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RESEARCH ARTICLE

Diel variability in fish assemblages in coastal wetlands and tributaries of the St. Lawrence River: a cautionary tale for fisheries monitoring

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Abstract Both coastal wetlands and tributaries of waterbodies provide important and distinct habitat for freshwater fishes. While diel migration into and out of these systems is known to occur for some species, the resulting changes in fish assemblage composition and dominance are less well understood. To evaluate diel changes in the fish assemblages of a coastal wetland and tributary, fish community surveys were completed at ten locations during the day (noon) and night (midnight) in Cooper's Marsh, a coastal wetland in the St. Lawrence River, and the Raisin River, a nearby tributary. Catch per unit effort (CPUE) and species richness were highest in the coastal wetland during the night sampling period. Species-specific differences were also apparent with high CPUE of Mimic Shiner in the marsh at night. Differences within the river were less pronounced, suggesting less diel variability in fish assemblage structure, possibly driven by the more constrained nature of fluvial systems. These results contribute to our understanding of diel movement patterns of fishes and the natural diel variability in species assemblages that can occur. These findings also emphasize that when engaged in environmental monitoring and assessment, it is important to consider how diel variation in fish assemblage structure will influence conclusions regarding the biotic components of a given aquatic ecosystem.

Keywords Catch per unit effort · Marsh · River · Fish community · Analysis of similarity

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Introduction

Freshwater fishes are a diverse group of aquatic species that generate numerous ecosystem services (Holmlund and Hammer 1999). However, freshwaters are among the most threatened ecosystems in the world (Strayer and Dudgeon 2010; Vörösmarty et al. 2010) with freshwater fishes being some of the most imperiled taxa (Richter et al. 1997; Ricciardi and Rasmussen 1999). Aquatic ecosystems have been severely degraded as a result of human alteration of landscapes (for agriculture, forestry, industry, urbanization, energy production, etc.; Dudgeon et al. 2006). Given that habitat is the foundation for productive and healthy aquatic ecosystems and fish populations (Lapointe et al. 2014), and given the immense value of fish (Holmlund and Hammer 1999), it is not surprising that great efforts have been devoted to the rehabilitation and enhancement of degraded aquatic habitat (Roni et al. 2006) and creation of new habitats (collectively herein referred to as restoration; Jackson et al. 1995; Hobbs and Harris 2001). While aquatic restoration objectives may be focused on a variety of goals, fish populations are often the primary target given their socio-economic importance (Holmlund and Hammer 1999), role as integrators within the aquatic ecosystem (Facey et al. 2005), and relative ease of sampling (Bonar et al. 2009).

Assessment of fish assemblage structure is needed to identify critical habitats (Rosenfeld and Hatfield 2006), establish fish–habitat relationships to inform restoration

(Bond and Lake 2003), identify and prioritize areas in need of restoration (Beechie et al. 2008), and to evaluate the effectiveness of restoration activities once complete (Ruiz-Jaen and Mitchell Aide 2005). Therefore, it is clear that robust information on fish community structure is needed to inform management actions. As a result, numerous studies have been conducted to develop and calibrate various methods for the quantification of fish assemblage structure in lentic (e.g., Weaver et al. 1993; Chow-Fraser et al. 2006; Ruetz et al. 2007; Sutela et al. 2008) and lotic (Pugh and Schramm 1998; Clavero et al. 2006; Lapointe et al. 2006) systems. It is well known that all gear types (e.g., electrofishing, active netting, and passive netting) have biases related to the capture of various species, different size groups, and the habitat being sampled (Cowx 1996). In addition to gear-biases, temporal variation in community assemblages and habitat associations occur naturally at both seasonal (e.g., spawning migrations of pelagic species) and daily scales (e.g., diel migration; Helfman 1986). Although biases associated with seasonal processes can be accounted for by limiting sampling to a specific time-period, diel changes can be harder to incorporate. Indeed, relative to the number of studies focused on gear bias and species-specific movement, there has been little work on the effects of fish movements, let alone diel movements, on the resulting catch and fish assemblage structure (Dolinsek et al. 2014).

Many fish species exhibit diel movement patterns that may change seasonally, or between life history stages (Lucas and Baras 2001; Rees et al. 2009). Telemetry has been used to track lateral and longitudinal fish movements (Clough and Ladle 1997; Lucas and Baras 2000; Cooke et al. 2013) and, more recently, diel vertical movements over multiple seasons (Bull Trout, *Salvelinus confluentus*, Gutowsky et al. 2013; Burbot, *Lota lota*, Harrison et al. 2013). Unfortunately, due to the high cost, technical constraints, and species limitations associated with telemetry equipment and tracking programs, this approach is unlikely to be incorporated into routine monitoring of the fish community or to inform sampling protocols. Although a number of gear comparison studies have used diel movement to support their findings (i.e., 24-h fyke nets catch a more representative population than day/night time electrofishing because of diurnal fish movement; Chow-Fraser et al. 2006; Ruetz et al. 2007), fewer studies have designed sampling protocols to specifically evaluate the effect of diel movement on fish catch using standardized sampling gear (see Pierce et al. 2001; McKenna 2008). Unbiased monitoring programs are essential for ensuring that sampling data provide an accurate representation of the fish assemblage in a given habitat (Harris 1995). In addition, diel habitat use is important for species-specific initiatives such as development of species-at-risk (SAR) recovery programs (e.g., Rosenfeld and Hatfield 2006) and invasive species monitoring (e.g., Round Goby, *Neogobius melanostomus*).

To aid in the development of effective monitoring programs, two sites, a coastal wetland and nearby tributary were sampled within the St. Lawrence River (Cornwall) Area of Concern (AOC). This is one of 43 AOCs designated by the International Joint Commission in the Laurentian Great Lakes. Legacy pollutants and contaminants, exotic species, and habitat loss and degradation have all contributed to the listing of this region as an AOC (Hartig and Thomas 1988). The development, implementation, and monitoring of mitigation or remediation strategies is a critical part of the AOC delisting process. The two primary goals of the study were therefore to determine: (1) how fish assemblages vary when sampling during the day versus at night using boat electrofishing, and (2) whether there are similar fish assemblages in a marsh and proximate tributary. Findings from this study will enhance our understanding of both diel patterns in the spatial ecology of fishes and daily variability in species assemblages in lentic and lotic systems. We recognize that by sampling with one gear type (boat electrofishing) we are assessing only the portion of the fish community that can be captured using this gear type and its associated settings. Nonetheless, since this is a common sampling method, we believe our findings will provide valuable information to environmental managers charged with the task of monitoring aquatic ecosystems for the purpose of establishing baseline conditions and identifying or quantifying changes in fish assemblages.

Materials and methods

Study site

The St. Lawrence River AOC spans areas downstream of the Moses Saunders power dam to the outflow of Lake St. Francis into the St. Lawrence River (Fig. 1). Lentic and lotic habitats are common within the AOC, therefore, to limit potentially confounding issues associated with regional variation, fish community assemblages in a coastal marsh (Cooper Marsh; CM) and nearby tributary (Raisin River; RR) were assessed.

Cooper marsh

Coastal wetlands sampled for this study are situated within CM, part of the larger Charlottenburgh Marsh wetland complex (Fig. 1). This coastal wetland complex has been designated as a provincially significant wetland (PSW) for exhibiting high biological value associated with large interior and edge coverage, habitat connectivity, and high

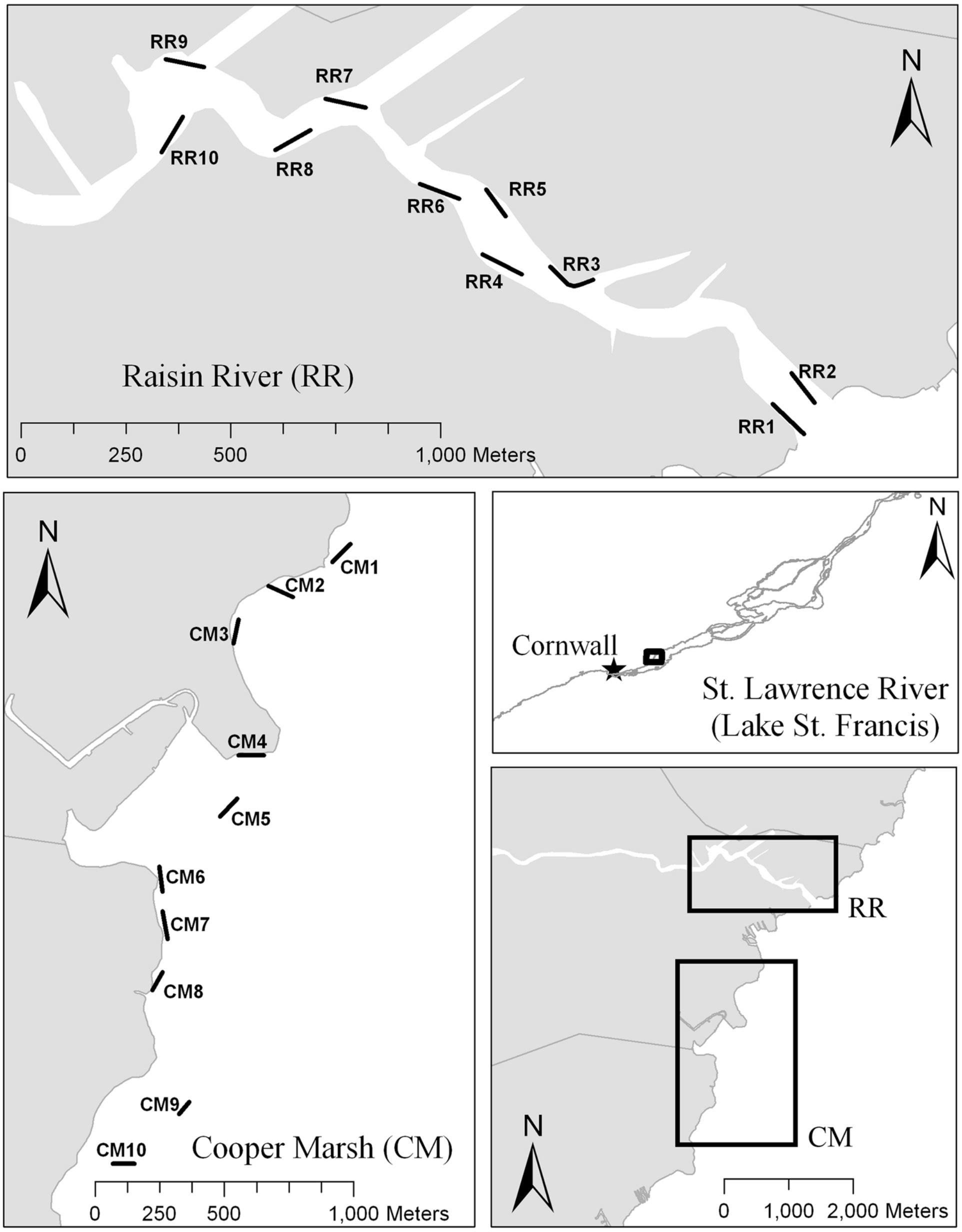

Fig. 1 Location of study site downstream of the City of Cornwall. *Black bars* show the location of transects sampled in Coopers Marsh (CM) and Raisin River (RR)

biodiversity (Burns 2004). This habitat is of regional significance as it has been documented to support Canadian species of concern such as Bridle Shiner (*Notropis bifrinatus)* and River Redhorse (*Moxostoma carinatum*), as well as provide critical nursery habitat for Walleye (*Sander vitreus*). Despite a loss of 4.2 ha of wetland area within Charlottenburgh Marsh over the past few decades, CM has not declined in size, which may be due to historic and ongoing habitat enhancement by Ducks Unlimited Canada and the Raisin River Conservation Authority (Burns 2004). CM also exists as a conservation area within Charlottenburgh Marsh, restricting the amount of shoreline development in the wetland. No development has occurred within proximity to the sampled areas.

The CM lentic ecosystem is characterized by open water marsh habitats and exposed shoreline wetland habitats. The vegetation community in open water habitats is dominated by bulrush (e.g., *Schoenoplectus* sp.), whereas shoreline sites are comprised of lilies (*Nymphaea*/*Nuphar* sp.), emergent cattails (*Typha latifolia*), and invasive common reed grass (*Phragmites australis australis*). Ten transects were sampled within CM (Fig. 1), representing a diversity of habitat types including exposed wetland fringes, open water areas and shoreline areas.

Raisin river

The lotic system sampled for this study was the RR, which drains a subwatershed covering 57,682 ha. This watercourse supports a warm water fishery of 43 species, including several species of concern in Canada, such as River Redhorse and Bridle Shiner. The main branch of the river is dominated by clay loam (poor drainage), with loam, silt-loam, and muck interspersed (Edwards et al. 2007).

The mouth of the RR is approximately 100 m wide at its confluence with Lake St. Francis (Fig. 1). A two-lane road bisects the RR nearly 300 m upstream of the river mouth, as well as a major four-lane highway an additional 1000 m upstream. The river width ranges between 75 and 120 m throughout this reach, with several side channels present, and the remnants of historic boat docking areas. These side channels range from 20 to 50 m wide and from 250 to 500 m in length, averaging a depth of 1–3 m throughout. In the river channel, the substrate is mucky to sandy with large boulders present, and water clarity generally does not exceed 1 m.

Shoreline development in the first 500 m of the river's lowest reach consists of residential land-use with numerous docks and small access channels, hardened shorelines as well as a small local marina. Beyond the first 500 m, the shoreline is undeveloped with large overhanging trees present. Throughout the sampled area aquatic vegetation such as submerged vegetation (e.g., *Potamogeton* sp.) and lilies (*Nuphar/Nymphaea* sp.) dominate the littoral zone. Near the shore sparse areas of emergent bulrush (*Schoenoplectus* sp.) are present, while within the artificial side channels cattails (*T. latifolia*) and invasive common reed grass (*P. australis*) dominate the shoreline and less submerged vegetation is present.

Fish assemblage sampling

Ten 100 m transects (Fig. 1) were sampled within the first 1500 m of the RR upstream of the confluence with Lake St. Francis. To sample as many distinct habitats as possible, transects were spread between east and west shorelines of varying vegetative cover and substrate, as well as two sites that were located in artificial side-channels. Similarly, ten 100 m transects (Fig. 1) were sampled across over 2000 m of shore along a north–south axis of CM. Transects in CM were also situated in different types of vegetative cover and varying degrees of exposure to wind and waves from the open lake. After surveys had been completed, rarefaction curves were used to determine whether our effort was sufficient to capture the expected assemblage at both CM and RR. For both locations at both times, over half of the species were encountered in 2–3 samples and curves began to level off at 5–6 samples indicating our efforts were sufficient (data not shown).

Sampling occurred during two distinct time periods, one during the day (sampled between 11:00 and 13:30) and one at night (sampled between 23:00 and 01:30). Day sampling in RR and CM occurred on the 2nd and 3rd of July 2013, respectively. Night sampling for RR was completed on July 4th, 2013 and was split over the evenings of July 4th and July 5th, 2013 for CM. Weather conditions were consistent across sample dates with clear skies and no precipitation. Conductivity was generally higher in RR (423 ± 16) than CM (306 ± 21). For each sampling time period and location, the ten 100 m transects were surveyed using boat electrofishing (5.3 m jon boat, 2.5 GPP Smith-Root Electroshocker with two hanging anodes located on booms at the bow of the boat). This unit uses pulsed DC-current and settings were typically set to 60 Hz at 80 % power, but when conductivity was high (e.g., RR), the pulse was increased to between 80 and 120 Hz at power of 60 %. Shocking effort ranged from 167 to 425 shock-seconds (mean = 307 ± 61). For each transect, all observed fish were netted and kept in a 110 L container of fresh water collected on site. Individuals were then identified to species (Hubbs et al. 2004; Holm et al. 2009) and the total length for the first 20 individuals of a species was measured; remaining individuals were included as part of the total catch. Vouchers of individuals that could not be readily identified in the field or were of interest to authorities were collected for later species determination and to comply with Federal species-at-risk legislation.

Statistical analysis

To account for differences in sampling effort (shock seconds) among sites, all metrics were standardized as catch per unit effort (CPUE), where effort was shock seconds. The first analysis focused on determining individual differences between night and day sampling within CM and RR. Paired *t* test with transect as the link term were used to compare differences in (1) the composition of capture by sampling location and time period [e.g., CPUE, species richness per unit effort (herein richness)], (2) differences in catch of a key group of species [e.g., centrarchid CPUE, centrarchid richness CPUE (herein centrarchid richness)], and (3) CPUE and size (length) differences in commonly occurring species [Pumpkinseed (*Lepomis gibbosus*) and Yellow Perch (*Perca flavescens*)] and an invasive species (Round Goby). All paired analyses were conducted in JMP 9.0 (SAS, Cary, NC, USA) and the distribution of the mean differences either met or was close to the assumption of normality as required (McDonald 2014).

Non-metric multidimensional scaling (NMDS) ordination with Analysis of similarity (ANOSIM) and similarity percentages (SIMPER) were used to identify species-specific differences in assemblage among locations and time periods (Clarke 1993). The ANOSIM was first used to determine whether there were significant differences between the four location-times. The SIMPER analysis provided species-specific proportions of contribution to determine which species were driving differences among location-times. Only species exhibiting an arbitrary dissimilarity value greater than 5 % were interpreted. These analyses were completed using the *vegan* (Oksanen et al. 2013) package in R (v2.14; R Development Core Team 2013). The function metaMDS was used for the NMDS and employed a Bray–Curtis distance measure applied to the non-transformed CPUE data. This function automatically attempts to find a stable solution based on the global optima using several random starting positions (Oksanen et al. 2013). The input matrix contained the CPUE of 27 species at 40 sites. Catch-per unit effort was used rather than abundance or presence absence to account for differences in sampling effort. Since the goal was to compare differences in assemblages, all species (including rare species) were included in the analysis (McCune and Grace 2002). This type of approach has been used to evaluate fish assemblage differences in coastal environments (Boys and Williams 2012; Lowry et al. 2014) and streams (Growns 2008; Geheber and Piller 2012). Rejection of the null hypothesis (α) for all tests was $P \leq 0.05$, and all values are reported as mean ± standard deviation (SD) where appropriate.

For species that were determined to contribute to the differences among assemblages, a non-parametric Kruskal–Wallis (KW) test was performed to determine whether there were significant differences in CPUE among location-times. If the KW was significant, a multiple comparison post hoc analysis was conducted using the *kruskalmc* function in the *pgirmess* R package (Giraudoux 2014) to determine differences between location-times.

Results

In total, 1348 individual fish were captured representing 48 species. More were captured in CM (794) relative to RR (554); however, both sites had the same species richness with 24 different species encountered.

Raisin river

Within RR, there were no differences in CPUE or richness between day and night ($p > 0.346$; Table 1). Similarly, there were no differences in richness of centrarchids ($p > 0.319$; Table 1) or CPUE of centrarchids, Pumpkinseed, Yellow Perch, or Round Goby ($p > 0.104$; Table 1). Finally, while length differences for Round Goby could not be evaluated due to insufficient capture rates at both day and night, there were no differences for Yellow Perch or Pumpkinseed in terms of fish size (i.e., total length) in RR ($p > 0.279$; Table 1).

Cooper's marsh

In CM, considering the whole population, there was no difference in CPUE during day and night ($p = 0.135$); however, richness was significantly higher at night (0.032 ± 0.0057)

Table 1 Summary of metrics by location-time period

Metrics (mean ± SD)	Location-time period			
	CM_{Day}	CM_{Night}	RR_{Day}	RR_{Night}
Shocking effort (seconds)	325 ± 28	323 ± 58	250 ± 25	329 ± 79
CPUE (#/second)	0.108 ± 0.148 (36.3)	0.214 ± 0.095 (70.0)	0.094 ± 0.049 (23.1)	0.095 ± 0.049 (32.3)
Richness	0.016 ± 0.007 (5.1)	0.032 ± 0.006 (10.2)	0.029 ± 0.009 (7.1)	0.026 ± 0.008 (8.2)
Centrarchid CPUE	0.010 ± 0.011 (3.5)	0.062 ± 0.042 (20.4)	0.051 ± 0.028 (12.6)	0.052 ± 0.028 (17.5)
Centrarchid richness	0.004 ± 0.004 (1.4)	0.011 ± 0.004 (3.6)	0.014 ± 0.004 (3.5)	0.012 ± 0.004 (3.9)
Pumpkinseed CPUE	0.004 ± 0.005 (5.0)	0.018 ± 0.024 (6.0)	0.025 ± 0.023 (7.9)	0.023 ± 0.016 (2.6)
Pumpkinseed length (mm)	88 ± 34	106 ± 24	87 ± 26	89 ± 27
Yellow Perch CPUE	0.010 ± 0.023 (3.7)	0.033 ± 0.038 (12.2)	0.027 ± 0.014 (6.6)	0.027 ± 0.020 (9.6)
Yellow Perch length (mm)	126 ± 27	127 ± 33	136 ± 42	128 ± 35
Round Goby CPUE	0.007 ± 0.010 (8.2)	0.020 ± 0.020 (62.5)	0.001 ± 0.002 (7.1)	0.002 ± 0.002 (23.5)
Round Goby length (mm)	65 ± 11	66 ± 11	72 ± 3	68 ± 5

With the exception of shocking effort, all values have been corrected for effort and are presented as mean metric per shocking seconds ± SD. In parentheses beside these values is the uncorrected mean (mean of each metric for each location-time period mean that does not account for unequal shocking effort). For CPUE, the uncorrected means represent the mean total catch for each location-time

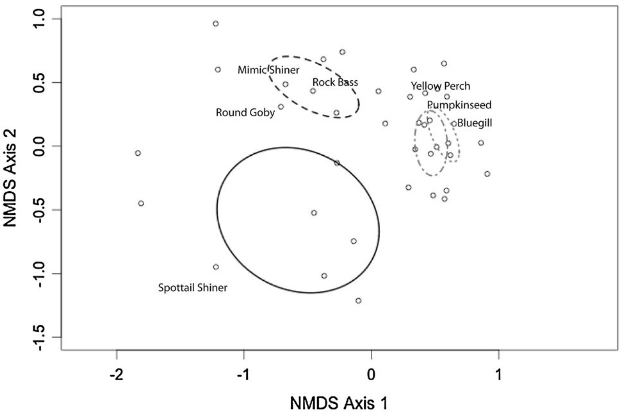

Fig. 2 Non-metric multidimensional scaling output of fish assemblage CPUE. The *open circles* represent the sample scores. *Ellipses* represent the 95 % confidence intervals, *black-dashed* CM_{Night}, *black-solid* CM_{Day}, *gray-dash-dots* RR_{Night}, *gray-dots* RR_{Day}. *Ellipses* that do not overlap represent distinct fish assemblages. The positions of the seven the species, which according to the SIMPER analysis, drove the main differences among location-times are shown

than day (0.016 ± 0.0067; $t_{(9)} = -8.018$, $p < 0.0001$). For centrarchids, both CPUE and richness were higher at night ($\text{CPUE} = 0.062 \pm 0.042$, $\text{richness} = 0.011 \pm 0.004$) compared to day ($\text{CPUE} = 0.010 \pm 0.012$, $\text{richness} = 0.004 \pm 0.004$; $t_{(9)} = -4.311$, $p = 0.0020$ and $t_{(9)} = -6.398$, $p = 0.0001$, respectively). Some species-specific patterns were also evident. For example, Round Goby had higher CPUE in CM at night (0.020 ± 0.020) compared to day (0.007 ± 0.010; $t_{(9)} = -2.51$, $p = 0.033$), and there were no differences in CM for Pumpkinseed and Yellow Perch ($p > 0.083$; Table 1). Finally, in terms of sizes, Pumpkinseed captured at night (120 ± 17 mm) were significantly larger than those caught during the day (77 ± 25 mm; $t_{(9)} = -2.61$, $p = 0.048$). There were no differences between diel periods with regards to size of Yellow Perch ($p = 0.913$; Table 1) and, similar to RR, low capture rates of Round Goby (particularly during the day) prevented a statistical comparison of size for this species.

Community comparisons

While there was considerable overlap of most species between locations and time periods, some species were only found in either the marsh (Banded Killifish, *Fundulus diaphanus*) or in the river (Greater Redhorse, *Moxostoma valenciennesi* and Walleye). Alternatively, some species were found in both locations, but only at night (Brook Silverside, *Labidesthes sicculus* and Smallmouth Bass, *Micropterus dolomieu*) or during the day (Spottail Shiner, *Notropis hudsonius*).

The ANOSIM was significant ($p = 0.001$; $R^2 = 0.376$), suggesting that there were differences in species assemblages among the location-times. From the NMDS output, the RR_{Day} and RR_{Night} species assemblages appeared to cluster closely together (Fig. 2). This link was confirmed by the SIMPER analysis where this pairing had the lowest dissimilarity value (46.3 %; Table 2). The NMDS output showed distinct assemblages for the CM_{Day} and CM_{Night}, which was confirmed with higher values of dissimilarity between these two time periods (Table 2).

For the SIMPER analysis, dissimilarity proportions for individual species greater than 5 % were included (Table 2). High rates of CPUE of Mimic Shiner (*Notropis volucellus*) at CM_{Night} was a main driver behind differences between this time period and the remaining location-times, accounting for close to 20 % of dissimilarity in all pairings. Indeed, the KW comparison for this species was significant ($\chi^2_{(3)} = 30.23$, $p < 0.0001$) and the post hoc analysis suggested CPUE was significantly higher at CM_{Night} than all other location times ($p < 0.05$). For CM_{Day}, high rates of capture of Spottail Shiner appeared to be an important driver of dissimilarity in all pairings (range = 10.2–13.4 %); however, despite significant results from the KW ($\chi^2_{(3)} = 13.35$, $p = 0.004$), no pairs were significantly different ($p > 0.05$). Otherwise dissimilarity at this location-time was driven by a low CPUE for many species, particularly for Yellow Perch (dissimilarity ranging from 11.5 to 17.1).

Differences during the day between CM and RR appeared to be driven by higher capture rates of Yellow Perch, Pumpkinseed, and Bluegill in RR relative to CM (dissimilarity rates of 17.1, 12.6, and 10.0, respectively). The KW found significant differences for all three of these species ($\chi^2_{(3)} = 10.356$, $p = 0.016$; $\chi^2_{(3)} = 13.38$, $p = 0.004$; $\chi^2_{(3)} = 28.97$, $p < 0.0001$, respectively) and confirmed significantly higher CPUE during the day for Yellow Perch and Pumpkinseed ($p < 0.05$) and significantly higher CPUE of Bluegill in RR for both the day and night ($p < 0.05$).

In contrast, differences at night appeared to be largely driven by higher capture rates in CM of Mimic Shiner, Rock Bass (*Ambloplites rupestris*), and Round Goby (dissimilarity rates of 19.5, 7.1, and 5.4, respectively; Table 2). As previously noted, the KW and multiple comparisons test confirmed significantly higher CPUE of Mimic Shiner relative to all other time periods. The KW for both Rock Bass and Round Goby were also both significant ($\chi^2_{(3)} = 10.62$, $p = 0.014$ and $\chi^2_{(3)} = 10.91$, $p = 0.012$, respectively), but CPUE at CM_{Night} was found to be higher only relative to CM_{Day} for Rock Bass and RR_{Day} for Round Goby ($p < 0.05$).

Table 2 Output from similarity percentages (SIMPER) analysis showing the contribution of key species to the overall dissimilarity between each location-time pairing

Location–time comparisons	Total dissimilarity	Species	Avg. A	Avg. B	Contribution
(A) CM$_{\text{Night}}$	82.2	Mimic Shiner	0.060	0.008	20.7
(B) CM$_{\text{Day}}$		Yellow Perch	0.033	0.010	11.5
		Spottail Shiner	*0.000*	*0.057*	*10.2*
		Rock Bass	0.028	0.003	7.8
		Round Goby	0.020	0.007	5.9
		Pumpkinseed	0.018	0.004	5.7
(A) CM$_{\text{Night}}$	72.0	Mimic Shiner	0.060	0.000	19.5
(B) RR$_{\text{Night}}$		Yellow Perch	0.033	0.028	9.2
		Pumpkinseed	*0.018*	*0.023*	*7.3*
		Rock Bass	0.028	0.004	7.1
		Round Goby	0.020	0.002	5.4
		Bluegill	*0.002*	*0.017*	*5.1*
(A) CM$_{\text{Night}}$	71.8	Mimic Shiner	0.060	0.000	19.4
(B) RR$_{\text{Day}}$		Yellow Perch	0.033	0.027	8.1
		Pumpkinseed	*0.018*	*0.025*	*7.9*
		Rock Bass	0.028	0.006	6.8
		Round Goby	0.020	0.001	5.5
(A) CM$_{\text{Day}}$	81.8	*Yellow Perch*	*0.010*	*0.028*	*16.4*
(B) RR$_{\text{Night}}$		Spottail Shiner	0.057	0.000	13.4
		Pumpkinseed	*0.004*	*0.023*	*12.2*
		Bluegill	*0.000*	*0.017*	*10.8*
		Brown Bullhead	0.008	0.007	5.1
		Round Goby	0.007	0.002	5.1
(A) CM$_{\text{Day}}$	83.7	*Yellow Perch*	*0.010*	*0.027*	*17.1*
(B) RR$_{\text{Day}}$		Spottail Shiner	0.057	0.000	13.4
		Pumpkinseed	*0.004*	*0.025*	*12.6*
		Bluegill	*0.000*	*0.015*	*10.0*
		Round Goby	0.007	0.001	5.1
		Brown Bullhead	0.008	0.006	5.0
(A) RR$_{\text{Night}}$	46.3	Pumpkinseed	0.023	0.025	10.3
(B) RR$_{\text{Day}}$		Yellow Perch	0.028	0.027	9.7
		Bluegill	0.017	0.015	6.2

The Avg. A represents the average CPUE for each species found in the location-time that is listed first, while Avg. B provides the same metric for the location-time that is listed second. Italicized averages are used to indicate when the Avg. B is significantly higher than Avg. A

Discussion

Effective assessment of fish community assemblages is integral to appropriate monitoring of community dynamics, as well as their application as indicators of freshwater ecosystem status and use in ecological restoration projects. Extensive research has concluded that gear selectivity can affect the composition of captured species (Lapointe et al. 2006; Ruetz et al. 2007; Cvetkovic et al. 2012) and there has been some research into horizontal and latitudinal movement in tributaries and lakes (Keast and Fox 1992; Reebs et al. 1995; McKenna 2008). However, a more thorough evaluation of daily variation in fish composition in two types of freshwater systems, a river tributary and proximate coastal marsh, is lacking. The current study revealed that within a river tributary there was minimal diel variability in fish community assemblages, richness or CPUE in summer when using boat electrofishing. In contrast, the assemblage and CPUE in a coastal marsh were both distinct from the tributary as well as distinct within the marsh during the day and night. Consequently, a complete evaluation of fish community assemblage in a coastal marsh during summer would necessitate sampling during both the day and night.

Diel freshwater migrations between nearshore and pelagic areas and within lotic systems are common occurrences. Studies assessing diurnal fish feeding habits in lake systems have established peak feeding times, and thus peak activity times, for certain species (Keast and Welsh 1968; Keast and Fox 1992), habitat preferences (Lewin et al. 2004) and predator–prey interactions (Jacobsen and Perrow 1998). Evidence for diel movement in tributaries is also provided by numerous studies and for various species (Reebs et al. 1995; Kubecka and Duncan 1998; Wolter and Freyhof 2004 McKenna 2008). However, the direction (i.e., lateral vs. longitudinal) and timing (i.e., inshore vs. offshore movements at night) of these movements is variable and dependent on species, habitat and life stage (reviewed in Lucas and Baras 2001), as well as the trophic dynamics within each system (e.g., abundance of planktivorous fish vs. piscivorous fish; Burks et al. 2002). There is no doubt that the interactions are complex, and that community structure, predator–prey relationships, competition, and trophic status all govern the behaviour of fishes, making comparisons among studies and between systems challenging.

In the present study, there was evidence of diel changes in fish community assemblages in a coastal marsh in the Cornwall AOC. Differences within the marsh between day and night were driven by greater richness and CPUE at night, which is similar to McKenna (2008), who found richness and diversity (as calculated by the Shannon–Wiener index) in tributaries and embayments of Lake Ontario were greater at night relative to day, but found no difference in abundance. Increased richness at night was partly driven by the presence of Mimic Shiners, which were often observed during evening surveys, particularly in CM, with CPUE increasing nearly tenfold from day to night. With the available data we are unable to evaluate the cause of this shift, but it may be related to the temporal availability of prey or predator avoidance. Regardless, these results clearly reveal that the fish assemblage structure within coastal marshes may be influenced by sampling time; therefore night and day sampling is necessary to gain a true picture of the presence and relative abundance of all species that utilize the marsh habitat.

While previous studies have documented diel shifts in littoral habitat utilization in lotic systems both laterally and longitudinally (Clough and Ladle 1997; Kubecka and Duncan 1998; Lucas and Baras 2001), we observed limited short-term diel variability in species composition in RR. Indeed, the three species that contributed to night/day dissimilarity in RR had absolute differences in CPUE less than 0.03, resulting in clear overlap between RR_{Day} and RR_{Night} in the NMDS. Overall, this would suggest that during the summer months there are minimal diel changes in fish assemblage in a freshwater tributary. These results may be constrained by gear bias (effective electrofishing depth), which may have limited our detection capacity of typical diel migrants such as Walleye that are known to occur in the RR. Further, given that the transects within RR encompassed a range of habitats, including vegetated tributaries as well as sections with minimal vegetation and artificial side-channels, any observation of diel movement occurring in one transect may have been lost when combined with observations from all transects in the analysis. Fishes found in transects with homogeneous habitat structure and substrate would have little to gain from diel movement and may have adapted to habitat conditions such that little migration occurs during the summer season. Without access to nearby open water or deeper areas as in CM, these fishes likely remain in more localized home ranges. Repeated sampling of transects within the tributary may help to conclusively determine whether there is any evidence within transects for diel shifts in community assemblage. However, since management units typically focus at the tributary level (Imhof et al. 1996), the current study suggests that within the summer months tributary sampling either during the day or at night may be sufficient to capture and therefore characterize the entire community assemblage, keeping gear sampling biases in mind.

In CM, there was significantly lower CPUE and centrarchid abundance at day relative to night, which was largely driven by changes in CPUE of Bluegill. This is not surprising given that high temperatures during the day can act to displace fish to areas of refuge, such as deeper, cooler waters or among structured habitat such as macrophyte communities (Chow-Fraser et al. 2006). Boat electrofishing is typically not effective below depths of 1.5–2.0 m (Brousseau et al. 2011), which may partially explain the lower catch rates during mid-day in CM if fish were migrating to deeper waters. While some fish species remain in the nearshore zone during the day and migrate towards the open water at night, other species exhibit the opposite movements (Lewin et al. 2004). The latter may be the case in CM because the predominant vegetation is dense cattails with the remainder of the marsh affording less cover and structure. Because there are numerous factors affecting the diel movements of fishes (e.g., food availability and diel horizontal and vertical movement of zooplankton and other invertebrates (Jacobsen and Perrow 1998; Lucas and Baras 2001; Burks et al. 2002; Lewin et al. 2004; Farrell and Hodgson 2012), without further information on habitat structure and prey dynamics in the marsh and river it is difficult to postulate on the specific drivers behind the patterns in fish assemblages we observed. Importantly, sampling a marsh only during the day can lead to inaccurate conclusions on the fish community structure.

Pumpkinseed had greater CPUE in the tributary relative to the marsh (significantly lower CPUE during the day). Because CM is a relatively exposed marsh with pockets of sparse macrophyte cover, the habitat may not be as ideal for Pumpkinseed compared to the quieter backwater areas that were sampled in the RR (Lewin et al. 2004; Cvetkovic 2008; Smyth 2008). RR is a slow-moving river with portions of heavily vegetated shorelines, which provide spawning and nursery habitat as well as increased zooplankton and zoobenthos abundance (Lewin et al. 2004). When they were found, Pumpkinseed were significantly larger at night in the marsh, which may be a result of diel movements by a portion of the population of this species. Pumpkinseed are known to show morphological divergence between littoral and pelagic individuals, particularly in areas where there is limited competition from congenerics (e.g., Bluegill; Robinson et al. 2000; Gillespie and Fox 2003), as is the case in CM.

While the paired analysis suggested that Yellow Perch CPUE was not significantly different among any of the location-times, their CPUE did contribute to the dissimilarity in community assemblages in the marsh at day and night with on average three times higher catch at night. This increase in capture is consistent with previous findings of higher CPUE of Yellow Perch in marshes at night (Helfman 1979; Pierce et al. 2001; Lewin et al. 2004). In Lake Opinicon in Ontario, Yellow Perch feeding times peaked in the evening and mid-morning (Keast and Welsh 1968). Other studies have shown that Yellow Perch seem to be active throughout the entire day and indeed, Helfman (1979), found that Yellow Perch behaviour varied from lake to lake, while in tributaries and embayments in Lake Ontario Yellow Perch were caught at all times of day and by all sampling gear (e.g., seines and trawls; McKenna 2008). If sampling programs are focused on Yellow Perch, both day and night electrofishing, or alternately overnight trap or fyke nets (as outlined in Chow-Fraser et al. 2006, Seilheimer and Chow-Fraser 2007, and Ruetz et al. 2007), are viable sampling options in coastal marshes.

Understanding the temporal location of invasive species is important for documenting their distribution and abundance to assist with controlling their expansion. We therefore evaluated the temporal CPUE of Round Goby, an invasive species commonly found throughout the Laurentian Great Lakes and their connecting water-bodies (Jude et al. 1992; Kornis et al. 2012). Gobies were captured between three and ten times more frequently in the coastal marsh relative to the tributary, with significantly higher CPUE at $\mathrm{CM_{Night}}$ than $\mathrm{RR_{Day}}$. This could be the result of the exposed nature of CM providing better foraging opportunities for Round Goby in the form of dreissenid mussels (one of their preferred prey; Kornis et al. 2012). The distribution of Round Goby within the Great Lakes is still being mapped, and while they seem to prefer rocky substrate and dreissenid mussels as food sources, they can invade areas with soft muddy substrate and eat a variety of other prey (Kornis et al. 2012).

Given their increasing presence in tributaries and areas with less rocky substrates (Bronnenhuber et al. 2011; Brownscombe and Fox 2012), the sampling methodology (boat electrofishing) combined with the geomorphology of the marsh and tributary may also have contributed to the observed differences in Round Goby abundance. Boat electrofishing has a limited effective depth such that benthic species such as Round Goby may have lower rates of capture, especially in turbid waters where clarity is low, as it was in the tributary (Bohlin et al. 1989; Brousseau et al. 2011). Indeed, in Muskegon Lake, Ruetz et al. (2007) found that small-mesh fyke nets were much more efficient at capturing Round Goby than night boat electrofishing. Given this, it is possible that there was a sampling bias in our study and different sampling gear may be more efficient in the tributary compared to the coastal marsh with regards to certain species such as the Round Goby. An alternate strategy to capturing fish that may be taking refuge in deep water is to use greater power output, which has been used for sampling large catfish in the Mississippi River (Schramm and Eggleton 2006). Based on this caveat we are hesitant to conclusively state that benthic species such as Round Goby were more prevalent in the coastal marsh than the tributary. Nonetheless, it is rather typical in monitoring programs to make such conclusions while using a single sampling gear.

Due to differences in effectiveness of sampling gear, it is critical to identify the question being addressed and subsequently ensure that the sampling methodology is effective. Active gears, such as seines or electrofishing, can be effective at capturing a representative sample of the fish assemblage and also can be used to easily estimate CPUE. However, as with any gear there are caveats, and deep water (both gears) and rough bottom (seines in particular) influence their utility. In these situations other methods may be more suitable for assessing fish assemblages.

A key component in the delisting process of Areas of Concern within the Laurentian Great Lakes is the effective monitoring of fish and wildlife populations in both degraded and reference locations. A complete understanding of species assemblages is also critical for the development of appropriate ecological restoration strategies, ensuring they will benefit the community in its entirety. The current study provides an evaluation of fish assemblages in two distinct aquatic ecosystems and suggests that while daily timing of sampling may not influence the recorded assemblage in a tributary in mid-summer, it has a profound impact on the recorded assemblage in a nearby coastal marsh. The major caveat of the present study is that, due to practical limitations and costs, it was only conducted during one season (summer) and each diel period was only sampled once per site. Variation in habitat use by fish species across seasons and among life history phases suggests that, by sampling only in the summer, we are only capturing a portion of the fish community (e.g., Pope and Willis 1996; Jordan et al. 1998). In order to gain a better understanding of fish community diel movement within these systems, an extensive sampling regime, including multiple sampling events per season and the incorporation of additional seasons, would be preferential. Nonetheless, managers responsible for these types of systems (e.g., Cornwall AOC and RRCA for this region) need to take these findings into consideration when conducting routine monitoring and comparisons with future and previous data. In order to elucidate the drivers behind the fish–habitat interactions and movement patterns that were noted in this study, future studies need to incorporate detailed habitat surveys including sampling substrate, vegetation (composition, percent cover), invertebrate community dynamics, hydrology, water-quality information, and broader, large-scale landscape characteristics.

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