

Use of urban marine habitats by foraging wading birds

Richard A. McKinney · Kenneth B. Raposa ·
Thomas E. Kutcher

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Abstract Wading birds (i.e., Ardeidae: herons, egrets, and bitterns) are a guild of waterbirds that forage in coastal habitats which in the US and Europe are often located in close proximity to urban centers. However, the use of urban marine habitats may have consequences for bird populations, as birds can be subject to stress from increased levels of passive and active human disturbance. We examined the effects of human disturbance, available foraging habitat, and prey abundance on wading bird density and species richness at 17 urban coastal sites in Narragansett Bay, Rhode Island USA. The sites represented a gradient of immediately adjacent residential and commercial land use (e.g., 0.0–67.7% urban land use within a 30.5 m buffer of the sites) within an urban matrix (i.e., all sites were located within a suburban center with a population of about 85,000 people). Wading bird density (0.62 ± 0.12 birds ha^{-1}) and species richness (average 4.49 ± 0.37 species across all sites) were not influenced by passive human disturbance as measured by the extent of urban land surrounding a site. However, wading bird density and species richness both decreased significantly as active disturbance (i.e., number of boats moored or docked upstream of the site) increased ($r = -0.56$, $F = 6.85$, $p = 0.019$ and $r = -0.73$, $F = 16.6$, $p = 0.001$, respectively). In addition, both density ($r = 0.72$, $F = 16.2$, $p = 0.001$) and species richness ($r = 0.72$, $F = 16.2$, $p = 0.001$) increased concomitantly with a prey index that combines the density of fish and invertebrates on which the birds feed with the amount of available shallow water foraging habitat at a site. Our results suggest that wading birds i) may not be negatively affected by urban land surrounding estuarine foraging areas in and of itself; and ii) may be

R. A. McKinney (✉)

Atlantic Ecology Division, US Environmental Protection Agency, Office of Research and Development, National Health and Environmental Effects Research Laboratory, 27 Tarzwell Drive, Narragansett, RI 02882, USA
e-mail: mckinney.rick@epa.gov

K. B. Raposa

Narragansett Bay National Estuarine Research Reserve, 55 South Reserve Drive, Prudence Island, RI 02872, USA

T. E. Kutcher

Rhode Island Natural History Survey, Room 101 Coastal Institute Kingston, 1 Greenhouse Rd., Kingston, RI 02881, USA

utilizing urban areas in the absence of high levels of active disturbance to take advantage of potentially enhanced prey resources. In the case where the benefits of foraging at a site outweigh the costs related to human disturbance, urban marine habitats may need to be considered for restoration or protection from further increases in active human disturbance.

Keywords Ardeidae · Estuarine habitats · Human disturbance · Narragansett Bay · Nekton · Urban marine habitats · Wading birds

Introduction

Waterbirds (i.e., wading birds, shorebirds, and waterfowl) often use shallow coastal waters associated with tidal wetlands to feed on fish and other aquatic species. Increasing human populations in coastal areas and subsequent increases in urbanization have placed many of these foraging and stopover sites in close proximity to highly populated areas. The resulting disturbance from human activities may have both local and larger-scale impacts on populations of waterbirds that utilize urban marine habitats for breeding, foraging, or as stopover sites during migration. While a number of studies have examined urbanization effects on songbirds in riparian and freshwater wetlands (e.g., Pearce et al. 2007; Valiela and Martinetto 2007), few have assessed the relative importance of human disturbance in the context of other landscape and biological factors that may influence waterbird use of habitats in urban estuaries. In order to make better-informed decisions regarding the conservation of urban habitats, we need to know more about the effects of human disturbance on these and other species of estuarine wildlife (Bulleri 2006; Burton 2007). This information, along with the impact of these alterations on the degradation of coastal habitats themselves and its effects on economically important fish and shellfish populations (e.g., Ambrose and Meffert 1999; Vanderklift and Jacoby 2003), can be considered when evaluating the costs and benefits of developing or protecting coastal areas. Assessing the impacts of urbanization and human disturbance takes on added urgency in light of recent findings that many species of coastal birds have undergone worldwide population declines (IWSG 2003).

Wading birds (i.e., Ardeidae: herons, egrets, and bitterns) that forage in estuarine wetlands need to acquire enough food to maintain fitness when breeding or, if using the site as a stopover during migration, accumulate sufficient body reserves to reach their breeding or wintering grounds. These birds are under the constant risk of not being able to acquire sufficient food to maintain fitness (which can potentially lead to starvation), while simultaneously facing threats from disease or predation (Goss-Custard et al. 2006). These inherent risks can be enhanced by both natural (e.g., being forced to take flight and disperse upon approach by a raptor or mammalian predator) and either active or passive human disturbance. Active human disturbance includes activities such as walking on the shoreline, using a motorbike on or near the shore, swimming, shellfishing, boating, or bait digging (Antos et al. 2007). The effects of active disturbance on wading birds have been examined; for example several studies have determined appropriate buffer zones for protecting foraging birds from disturbance (Rodgers and Smith 1997; Rodgers and Schwikert 2002; Sabine et al. 2008). Passive human disturbance results from development and urbanization of coastal areas and can include the degradation and loss of habitats, impaired water quality conditions, loss of buffer vegetation (e.g., trees and shrubs), eutrophication resulting from increased nutrient inputs, and increased sedimentation and surface water flow. Each of these passive forms of disturbance can potentially affect foraging wading birds.

A number of studies have examined human disturbance effects on birds using estuarine wetlands in urban landscapes. Earlier studies focused on the ecology of waterbird species in urban habitats (e.g., Figley and VanDruff 1982; Adams et al. 1985). Recent studies have focused more on modeling human disturbance effects, for example by developing individual-based behavioral models to look at the effects of active human disturbance on populations of wintering waterbirds (reviewed in Stillman et al. 2007). For example, Goss-Custard et al. (2006) applied behavioral models to non-breeding wading birds in the Baie de Somme, France in order to establish critical thresholds of active human disturbance (shellfishing and recreational activities). Behavioral models have also been employed to investigate the effect of magnitude and frequency of active disturbance on individual bird survival (West et al. 2002), and to determine minimal (human) approach distances and their effects on bird behavior (de Boer and Longamane 1996). Models of this sort provide important information on the impact of human disturbance on bird fitness, and can be used to generate rigorous estimates of the effects of disturbance on the survival of individuals. However, the interpretation and extrapolation of these data to predict population effects can be problematic (Gill et al. 2001), and data of this sort have only recently been applied to predict impacts of the loss or degradation of specific habitats through human disturbance (West and Caldow 2006).

Studies assessing the effects of passive human disturbance and urbanization on waterbirds have included those that focus on the importance of larger scale landscape structure on habitat use (e.g., Shriver et al 2004), and local studies that examine the effects of landscape setting and surrounding land use. While large-scale studies generally demonstrate altered use of habitats in urban settings by waterbirds on a regional scale, more localized studies have begun to show conflicting patterns of habitat use in the presence of human disturbance. For example, DeLuca et al. (2004) found that marsh bird community integrity in Chesapeake Bay (Maryland, USA) was significantly reduced when the urban development exceeded one-quarter of the land use within 1000 m of a wetland. However, a study of urban impacts on waterbirds foraging at four lakes in central Florida, USA showed that birds preferred areas with developed land cover within 20 m of the lake shore, a phenomenon the authors attributed to the absence of tall emergent vegetation in these areas (Traut and Hostetler 2004). These results illustrate that confounding landscape and biological factors that may potentially influence bird distribution and abundance must be considered when assessing the impacts of human disturbance. Indeed, O'Connell et al. (2007) have called for the need to include other factors such as prey abundance and availability when developing models to assess human disturbance impacts on waterbird distribution. One example of this approach is a study that examined the effects of surrounding landscape composition on waterfowl abundance and richness at a series of urban wetlands while also accounting for habitat characteristics such as prey density and shoreline configuration (McKinney et al. 2006).

In this study, we examine the effects of both active and passive human disturbance on the density and species richness of wading birds at a series of urban estuarine sites in the northeast US. We build on existing landscape-scale studies by also incorporating habitat characteristics such as prey abundance and the extent of available foraging habitat in our analysis. Our specific objectives are to examine patterns of habitat use by wading birds in the presence of human disturbance, and also to identify local-scale habitat characteristics that may be useful in identifying and prioritizing urban estuarine habitats for protection and restoration.

Methods

Study sites

Our study sites were 17 salt marshes in Greenwich Bay, an 11.7 km² sub-estuary of Narragansett Bay, RI (Fig. 1). Narragansett Bay is a temperate coastal plain estuary in the northeast US formed by rising sea level that accompanied glacial retreat at the end of the last ice age. Estuaries in the northeast US in general are characterized by relatively short warm seasons when prey resources are most productive, predictable semi-diurnal tides of 1–2 m, and generally high water clarity (Roman et al. 2000). Salt marshes in northeast US estuaries tend to be smaller and less extensive than the rest of the US east coast, leading to relatively greater effects from surrounding land use patterns (Roman et al. 2000). Greenwich Bay is surrounded by the city of Warwick, RI, a suburban center with a population of about 85,000 people. Its watershed is highly urbanized and is characterized by dense neighborhoods of single-family homes along with accompanying transportation infrastructure and commercial centers. However, despite this urban setting the landuse immediately surrounding Greenwich Bay (i.e., within 100 m of the shoreline) is varied, consisting of a number of forested and vegetated areas, along with marinas, shipyards, and other types of urban land (RI CRMC 2005). The same is true of our study sites: landuse immediately adjacent to the marshes ranges from urban to forested habitat types. The size of our study sites ranged in area from 0.6–10.9 ha

Fig. 1 Location of study sites in Greenwich Bay, RI USA.

1 = Mitchell Court; 2 = Warwick Cove Fringe; 3 = First Point; 4 = Second Point; 5 = Paine Road; 6 = Brush Neck East Inner; 7 = Brush Neck East Fringe; 8 = Warwick Cove Mouth; 9 = Brush Neck West Fringe; 10 = Brush Neck West Inner; 11 = Apponaug Inner; 12 = Apponaug Fringe; 13 = Thatch Cove; 14 = Chepiwanoxet North; 15 = Chepiwanoxet South; 16 = Greenwich Cove; 17 = Warwick Cove Inner



(average 4.9 ha; Table 1). Two of our sites were classified as salt meadow marshes, while the remaining consisted of roughly equal numbers of meadow-fringe, wide fringe, and narrow fringe marshes (Table 1).

Biotic and abiotic parameter selection

At all sites, counts were made of wading birds (composition, density, and richness), prey abundance, amount of available foraging habitat, and passive and active disturbance. The latter parameters were selected after a review of extant literature on wading bird habitat requirements for use in regression models with wading bird density and species richness across the study sites. Prey abundance (i.e., nekton density measured using throw traps) was included since all wading bird species included in this study feed on small fish or crustaceans. Wading birds feed in shallow water and on exposed intertidal areas; accordingly we measured the extent of shallow, subtidal water <15 cm in depth at our sites (Gawlik 2002) and converted this to a percent of the entire area at each site. We combined these two metrics into an index in order to estimate the availability of prey for foraging wading birds at each site. We estimated the extent of passive human disturbance at a site by measuring the proportion of urban land in a 30.5 m buffer around each site, and active disturbance by counting the number of boats either moored or docked upstream of the site (a surrogate for active disturbance by boat traffic).

Table 1 Locations (latitude / longitude in degrees minutes.decimal minutes), marsh, and intertidal areas of 17 salt marsh study sites in Greenwich Bay, R.I. USA. Site morphology was classified according to McKinney and Wigand (2006)

Site #	Site name	Location	Area (ha)	Intertidal area (ha)	Site morphology ^a
1	Mitchell Court	41° 41.71' N / 071° 23.15' W	2.05	0.48	WF
2	Warwick Cove Fringe	41° 41.45' N / 071° 23.38' W	0.60	0.12	NF
3	First Point	41° 41.82' N / 071° 23.53' W	1.49	0.24	WF
4	Second Point	41° 41.72' N / 071° 23.70' W	5.31	1.66	MF
5	Paine Road	41° 41.29' N / 071° 23.42' W	3.08	0.43	MF
6	Brush Neck East Inner	41° 41.94' N / 071° 24.97' W	10.6	3.84	MF
7	Brush Neck East Fringe	41° 41.76' N / 071° 24.40' W	3.51	1.65	WF
8	Warwick Cove Mouth	41° 41.03' N / 071° 23.37' W	5.63	0.80	NF
9	Brush Neck West Fringe	41° 41.33' N / 071° 24.46' W	3.22	0.51	WF
10	Brush Neck West Inner	41° 41.33' N / 071° 24.88' W	3.70	1.46	MF
11	Apponaug Inner	41° 41.82' N / 071° 27.43' W	6.17	2.97	NF
12	Apponaug Fringe	41° 41.56' N / 071° 27.01' W	4.10	0.54	NF
13	Thatch Cove	41° 41.35' N / 071° 27.08' W	10.9	2.39	SM
14	Chepiwanoxet North	41° 40.44' N / 071° 26.65' W	3.65	0.51	SM
15	Chepiwanoxet South	41° 40.37' N / 071° 26.63' W	5.49	0.38	NF
16	Greenwich Cove	41° 38.91' N / 071° 27.22' W	5.94	2.46	MF
17	Warwick Cove Inner	41° 41.92' N / 071° 22.99' W	7.02	1.65	MF

^a SM = salt meadow marsh; MF = meadow fringe marsh; WF = wide fringe marsh; NF = narrow fringe marsh. Detailed descriptions of marsh morphologies can be found in McKinney and Wigand 2006

Wading bird surveys

Wading bird abundance and species richness data were collected during biweekly surveys from May–September in 2006 and 2007. At each site we were positioned so we could observe the entire surface of the site from land, and we scanned the sites using a 32x60 spotting scope or 10x50 binoculars. Sites were surveyed on randomly chosen days, and each survey event (consisting of a full survey of all 17 sites) was conducted within a period of time from 2 h before to 2 h after low tide. We completed a total of 17 full survey events during the course of the study. The number and species of wading birds present at the sites were recorded during each survey. Bird abundances (number per survey) are reported as means \pm 1 standard error (SE). We calculated species richness from the observed number of species at a site using a jackknife procedure (Heltshel and Forrester 1983). Since we did not observe significant between-year differences, all data were pooled to provide an average wading bird abundance across both years of the study. We did, however, see a significant relationship between bird abundance and site area, so we used wading bird density (\pm 1 SE) in our regression models.

Prey availability

To estimate the availability of prey for foraging wading birds we developed an index that combined nekton density with the proportion of available foraging habitat at a site. Nekton was sampled at each of the 17 sites with 1-m² × 1-m high throw traps (3-mm mesh) using protocols based on Raposa and Roman (2001). All sites were sampled in August of 2006 and 2007. The total number of stations sampled at a site ranged from 9 to 16 each year depending on local conditions and nekton densities. Sampling stations were randomly established in shallow water prior to field sampling using geographic information system (GIS) software. Stations close to shore were approached by foot; those in open water were approached by canoe. Sampling commenced at mid-ebb tide when water had completely drained off any emergent marsh vegetation, and continued through the next mid-flood stage. To collect a sample, the trap was quickly thrown into the water and pushed into the sediment to prevent any organisms from escaping under the bottom of the trap. Once secure, the trap was then emptied using a 1-m wide dip net with 3-mm mesh. Dip netting was conducted from all four sides of the trap until three consecutive scoops did not produce any additional nekton. All captured nekton were identified to species (or the lowest possible taxon), counted, measured (total length for fish and shrimp; carapace width for crabs), and returned to the field. Ancillary data collected with each throw trap sample included water depth and percent cover of vegetation.

We estimated the area of shallow, subtidal water <15 cm by making direct depth measurements at mean low tide. First, during August 2007 the extent and area of the entire subtidal zone was determined for each site by photointerpreting and field-checking 2004 and 2006 low-tide orthophotography. Next, a grid containing 50 crosshairs was created and overlaid over a map of the subtidal zone at each site. Finally, a field map of each site was produced that included the grid, the subtidal zone, and the original orthophotographic image. Using the field map and shoreline landmarks, water depth was directly measured in the field with meter sticks around the time of low tide at each crosshair. Since all measurements were not taken at precisely mean low tide, they had to be adjusted for changes in tide height that occurred during depth measurements. This was facilitated by measuring changes in tide height at regular intervals from an additional tide meter at the same time that depth measurements were taken. Low tide was determined from the lowest

water reading on the additional tide meter, and depth measurements were then adjusted accordingly (e.g., if a depth measurement was taken 10 minutes before low tide, and it was found from the tide meter that overall depth decreased by 3 cm during those 10 min, then 3 cm was subtracted from the original depth measurement for an approximate low tide depth). The area of open water <15 cm in depth was then quantified for each site as:

$$A_c = P_c + A_t$$

where

A_c estimated area of open water <15 cm

P_c proportion of samples within the area of open water <15 cm, and

A_t total area of subtidal zone

A prey index was then calculated by first ranking the sites from lowest to highest in terms of the area of open water <15 cm, with the lowest ranking corresponding to the lowest area of available foraging habitat. Next we ranked the sites from lowest to highest in terms of nekton density, again with the lowest ranking corresponding to the lowest nekton or potential prey density. We then calculated the prey index by summing the two ranks. Sites with low prey index values had low amounts of nekton density and available foraging habitat.

Human disturbance characteristics

The percent cover of urban land within a 30.5 m buffer surrounding the sites was used to estimate passive human disturbance. We chose a 30.5 m buffer in order to assess the effects of land use and land cover in close proximity to the sites, and because it corresponds to a 100 ft state protected zone that extends landward the shoreline. First, the marsh-upland interface was identified at each site using GIS. This interface was then buffered to 30.5 m to create polygon donuts around each site. The buffer polygons were then laid over 2006 1:12,000 georectified, leaf-on aerial photography and 2003 1:5,000 leaf-off digital orthophotography. Using a minimum mapping unit of 0.1 ha, land cover classes were photointerpreted and classified based on the National Estuarine Research Reserve Classification Scheme (Kutcher et al. 2005) to characterize habitat and land use structure. From the resulting dataset, the amount of urban land in this 30.5 m buffer zone (i.e., residential, commercial/industrial and associated land use types; Kutcher et al. 2005), was derived for each site. To standardize this across all sites, the absolute area of urban land was converted to a percentage based on the total buffer area around each site.

To estimate active human disturbance (in the form of boat traffic), we counted the number of boats either moored or docked upstream of the site during June and July, 2007. We used a small boat to conduct field surveys of the open water areas adjacent to and upstream from each site and noted the total number of docked or moored boats, as well as unoccupied dock slips and moorings. In the heavily populated areas in which our sites are located, dock and mooring space is at a premium and we felt that the few unoccupied slips and moorings we came across were most likely occupied by boats that were at sea during our surveys.

Data analysis

The effects of surrounding urban land, the number of boats, and the prey index on wading birds were examined using simple linear regressions with wading bird density and species richness as the dependent variables. The number of boats either moored or docked upstream

of the site was log transformed to minimize the influence of extreme values of this metric that were obtained at several of the sites. We then used multiple linear regression, again with wading bird density and species richness as the dependent variables, to construct models to explain habitat use based on our measured habitat characteristics. An information-theoretic approach was used to examine the relative importance of the characteristics in explaining overall habitat use by wading birds. Best habitat characteristic models were determined using small sample Akaike Information Criteria (AIC_c), which accounts for biases that might arise from relatively small sample size (Burnham and Anderson 2002). Models were compared by computing AIC_c differences ($\Delta_i = AIC_{ci} - AIC_{cmin}$; Burnham and Anderson 2002). Candidate models with Δ_i between 0 – 4 were considered candidate models, whereas candidate models with $\Delta_i > 4$ were not well supported. The w_i are considered the weight of evidence in favor of model i and provide an effective way to scale and interpret the Δ_i (Burnham and Anderson 2002). We used model averaging to arrive at a single regression model to explain wading bird density, as well as a single regression model to explain species richness (Burnham and Anderson 2002). Statistical analyses were performed with SAS for Windows ver. 6.12 (SAS Institute, Inc., Cary, NC, USA).

Results

The wading bird community at our sites was comprised of 6 species and was dominated by snowy egrets *Egretta thula* and great egrets *Aldea alba* (average 1.55 ± 0.51 and 0.95 ± 0.35 birds per site, and 0.28 ± 0.06 and 0.17 ± 0.06 birds ha^{-1} , respectively). The average abundance of wading birds across all sites was 3.35 ± 0.89 birds per site, while the average density and richness of wading birds was 0.62 ± 0.12 birds ha^{-1} and 4.49 ± 0.37 species, respectively (Table 2). Average per site abundances and densities were relatively low, but we would occasionally see higher numbers of birds at the sites, particularly during the late summer and early fall. The maximum number of wading birds observed during an individual survey was 49 great egrets at Apponaug Inner (site 11, August 2006).

Open water <15 cm comprised between 10 and 34% of the area of roughly half of our sites; the remaining sites had <10% open water <15 cm (Table 3). Apponaug Inner (site 11) and Greenwich Cove (site 16) had the highest proportion of open water <15 cm, and also the greatest per-site abundances of wading birds (Tables 2 and 3). Nekton density varied over an order of magnitude, and the number of boats either moored or docked upstream of a site over three orders of magnitude, across the sites (Table 3). The percent urban land in a 30.5 m buffer surrounding the sites was somewhat less variable, and ranged from 0–68% (Table 3).

We found that wading bird density at our sites significantly increased with increasing prey index ($r=0.51$, $F=5.31$, $p=0.04$), and decreased significantly with active disturbance (i.e., increasing number of boats; $r=-0.56$, $F=6.85$, $p=0.019$; Fig. 2). Wading bird species richness also increased with increasing prey index ($r=0.65$, $F=11.1$, $p=0.005$), and decreased with increasing number of boats ($r=-0.73$, $F=16.6$, $p=0.001$; Fig 3a, b). The model averaged regression model for wading bird density showed that density increased with increasing prey index, but decreased with increasing number of boats and surrounding urban land (Table 4). On the other hand, the model averaged regression model for wading bird species richness showed increasing species richness with increasing passive disturbance (i.e., surrounding urban land), and also with increasing prey index, but decreasing species richness with increasing number of boats (Table 4).

Table 2 Mean abundance (mean number of birds per survey \pm SE) for individual wading bird species and summed for all wading birds at a site, average density (birds / survey / ha \pm SE) and species richness of wading birds during biweekly surveys ($n=17$) at 17 salt marsh study sites in Greenwich Bay from 2006 – 2007

Site #	Abundance				Wading bird density				Wading bird richness
	Snowy ^a egret	Great egret	Black-crowned night heron	Great blue heron	Green heron	Little blue heron	Ardeid abundance	Wading bird density	
1	0.42±0.17	0.18±0.10	0.06±0.04	0.12±0.08	0.25±0.09	0.12±0.08	1.15±0.57	0.56±0.28	6.94
2	0.11±0.08	0.00±0.00	0.00±0.00	0.00±0.00	0.00±0.00	0.00±0.00	0.11±0.08	0.19±0.13	1.94
3	0.18±0.04	0.00±0.00	0.00±0.00	0.00±0.00	0.25±0.09	0.00±0.00	0.31±0.13	0.21±0.09	2.94
4	0.93±0.27	0.67±0.19	1.19±0.46	0.19±0.09	0.53±0.16	0.00±0.00	3.50±1.17	0.66±0.22	5.00
5	0.85±0.27	0.18±0.10	0.00±0.00	0.00±0.00	0.00±0.00	0.00±0.00	1.03±0.36	0.33±0.12	2.00
6	1.33±0.50	2.55±0.93	0.17±0.10	1.17±0.32	0.30±0.14	0.00±0.00	5.53±2.00	0.52±0.19	5.00
7	1.39±0.51	2.01±0.80	0.17±0.10	0.60±0.26	0.00±0.00	0.00±0.00	4.18±1.65	1.19±0.47	4.00
8	0.24±0.14	0.00±0.00	0.00±0.00	0.00±0.00	0.06±0.04	0.00±0.00	0.30±0.18	0.05±0.03	2.94
9	0.25±0.09	0.63±0.18	0.00±0.00	0.31±0.11	0.06±0.04	0.00±0.00	1.25±0.43	0.39±0.13	4.94
10	1.25±0.52	0.95±0.50	0.68±0.38	0.79±0.40	0.06±0.04	0.00±0.00	3.74±1.84	1.01±0.50	5.94
11	5.86±2.03	5.79±2.50	0.94±0.35	0.56±0.19	0.13±0.06	0.00±0.00	13.3±5.13	2.15±0.83	5.00
12	1.31±0.41	0.31±0.13	0.00±0.00	0.06±0.04	0.00±0.00	0.00±0.00	1.67±0.58	0.41±0.14	3.94
13	8.01±2.43	0.65±0.27	0.19±0.06	1.15±0.46	0.24±0.11	0.11±0.05	10.4±3.39	0.95±0.31	6.00
14	1.04±0.33	0.00±0.00	0.06±0.04	0.18±0.10	0.13±0.06	0.00±0.00	1.41±0.52	0.39±0.14	4.94
15	0.61±0.28	0.00±0.00	0.06±0.04	0.00±0.00	0.00±0.00	0.00±0.00	0.67±0.32	0.12±0.06	2.94
16	1.65±0.57	1.10±0.48	0.22±0.16	2.32±0.67	0.06±0.04	0.00±0.00	5.36±1.93	0.90±0.32	6.88
17	1.06±0.22	1.08±0.39	0.40±0.19	0.49±0.18	0.12±0.08	0.00±0.00	3.15±1.06	0.45±0.15	5.00

^a Snowy egret = *Egretta thula*; great egret = *Aldea alba*; black-crowned night heron = *Nycticorax nycticorax*; great blue heron = *Aldea herodias*; green heron = *Butorides virescens*; little blue heron = *Egretta caerulea*

Table 3 Habitat characteristics used in regression models for the 17 salt marsh study sites in Greenwich Bay. All variables except nekton density were calculated from single measurements; nekton density is reported \pm SE

Site #	OPWAT ¹	NEKTON	BOATS	URBAN	PREY
1	17.0	374 \pm 68.0	1.08	47.0	22
2	8.33	381 \pm 43.1	2.11	65.3	16
3	4.69	761 \pm 50.8	1.79	49.3	18
4	8.66	263 \pm 14.2	1.97	53.6	13
5	2.27	120 \pm 7.9	2.44	56.8	5
6	23.1	573 \pm 129	0.00	23.4	28
7	9.39	522 \pm 34.1	0.85	44.7	21
8	5.86	69.9 \pm 11.9	3.45	26.4	8
9	10.3	320 \pm 14.6	0.78	0.00	18
10	14.9	594 \pm 93.5	0.85	8.85	27
11	34.2	264 \pm 4.19	0.60	67.7	23
12	4.89	762 \pm 125	2.58	36.2	18
13	12.1	433 \pm 90.6	0.00	50.6	22
14	7.95	50.1 \pm 1.61	0.95	26.2	7
15	7.47	117 \pm 12.3	1.80	17.6	8
16	31.2	444 \pm 32.5	0.00	7.86	27
17	10.3	893 \pm 172	1.20	38.0	27

¹ OPWAT = percent of the site area comprised of shallow open water less than 15 cm; NEKTON = nekton density (organisms m^{-2}); LOG(BOATS) = log number of boats moored or docked upstream from the site; URBAN = percent of a 30.5 m buffer surrounding the site comprised of urban land cover; PREY = index of prey availability, a metric that incorporates nekton density and available foraging habitat

Discussion

Our results suggest that: i) wading bird density and species richness is negatively impacted by active human disturbance (i.e., increased boat traffic), ii) in the absence of active disturbance, the extent of urban land surrounding a site may have a positive effect on wading bird density, and iii) the amount of available prey and foraging habitat at the sites (represented by the prey index) had a significant positive effect on both density and species richness at the sites. One interpretation of these results is that active human disturbance may limit the use of highly disturbed sites to species or perhaps even individuals that can tolerate higher levels of disturbance. For these birds, the choice of foraging sites may not be negatively influenced by passive human disturbance, but rather by the inherent characteristics of the habitat (e.g., the extent of available foraging habitat and prey abundance). They may be taking advantage of highly urbanized sites where prey abundance and availability are high.

Active human disturbance

Our finding that active human disturbance, in our case from potential boat traffic, negatively impacts wading bird use of foraging sites is consistent with other waterbird studies that have demonstrated disturbance effects from human activities (e.g., shellfishing) in close proximity to foraging birds (Urfi et al. 1996; Gill et al. 2001; Blumstein et al.

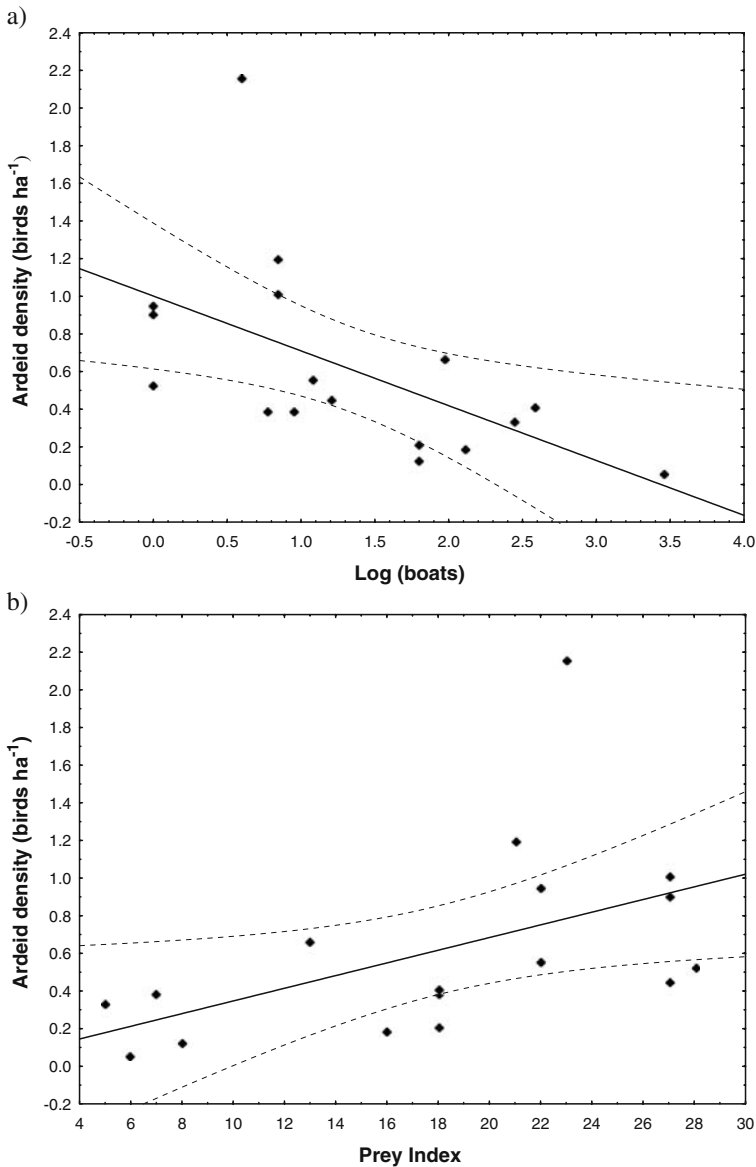


Fig. 2 Relationship between wading bird density (birds ha⁻¹) and a) log number of boats moored or docked upstream from the site, and b) prey index, a metric that incorporates nekton density and available foraging habitat, at 17 salt marsh study sites in Greenwich Bay. Dotted lines represent 95% confidence intervals

2003). At our study sites boat traffic is predominantly recreational, consisting of pleasure craft (e.g., outboard-powered boats) generally less than 8 m in length. Several of our sites with high numbers of upstream boats (Warwick Cove Fringe, Paine Road, Warwick Cove Mouth, Apponaug Cove) were bordered by relatively narrow waterways (10–50 m width) that tended to concentrate boat traffic near marshes where birds were foraging. These distances were often less than the average flush distances reported by Rodgers and

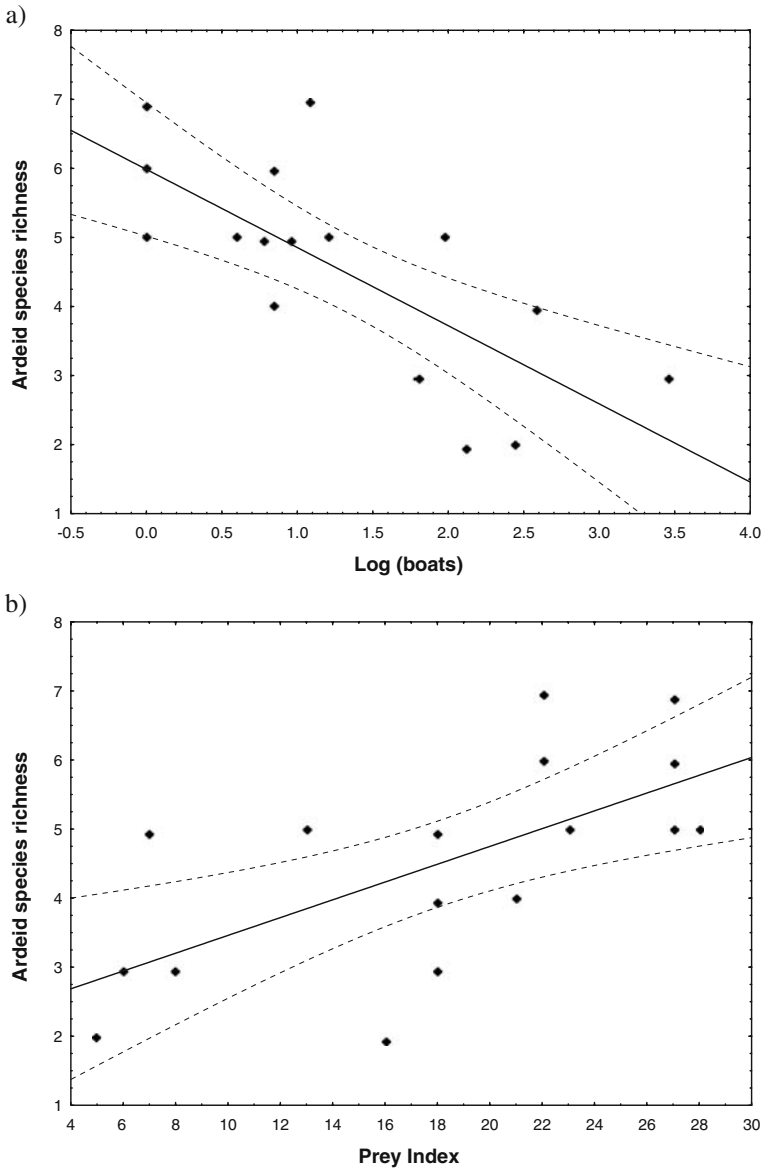


Fig. 3 Relationship between wading bird species richness and a) log number of boats moored or docked upstream from the site, and b) prey index, a metric that incorporates nekton density and available foraging habitat, at 17 salt marsh study sites in Greenwich Bay. Dotted lines represent 95% confidence intervals

Schwikert (2002) for some waterbird species, and in most cases less than the recommended 100 m buffer from boat traffic based on a study of waterbirds in Florida, USA (Rodgers and Smith 1997). Since shellfishing is restricted by state statute at several of our sites, disturbance from boating may be the primary human disturbance threat to wading birds. Similar restrictions are found in many urban estuaries, leading to similar disturbance scenarios (Chmura and Ross 1978).

Table 4 Akaike Information Criteria candidate models ($\Delta_i < 4.00$) for three habitat characteristics versus wading bird species richness and density at the 17 salt marsh study sites in Greenwich Bay. Habitat characteristics are described in Table 3. Coefficients within the model averaged regression models are listed in order of their relative importance

Models ^a	Wading bird species richness						Wading bird density					
	Maximized			Models			Maximized			Models		
	\log_e^b	AIC _c	Δ_i	w_i			\log_e^b	AIC _c	Δ_i	w_i		
SR = 5.99 – 1.13(boats)	-0.496	8.84	0.00	0.99	D = 0.70 + 0.01(urban) – 0.34(boats)		17.18	-23.02	0.00	0.61		
SR = 5.99 – 1.13(boats) + 0.06(preys)	0.209	10.9	2.01	0.35	D = 1.00 – 0.29(boats)		15.04	-22.24	0.77	0.41		
SR = 6.37 – 0.01(urban) – 1.07(boats)	0.005	11.3	2.49	0.29	D = 0.01 + 0.34(preys)		14.42	-21.00	2.02	0.22		
SR = 2.17 + 0.13(preys)	-2.37	12.1	3.28	0.19	D = -0.33 + 0.01(urban) + 0.04(preys)		15.66	-19.98	3.03	0.13		
Model averaged: SR = 5.65 ± 4.92 – 1.12 ± 0.03(boats) – 0.01 ± 0.00(urban) + 0.08 ± 0.09(preys)					D = 0.37 + 0.01(urban) – 0.27(boats) + 0.13(preys)		17.49	-19.53	3.49	0.11		
					D = 0.62 – 0.21(boats) + 0.02(preys)		15.37	-19.41	3.6	0.10		
					Model averaged: D = 0.57 ± 0.34 – 0.31 ± 0.24(boats) + 0.01 ± 0.00(urban) + 0.17 ± 0.37(preys)							

Passive human disturbance

Our models also indicate that the amount of urban land directly adjacent to a site has a positive relationship with wading bird density (i.e., increasing wading bird density with increasing extent of urban land). This finding is contrary to that of larger-scale human disturbance studies (e.g., Shriver et al. 2004; DeLuca et al. 2004) but similar to studies of bird use at a series of urbanized lake shorelines (Traut and Hostetler 2004). Wading birds at our sites may be to some extent tolerating increased human presence at our urban sites in order to take advantage of abundant resources, foraging habitat, and in some cases lessened levels of active human disturbance (Donaldson et al. 2007). Wading birds have been shown to feed opportunistically on abundant resources, although not necessarily in the presence of human disturbance (Kushlan 1976; Davis 1993). However, several studies have demonstrated tolerance of human disturbance by birds in areas of high resource availability. For example, Yasue (2006) found that human disturbance was not influencing habitat selection in shorebirds using coastal habitats in British Columbia, Canada, and in fact the time taken for the birds to resume feeding after a disturbance was less in areas of high prey availability. Evidence of habituation to human disturbance has also been noted in nesting shorebirds and piscivorous waterbirds (Newbrey et al. 2005; Baudains and Lloyd 2007). In our study, wading bird density was greater at sites with more shallow open water (i.e., open water <15 cm) and higher prey density (i.e., high prey availability), suggesting that the birds may be to some extent tolerating the effects of surrounding urbanization to utilize enhanced prey resources. While increased bird abundance has been noted in previous studies (Cam et al. 2000; Chace and Walsh 2006), our study provides a potential link to increases in resource availability.

Birds may be prevented from attaining enough food to maintain fitness or accumulate energy reserves through natural processes such as competition, predation, or predation-related disturbance. Recent studies have also shown that human disturbance can exacerbate these threats and, in areas where there is sufficient habitat available, may determine observed patterns of bird distribution (Burton et al. 1996, 2002; Gill et al. 2001). Because the effects of competition and predation at our sites was probably low (McKinney personal observation), our study area was well suited for studying human disturbance effects on wading birds without the potentially confounding effects of other types of natural disturbance. For example, wading bird densities were generally <1 ha⁻¹ at our study sites (located near the northern limit of their range); these values are considerably lower than that of more southern US marshes in the central part of their range where densities of up to 13.3 birds ha⁻¹ have been reported for some species (Traut and Hostetler 2004). As a result, intra- and inter-specific competition and density-dependent phenomena may not be as prominent at our sites. Also, the urban setting of our study area may have supported lower densities of both mammalian and avian predators, thus lowering the predation risk for birds foraging at our sites (Johnsen and vanDruff 1987; Blair 1996; Baudains and Lloyd 2007). However, there is also the possibility that the absence of top-level predators in urban landscapes may lead to higher densities of mammalian predators such as red fox *Vulpes vulpes* coyote *Canis latrans* that are known to frequent coastal marshes.

Site-specific effects

The highest density of wading birds during our study was found at the Apponaug Inner site (site # 11), a moderately-sized fringing marsh that was surrounded by the highest proportion of urban land (Table 3). However, this site also had the highest proportion of

shallow open water, along with a moderately high nekton density, which led to a relatively high prey index value. At low tide when utilization of the site by wading birds was greatest (Raposa, personal communication), nekton was concentrated into a series of readily-accessible shallow pools of water surrounded by intertidal flats. The resulting high concentrations of nekton in water of optimal foraging depth were utilized by relatively large numbers of foraging wading birds, despite the high density of surrounding urban land. Because of the very shallow water at low tide, there is little to no boating activity at this site. Also, at this and other sites surrounded by high proportions of urban land, water quality issues have led to the prohibition of shellfishing, which further restricts active human disturbance (RI DEM 2007). However, the proximity of the site to an urban center results in other forms of disturbance, such as from people using the marsh for recreation (e.g., dog walking) and traffic along roads near the site, that must be tolerated by the foraging birds. A similar scenario was seen at Brush Neck East Fringe (site # 7) and Thatch Cove (site # 13), which showed high wading bird densities in spite of a high proportions of surrounding urban land. Although neither site had particularly high proportions of shallow open water habitat, both had among the highest prey index values.

Prey index

High nekton densities at sites with high proportions of surrounding urban land may be a subsequent effect of cultural eutrophication of the sites, or the introduction of nutrients (principally reactive forms of nitrogen such as nitrate or ammonia) via non-point source human wastewater inputs to the sites. Increased eutrophication of salt marsh sites in Portugal led to higher primary productivity of several species including emergent *Spartina* species (Simas and Ferreira 2007), and differences in productivity were seen between vegetated and un-vegetated urban salt marshes in the Gulf of Mexico (Rozas and Zimmerman 2000). Increased primary productivity at our sites could lead to increased levels of detritus, which could in turn lead to higher densities of detritivores including species of salt marsh fish and crustaceans that are prey for wading birds. In support of this hypothesis, increased resource availability along a gradient of human residential land use was attributed to eutrophication effects, and hypothesized as a driver of greater utilization of disturbed sites by juvenile winter flounder *Pseudopleuronectes americanus* (Meng et al. 2005).

Management implications

Our study of small tidal wetlands in a highly urbanized landscape setting showed that some of these sites support relatively large numbers and high species richness of foraging wading birds suggesting that, at a local level, urban sites may be providing access to enhanced prey resources that may offset the energetic costs of foraging in a human-dominated environment. Higher productivity in some urban habitats resulting from anthropogenic influences (such as increased nutrient input) may override classic diversity-productivity relationships for species such as birds that are able to take advantage of increased resources (McKinney 2002; Shochat et al. 2006). Wading birds using urban coastal habitats may also encounter lower numbers of natural predators and decreased competition as a result of the altered landscape, making these habitats more appealing to foraging birds. In light of these findings, urban marine sites that provide foraging habitat for resident and migratory waterbirds may need to be identified and considered for restoration or protection from further active human disturbance (McKinney 2009). However, it is important to consider

that despite their apparent suitability for foraging wading birds, urban coastal habitats that are structurally degraded (i.e., those that exhibit detrimental changes in vegetation patterns, hydrology, or area) may be of little value to species that traditionally used these areas for breeding (e.g., seaside sparrow *Ammodramus maritimus*, willet *Catoptrophorus semi-palmatus*). In addition, utilizing these resources may come at a cost: the fitness of birds that utilize urban foraging habitats may be impacted both by the increased levels of human disturbance and exposure to higher levels of human-derived pollution, including persistent organic contaminants and trace metals (Gill et al. 1996; Rattner et al. 2000; Stillman et al. 2001; Kight and Swaddle 2007). Further studies will be needed to compare the energetic costs of foraging at these sites versus the energetic gains from abundant prey in order to ascertain whether urban marine habitats are providing needed resources, or are simply functioning as sink habitats for birds that have been displaced from optimal foraging sites.

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