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Coastal urbanization and the integrity of estuarine waterbird communities: Threshold responses and the importance of scale

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ABSTRACT

Estuarine ecosystems are becoming increasingly altered by the concentration of human populations near the coastline, however a robust indicator of this change is lacking. We developed an index of waterbird community integrity (IWCI) and tested its sensitivity to anthropogenic activities within 28 watersheds and associated subestuaries of Chesapeake Bay, USA. The IWCI was used as a tool to gain insight into how human land use affects estuarine ecosystem integrity. Based on Akaike's information criteria (AIC), a single variable model including percent developed land in estuarine watersheds was thirteen (2002) and twenty-six (2003) times more likely than models including percent agriculture and forest cover to fit the IWCI data. Consequently, we examined how suburban, urban, and total development shaped IWCI scores at three spatial scales: (1) watershed; (2) inverse-distance-weighted (IDW) watershed (land cover near the coastline weighted proportionally greater than that farther away); (3) local (land cover within 500 m of the coastline). Suburban, urban, and total development were all significant predictors of IWCI scores. Relationships were stronger at the IDW and local scales than at the whole watershed scale. Nonparametric changepoint analysis revealed a >80% probability of a threshold in IWCI scores when as little as 3.7% (2002) and 3.5% (2003) of the IDW land cover within the watershed was urban. Our results indicate that, of the landscape stressors we examined, development near estuarine coastlines is the primary stressor to estuarine waterbird community integrity, and that estuarine ecosystem integrity may be impaired by even extremely low levels of coastal urbanization.

1. Introduction

Estuaries are one of the most biologically productive and threatened ecosystems in the world (Kennish, 2002). Although estuarine structure and function can be compromised by a variety of factors, degradation can often be traced to stressors arising from human development of coastal landscapes. For example, coastal development can alter benthic

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(Hale et al., 2004; King et al., 2005a), fish (Sanger et al., 2004) and marsh bird (DeLuca et al., 2004) communities. Furthermore, eutrophication of coastal waters, frequently the result of anthropogenic nutrient influxes (Nixon, 1995), can disturb estuarine food web structure, potentially compromising both ecological and economic integrity (Baird et al., 2004; Keats et al., 2004). With 75% of the world's population expected to live within 60 km of the coast by 2020 (Roberts and Hawkins, 1999), refining our understanding of how modernizing coastal landscapes shape estuarine condition will be crucial for planning long term, sustainable land use strategies.

Anthropogenic disturbances span local and regional political boundaries and pose difficult conservation dilemmas. Because planning goals, policies, and laws often differ across such borders, cooperation among stakeholders can be the primary obstacle to implementing effective management initiatives (Brody et al., 2004). Two approaches can help ameliorate this situation. First, information about the critical scale at which human activities disrupt ecosystems should be an integral part of land use planning because the scale of disturbance will also determine the scale at which action should be taken (e.g. community, county, state, etc.) (Lovell et al., 2002; Jackson et al., 2004). Second, identifying quantitative thresholds in the response of biota to disturbances can provide conservation planners with simple, numerical targets that can be easily communicated to nonscientists (With and Crist, 1995; DeLuca et al., 2004; Guénette and Villard, 2005). Thus, understanding the scale at which disturbances are influencing ecosystems is particularly important to identifying the numerous political and management agencies that could potentially be involved with conservation actions. Such methods can facilitate the process of conveying sound scientific findings into practical conservation practices.

Bird communities have proven to be effective indicators of ecological condition in research where land cover modifications were hypothesized to affect ecosystem integrity (O'Connell et al., 2000; Bryce et al., 2002; Hausner et al., 2003; Glennon and Porter, 2005). DeLuca et al. (2004) previously demonstrated that even low levels of development near coastal marshes resulted in a threshold response beyond which marsh ecosystem integrity significantly declined. The present study expands on the methods developed for calculating indices of community integrity in DeLuca et al. (2004) and applies them to an aquatic ecosystem. This application enabled us to pursue several novel inquiries from those presented in DeLuca et al. (2004). First, the waterbird community is more directly dependent upon estuarine condition than marsh or near-shore terrestrial bird communities. For example, the presence of breeding terrestrial birds is typically tied to territory locations that may be dependent upon factors other than the current integrity of the site. Such factors include previous breeding success, patch size, social systems, and vegetation structure. Conversely, due to the lack of territoriality of most breeding waterbirds, their presence is more likely related to the current state of food resources at that location. Thus, an index based on the waterbird community is likely to reflect conditions at lower trophic levels and abiotic conditions at survey locations (Takekawa et al., 2006). Second, because waterbirds are part of the aquatic food web of estuaries, this community offers a reliable method to assess the importance of scale within a watershed framework. Disturbances within the watershed have the potential to alter aquatic systems via direct hydrological connectivity. Finally, relatively recent innovations in GIS modeling (i.e. inverse-distance weighting) enabled us to conduct a detailed analysis accounting for local and watershed scales simultaneously, resulting in a refined resolution of the scale at which human disturbance affects waterbirds.

We developed an index of waterbird community integrity (IWCI) and used it as a tool to evaluate whether coastal anthropogenic landscape disturbances alter estuarine ecosystems. We first determined which land cover types were significant stressors to the IWCI and then evaluated how these land cover types affected the IWCI at three geographic scales: watershed, inverse-distance-weighted (IDW) watershed (emphasizing land cover near the shoreline to account for within-watershed spatial arrangement), and local (within 500 m of the subestuary). Finally, we tested the hypothesis that nonlinear relationships between land cover and IWCI scores represented ecological thresholds.

2. Study site and methods

2.1. Study area

Field work was conducted in subestuaries of Chesapeake Bay, USA (39° 23′ N; 36° 48′ N–76° 45′ W; 75° 44′ W). The periphery of Chesapeake Bay is dominated by subestuaries which are small, shallow estuarine embayments, many of which are fed by third through fifth order streams. Chesapeake Bay is one of the largest and most productive estuaries in the world. It is characterized by 7400 km of tidal shoreline, shallow waters, approximately 101,000 ha of estuarine wetlands, and diverse floral and faunal communities (Tiner and Burke, 1995; Lippson and Lippson, 1997). Land cover within the Chesapeake Bay watershed is varied, but spatially aggregated. Industrial and high-density urban development are concentrated on the western shore of the bay near Baltimore, Maryland and Portsmouth, Virginia. Forest cover is highest in the vicinity of the Patuxent River on the western shore, but declines as it becomes increasingly interspersed with urban/suburban development to the north and low-density agriculture to the south. Commercial agriculture dominates the eastern shore of the bay and consists of row crops, poultry farms, and pasture.

2.2. Site selection and classification

We selected 28 subestuaries (Fig. 1) based on land cover characteristics, geomorphology, and hydrology of surrounding watersheds. Watershed boundaries were delineated using techniques described by King et al. (2005a). We used National Land Cover Data (USEPA, 2000) to select watersheds that best represented land cover types present in the study area, while minimizing confounding effects of spatial distribution unrelated to land cover (King et al., 2005b). We required that watersheds contain a third through fifth order stream that drained to a well-defined subestuary. These conditions were necessary because we wished to maximize hydrological connectivity between watershed land cover and biological processes in

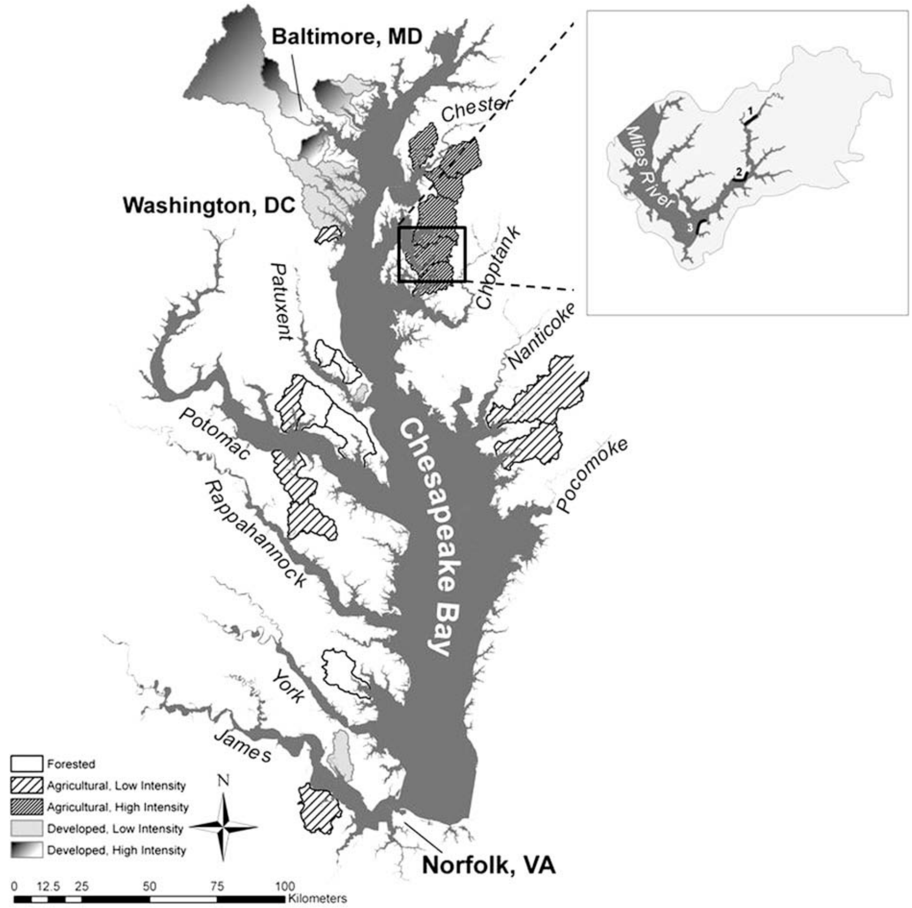

Fig. 1 – Map of study site, Chesapeake Bay, USA showing the distribution of 28 watersheds used in our study. The inset is an example of a study subestuary, depicting the distribution of waterbird sampling transects.

subestuaries. Furthermore, watersheds were selected so that, among the three major land cover types (forest, development, and agriculture), a gradient of values existed (see Table 1 in Deluca et al., 2004). We considered the disturbance gradient to be within land cover categories and did not consider the gradient to exist among categories. That is, a watershed with 50% agriculture and 50% forest was equally disturbed as a watershed with 50% development and 50% forest. More information regarding study watersheds and their subestuaries can be found in DeLuca et al. (2004) and King et al. (2004, 2005a). We sampled 17 subestuaries in 2002, 20 in 2003, and a subset of 9 in both years.

We considered five land cover categories when testing impacts on the waterbird community: (1) forest; (2) agriculture (cropland and pastures); (3) suburban/rural (low-density residential development); (4) urban (high-density residential, commercial, and industrial development); and (5) total development (the sum of urban and suburban/rural).

We measured land cover at three different geographic scales. First, we determined watershed land cover as the percentage of land cover category within the total catchment area of a subestuary's watershed. Second, we calculated the inverse-distance-weighted (IDW) percentage of land cover within the watershed. The IDW method allowed us to con-

Table 1 – Bird species attributes and criteria used to develop index of waterbird community integrity (IWCI) scores

Species attributes	Score Generalist→Specialist				
	1	2	2.5	3	4
Foraging niche breadth	Generalist	Aquatic generalist	–	Moderate specialist	Specialist
Nesting sensitivity	Tolerant	–	Moderately tolerant	–	Sensitive
Migratory status	Resident	Partial	–	Temperate	Neotropical
Breeding range	Global	–	North America	–	East coast of North America
State listing	Not listed	Special concern	–	Threatened	Endangered
Native status	Non-native*	Native	–	–	–

* An attribute score of 0 is given for non-native species.

sider land cover within the entire watershed while emphasizing land cover closer to the subestuary shoreline, thus accounting for the spatial arrangement and proximity of land cover to the subestuary (Comeleo et al., 1996; Soranno et al., 1996; King et al., 2004, 2005b). IDW percentages within the five land cover classes were calculated by measuring the linear distance of each 30×30 m cell to the shoreline. Pixels were aggregated (by land cover class) into distance classes, weighted by the squared inverse of their distance to the shoreline, and summed for a distance-weighted pixel count for the entire watershed. The process was repeated for all pixels in the watershed (irrespective of land cover class). The sum of distance-weighted land cover class pixels was divided by the sum of distance-weighted total land in the watershed to yield distance-weighted percentage land cover (King et al., 2004, 2005b). Third, we measured local land cover by determining the percentage of each land cover category within 500 m of the subestuary shoreline.

2.3. *Waterbird community sampling*

For the purposes of this study, we defined waterbirds as all species that forage exclusively or opportunistically on aquatic estuarine organisms (i.e. gulls, terns, waders, raptors, kingfishers, and waterfowl). We sampled the waterbird community using three 1-km transects in each subestuary, resulting in 84 total transects. Transects were positioned in the upper, middle, and lower thirds of subestuaries (Fig. 1). Transects were located 100 m from the shoreline, and the distance among adjacent transects within a subestuary was >500 m.

Waterbirds were surveyed from a boat traveling at three knots along transects. We used the double observer approach (Nichols et al., 2000) to survey waterbirds that occurred within 100 m of the transect. All individuals on the shore, in the air, or perched within the survey area (20 ha/transect) were counted. To minimize the effect of tidal stage on waterbird sampling, surveys were not conducted during extreme high or low tides. Subestuary morphology and tidal influence was such that exposed mudflats were typically <1 m wide and varied little. Surveys were conducted in 2002 and 2003 from 15 May to 15 August between 0600 h and 1300 h. Surveys were done three times a year with a minimum of 14 days between counts. Abundance values for each species were entered into program DOBSERV to calculate estimates of abundance corrected for differences in observer detection probabilities and species specific detection probabilities that are less than one (Nichols et al., 2000). Corrected abundance estimates were averaged between the three surveys at each transect.

2.4. *Index of waterbird community integrity*

The initial step in IWCI development was to calculate a score for each species detected during the study (S_{IWCI}):

$$S_{IWCI} = \sum L_S \quad (1)$$

where L_S was the cumulative score of six species attributes (DeLuca et al., 2004) on a scale of 1 (generalist) to 4 (specialist). The six species attributes that were scored are: (1) foraging niche breadth; (2) nesting sensitivity; (3) migratory status; (4) breeding range; (5) state conservation listing; and (6) native or non-native status (Table 1). We included foraging niche breadth, nesting sensitivity, and migratory status because the presence of birds with specialized foraging and nesting strategies and the occurrence of long distance migrants have been shown to be sensitive to human disturbance (O'Connell et al., 2000; Bryce et al., 2002; DeLuca et al., 2004). We selected breeding range because it is likely that birds with limited ranges are adapted to conditions specific to that geographic region. Thus, by measuring this attribute, we capture a species' capacity to respond to regional disturbances. We scored species' state conservation listing to enable the index to reflect local conservation concerns. Finally, a species was defined as either native or non-native because exotic species are often most successful when exploiting disturbed systems (Duncan et al., 2003). See Deluca et al. (2004) for further explanation and rationale for species attribute consideration.

The scores of species detected during this study ranged from 5 to 21. Species with scores <10 were considered disturbance tolerant and species with scores of >10 were considered sensitive to disturbance. The split between tolerant and sensitive species was based on natural history information (Poole and Gill, 1999; Appendix A). Species' abundance estimates were used to develop an abundance score (A_I) for each species along each transect. A higher representation of disturbance tolerant species in a bird community is typically indicative of a disturbed system, whereas a higher representation of disturbance sensitive species typically signals an undisturbed system (O'Connell et al., 2000; Bryce et al., 2002; Hausner et al., 2003; DeLuca et al., 2004; Glennon and Porter, 2005). The range of each species' abundance across all transects was divided into quartiles. A species' abundance at a transect was then scored based on its placement within those quartiles so that a disturbance tolerant species received a higher score of 3 or 4 if it's transect abundance was within the lower quartiles and a low score of 1 or 2 if its abundance was in the upper quartiles. Disturbance sensitive species were scored higher for abundances in the upper quartiles and lower in the lowest quartiles. A_I was then calculated for each transect by taking the mean species abundance score for that transect. The calculation of the IWCI improves on the IMBCI described in DeLuca et al. (2004) in that the IMBCI scored a species based on its presence and was not weighted by the abundance of disturbance tolerant or sensitive species. Finally, a score was calculated for each transect (T_{IWCI}):

$$T_{IWCI} = \left(\sum S_{IWCI}/S_N\right) + (2)A_I \quad (2)$$

where S_N is the total number of species detected at a transect. A_I was doubled to give it comparable weight to other variables in the equation. IWCI scores for the entire subestuary (E_{IWCI}) were calculated by taking the mean of the three T_{IWCI} scores within a subestuary.

2.5. *Data analysis*

E_{IWCI} was the dependant variable for all analyses except the redundancy analysis ($n = 17$ in 2002 and $n = 20$ in 2003). Data were analyzed separately by year because of annual differences in rainfall. In 2002, a region-wide drought year, freshwater inflow into Chesapeake Bay from January to August was 10,724 cubic meters/s (USGS, 2005). In contrast, 2003 had above average rainfall with a freshwater inflow of 29,749 cubic meters/s between January and August (USGS, 2005). Such annual variation in rainfall could alter nutrient discharge from watersheds (Correll et al., 1999), potentially influencing estuarine eutrophication and, ultimately, waterbird food resources and abundances.

We used redundancy analysis (RDA) on $\log_{10}(x + 1)$ species abundance data for 2002 and 2003 to examine whether species identified as disturbance tolerant were associated with disturbed landscapes (watersheds with high development or agriculture) and whether species identified as disturbance sensitive were associated with relatively undisturbed landscapes. Forward selection was used to assess the importance of forest, agriculture and development variables at the IDW watershed scale and to fit them as vectors in the ordination (Ter Braak and Verdonschot, 1995). The IDW watershed scale was selected based on the results of the least-squares regression analysis. Significance of the landscape variables included in the ordination was assessed with Monte Carlo permutation tests (1000 permutations). RDA was performed in CANOCO version 4.5 for Windows (Ter Braak and Smilauer, 1998).

To determine which land cover types were significant stressors to the IWCI, we tested seven candidate models relating IWCI scores to percentage of forest, agriculture, and development within each watershed. We evaluated a global model, models with all two-predictor combinations, models based on single predictors, and a null model. Interaction terms were not included in any of the models because we had no *a priori* evidence for such effects. We used a small-sample version of Akaike's information criteria (AIC_c) for model selection. Models with ΔAIC_c values >2 were considered to have strong support, between 4 and 7 to have some support, and >10 to have little support as the best model (Burnham and Anderson, 1998). To aid in selecting the best model, we also examined AIC_c weights (w_i), which can be interpreted as the probability that a given model provides the best fit to the data (Burnham and Anderson, 1998). AIC_c were generated in PROC MIXED using SAS 8.2 (SAS, 1999).

Next, we used least-squares regression to examine the relationship between the AIC_c best land cover model (total development) and IWCI scores by testing the influence of total development, suburban/rural development and urban development on IWCI scores at three geographic scales: watershed, IDW watershed, and local. When relationships between land cover and IWCI scores failed to meet assumptions of linearity, we used a nonparametric changepoint analysis to test for a nonlinear, threshold response of IWCI scores to land cover (King and Richardson, 2003; Qian et al., 2003). Nonparametric changepoint analysis estimates the numerical value of a predictor, x, resulting in a threshold in the response variable, y. The changepoint method employs a bootstrapping (resampling) technique to estimate a percentile confidence interval around the observed threshold. We plotted the cumulative distribution of the empirical percentile confidence limits on each predictor as a measure of the cumulative probability of a threshold (e.g., DeLuca et al., 2004; King et al., 2005b). We also estimated the probability that the observed variance explained by the changepoint was not different from zero (deviance reduction = 0), providing a further test for significance of nonlinear responses.

3. Results

Twenty-three total species were detected between both years across all sites and were scored in the IWCI. We detected 5.76 ± 3.13 bird species per subestuary in 2002 ($n = 17$) and 10.75 ± 2.57 species in 2003 ($n = 20$). In 2002, IWCI scores ranged from 10.70 to 17.73 (14.30 ± 2.06), and in 2003 from 12.17 to 15.55 (13.96 ± 0.99). The RDA explained 20% of the variation in waterbird community composition in 2002, and 27% in 2003. A Monte Carlo forward selection permutation test indicated a significant relationship of waterbird species to total development (2002: $\lambda_A = 0.13$, $P = 0.01$; 2003: $\lambda_A = 0.16$, $P = 0.001$), but not to agriculture (2002: $\lambda_A = 0.04$, $P = 0.79$; 2003: $\lambda_A = 0.04$, $P = 0.51$) or forest (2002: $\lambda_A = 0.03$, $P = 0.86$; 2003: $\lambda_A = 0.07$, $P = 0.14$). Although development was the only significant land use variable, we retained both agriculture and forest in the ordination to illustrate associations between individual species and each of the three land uses (Fig. 2). Of the seven species we identified as disturbance tolerant, the majority were associated with disturbed landscapes (2002: 5/7; 2003: 6/7).

A single variable model including total development was the best AIC_c model in both 2002 and 2003 (Table 2). Based on AIC_c weights (w_i), this model was thirteen times (2002) and twenty-six times (2003) more likely than the seven other candidate models to fit the IWCI data. Because total development was the best-supported predictor in both years, we focused subsequent analyses on this land cover and its constituent parts, urban and suburban development.

In both 2002 and 2003, IWCI scores were lower in subestuaries where total development occupied a larger portion of the landscape at multiple scales (Table 3). High levels of suburban/rural development also led to reduced IWCI scores in both years, but the relationship was weaker in comparison (Table 3). For both types of land cover, model fit improved when the two geographic scales emphasizing land cover near the estuarine coastline (IDW and 500 m) were used as predictors (Table 3). Extensive urban land cover resulted in low IWCI scores at the watershed scale, however the relationship between IWCI scores and urban land cover was not linear at the IDW and 500 m scales and was therefore examined using nonparametric changepoint analysis.

In 2002, changepoint analysis indicated a > 90% probability of a threshold response in the IWCI when as little as 3.7% of the IDW land cover within a watershed was urban (Fig. 3a). Comparable levels of urban development at the 500 m scale produced a weaker effect, with only a $\sim$30% probability of a threshold (Fig. 3b). However, when 4.1% of local land cover was urban, the probability of a threshold response rose to >85% (Fig. 3b). All changepoint values in 2002 were significantly different from 0 ($P \leqslant 0.05$).

In 2003, there was a >80% chance of a threshold response in IWCI scores when 3.5% of IDW land cover was urban and a 99.9% probability of a threshold at 4.6% urban development (Fig. 3c). Unlike the previous year, the effect in 2003 was

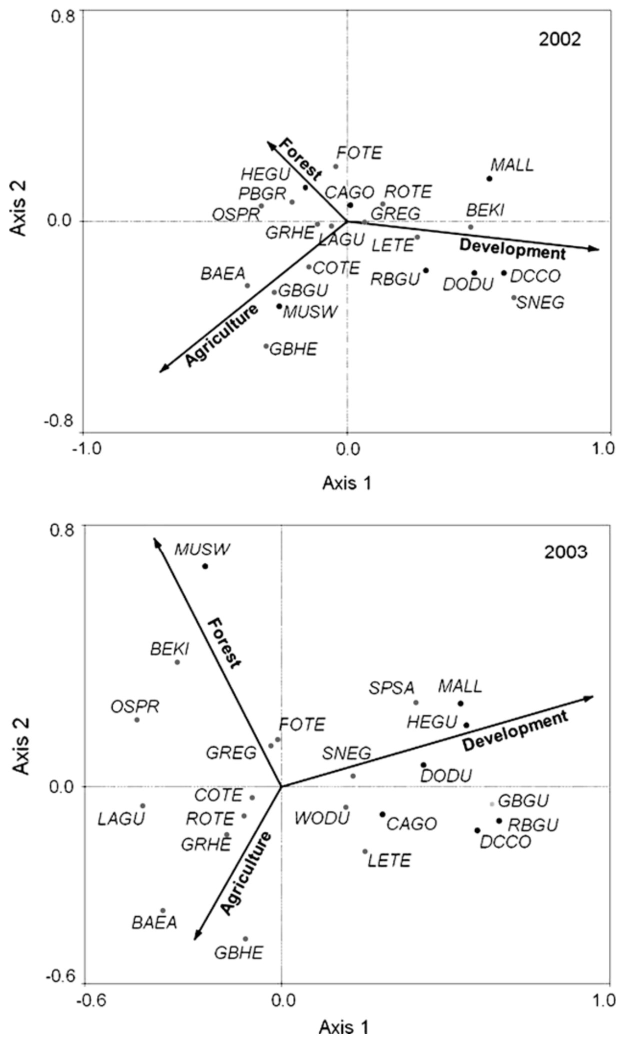

Fig. 2 – Biplot from a redundancy analysis (RDA) with waterbird abundance and inverse-distance-weighted (IDW) watershed land cover in 2002 and 2003. Species with gray circles were considered disturbance sensitive in the index of waterbird community integrity (IWCI) and species with black circles were considered disturbance tolerant. Species codes are listed in the Appendix A.

strongest at the local scale, with a 50% chance of a threshold response IWCI scores when 2.1% of the local land cover was urban and a 99.9% probability of a threshold at 3.9% urban development (Fig. 3d). All changepoint values in 2003 were significantly different from 0 ($P \leqslant 0.05$).

4. Discussion

The IWCI clearly identified developed land cover as the primary stressor influencing waterbird community integrity in Chesapeake Bay. In fact, no other land cover or combination of land covers explained more variation in IWCI scores than the null model. Many studies have identified development as a major contributor to coastal ecosystem impairment. For example, measures of development have led to decreases in the condition of benthic communities (Dauer et al., 2000; Hale et al., 2004; Bilkovic et al., 2006), blue crab (*Callinectes sapidus*) abundances (King et al., 2004) and tidal marsh plant communities (Bertness et al., 2002; Silliman and Bertness, 2004; King et al., 2007). Additionally, estuarine marsh bird community integrity has been shown to decrease significantly with increasing local development (DeLuca et al., 2004) and local road density has also been documented to influence marsh bird habitat occupancy (Shriver et al., 2004). It is clear that waterbird communities are sensitive to anthropogenic disturbance and to development in particular. However, few studies have identified significant nonlinear responses and quantified ecological thresholds in landscape disturbances beyond which bird communities are severely altered.

We detected a nonlinear, threshold response in waterbird community integrity at low levels of urban development (<5%) at local scales. Interestingly, the response of waterbird community integrity to suburban and total development at local scales was linear. It is likely that high-density residential buildings coupled with the commercial and industrial activities of urban land cover are significant contributors to the mechanisms that are negatively impacting waterbird community integrity, resulting in a large and abrupt shift in the state of the ecosystem (Scheffer et al., 2001). Such nonlinear responses can be moderated by both scale and intricacies in land use changes.

DeLuca et al. (2004) found a similar nonlinear response of marsh birds to local development; however several important distinctions exist between their findings and those reported here. DeLuca et al. (2004) examined marsh bird community integrity at independent marshes as a function of land use at the 500 m, 1000 m, and watershed scales. They found a 95% probability that thresholds occurred when 14% of land within 500 m of the marsh was developed and when 25% of land was developed within 1000 m. In contrast, marsh bird community integrity was not affected by land use at the watershed scale (DeLuca et al., 2004). The thresholds found by DeLuca et al. (2004) at higher levels of disturbance, coupled with a lack of response at the watershed scale, suggests that marshes and, in particular, the marsh bird community are primarily vulnerable to disturbances at local scales. Our findings of thresholds at much lower levels (<5% development), at the inverse-distance-weighted watershed scale suggest that estuarine waterbird community integrity, unlike estuarine marsh birds, are driven by mechanisms that operate at both local and watershed scales. These findings underscore the importance of incorporating multiple scales when considering stressors to estuarine ecosystems, particularly when spatial scale may provide insight into the mechanisms driving these responses.

When watershed land cover was weighted by its inverse-distance to the shoreline it consistently explained more variation in IWCI scores than both land cover within 500 m of the shoreline and at the unweighted watershed scale. Many studies have shown that either local (DeLuca et al., 2004; Shriver et al., 2004, etc.) or whole watershed (Dauer et al., 2000; Hale et al., 2004) scales are important when characterizing estuarine condition. Evidence is accumulating that models describing components of estuarine ecosystems can be improved when both local and watershed scales are integrated into a single predictor (Comeleo et al., 1996;

Table 2 – Relative ranking of models using land cover variables to describe variation in index of water bird community integrity (IWCI) scores

Year	Model	K	AIC_c	ΔAIC_c	w_i
2002					
	Development	3	68.5	0.0	0.752
	Null	2	73.7	5.2	0.056
	Development + forest	4	74.1	5.6	0.046
	Development + agriculture	4	74.1	5.6	0.046
	Agriculture	3	74.3	5.8	0.041
	Agriculture + forest	4	74.8	6.3	0.032
	Development + agriculture + forest	5	76.5	8.0	0.014
	Forest	3	76.6	8.1	0.013
2003					
	Development	3	52.1	0.0	0.903
	Null	2	58.6	6.5	0.035
	Development + forest	4	59.7	7.6	0.020
	Development + agriculture	4	59.7	7.6	0.020
	Development + agriculture + forest	5	61.9	9.8	0.001
	Agriculture	3	62.3	10.2	0.001
	Agriculture + forest	4	62.6	10.5	0.001
	Forest	3	62.8	10.7	0.000

Columns give model notation, number of estimable parameters (K), second-order Akaike's information criterion values (AIC_c), AIC_c differences (ΔAIC_c), and AIC_c weights (w_i).

Table 3 – Results of linear regressions for IWCI scores compared to developed, suburban/rural, and urban land cover types (suburban/rural and urban land covers are sub-units of development) at three different geographic extents in a dry (2002) and wet (2003) year

Land cover	Geographic extent					
	Watershed		Watershed (IDW)		500-m Buffer	
	2002	2003	2002	2003	2002	2003
Development	0.51, 0.001	0.54, <0.001	0.57, <0.001	0.60, <0.001	0.54, 0.001	0.57, <0.001
Suburban/rural	0.43, 0.004	0.47, <0.001	0.54, 0.001	0.57, <0.001	0.55, 0.001	0.54, <0.001
Urban	0.40, 0.007	0.51, <0.001	NL*	NL*	NL*	NL*

Results are summarized as r^2 and P-value. Slopes for all regressions were negative.

* Relationships between urban land cover and IWCI scores were not linear and were therefore analyzed with a changepoint analysis to test for the presence of an ecological threshold (see Fig. 3).

King et al., 2004, 2005a, 2007). We suggest that both watershed and local scale processes be considered when evaluating the health of estuarine ecosystems, but that landscape alterations near the coastline are most important. Despite substantial increases in the demand for coastal real estate in the mid-Atlantic region of North America, most restrictions on development near aquatic ecosystems have been focused on riparian zones (Miltner et al., 2004). Our findings offer compelling evidence that limits on urban development near estuarine shorelines should also be implemented.

Estuarine waterbird communities may be influenced by developed land cover through two potential pathways. First, development in coastal watersheds can contribute significant levels of nutrients and contaminants via point sources and hydrological processes of the watershed, thereby causing eutrophic and potentially hypoxic conditions (Boesch et al., 2001; Scavia and Bricker, 2006). Such conditions may impair estuarine organisms at lower trophic levels, such as benthic invertebrate and fish communities (Dauer et al., 2000; Eby and Crowder, 2002; Bilkovic et al., 2006). The estuarine waterbird community, positioned at the top of the food web, may be vulnerable to these disturbances via bottom-up controls (Baird et al., 2004). Contaminants such as PCBs and heavy metals present in the food web may compound the problem via biomagnification, imposing unfavorable physiological burdens on the waterbird community (Larsen et al., 1996; Rattner et al., 1997; Frank et al., 2001).

Second, development near the estuarine shoreline may result in the loss, fragmentation, and isolation of essential adjacent terrestrial habitats. Disturbances such as these are known to impact bird communities by reducing connectivity between habitat patches and increasing access for predators (Faaborg et al., 1995). Furthermore, development in close proximity to the coastline may also reduce shoreline refugia and increase competition with terrestrial generalists that are more abundant in disturbed landscapes (Blair, 1996), such as American crow (*Corvus brachyrhynchos*) and common

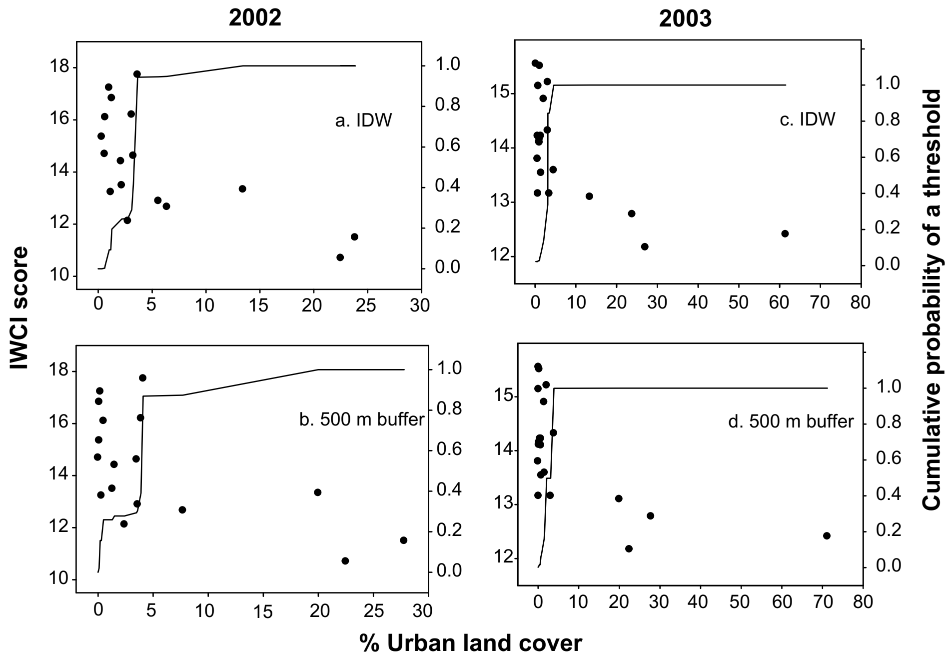

Fig. 3 – Results of changepoint analyses for percent urban development and index of waterbird community integrity scores (IWCI) in 2002 (a, b) and 2003 (c, d) for two different geographic extents; inverse-distance-weighted (IDW) land cover within the watershed (a, c) and a 500-m buffer around the subestuary (b, d). The solid lines depict the cumulative probability that an ecological threshold will occur with increasing urban development.

grackle (*Quiscalus quiscula*). Continued research aimed at determining the relative importance of aquatic and terrestrial pathways for estuarine waterbird community integrity could yield important insights into future watershed management practices.

The IWCI developed in this study is versatile and can be applied to other biological communities and regions. To use a modification of the IWCI to assess other ecosystems, species attributes and the scoring of those attributes should reflect the components of an intact ecosystem, while addressing regional management concerns. It is important to appreciate that the IWCI is meant to assess the entire community and not the relationship between a stressor and any one species. For example, in our study, reduced waterbird community integrity in areas with high indices of development was the result of low diversity and abundance of species with specialist attributes, such as terns (*Sterna*), and/or high diversity and abundance of species with generalist attributes, such as herring (*Larus argentatus*) and ring-billed gulls (*Larus delawarensis*). Therefore, IWCI scores were typically the result of a combination of generalists and specialist scores and were not driven by the presence or absence of any one species.

We used the IWCI to identify specific land cover types that are most detrimental to the Chesapeake Bay waterbird community and supporting estuarine ecosystem. Furthermore, we identify precise land cover thresholds, beyond which waterbird community integrity is severely impaired. This information, coupled with an understandable score representing waterbird community integrity, can be easily interpreted and applied by conservation decision makers and watershed managers. Resolving the pathway through which urban development harms estuarine integrity promises to illuminate the mechanisms and scaling relationships between disturbances and indicators. This information, in turn, will enhance monitoring efficiency and focus efforts on key stressors. With the recognition that estuarine degradation is not just an environmental but also an economic concern, cooperation between disparate stakeholders to address these challenges is now more likely than ever.

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Appendix A

List of bird species detected during transect surveys in Chesapeake Bay subestuaries, including scores for each species attribute used to calculate the total species score (S_{IWCI}). Alpha codes used in the redundancy analysis (RDA) are in parentheses.

Common name	Scientific name	Foraging niche breadth	Nesting sensitivity	Migratory status	Breeding range	State listing	Native status	S_{IWCI}
Pied-billed grebe (PBGR)	*Podilymbus podiceps*	3	2.5	1	1	1	2	10.5
Double-crested Cormorant* (DCCO)	*Phalacrocarax auritus*	2	1	1	2.5	1	2	9.5
Great blue heron (GBHE)	*Ardea herodias*	2	2.5	1	2.5	1	2	11
Great egret (GREG)	*Ardea alba*	2	2.5	2	1	1	2	10.5
Snowy egret (SNEG)	*Egretta thula*	2	2.5	3	1	1	2	11.5
Green heron (GRHE)	*Butorides virescens*	2	1	4	1	1	2	11
Mute swan* (MUSW)	*Cygnus olor*	3	2.5	1	1	1	0	8.5
Canada goose* (CAGO)	*Branta candadensis*	1	1	1	1	1	2	7
Wood duck (WODU)	*Aix sponsa*	4	4	3	2.5	1	2	16.5
Mallard* (MALL)	*Anas platytyrhynchos*	1	1	1	1	1	2	7
Domestic duck* (DODU)	*Anas platytyrhynchos*	1	1	1	1	1	0	5
Bald eagle (BAEA)	*Haliaeetus leucocephalus*	2	4	3	2.5	3	2	16.5
Osprey (OSPR)	*Pandion haliaetus*	4	1	4	1	1	2	13
Spotted sandpiper (SPSA)	*Actitis macularia*	3	1	4	2.5	1	2	13.5
Laughing gull (LAGU)	*Larus atricilla*	2	2.5	4	4	1	2	15.5
Ring-billed gull* (RBGU)	*Larus delawarinsis*	1	1	1	2.5	1	2	8.5
Herring gull* (HEGU)	*Larus argentatus*	1	1	1	2.5	1	2	8.5
Great black-backed gull (GBGU)	*Larus marinus*	2	2.5	2	4	1	2	13.5
Royal tern (ROTE)	*Sterna maxima*	4	4	3	4	4	2	21
Common tern (COTE)	*Sterna hirundo*	4	4	4	1	1	2	16
Forster's tern (FOTE)	*Sterna forsteri*	4	4	1	2.5	1	2	14.5
Least tern (LETE)	*Sterna antillarum*	4	4	4	2.5	2, 3	2	18.5, 19.5
Belted kingfisher (BEKI)	*Ceryle alcyon*	4	2.5	1	2.5	1	2	13

* Disturbance tolerant species.

REFERENCES

Baird, D., Christian, R.R., Peterson, C.H., Johnson, G.A., 2004. Consequences of hypoxia on estuarine ecosystem function: energy diversion from consumers to microbes. Ecological Applications 14, 805–822.

Bertness, M.D., Ewanchuk, P.J., Silliman, B.R., 2002. Anthropogenic modification of New England salt marsh landscapes. Proceedings of the National Academy of Sciences of the United States of America 99, 1395–1398.

Bilkovic, D.M., Roggero, M., Hershner, C.H., Havens, K.H., 2006. Influence of land use on macrobenthic communities in nearshore estuarine habitats. Estuaries and Coasts 29, 1185–1195.

Blair, R.B., 1996. Land use and species diversity along an urban gradient. Ecological Applications 6, 506–519.

Boesch, D.F., Brinsfield, R.B., Magnien, R.E., 2001. Chesapeake Bay eutrophication: scientific understanding, ecosystem restoration, and challenges for agriculture. Journal of Environmental Quality 30, 303–320.

Brody, S.D., Highfield, W., Carrasco, V., 2004. Measuring the collective planning capabilities of local jurisdictions to manage ecological systems in southern Florida. Landscape and Urban Planning 96, 33–50.

Bryce, S.A., Hughes, R.M., Kaufmann, P.R., 2002. Development of a bird integrity index: using bird assemblages as indicators of riparian condition. Environmental Management 30, 294–310.

Burnham, K.P., Anderson, D.R., 1998. Model Selection and Inference: A Practical Information–Theoretic Approach. Springer-Verlag, New York, NY.

Comeleo, R.L., Paul, J.F., August, P.V., 1996. Relationships between watershed stressors and sediment contamination in Chesapeake Bay estuaries. Landscape Ecology 11, 307–319.

Correll, D.L., Jordan, T.E., Weller, D.E., 1999. Effects of interannual variation of precipitation on stream discharge Rhode River subwatersheds. Journal of the American Water Resources 35, 73–82.

Dauer, D.M., Ranasinghe, J.A., Weisberg, S.B., 2000. Relationships between benthic community condition, water quality, sediment quality, nutrient loads, and land use patterns in Chesapeake Bay. Estuaries 23, 80–96.

DeLuca, W.V., Studds, C.E., Rockwood, L.L., Marra, P.P., 2004. Influence of land use on the integrity of marsh bird communities of Chesapeake Bay, USA. Wetlands 24, 837–847.

Duncan, R.P., Blackburn, T.M., Sol, D., 2003. The ecology of bird introductions. Annual Review of Ecology and Systematics 34, 71–98.

Eby, L.A., Crowder, L.B., 2002. Hypoxia-based habitat compression in the Neuse River Estuary: context-dependent shifts in behavioral avoidance thresholds. Canadian Journal of Fisheries and Aquatic Sciences 59, 952–965.

Faaborg, J., Brittingham, M.C., Donovan, T.M., Blake, J.G., 1995. Habitat fragmentation in the temperate zone. In: Martin, T.E., Finch, D.M. (Eds.), Ecology and Management of Neotropical Migratory Birds: A Synthesis and Review of Critical Issues. Oxford University Press, New York, pp. 357–380.

Frank, D.S., Mora, M.A., Sericano, J.L., Blankenship, A.L., Kannan, K., Giesy, J.P., 2001. Persistent organochlorine pollutants in eggs of colonial waterbirds from Galveston Bay and East Texas, USA. Environmental Toxicology and Chemistry 20, 608–617.

Glennon, M.J., Porter, W.F., 2005. Effects of land use management on biotic integrity: an investigation of bird communities. Biological Conservation 126, 499–511.

Guénette, J.S., Villard, M.A., 2005. Thresholds in forest bird response to habitat alteration as quantitative targets for conservation. Conservation Biology 19, 1168–1180.

Hale, S.S., Paul, J.F., Heltshe, J.F., 2004. Watershed landscape indicators of estuarine benthic condition. Estuaries 27, 283–295.

Hausner, V.H., Yoccoz, N.G., Ims, R.A., 2003. Selecting indicator traits for monitoring land use impacts: birds in northern coastal birch forests. Ecological Applications 13, 999–1012.

Jackson, S.F., Kershaw, M., Gaston, K.J., 2004. The performance of procedures for selecting conservation areas: waterbirds in the UK. Biological Conservation 118, 261–270.

Keats, R.A., Osher, L.J., Neckles, H.A., 2004. The effect of nitrogen loading on a brackish estuarine faunal community: a stable isotope approach. Estuaries 27, 460–471.

Kennish, M.J., 2002. Environmental threats and environmental future of estuaries. Environmental Conservation 29, 78–107.

King, R.S., Richardson, C.J., 2003. Integrating bioassessment and ecological risk assessment: an approach to developing numerical water-quality criteria. Environmental Management 31, 795–809.

King, R.S., Beaman, J.R., Whigham, D.F., Hines, A.H., Baker, M.E., Weller, D.E., 2004. Watershed land use is strongly linked to PCBs in White Perch in Chesapeake Bay subestuaries. Environmental Science and Technology 38, 6546–6552.

King, R.S., Hines, A.H., Craige, F.D., Grap, S., 2005a. Regional, watershed, and local correlates of blue crab and bivalve abundances in subestuaries of Chesapeake Bay, USA. Journal of Experimental Marine Biology and Ecology 319, 101–116.

King, R.S., Baker, M.E., Whigham, D.F., Weller, D.E., Jordan, T.E., Kazyak, P.F., Hurd, M.K., 2005b. Spatial considerations for linking watershed land cover to ecological indicators in streams. Ecological Applications 15, 137–153.

King, R.S., DeLuca, W.V., Whigham, D.F., Marra, P.P., 2007. Threshold effects of coastal urbanization on *Phragmites australis* (common reed) abundance and foliar nitrogen in Chesapeake Bay. Estuaries and Coasts 30, 469–481.

Larsen, J.M., Karasov, W.H., Sileo, L., Stromborg, K.L., Hanbidge, B.A., Giesy, J.P., Jones, P.D., Tillitt, D.E., Verbrugge, D.A., 1996. Reproductive success, developmental anomalies, and environmental contaminants in double-crested cormorants (*Phalacrocorax auritus*). Environmental Toxicology and Chemistry 15, 553–559.

Lippson, A.J., Lippson, R.L., 1997. Life in the Chesapeake Bay. The Johns Hopkins University Press, Baltimore, MD.

Lovell, C., Mandondo, A., Moriarty, P., 2002. The question of scale in integrated natural resource management. Conservation Ecology 5, 1–30.

Miltner, R.J., White, D., Yoder, C., 2004. The biotic integrity of streams in urban and suburbanizing landscapes. Landscape and Urban Planning 69, 87–100.

Nichols, J.D., Hines, J.E., Sauer, J.R., Fallon, F.W., Fallon, J.E., Heglund, P.J., 2000. A double observer approach of estimating detection probability and abundance from point counts. The Auk 117, 393–408.

Nixon, S.W., 1995. Coastal marine eutrophication: a definition, social causes, and future consequences. Ophelia 41, 199–219.

O'Connell, T.J., Jackson, L.E., Brooks, R.P., 2000. Bird guilds as indicators of ecological condition in the central Appalachians. Ecological Applications 10, 1706–1721.

Poole, A., Gill, F., 1999. The Birds of North America. The Birds of North America, Inc, Philadelphia, PA.

Qian, S.S., King, R.S., Richardson, C.J., 2003. Two statistical methods for the detection of environmental thresholds. Ecological Modeling 166, 87–97.

Rattner, B.A., Melancon, M.J., Rice, C.P., Riley, W., Eisemann, J., Hines, R.K., 1996. Cytochrome P450 and organochlorine contaminants in black-crowned night-herons from Chesapeake Bay region, USA. Environmental Toxicology and Chemistry 16, 2315–2322.

Roberts, C.M., Hawkins, J.P., 1999. Extinction risk in the sea. Trends in Ecology and Evolution 14, 241–246.

Sanger, D.M., Holland, A.F., Hernandez, D.L., 2004. Evaluation of the impacts of dock structures and land use on tidal creek ecosystems in South Carolina estuarine environments. Environmental Management 33, 385–400.

SAS Institute, 1999. SAS/STAT Users Guide, Version 8.2. SAS Institute, Cary, North Carolina, USA.

Scavia, D., Bricker, S.B., 2006. Coastal eutrophication assessment in the United States. Biogeochemistry 79, 1878–2208.

Scheffer, M., Carpenter, S., Foley, J.A., Folke, C., Walker, B., 2001. Catastrophic shifts in ecosystems. Nature 413, 591–596.

Shriver, W.G., Hodgman, T.P., Gibbs, J.P., Vickery, P.D., 2004. Landscape context influences salt marsh bird diversity and area requirements in New England. Biological Conservation 119, 545–553.

Silliman, B.R., Bertness, M.D., 2004. Shoreline development drives invasion of *Phragmites australis* and the loss of plant diversity on New England salt marshes. Conservation Biology 18, 1424–1434.

Soranno, P.A., Hubler, S.L., Carpenter, S.R., 1996. Phosphorous loads to surface waters: a simple model to account for spatial pattern of land use. Ecological Applications 6, 865–878.

Takekawa, J.Y., Miles, A.K., Schoellhamer, D.H., Athearn, N.D., Saiki, M.K., Duffy, W.D., Kleinschmidt, S., Shellenbarger, G.G., Jannusch, C.A., 2006. Trophic structure and avian communities across a salinity gradient in evaporation ponds of the San Francisco Bay estuary. Hydrobiologia 567, 307–327.

Ter Braak, C.J.F., Verdonschot, P.F.M., 1995. Canonical correspondence analysis and related multivariate methods in aquatic ecology. Aquatic Sciences 57, 153–187.

Ter Braak, C.J.F., Smilauer, P., 1998. Canoco 4. Microcomputer Power. Ithaca, New York.

Tiner, R.W., Burke, D.G., 1995. Wetlands of Maryland. U.S. Fish and Wildlife Service, Ecological Services, Hadley, MA and Maryland Department of Natural Resources, Annapolis, Maryland.

United States Environmental Protection Agency (USEPA), 2000. Multi-Resolution Land Characteristics Consortium (MRLC) Database. <http://www.epa.gov/mrlcpage> (accessed 15.05.07.).

United States Geological Survey (USGS), 2005. Estimated Streamflow Entering Chesapeake Bay. <http://md.water.usgs.gov/monthly/bay1.html> (accessed 15.05.07.).

With, K.A., Crist, T.O., 1995. Critical thresholds in species' responses to landscape structure. Ecology 76, 2446–2459.