Sources and fate of organic matter in constructed versus natural coastal waterways

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ABSTRACT

Coastal wetlands are increasingly being converted into canal estates with potential consequences for ecosystem functioning. We compared the sources and fate of organic matter and water quality at four types of canal habitats (entrances and ends of canals, canal lakes and lake edges) and shallow and deep natural habitats (four replicates of each habitat). The fate of labile organic matter was assessed by measuring rates of scavenging of carrion. Surface sediments were analysed for organic carbon content and stable carbon isotopes, fatty acid biomarkers and compound specific stable isotope analysis of selected fatty acids were used to elucidate sources of sedimentary organic matter. Canal lakes differed from other habitats and were characterised by negligible scavenging, larger quantities of organic matter comprised of higher contributions from diatoms, and hypoxia. Despite some trends, natural habitats were statistically indistinguishable from canal entrances and ends. Variation among replicate habitats was large.

1. Introduction

Coastal wetlands, such as mangroves, saltmarshes and seagrass meadows, provide important ecosystem services. For example, they are habitat for macrofauna, support coastal food webs, maintain water quality by trapping sediments and organic matter, and efficiently recycle organic matter or sequester it (Barbier et al., 2011; McLeod et al., 2011). Growing demand for waterfront residential property has seen increasing areas of coastal wetlands being converted to residential canal estates and 4000 linear km of canal estates now occur globally (Waltham and Connolly, 2011). These conditions make canal estates less hospitable for fish and macroinvertebrates and the biomass of these taxa is usually less than in nearby unmodified coastal waterways (Maxted et al., 1997; Morton, 1989).

Several of the important primary producers in coastal wetlands, including mangroves, saltmarsh plants and seagrasses are usually absent in constructed waterways. Primary producers in canal estates are thus dominated by phytoplankton, benthic microalgae and some novel sources, such as urban grasses (Connolly, 2003). This can lead to food webs within constructed waterways being supported by basal carbon sources that are different from those in natural waterways (Connolly, 2003; Waltham and Connolly, 2006). Moreover, the particulate organic matter that is deposited in sediments in constructed vs natural waterways may differ in lability, since phytoplankton and microalgae typically have lower C:N than highly refractory mangrove leaves and seagrasses (Enriquez et al., 1993).

A variety of methods can be used to identify the sources of carbon within sediments including stable isotopes and lipid biomarkers. Stable isotopes are useful when different sources have distinct isotopic signatures (Fry, 2013). Fatty acid biomarkers can usually identify a larger range of sources than stable isotopes (e.g. bacteria and different types of...
algae) but the assignment of different types of fatty acids to their sources is sometimes ambiguous (Dalsgaard et al., 2003). Isotopic analysis of particular fatty acids (i.e. compound-specific isotopic analysis) can more reliably resolve the sources of different fatty acids. Moreover, isotopic analysis of fatty acids unique to bacteria (e.g. odd carbon-numbered and branched-chain fatty acids and the mono-unsaturated fatty acid (MUFA) 18:1ω7) can help elucidate the source of organic matter being used by bacteria (Boschker et al., 1999).

Recycling of particulate organic matter is a critical ecosystem function. Particulate organic matter that accumulates on the benthos may have several fates: it may be consumed by scavengers or re-mineralised by microbes and thus recycled to the environment, or it may be buried in sediments. In natural waterways that support an abundant and diverse community of scavengers, most detritus is probably consumed. Because canals support fewer fish and macro-invertebrates overall and fewer detritivores (Maxted et al., 1997; Morton, 1989), less detritus is probably scavenged and a larger proportion of the decaying organic material may be remineralised by microbes or accumulate in sediments. Scavenging rates may vary through time, however, as consumption rates of organisms often vary seasonally (e.g. Micheli, 1997). Moreover, deep lakes and the ends of canals that are poorly flushed may act as sinks for fine sediments and organic matter (Cosser, 1989) and if the organic matter content is high, the deep lakes may experience net respiration and hypoxia. Consequently, compared to natural waterways sediments in artificial waterways potentially contain more organic matter and bacterial biomass, have a different composition of organic matter and their bottom waters may be more prone to hypoxia.

The Gold Coast region of southeast Queensland, Australia, supports > 150 linear km of artificial waterways (Waltham and Connolly, 2011). Canal systems have been established within the Nerang River, Tallebudgera River and Currumbin River estuaries and have replaced extensive mangrove forests, saltmarshes and seagrass meadows. The canal estates are comprised of flow-through and dead-end canals and deep lakes (some exceeding 25 m depth). The lakes were created when sediment was excavated during the construction of the canal system and used to elevate the adjacent low-lying residential land. The hydrology of the lakes is different to the canals and natural waterways since they are much deeper and some are located behind tidal gates that restrict tidal flows (Zigic et al., 2002; Waltham and Connolly, 2013).

Similar species of fish inhabit the Gold Coast canals and nearby natural waterways but the relative abundances of species vary (Morton, 1989; Morton, 1992; Morton et al., 1987). In particular, the dead-ends of canals support fewer macrobenthic carnivores (e.g. sparids and tetraodontids) and detritivores (mugilids) than canal entrances or rivers (Morton, 1989; Morton, 1992) and abundances of these groups are also reduced compared to nearby natural waterways (Morton et al., 1987). The ends of canals also support reduced species richness and diversity of benthic macroinvertebrates (Cosser, 1989). Fish assemblages are depauperate and sometimes absent in the deep lakes (Waltham and Connolly, 2013). Consequently, rates of scavenging in the canal system, and in the deep lakes in particular, are likely to be lower than in natural waterways.

The objective of this study was to investigate the sources and fate of particulate organic matter in different types of habitats within artificial and natural waterways of southeast Queensland, Australia to determine how the proliferation of artificial waterways affects coastal ecosystem functioning. We tested the following hypotheses:

1. That rates of scavenging of carrion would be greater in natural than canal habitats and lowest in deep lakes and that rates of scavenging would be greater in summer than in winter.
2. That canal habitats would contain more sedimentary organic matter, finer sediments and be more prone to hypoxia than natural habitats, with the most extreme values occurring in the deep lakes.
3. That sediment organic matter within the constructed waterways would have greater contributions from bacteria and microalgae and less from macrophytes than natural waterways.

2. Materials and methods

The study was done in the Gold Coast region of southeast Queensland, Australia. Six types of habitat were sampled: 1) canal lakes (depth range 8–23 m); 2) edges of canal lakes (< 2 m); 3) entrances of canals (where a canal intersected the river; depth range 1.4–5 m); 4) ends of canals (> 500 m from the canal entrance; depth range 1.5–2.1 m); 5) natural deep habitats (depth range 3.5–4.3 m); and 6) shallow natural shallow habitats (depth range 0.7–1.0 m) (for examples see Fig. 1). Four independent replicate locations were sampled for each type of habitat from across the Gold Coast region (24 locations overall spread over 40 km of coastline). Except for deep lakes and edges of deep lakes, all locations were > 500 m apart. Latitude and longitudes of all 24 locations are provided in Supplementary Table 1. Natural habitats were located within the Broadwater of southern Moreton Bay and consisted of a network of channels containing extensive seagrass meadows (predominantly Zostera muelleri) fringed by mangroves (predominantly Avicennia marina) and saltmarsh flats (predominantly Sporobolus virginicus).

2.1. Rates of scavenging

Rates of scavenging were assessed at all habitats and locations, except for the edges of the deep lakes. Scavenging rates were measured twice during summer (December 2009 and February/March 2010) and twice during winter (June and July/August 2010) to assess temporal variation within one year. Scavenging was assessed using a commonly-employed assay (e.g. Porter and Scanes, 2015) by quantifying the mass of carrion consumed over 1 h. Fifteen ‘dillies’ (300 mm diameter flat rings covered in 20 mm mesh) were baited with a known mass (~50 g) of dead pilchards (Sardinops sagax) and lowered to the benthos. Any carrion remaining on the dillies was re-weighed after retrieval.

2.2. Sampling and analyses of sediments

Surface sediments (~15 cm depth) were sampled between December 2012 and January 2013 for analysis of % of particulate organic matter (%POM), percentage of organic carbon (%OC), δ13C, atomic C:N, sediment grain size distributions, profiles of fatty acid methyl esters (FAMES) and compound-specific isotopic analyses (CSIA; δ13C) of selected bacterial fatty acids. At every location, five samples of unvegetated surface sediment were collected using a van Veen grab and immediately cooled and then frozen when returned to the laboratory.

The percentage of organic matter was determined by wet-sieving sub-samples of sediment through a 2 mm sieve. Organic material retained on the sieve and sediments < 2 mm that passed through the sieve were retained and dried in an oven at 60 °C until constant weight. The %POM was determined as the proportion of the organic material relative to the total weight of the dried sample. Sediments analysed for %OC, δ13C and atomic C:N were dried at 60 °C until constant weight, homogenised and subsamples were extracted. Sub-samples were acidified with 1 M HCl to remove inorganic carbonates and redried. Samples were then ground using a mortar and pestle, weighed and combusted on a Sercon Hydra 20–22 mass spectrometer. Isotope results were presented using standard δ notation (per mil ‰), defined as:

$$\delta^{13}C = \left(\frac{R_{sample}}{R_{stand}}\right) - 1 \times 10^3$$

where R represents 13C/12C. PDB limestone was the standard reference for carbon. Atomic C:N was calculated based on percentage dry weight of the two elements. Sediment grain size distribution was determined from volumetric particle size distribution (0.1–2000 μm) measurements. The five replicate samples collected at each location
were pooled and homogenised. A sub-sample was taken, ultrasonically treated and measured using laser diffraction particle size analysis (Malvern Mastersizer, 3000).

2.3. Preparation of fatty acid methyl esters

Lipids were extracted from sediments using an accelerated solvent extractor. 10 g (dry weight) of sediment was transferred to an 11 ml extraction cell and any void was filled with glass beads or acid-washed sand. ASE extraction was performed at 100 °C and 1500 psi using a chloroform:methanol ratio of 2:1. The extract was concentrated to 1 ml by rotary evaporation at 50 °C and the extracts were transferred to clean tubes and dried under nitrogen gas. Samples were saponified by adding 1 ml of NaOH and 2 ml of methanol. Tubes were heated in boiling water for 2 h and cooled before being acidified with 0.4 ml of 37.5% HCl. 2 ml of chloroform was added and samples were centrifuged at 3000 rpm for 5 min. Samples were transferred to new tubes and the aqueous phase was extracted again with another 2 ml of chloroform. The chloroform extracts were combined and evaporated and dried under nitrogen gas. Samples were methylated by adding 1 ml of 14% BF₃-methanol into the liquid residue. The tube was purged with N₂, closed, heated at 90 °C for 10 min and cooled. To isolate FAMES 2 ml of hexane and 2 ml of water were added and the samples mixed on a vortex for 1 min and centrifuged at 2500 rpm for 3 min. The upper hexane layer was extracted and the extraction was repeated with another 2 ml of hexane. The combined hexane phases were stored together and concentrated under nitrogen before being stored at −20 °C until analysed. Samples were analysed by a gas chromatograph using an Omegawax 320 *Supelco column (30 m × 0.32 mm internal diameter, 0.25 μm film thickness). N₂ was used as the carrier gas with a flow rate of 1.5 ml min⁻¹. After injection at 24 °C the oven temperature was raised to 60 °C for 1 min, 150 °C at 40 °C min⁻¹ and held for 3 min, then 240 °C at 2 °C min⁻¹ and held for 10 min. Bacterial fatty acids were identified by comparing the samples with that of bacterial acid methyl ester (BAME, Sigma) and the percentage contribution of each bacterial FAME to total FAMEs extracted from each sample were calculated as following:

\[
\text{FA\%} = \left( \frac{A_{\text{FA}}}{A_{\text{Total}}} \right) \times 100
\]

where \( A_{\text{FA}} \) = Area of FAME of interest and \( A_{\text{Total}} \) = Total area of all FAMES in the sample.

CSIA of 15:0 iso and 15:0 anteiso was done at the Stable Isotope Facility at the University of California, Davis USA. Replicate samples from within each location were pooled and locations were used as replicates. FAME extracts were combusted to gas and analysed on a Thermo GC/C-IRMS coupled to a Delta V Advantage isotope ratio mass spectrometer.

Fig. 1. Examples of the six types of habitat (four canal habitats and two natural habitats) sampled within the canal system and within unmodified natural waterways of the Gold Coast, Australia. Four replicate locations were sampled for each type of habitat from across the Gold Coast region. See Supplementary Table 1 for geographic coordinates of all 24 locations.
were used to identify which levels differed.

Spatial and temporal variability in rates of scavenging and water quality parameters measured during 2009/2010 (%DO, temperature and salinity) were analysed using four-way PERMANOVAs. The factors were Season (a fixed factor with two levels; summer and winter); Time (a random factor that was nested within season and had two levels); Habitat (a fixed factor with five levels) and Location (a random factor that was nested within habitat and had 4 levels).

Overall differences among habitats were analysed using a composite multivariate data set comprising 2012/2013 data on water quality (%DO, temperature, salinity, pH) and sediment characteristics (%POM, %OC, δ13C, C:N, median grain size and the scores of the first axis of the principle components of FAMES (which explained 41% of the total variation in FAMEs)). Average values of each variable were compiled for each location and locations were used as replicates in the analysis.

Data were normalised to account for differences in units of measurement among variables. Differences among habitats were analysed using a one-way PERMANOVA. Canonical analysis of principal coordinates (CAP) was used to graphically represent differences among habitats. CAP is a constrained ordination of multivariate points that uses a priori hypotheses to produce the plot (Anderson and Willis, 2003).

Two-way PERMANOVAs were used to analyse overall FAME profiles. Univariate two-way PERMANOVAs were also used to analyse 2012/2013 water quality parameters (%DO, temperature, salinity and pH), sediment parameters (%POM, %OC, C:N, δ13C), total bacterial fatty acids (Σ 15:0+15:0 iso + 15:0 anteisio + 16:0 iso + 17:0 + 17:0 iso + 18:1aa7), the ratio of even-numbered very long chain fatty acids (VLFCFA; Σ 22:0 + 24:0) to even-numbered long chain fatty acids (LCFA; Σ 14:0 + 16:0 + 18:0); an indicator of the relative contribution of aquatic and terrestrial organic matter; (Meyers, 1997)) and the ratio of 16:1/16:0 (which reflects differences between diatom and dinoflagellates sources and values > 1.6 indicate a predominance of diatom sources (Budge and Parrish, 1998)). The factors were ‘Habitat’, a fixed factor, and ‘Location’, a random factor that was nested within habitat.

CSIA of 15:0 iso and 15:0 anteisio and sediment grain size distributions among habitats were analysed using a one-way PERMANOVAs.

3. Results

Rates of scavenging varied among habitats but patterns were not consistent between summer and winter (Supplementary Table 1). During summer average rates of scavenging in natural habitats exceeded 50% during the one-hour deployments but scavenging rates were < 15% in artificial waterways and were negligible in the canal lakes (Fig. 2A). Rates of scavenging were lower and more variable during winter. Average rates of scavenging were greatest and exceeded 30% in deep natural habitats and were lowest in canal lakes (8%) but varied between 15 and 20% across all other habitats (Fig. 2A).

During the 2009/2010 sampling period, dissolved oxygen saturation was consistently lower in the canal lakes than in all other habitats (Fig. 2B; Supplementary Table 1). Patterns of %DO were consistent between summer and winter but varied among times and locations within habitats (Supplementary Tables 1, 2). Water temperature was cooler in winter than summer (18.2 ± 0.2 °C vs 26.7 ± 0.3 °C) at all habitats, except for the deep lakes, where the temperature of bottom waters did not differ significantly between seasons (summer = 24.3 ± 1.1 °C; winter = 21.4 ± 0.9 °C; Supplementary Tables 1 and 2). Salinity varied among locations but patterns were not consistent through time (Supplementary Tables 1 and 2).

Analysis of the composite multivariate dataset of sediment and water quality parameters collected in 2012/2013 indicated variation among habitats (Pseudo-F = 3.187; Pseudo-P = 0.007; Fig. 3). Post-hoc analyses revealed that canal lakes differed from all other habitats, except the edges of the canal lakes. No significant differences were detected among the remaining habitats although there was a trend for ends of canals to differ from both deep (Pseudo-P = 0.055) and shallow
Sediments were dominated by silt (2–50 μm; 40%), medium sand (250–500 μm; 29%) and fine sand (100–250 μm; 21%), very fine sand (50–100 μm) and coarse sand (500–1000 μm) contributed < 5% each, on average. The distribution of grain sizes did not vary among habitats (Pseudo-F = 1.513; Pseudo-P = 0.231). The % POM varied greatly among habitats and locations within habitats (Table 1). Canal lakes contained large amounts of POM (> 70%) whereas both natural locations and canal entrances contained relatively little (< 20%; Fig. 4A). The spatial variability in the %OC in sediments among habitats mirrored that of the %POM and was lowest in the natural habitats and at the entrances of canals (< 0.75%), greatest in the deep lakes (> 2%), and intermediate at the ends of canals and at the edges of the deep lakes (Table 1; Fig. 4B). Sediments in deep natural waterways were most enriched in 13C (−21.5‰) but were similar across all other habitats (range from −23.4‰ to −24.6‰) (Table 1; Fig. 4C). C:N did not vary among habitats but varied substantially among locations within habitats. C:N ranged between 5.6 and 17.2 (average 10.6) across all habitats.

Sixteen individual fatty acids were isolated and expressed as a percentage of the overall FAMEs extracted from each sample. FAMEs comprised saturated fatty acids (SAFAs), including long chain (LCFAs; C14–C20) and very long chain fatty acids (VLCFAs C22–C24), four branched fatty acids (BAFAs) and four monounsaturated fatty acids (MUFAs). No polyunsaturated fatty acids were detected. FAMEs were dominated by palmitic acid (16:0), which contributed, on average, 41% of all FAMEs. Bacterial fatty acids (15:0, iso- and anteiso-15:0, iso-16:0, 17:0, iso-17:0, 18:1ω7) were also abundant and contributed 15.9% of total FAMEs. VLCFAs were relatively scarce (5.9% of total FAs).

Despite a strong trend, overall FAME profiles did not vary significantly among habitats but did vary among locations within habitats (Table 1). Total bacterial FAMEs varied among habitats and locations within habitats but pair-wise comparisons were unable to determine which habitats differed (Table 1; Fig. 5A). The bacterial marker 15:0 anteiso was the only individual bacterial FAME that varied among habitats and locations (Table 1). The percentage contribution of 15:0 anteiso in the deep natural habitat was lower than all other habitats, except the lake edge (Fig. 5B). The diatom marker 14:0 varied among habitats and locations within habitats (Table 1). 14:0 contributed a greater percentage of the FAME profile in the deep lakes compared to the canal entrances and both shallow habitats (Fig. 5C). Neither the ratio of SCFA to VLCFAs nor the ratio of 16:1/16:0 varied among habitats but both varied among locations within habitats (Table 1). δ13C of the bacterial markers 15:0 iso (Pseudo-F = 1.143; Pseudo-P = 0.393) and very long chain fatty acids (VLCFAs C22–C24), four branched fatty acids (BAFAs) and four monounsaturated fatty acids (MUFAs). No polyunsaturated fatty acids were detected. FAMEs were dominated by palmitic acid (16:0), which contributed, on average, 41% of all FAMEs. Bacterial fatty acids (15:0, iso- and anteiso-15:0, iso-16:0, 17:0, iso-17:0, 18:1ω7) were also abundant and contributed 15.9% of total FAMEs. VLCFAs were relatively scarce (5.9% of total FAs).
and 15:0 anteiso (Pseudo-F = 0.670; Pseudo-P = 0.687) did not vary among habitats. The average $\delta^{13}$C of 15:0 anteiso was $-26.4\%$ (± 0.65) and of 15:0 iso was $-34.4\%$ (± 1.49).

Trends in dissolved oxygen saturation sampled in 2012/2013 were similar to those sampled in 2009/2010, with oxygen saturation of canal lakes being lower than all other habitats, except the edges of the lakes (Table 1; Fig. 6A). pH was lowest in the canal lakes (< 7.1) and highest in the two natural habitats (> 7.8; Fig. 6B). Salinity and temperature did not vary among habitats, but varied among locations within habitats (Table 1; Supplementary Table 2).

4. Discussion

Canal lakes stood out as being different from all other habitats, except the edges of the lakes. The environmental conditions in the bottoms of lakes were characterised by almost negligible amounts of scavenging, large quantities of organic matter, low levels of dissolved oxygen and very low pH. Hence our hypothesis that environmental conditions would be most extreme in the canal lakes was shown. Despite strong trends in some variables, the entrances and ends of canals and the edges of the canal lakes were statistically indistinguishable from the natural habitats, indicating that the overall environmental conditions of those canal habitats were not significantly different to those in unmodified waterways.

One of the clearest patterns to emerge was the difference in the amounts of organic matter among habitats. The canal lakes contained large amounts of POM but levels were low in the natural habitats and at the entrances of canals. Not surprisingly, the percentage of organic carbon in the sediments mirrored that of the particulate matter and exceeded 2% in the canal lakes, which is much greater than even the highest measurements made in a nearby shallow vegetated coastal lagoon with limited tidal exchange (Dunn et al., 2008). The bathymetry of the lakes, which are localised deep points in the canal system, would facilitate the accumulation of organic material. Indeed, extreme vertical stratification of salinity in the canal lakes indicates minimal exchange of bottom-waters (Lemckert, 2006), hence organic matter that accumulates in the basins is probably not removed via tidal flushing.

The accumulation of organic matter in poorly flushed systems facilitates hypoxia because microbial remineralisation of the organic material depletes oxygen from the water column (Cai et al., 2011). Our observations of oxygen saturation were consistent with this, with oxygen saturation being consistently lowest in the canal lakes, which had the highest organic loads, and highest in the natural habitats and canal entrances, where organic loads were smallest. The limited tidal flushing of the deep lakes, coupled with their depth and stratification (Lemckert, 2006), may limit replenishment of oxygen in the bottom waters through tidal movements or wind-driven reaeration. Moreover, although hypoxia can occur naturally, the dissolved oxygen saturation of the natural habitats always exceeded that of the canal lakes and the persistence of the trend suggests that the localised hypoxia in the canal lakes reflected their hydrological environment. The pH of the canal
lakes was also much lower than in the natural habitats and there was a
trend (although not statistically significant) for it to be lower than the
other canal habitats. This may have reflected the high rates of re-
spiration that occur in hypoxic regions, which elevates pCO₂ and re-
duces pH (Cai et al., 2011). pH may also be reduced in canal systems
because, during construction, canals may cut into aquifers and facilitate
outgassing of CO₂ through the release of groundwater (Macklin et al.,
2014). High concentrations of organic acids (e.g. humic and fulvic),
which were not measured in this study, might also reduce pH. Moreover,
because measurements were made during daytime, photosynthesis of
seagrasses in natural habitats may have enhanced O₂ and depleted CO₂
and exacerbated differences in dissolved oxygen and pH between nat-
ural and artificial habitats. Overall, the average pH of the canal system
was comparable to previous measurements from the same canal system
(Macklin et al., 2014).

A combination of stable isotopes, FAMEs, and compound specific
stable isotope analyses were used to try to elucidate the sources of or-
ganic material in the sediments of the canal and natural habitats.
Specifically, we had predicted that organic matter in the canal habitats
would have a higher prevalence of bacteria and microalgae and a
smaller contribution of macrophytes than the natural habitats. Sediments in all four canal habitats were depleted in 13C, with 8δ13C ranging from −23.3% to −24.3%. Connolly (2003) sampled the 8δ13C of the dominant autotrophs in natural waterways and the canal system of the Gold Coast and seston (including phytoplankton) and benthic microalgae were the most depleted sources in the canals (−24.8% and −23.1% respectively). The strong similarity in the 8δ13C of the sediments in the current study and seston and microalgae of canals reported by Connolly (2003) indicates that sedimentary organic pools were probably dominated by seston and/or microalgae and that mac-
roalgae and urban grasses, both of which are enriched in 13C, made
minor or no contributions. Shallow natural habitats displayed similar
values to those of artificial habitats and were consistent with con-
tributions from seston, microalgae and, potentially, some input from
mangroves, which are depleted in 13C and are abundant in the natural
waterways but very scarce in the canal system. However, despite not
being statistically different to other habitats, deep natural habitats ex-
hibited a strong trend (pairwise test = 0.057) to be more enriched in
13C than the natural shallow habitats, which might reflect a greater
contribution from seagrass or saltmarsh grasses, both of which are
highly enriched in 13C. Given that all habitats sampled were sub-tidal,
however, seagrasses rather than saltmarsh grasses are more likely
to have contributed to the enriched signature in deep natural habitats
(Connolly and Waltham, 2015). The apparent difference between the
deep and shallow habitats may reflect that deep areas may act as basins
where seagrass detritus can accumulate whereas shallow areas do not.

Stable isotopes were limited in their ability to identify the organic
sources in the sediments because the number of elements available for
analysis was less than the number of possible sources (Fry, 2013).
Isotope analyses were, therefore, complemented with FAME analyses to
further elucidate carbon sources. The diatom marker 14:0 was abun-
dant in all habitats (> 4%) but was most abundant in the canal lakes,
which was consistent with our hypothesis that microalgae would
further elucidate carbon sources. The diatom marker 14:0 was abun-
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further elucidate carbon sources. The diatom marker 14:0 was abun-
dant in all habitats (> 4%) but was most abundant in the canal lakes,
system and nearby natural waterways (Morton, 1989; Morton, 1992; Morton et al., 1987). During summer, deep lakes were the exception, with negligible amounts of carrion scavenged and organic matter in the canal lakes is more likely to be buried within sediments or recycled by microbes, creating a feedback that would perpetuate hypoxia and reduce pH. Animal organic matter, however, is typically more labile than plant organic matter and so further studies are needed to confirm whether similar patterns hold for less labile detritus.

Substantial variation among locations within habitats was detected for most variables measured indicating that local factors may have a greater influence on environmental conditions at a location than their overall position in the waterway. Habitats were defined by a rigid set of criteria (e.g. distance from river and depth) but the flushing rates of the locations sampled within each habitat type may have differed, which could influence water quality and sediment dynamics. Other factors that may have increased heterogeneity among locations include the proximity of the location to storm water discharges and the history of maintenance dredging. Although no dredging was observed during the study, differences in the history of maintenance dredging among locations may have influenced parameters such as sediment grain size, which did not vary among habitats, despite other studies reporting that finer sediments occur at the ends of canals (Cosser, 1989; Morton, 1989).

For some variables (e.g. C/N and overall FAME profiles) strong trends among habitats were apparent, despite not being statistically significant at α = 0.05. In other cases, such as the composite multivariate dataset of water quality and sediment parameters (and, in particular for % organic matter and pH), statistically significant differences were detected only between the deep lakes and natural habitats, despite trends for canal ends and, sometimes, canal entrances to be intermediate between natural and deep lake habitats. In environmental impact studies, Type II errors (i.e. conclusion of no difference where one exists) are often considered more problematic than Type I errors (i.e. erroneous conclusion of differences where none exist) (Mapstone, 1995). This is because the consequences of not identifying an impact may be more detrimental for the environment than erroneously concluding one exists. Consequently, for those parameters where statistical power may have been limited, more detailed studies, such as comparing ends and entrances of canals with variable flushing rates or proximity to storm water discharges, may be required before concluding that no differences between canal ends and entrances and natural habitats exist.

5. Conclusions

The conversion of coastal wetlands into canal estates will continue. Whilst canal estates cannot function exactly like natural waterways they replace, their design should be maximised to optimise their ecological value. Canal developments must, therefore, avoid the inclusion of deep lakes since they act as traps for organic material, are characterised by hypoxia, and their ecological functioning is greatly compromised. Whilst rates of scavenging were also diminished in the remaining canal habitats, relative to natural habitats, scavenging still occurred and this important ecosystem process was preserved within the canal estates. Moreover, the sources of organic matter in the canal systems were mostly indistinguishable from nearby natural waterways. Designing canal estates to maximise their flushing rates and minimise areas prone to sedimentation and hypoxia is critical to maximise their ecological integrity.

Supplementary data to this article can be found online at https://doi.org/10.1016/j.marpolbul.2018.07.044.

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