) and the closest Canadian Hydrological Service (CHS) tidal station in Abram’s River, NS (43°49’57″N; 65°56’20″W) show tidal highs of up to ~ 3.8 m. It can be assumed that the currents and tidal influence in the lake are smaller due to its enclosed nature. Salinity in the lake fluctuates depending on environmental factors such as weather, tidal cycle, depth, etc. Water parameters were monitored in the lake at a fouled oyster lease (43°49’39.263″N; 65°54’31.226″W).
Monitoring of invasive *Hydroides dianthus* in Eel Lake, Argyle, Nova Scotia

Hall et al. (2022), *Management of Biological Invasions* 13(2): 326–343, https://doi.org/10.3391/mbi.2022.13.2.05

**Figure 1.** Site map of Eel Lake, NS, with sampling locations in summer of 2013. Site 1 – Non-fouled (43°49′3.569″N; 65°53′59.694″W), Site 2 – Fouled (43°49′31.436″N; 65°54′25.65″W), Site 3 – Fouled (43°49′35.67″N; 65°54′29.048″W), Site 4 – Fouled (43°49′41.243″N; 65°54′29.048″W). NS – Nova Scotia; NB – New Brunswick, PEI – Prince Edward Island.

on three dates (June 27, July 8, and September 10) by Fisheries and Oceans Canada. Water parameters were measured in the water column up to 9.1 m and found that salinity fluctuated between 12–22‰, with an apparent vertical salinity gradient. Temperature data was also recorded in the summer of 2013 by Fisheries and Oceans Canada. A HOBO® temperature data logger was deployed on the fouled aquaculture lease in the summer of 2013 into Eel Lake from June 27th–October 13th, 2013, which recorded the temperature data every hour.

Water samples were collected weekly from June to the end of August in 2013. Sampling occurred at four locations within the lake three fouled sites that were ~ 0.15 km apart and one non-fouled site which was ~ 1 km from the closest fouled site (Figure 1). The fouled sites (2–4) were in close proximity to where water parameters were collected. At each site, one water sample was obtained from three depths (0.3 m, 1.5 m, and 3.0 m) in the water column (three samples total). At each site and depth, the submersible pump was placed in the water and 150 l of water was pumped through a 60-micron sieve creating a highly concentrated sample. The sample was placed in a 50 ml test tube and was immediately frozen.

**eDNA sample processing**

Water samples were thawed and centrifuged at 3000 g for five minutes until a pellet formed at the bottom of the 50 ml test tube. The pellet was pipetted out of the sample and placed into a 1.5 ml microcentrifuge tube. DNA was extracted as above. Optimized species-specific primers described above were used to amplify the DNA extracted from the water samples.
using PCR, then agarose gel electrophoresis was performed to determine if *H. dianthus* DNA was present in the water sample collected. Technical replicates were not obtained at each site and depth, and therefore the presence/absence of amplification of one sample was used to determine if the sample was positive for *H. dianthus* for a particular site and depth.

**Statistical analysis**

Three separate statistical analysis were completed in this study to determine if there were significant differences between the sites (site 1 (non-fouled), site 2 (fouled), site 3 (fouled), site 4 (fouled), depths (0.5 m, 1.5 m, 3.0 m), and sampling months (June, July, August). Differences between sites and depths were evaluated based on the overall % positive PCR results over the entire sampling period using a one-way ANOVA. The differences in % positive PCR samples between months was also evaluated using a one-way ANOVA. The interactions between sites, depths, and months could not be evaluated due to lack of replication. The assumptions of normality and equal variance were assessed using Anderson-Darling test of normality and Levene’s test of equal variance. All estimates are reported as mean ± standard error (SE), unless otherwise specified.

**Results**

**Sequencing and species confirmation**

Sequencing resulted in only one 416-bp sequence of the mitochondrial COI gene from a *H. dianthus* specimen collected in Eel Lake, NS (Accession #: ON003460.1). Species confirmation using blastn search resulted in the segment of COI gene definitively identified as *H. dianthus* with the top blast hit associated with two specimens KY605381.1 from China and MT044497.1 from Majorca, Mediterranean, both hits receiving a query coverage score of 100%, e-value of ~ 0.0 and percent identity of 99.28%.

**Species specific molecular assay**

**Primer development**

Comparison of *H. dianthus* Eel Lake specimen with 10 *Hydroides* spp. COI sequences (Table 1) resulted in the development of five species-specific primers (Table 3). Primer amplicon length was 72–125-bp with optimal annealing temperatures ranging from 51 to 53.2 °C. Each primer was verified with the NCBI database using the Primer-BLAST tool to ensure primers would not result in amplification of non-target organisms.

**Efficacy, sensitivity, and specificity tests**

Amplification of *H. dianthus* was successful for all COI primer sets for the two positive controls and did not result in amplification in negative control
Table 3. Species specific primers developed for *Hydroides dianthus* Eel Lake specimens from the COI gene.

| Primer            | Sequence 5’-3’                        | Location  | Length | Annealing |
|-------------------|---------------------------------------|-----------|--------|-----------|
| H. dianthusCOI1F  | GGGTCGGTTGGATCGTATAGC                 | 275–295   | 97     | 53 °C     |
| H. dianthusCOI1R  | AGTCAACCCACCCAGCCAAC                  | 371–353   |        |           |
| H. dianthusCOI2F  | GTCCAGAGGGTCCGGTTGGA                  | 268–286   | 103    | 53.2 °C   |
| H. dianthusCOI2R  | GTCAACCCACCCAGCCAAC                  | 352–370   |        |           |
| H. dianthusCOI3F  | GGCGATTGGGGTGACAACT                  | 308–326   | 72     | 53 °C     |
| H. dianthusCOI3R  | AAAAGCATAGTCAACCCACCACA              | 359–379   |        |           |
| H. dianthusCOI4F  | AATCGGGGCATCAGACATAA                 | 8–28      | 125    | 51 °C     |
| H. dianthusCOI4R  | AAAGTTCACCCCGTTGGAAGC                | 132–113   |        |           |
| H. dianthusCOI5F  | ATCGGGGCATCAGACATAAT                 | 9–29      | 125    | 51.2 °C   |
| H. dianthusCOI5R  | AAAGTTCACCCCGTTGGAAGC                | 113–133   |        |           |

(Figure 2A). All primers could detect down to the lowest concentration (0.1 ng/l) (Figure 2B) and all primers were specific and did not amplify any other *Hydroides* spp. other than the target species (Figure 2C). This indicated that all primer sets were optimal for this research. H.dianthusCOI4F/R was chosen as the best candidate as it resulted in high quality amplification and had a large amplicon size, therefore, was used for spatial and temporal monitoring in this study.

Spatial and temporal monitoring and analysis

**Eel Lake depths, sites, and temperature recording**

There were significant differences in the % positive PCR samples between months (June to August) (*p*-value < 0.001). Each month had significantly different % positive PCR samples with June having a mean of 18.7 ± 0.1%, July having a mean of 52.7 ± 4.1%, and August having a mean of 65.0 ± 2.9%. Figure 3 shows the trend in positive PCR results over the sampling period at each site and depth. On the first week of sampling (June 3), only one water sample tested positive for *H. dianthus*, site 2 (fouled) at depth 0.3 m. No water samples were found positive again until the week of June 20 when all sites, including the non-fouled site, tested positive in at least one depth. After this, there were no positive results until July 4 when site 1 tested positive at the 1.5 m depth and site 3 tested positive at 3.0 m depth. Then, on July 11, all sites and depths tested positive for *H. dianthus*, with exception of site 3 at depth 0.3 m. From this date, positive PCR results occurred more frequently, as seen with higher mean % positive water samples for the months July and August.

The effects of depths were not significantly different (*p*-value = 0.5) and *H. dianthus* was detected in all levels. Overall, it was found that the mean % positive PCR result were 45.8 ± 5.4% in 0.3 m depth, 54.2 ± 5.4% in 1.5 m depth, and 52.1 ± 4.0% in 3 m depth. Despite this, an oscillating trend was observed after July 11 in the 0.3 m depth, with one week testing positive for *H. dianthus* in most sites (July 11, July 25, August 8, August 22) followed by a week of negative detection of *H. dianthus* in all sites (or most) (July 18, August 1, August 15). The positive PCR results in the 0.3 m depth occurred
Monitoring of invasive *Hydroides dianthus* in Eel Lake, Argyle, Nova Scotia

Hall et al. (2022), *Management of Biological Invasions* 13(2): 326–343, https://doi.org/10.3391/mbi.2022.13.2.05

Figure 2. Agarose gel of PCR amplicons created using all newly designed primer assays for:

A – Efficacy testing for assay developed for COI gene: Ln 1–3 = H.dianthusCOI1F/R; Ln 4–6 = H.dianthusCOI2F/R; Ln 7–9 = H.dianthusCOI3F/R; Ln 10–12 = H.dianthusCOI4F/R; Ln 13–15 = H.dianthusCOI5F/R. Positive controls are located in Ln 1, 2, 4, 5, 7, 8, 10, 11, 13, 14. Negative controls are located in: Ln 3, 6, 9, 12, 15. B – Sensitivity test for primer assays developed: Ln1–5 = H.dianthusCOI1F/R; Ln6–10 = H.dianthusCOI2F/R; Ln11–15 = H.dianthusCOI3F/R; Ln16–20 = H.dianthusCOI4F/R; Ln21–25 = H.dianthusCOI5F/R. Each primer set was tested for sensitivity using a series of dilutions on *H. dianthus* positive controls: 100 ng/l = Ln 1, 6, 11, 16, 21; 10 ng/l = Ln 2, 7, 12, 17, 22; 1 ng/l = Ln 3, 8, 13, 18, 23; 0.1 ng/l = Ln 4, 9, 14, 19, 24. Negative controls are located in: Ln 5, 10, 15, 20, 25. C – Specificity test using H.dianthusCOI4F/R assay: Ln 1 = *Hydroides dianthus* positive control; Ln 2 = negative control; Ln 3 = *Hydroides brachyacantha*; Ln 4 = *Hydroides dirampha*; Ln 5 = *Hydroides elegans*; Ln 6 = *Hydroides ezoensis*; Ln 7 = *Hydroides heterocera*; Ln 8 = *Hydroides homoceros*; Ln 9 = *Hydroides longispinosa*; Ln 10 = *Hydroides minax*; Ln 11 = *Hydroides recurvispina*; Ln 12 = *Hydroides trivesiculosa*.

...a few days after the full or new moon (spring tide) (Figure 3). Statistical analysis associated with this oscillating trend could not be completed as more replicates would be necessary. Sites were also not found significantly different (*p*-value = 0.8) with all locations having positive PCR results. Overall, mean % positive PCR results in site 1 (non-fouled) was 50.0 ± 4.8%, site 2 (fouled) was 47.2 ± 2.8%, site 3 (fouled) was 50.0 ± 8.3%, and site 4 (fouled) was 55.6 ± 7.3%.
Temperature recording occurred in the lake starting on the week of June 27, therefore, temperatures for the first week when *H. dianthus* was detected (June 20) could not be determined. However, during the second major peak of the summer (July 11), the temperature in the water column was \( \sim 18.3 \, ^\circ C \). From this point on, water temperature increased until the end of the sampling period when the temperatures were \( \sim 22.2 \, ^\circ C \), and detection of *H. dianthus* occurred frequently (Figure 3).

**Discussion**

The frequent positive PCR results in all sites and depths over the summer of 2013 indicate that *H. dianthus* spawned multiple times. This finding is
consistent with the patterns demonstrated for other members of the genus *Hydroides*, which spawn several times over their lifetime, typically, continuously over their reproductive season (Kupriyanova et al. 2001). Grave (1933) documented that *H. dianthus* (as *Hydroides hexagonus* (see Read et al. 2017)) spawned from June to late October in Woods Hole, Massachusetts, and Zuraw and Leone (1968) found that *H. dianthus* could spawn every two to four weeks at 23 °C during their reproductive season. The temperatures in Eel Lake within the study period was between ~ 18–22 °C, indicating that *H. dianthus* was able to spawn at least once but likely multiple times within this temperature range, as the prevalence of positive PCR results significantly increased from June to August. A longer study period, along with visual observation of larvae through sampling would be necessary to understand whether spawning took place in Eel Lake at temperatures above or below this range.

This study hypothesized that heavy biofouling of *H. dianthus* could be mitigated by identifying an optimal depth to hold oyster cages when spawning was occurring. However, there were no significant differences between depths and positive PCR results were found in all examined depths through out the summer. Interestingly, an oscillating trend was found within the 0.3 m depth, where positive PCR results were obtained on a biweekly basis after July 4th. It is known that *H. dianthus* larvae exhibits photopositivity in laboratory setting (Zeleny 1905; Kupriyanova et al. 2001), but nothing is known about its behaviour in a natural environmental setting. This oscillating trend could be an indicator of photopositive behaviour when positive PCR results were obtained in surface water. On the weeks where *H. dianthus* tested negative in surface water, it was found positive frequently at the 1.5 m and 3 m depth and could be indicator of photonegative behaviour. While it is not known if this trend was produced by coincidence, it is also not uncommon for pelagic larvae to exhibit this photopositive response followed by photonegative behaviour (Thorson 1964) and has been found in other serpulids (Miura and Kajihara 1981; Kupriyanova et al. 2001). In theory, if *H. dianthus* is found less frequently in the surface waters, then the optimal area to maintain cages would be in the surface water. Unfortunately, oyster bags/cages in Eel Lake were already maintained at the surface, but biofouling from *H. dianthus* persisted. It is likely that important settlement cues, such as an optimal biofilm and presence of conspecifics on bags/cages triggered the associative gregarious settling well-documented for *H. dianthus* (Toonen and Pawlik 1994, 2001). Therefore, there was not an optimal depth to maintain oyster cages during *H. dianthus*’ reproductive season.

In Atlantic Canada, prevention of biofouling on oyster aquaculture leases by barnacles (*Balanus improvisus*) and mussels (*Mytilus edulis*) can be mitigated by completing a mid-August flip and early October flip during settlement periods of these species (Mallet et al. 2009). However, in areas
Monitoring of invasive *Hydroides dianthus* in Eel Lake, Argyle, Nova Scotia

Hall et al. (2022), Management of Biological Invasions 13(2): 326–343, https://doi.org/10.3391/mbi.2022.13.2.05

with biofouling from invasive solitary tunicates such as *Ciona intestinalis*, treatment may be needed in mid-July and mid-August (Ramsay et al. 2009), or as frequent as biweekly during times of heavy biofouling for invasive colonial tunicates such as *Botryllus schlosseri* and *Botrylloides violaceus* (Government of New Brunswick 2018). Biofouling by *H. dianthus* results in an additional challenge, as their calcium carbonate tube is difficult to remove. Other biofouling organisms that are difficult to remove, such as barnacles (*Balanus improvisus*) and oysters (*Crassostrea virginica*), can be controlled through desiccation during their initial settlement period (Adams 1991), which is easier to time properly because barnacles and oysters only have one major spawning event per reproductive period. In contrast, frequent spawning events documented for *H. dianthus* in this study indicate that this flipping would have to be done on a weekly or biweekly basis to remove newly settled larvae, as *H. dianthus* has been found ready to settle and undergo metamorphosis as early as 5 days post-fertilization at 24 °C (Scheltema et al. 1981), or 10–14 days (no temperature specified, likely < 24 °C) (Grave 1933, 1937). Biweekly flipping was already used in Eel Lake and was not successful at preventing *H. dianthus* biofouling, indicating that weekly flipping of cages may be necessary to mitigate fouling. The persistence of fouling despite biweekly flipping might be the result of fast growth rate and strong gregarious settlement response. Once settled, the tube of *H. dianthus* can grow up to 54 mm in their first three months (Grave 1933), indicating that biofouling can rapidly accumulate through gregarious settling behaviour (Toonen and Pawlik 1994, 2001). Therefore, integration of weekly flipping of cages during peak spawning/settling activity and frequent cleaning of cages may be the only method to mitigate biofouling of *H. dianthus* on the oyster lease.

Interestingly, positive PCR results obtained in the 0.3 m depth on a biweekly basis also appear to occur a few days after a spring tide (i.e. after full or new moon) (Figure 3). Marine organisms have been found to exhibit semilunar rhythmicity (Naylor 2013). Some serpulids from the genus *Spirorbis* synchronize their release of larvae at the neap tide (Garbarini 1933; Knight-Jones 1951; Rothlisberg 1974). For example, *Spirorbis marioni* Caullery and Mesnil, 1897 synchronizes its spawning to the spring tide but release of larvae occurred during the neap tide (Rothlisberg 1974). Release of larvae on neap tides may have benefits to marine organisms as reduced tidal influence mitigates the risk of larvae being carried away to more hostile environments (Naylor 2010). In contrast, some species of corals spawn on spring tides so that their larvae are swept away and allow for wider dispersal (Naylor 2010). *Hydroides dianthus* is a broadcast spawner with external fertilization and planktotrophic development in the water column (Scheltema et al. 1981), therefore the breeding behaviour of this species differs from that found in the genus *Spirorbis* that brood lecithotrophic larvae (Rothlisberg 1974). However, if positive PCR results at 0.3 m depth
Monitoring of invasive *Hydroides dianthus* in Eel Lake, Argyle, Nova Scotia

Hall et al. (2022), *Management of Biological Invasions* 13(2): 326–343, https://doi.org/10.3391/mbi.2022.13.2.05

are associated with early-stage photopositive larvae it could indicate that spawning occurs closer to a spring tide, where *H. dianthus* could benefit from dispersal during stronger tides to establish new populations, but this hypothesis needs further confirmation. This study also found that there were no significant differences between sites and all sites had positive PCR results for *H. dianthus* throughout the summer, including the non-fouled site. Unfortunately, this could be indicative that the site was recently fouled or that larvae were dispersing toward that area of the lake. Once in a new location, *H. dianthus* colonises new substrates and give off chemical cues to other conspecifics, to settle and form new populations (Toonen and Pawlik 1994).

It is unknown how *H. dianthus* spread to Eel Lake, NS. Aquatic invasions are usually attributed to ballast water systems, shellfish transfers, ship hulls, recreational boating, floating marine infrastructure and debris (Mckindsey et al. 2007; Darbyson et al. 2009; Davidson et al. 2010; Ruiz et al. 2011; Sylvester et al. 2011; Briski et al. 2012). However, ballast water systems have never been a proven method of transport of serpulid larvae (Capa et al. 2021), and international shellfish transfers of oysters are not permitted due to presence of reportable diseases such as MSX (*Haplosporidium nelsoni* (Haskin et al. 1966)) and Dermo disease (*Perkinsus marinus* (Mackin et al. 1950)). It can’t be completely ruled out that the invasion was through a natural range extension, as Eel Lake is adjacent to its northern most point of distribution in Cobscook Bay, Maine, USA (Trott 2004) (~150 km away from Eel Lake) which could have been facilitated by floating marine debris that made its way into the lake. However, it is unlikely, as the major currents that circulate around the Bay of Fundy are typically counter-clockwise including the Scotian Shelf Coastal Current and Bay of Fundy Gyre (Aretxabaleta et al. 2009) that would make it difficult for natural currents to transport marine debris from Cobscook Bay to Eel Lake. It is therefore more feasible that *H. dianthus* was transported by anthropogenic means such as by recreational boating with attached fouling.

*Hydroides* spp. biofouling has been known to have consequences to the aquatic ecosystem, aquaculture leases and marine infrastructure. *Hydroides dianthus* has been found to smother juvenile oysters from biofouling (Eno et al. 1997) and is concerning for shellfish growers that rely on wild spat collection. Removal of biofouling can also lead to scars that impacts the aesthetics of shells, which can impact market value of shellfish (Watson et al. 2009; Dürr and Watson 2010). Therefore, mitigating the spread of *H. dianthus* in Atlantic Canada, outside of Eel Lake will be important in the future, and the use of molecular assay using eDNA shows promise in monitoring programs (LeBlanc et al. 2020). The molecular assay developed and applied in this study was useful in understanding the spatial and temporal patterns of *H. dianthus* larvae in eDNA water samples in the summer of 2013; however, it has the potential to be used in the future to monitor spread elsewhere in Atlantic Canada. Future studies should also
include visual observation along side this method of detection to ensure that the detection of *H. dianthus* was from larvae and not other tissues from *H. dianthus* such as sloughed cells.

**Conclusions**

Monitoring of *H. dianthus* in eDNA samples from Eel Lake reiterated the challenges associated with serpulid biofouling. Frequent spawning and presence of *H. dianthus* in all depths indicated that there was no optimal depth for oyster cages to be sunk to, and that mitigation of recruitment is likely to be highly challenging. More research in the aquaculture industry is needed to find solutions for biofouling by serpulids. The potential impacts to other shellfish leases in Atlantic Canada warrant the need to mitigate and monitor the spread of this species in the region and the molecular assay developed in this study may be an excellent tool to assist with this.

**Acknowledgements**

We thank Dawn Sephton and Bénédikte Vercaemer from Fisheries and Oceans Canada, Maritime Region, for providing water temperature data and information on the initial identification within Canada, and Harry ten Hove (Naturalis, the Netherlands) for the final confirmation of *Hydroides dianthus* identification. We thank Vicki Savoie-Swan for assisting with the DNA extractions and amplifications. Thanks are due to Nolan and Colton D’Eon from Eel Lake Oyster Ltd for collection of water samples at Eel Lake. We would like to thank Steve Keable at the Australian Museum for organizing the donation of *Hydroides* spp. samples. Finally, we thank Andrea Locke and two anonymous reviewers for their helpful comments.

**Funding declaration**

This study was funded using a combination of IRAP-STC funding and start up funds for Dr. Sarah Stewart-Clark from Dalhousie University and by the Australian Biological Resources Study (ABRS) grant RF213-19 to Dr. Elena Kupriyanova.

**Authors’ contribution**

Stephanie Hall: investigation and data collection; data analysis and interpretation; writing – original draft; writing – review and editing; Sarah Stewart Clark: research conceptualization; sample design and methodology; investigation and data collection; data analysis and interpretation; writing – review and editing; Elena Kupriyanova: samples and methodology; writing – review and editing.

**References**

Adams MP (1991) Eliminating spat settlement on oysters cultured in coastal Georgia: A feasibility study. *Journal of Shellfish Research* 10: 207–213

Ardura A, Zaiko A, Martinez JL, Samuloviene A, Semenova A, Garcia-Vazquez E (2015) eDNA and specific primers for early detection of invasive species - A case study on the bivalve *Rangia cuneata*, currently spreading in Europe. *Marine Environmental Research* 112: 48–55, https://doi.org/10.1016/j.marenvres.2015.09.013

Aretxabaleta AL, McGillicuddy DJ, Smith KW, Manning JP, Lynch DR (2009) Model simulations of the Bay of Fundy Gyre: 2. Hindcasts for 2005-2007 reveal interannual variability in retentiveness. *Journal of Geophysical Research* 114: C09005, https://doi.org/10.1029/2008JC004948

Bastida-Zavala JR, ten Hove HA (2002) Revision of *Hydroides* Gunnerus, 1768 (Polychaeta: Serpulidae) from the Western Atlantic Region. *Beaufortia* 52(9): 103–178

Bastida-Zavala JR, Salazar-Vallejo SI (2000) Serpulídeos (Polychaeta: Serpulidae) del Caribe noroccidental: *Hydroides y Serpula*. *Revista de Biología Tropical* 48: 841–858

Briski E, Ghabooli S, Bailey SA, MacIsaac HJ (2012) Invasion risk posed by macroinvertebrates transported in ships’ ballast tanks. *Biological Invasions* 14: 1843–1850, https://doi.org/10.1007/s10530-012-0194-0
Hall et al. (2022), Management of Biological Invasions 13(2): 326–343, https://doi.org/10.3391/mbi.2022.13.2.05
Kupriyanova EK, Nishi E, ten Hove HA, Rzhavsky AV (2001) Life-history patterns in serpulimorph polychaetes: ecological and evolutionary perspectives. Oceanography and Marine Biology: an Annual Review 39: 1–101

Larkin MA, Blackshields G, Brown NP, Chenna R, McGettigan PA, McWilliam H, Valentin F, Wallace IM, Wilm A, Lopez R, Thompson JD, Gibson TJ, Higgins DG (2007) Clustal W and Clustal X version 2.0. Bioinformatics 23: 2947–2948, https://doi.org/10.1093/bioinformatics/btm604

LeBlanc F, Belliveau V, Watson E, Coomber C, Simard N, Dibacco C, Bernier R, Gagné N (2020) Environmental DNA (eDNA) detection of marine aquatic invasive species (AIS) in Eastern Canada using a targeted species-specific qPCR approach. Management of Biological Invasions 11: 201–217, https://doi.org/10.3391/mbi.2020.11.2.03

Locke A, Hanson JM, Ellis KM, Thompson J, Rochette R (2007) Invasion of the southern Gulf of St. Lawrence by the clubbed tunicate (Styela clava Herdman): Potential mechanisms for invasions of Prince Edward Island estuaries. Journal of Experimental Marine Biology and Ecology 342: 69–77, https://doi.org/10.1016/j.jembe.2006.10.016

Ma Y, Liu P, Yu S, Li D, Cao S (2009) Inhibition of common fouling organisms in mariculture by epiphytic bacteria from the surfaces of seaweeds and invertebrates. Acta Ecologica Sinica 29: 222–226, https://doi.org/10.1016/j.actasina.2009.08.004

Ma KCK, Simard N, Stewart-Clark SE, Bernier RY, Nadeau M, Willis J (2016) Early detection of the non-indigenous colonial ascidian Diplosoma listerianum in eastern Canada and its implications for monitoring. Management of Biological Invasions 7: 365–374, https://doi.org/10.3391/mbi.2016.7.4.06

Mackin JG, Owen MH, Collier A (1950) Preliminary note on the occurrence of a new protistan parasite, Dermoeyctidium marinum n. sp. in Crassostrea virginica (Gmelin). Science 111: 328–329, https://doi.org/10.1126/science.111.2983.328

Mallet AL, Carver CE, Hardy M (2009) The effect of floating bag management strategies on biofouling, oyster growth and biodeposition levels. Aquaculture 287: 315–323, https://doi.org/10.1016/j.aquaculture.2008.10.023

McKenzie CH, Matheson K, Caines S, Wells T (2016) Surveys for non-indigenous tunicate species in Newfoundland, Canada (2006-2014): a first step towards understanding impact and control. Management of Biological Invasions 7: 21–32, https://doi.org/10.3391/mbi.2016.7.1.04

Mckindsey CW, Landry T, O’Brien FX, Davies IM (2007) Bivalve aquaculture and exotic species: A review of ecological considerations and management issues. Journal of Shellfish Research 26: 281–294, https://doi.org/10.2983/0730-8000(2007)26[281:BAASEA]2.0.CO;2

Miura T, Kajihara, T (1981) The development of a serpulid worm, Hydroides ezoensis (Innelda, Polychaeta). Proceedings of the Japanese Society of Systematic Zoology 20: 7–20

Mörch OA (1863) Revisio critica Serpulidarum. Et bidrag til røror -menes naturhistorie. Naturhistorisk Tidsskrift Henrik Kroyer Kopenhavn 3(1): 347–470

Naylor E (2010) Chapter 5: Lunar and semilunar biorhythms. In: Naylor E (ed), Chronobiology of Marine Organisms. Cambridge University Press. pp 89–115, https://doi.org/10.1017/CBO9780511803567.006

Naylor E (2013) Rhythmic behaviour and reproduction. In: Newell RC (ed), Adaptation to environment: essays on the physiology of marine animals. Butterworth & Co Ltd., pp 393–418, https://doi.org/10.1016/B978-0-408-70778-7.50010-X

Okuda S (1934) Some tubicolous annelids from Hokkaido. Journal of the Faculty of Science Hokkaido University (Series 6). Zoology 3: 233–246

Pixell HLM (1913) Polychaeta of the Indian Ocean, together with some species from the Cape Verde Islands. The Serpulidae, with a classification of the genera Hydroides and Eupomatia. Transactions of the Linnean Society of London (Series 2) 16: 69–92

Ramaoy A, Davidson J, Bourque D, Stryhn H (2009) Recruitment patterns and population development of the invasive ascidian Ciona intestinalis in Prince Edward Island, Canada. Aquatic Invasions 4: 169–176, https://doi.org/10.3391/ai.2009.4.1.17

Read GB, ten Hove HA, Sun Y, Kupriyanova EK (2017) Hydroides Gunnerus, 1768 (Annelida, Serpulidae) is feminine: a nomenclatural checklist of updated names. Zookeys 642: 1–52, https://doi.org/10.3897/zookeys.642.10443

Rioja E (1941) Estudios anelidológicos, 2. Observaciones de varias especies del género Hydroides Gunnerus (sensu Fauvel) de las costas mexicanas del Pacífico. Anales del Instituto de Biologia. Universidad Nacional Autónoma de México 12: 161–175

Rothlisberg PC (1974) Reproduction in Spirorbis (Spirorbella) marioni Caullery & Mesnil (Polychaeta: Serpulidae). Journal of Experimental Marine Biology and Ecology 15: 285–297, https://doi.org/10.1016/0022-0981(74)90051-3

Ruiz GM, Fofonoff PW, Steves B, Foss SF, Shiba SN (2011) Marine invasion history and vector analysis of California: a hotspot for western North America. Diversity and Distributions 17: 362–373, https://doi.org/10.1111/j.1472-4642.2011.00742.x

Sephton D, Vercaemer B, Nicolas JM, Keays J (2011) Monitoring for invasive tunicates in Nova Scotia, Canada (2006-2009). Aquatic Invasions 6: 391–403, https://doi.org/10.3391/ai.2011.6.4.04
Septon D, Martin JL, LeGresley M, Godin K (2016) Biofouling monitoring for aquatic invasive species (AIS) in DFO Maritimes Region, southwest New Brunswick: 2006-2011. Canadian Technical Report of Fisheries and Aquatic Sciences 3140, vi + 68 pp

Scheltema RS, Williams IP, Shaw MA, Loudon C (1981) Gregarious settlement by the larvae of Hydroides dianthus (Polychaeta: Serpulidae). Marine Ecology Progress Series 5: 69–74, https://doi.org/10.3354/meps00569

Simard N, Pereira S, Estrada R, Nadeau M (2013) State of marine invasive species in Quebec. Canadian Manuscript Report of Fisheries and Aquatic Sciences 2030, vii + 61 pp

Stewart-Clark SE, Siah A, Greenwood SJ, Davidson J, Berthe FC (2009) Development of 18S rDNA and COI gene primers for the identification of invasive tunicate species in water samples. Aquatic Invasions 4: 575–580, https://doi.org/10.3391/ai.2009.4.4.2

Stewart-Clark SE, Davidson J, Greenwood SJ (2013) Optimization and validation of molecular assays for invasive tunicate monitoring in environmental water samples. Aquatic Science and Technology 1: 143–166, https://doi.org/10.5296/ast.v1i1.2979

Straughan D (1967) Some Serpulidae (Annelida: Polychaeta) from Heron Island, Queensland. University of Queensland Papers 1: 27–45

Sun Y, Kupriyanova EK, Qiu JW (2012) COI barcoding of Hydroides: a road from impossible to difficult. Invertebrate Systematics 26: 539–547, https://doi.org/10.1071/IS12024

Sun Y, Wong E, Keppel E, Williamson J, Kupriyanova E (2017) A global invader or a complex of regionally distributed species? Clarifying the status of an invasive serpulid Hydroides dianthus (Verrill, 1873) (Polychaeta: Serpulidae) using DNA barcoding. Marine Biology 164: 1–12, https://doi.org/10.1007/s00227-016-3058-9

Sylvester F, Kalaci O, Leung B, Lacoursière-Roussel A, Murray CC, Choi FM, Bravo MA, Wisely B (1958) The development and setting of a serpulid worm, Hydroides norvegica (Gunnerus). Journal of Experimental Zoology 162: 1–102, https://doi.org/10.1002/jez.1400020102

Trott TJ (2004) Cobcook Bay inventory: a historical checklist of marine invertebrates spanning 162 years. Northeastern Naturalist 11: 261–324, https://doi.org/10.1656/1092-6194(2004)11[261:CBIBAC]2.0.CO;2

Verrill AE (1873) Results of recent dredging expeditions on the coast of New England; No. 6. American Journal of Science 3: 435–441, https://doi.org/10.2475/ajs.s3-7.40.405

Verrill AE (1873) Results of modern dredging expeditions on the coast of New England; No. 6. Ophelia 1: 167–208, https://doi.org/10.1080/00785326.1964.10416277

Weisly B (1958) The development and setting of a serpulid worm, Hydroides norvegica Gunnerus (Polychaeta). Marine and Freshwater Research 9: 351–361, https://doi.org/10.1071/ MF9580351

Zeleny C (1905) Compensatory regulation. Journal of Experimental Zoology 2: 1–102, https://doi.org/10.1002/jez.1400020102

Zuraw EA, Leone DE (1968) Laboratory culture of the tubeworm, Hydroides (Eupomatus) dianthus Verrill 1873. General Dynamics, Electric Boat Division Publication, Groton, Connecticut, 96 pp