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Salt marsh width positively affects the occurrence of Least and Pectoral Sandpipers in the St. Lawrence River Estuary during fall migration

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Abstract

Salt marshes are vulnerable to climate change-associated sea-level rise and storm-induced surges. Their degradation will likely affect shorebirds relying on this ecosystem. Least Sandpiper (*Calidris minutilla*) and Pectoral Sandpiper (*Calidris melanotos*) migrating along coastline habitats typically use salt marshes to rest and replenish their body reserves. Our objective was to test if width of the different vegetation zones within salt marshes affects the occurrence of Least and Pectoral Sandpipers stopping along the St. Lawrence River Estuary, Quebec, Canada, during fall migration. We established 26 survey sites, each 600 m in length, along the shoreline. Shorebird surveys were conducted in 2011 and 2012. We characterized salt marshes by measuring the width of each vegetation zone (lower marsh and upper marsh). We analyzed shorebird presence/not detected data with generalized estimating equations to test the predictions that occurrence of Least Sandpipers and Pectoral Sandpipers increases with width of both the lower and upper marsh. Upper marsh width was positively associated with probability of occurrence in each species. Our results highlight the importance of protecting the integrity of salt marshes for these two species. In the St. Lawrence River Estuary, where landward migration of salt marshes is no longer possible (coastal squeeze), effective management of shorelines is much needed. Otherwise, salt marshes and these two species could be locally jeopardized.

Key words: Least Sandpiper; Pectoral Sandpiper; shorebird migration; stopover site, salt marsh, St. Lawrence River Estuary

Résumé

Les marais salés sont menacés par la hausse du niveau des océans et par les tempêtes côtières associées aux changements climatiques. Leur dégradation aura vraisemblablement un impact négatif sur les oiseaux de rivage qui les fréquentent. Le bécasseau minuscule (*Calidris minutilla*) et le bécasseau à poitrine cendrée (*Calidris melanotos*) migrant le long des côtes utilisent de manière importante cet habitat pour le repos et l'acquisition de réserves corporelles. Nous avons voulu vérifier si la largeur du bas marais et celle du haut marais avaient un effet sur l'occurrence de ces deux espèces dans l'estuaire St-Laurent au cours de la migration automnale. Nous avons disposé 26 sites d'inventaire d'une longueur de 600 m le long du littoral. Des inventaires d'oiseaux de rivage y ont été réalisés, en 2011 et en 2012. Nous avons mesuré dans ces sites la largeur du bas marais et celle du haut marais. Nous avons analysé des données de présence/absence pour le bécasseau minuscule et le bécasseau à poitrine cendrée à l'aide d'équations d'estimations généralisées, afin de vérifier si leur probabilité de présence augmentait avec la largeur du bas marais et celle du haut marais. La largeur du haut marais avait un effet positif sur l'occurrence de ces espèces. Ces résultats démontrent l'importance de protéger les marais salés pour celles-ci. Dans l'estuaire du fleuve St-Laurent, là où la migration vers l'intérieur des marais salés n'est plus possible (coincement côtier), des mesures de conservation sont requises. Sans ces mesures, les marais salés et ces deux espèces pourraient être localement menacés.

Mots-Clés: Bécasseau minuscule; Bécasseau à poitrine cendrée; migration des oiseaux de rivage; halte migratoire; marais salé; estuaire du fleuve St-Laurent

Introduction

Climate change impacts on coastal ecosystems, adjacent infrastructure, and low-lying communities is one of the most significant challenges of our time (United Nations 2020; World Wildlife Fund 2020). Indeed, the effects on coastal ecosystems of climate change-associated sea-level rise, as well as increasingly severe and frequent storm-induced surges, are now well documented (Hoegh-Guldberg and Bruno 2010; Passeri *et al*. 2015; Campbell and Wang 2020). Unfortunately, based on greenhouse gas emission scenarios, this situation is unlikely to improve in the short term (Nicholls and Cazenave 2010; Hinkel *et al*. 2014; Taherkhani *et al*. 2020). In salt marsh ecosystems, wave lateral erosion on the seaward edge and drowning, due to insufficient vertical surface accretion to compensate relative sea-level rise, both contribute to salt marsh degradation or losses (Watson *et al*. 2017; Cahoon *et al*. 2019; Payne *et al*. 2019).

Increasingly severe weather events and salt marsh degradation will likely affect animal populations, including shorebirds (Hunter *et al*. 2015; Correll *et al*. 2017; Von Holle *et al*. 2019). Unfortunately, many shorebird populations worldwide are already declining (Andres *et al*. 2012; Sutherland *et al*. 2012; Galbraith *et al*. 2014). In North America, shorebird populations have decreased since 1970 (Rosenberg *et al*. 2019). Some species are now considered Endangered in Canada (e.g., Piping Plover [*Charadrius melodus*], Red Knot rufa subspecies [*Calidris canutus rufa*]; SARA Registry 2021a,b). The underlying causes of global shorebird decline are not fully understood, but several mechanisms are likely involved (Munro 2017). On their subarctic and Arctic breeding grounds, rapidly changing climate conditions and associated mismatch between chick needs and peak insect emergence (van Gils *et al*. 2016; Kwon *et al*. 2019), degradation of tundra breeding sites by now overabundant Snow Goose (*Anser caerulescens*; Koons *et al*. 2014; Flemming *et al*. 2016), and increased predation of shorebird nests by Arctic Fox (*Vulpes lagopus*), and aerial predators attracted by conspicuous Snow Goose nests (Lamarre *et al*. 2017; Flemming *et al*. 2019a,b) have been invoked. Further south, hunting in the Caribbean and northern South America (Watts and Turrin 2016; Reed *et al*. 2018), human disturbance (Finney *et al*. 2005; Liley and Sutherland 2007), harvesting of marine resources (van Gils *et al*. 2006; Atkinson *et al*. 2007), pollution (Hua *et al*. 2015; Perkins *et al*. 2016; Pratte *et al*. 2020), and coastal development (Piersma *et al*. 2016; Chan *et al*. 2019; Mu and Wilcove 2020) are other likely drivers of shorebird decline.

Climate change effects on salt marsh integrity may exacerbate this ongoing decline if migrating shorebirds relying on salt marshes can no longer find adequate stopover and staging habitats. Stopover and staging sites are essential to migrating shorebirds to rest and replenish their body reserves throughout their route on predictable and abundant prey (Warnock 2010). Least Sandpiper (*Calidris minutilla*) and Pectoral Sandpiper (*Calidris melanotos*) migrating along coastline habitats typically use salt marshes (Bent 1962; Farmer *et al*. 2020; Nebel and Cooper 2020). Along the St. Lawrence River Estuary shoreline during fall (post-breeding) migration, Least Sandpiper and Pectoral Sandpiper *en route* to their wintering grounds, located mainly in South America, are observed almost exclusively in, or close to, salt marshes. While Pectoral Sandpiper populations are considered stable (BirdLife International 2020a), Least Sandpiper populations are decreasing (BirdLife International 2020b). Therefore, it is essential to readily identify salt marsh characteristics selected by these species during migration to achieve proper protection or restoration of these ecosystems where needed. These actions would help meet shorebird conservation objectives (North American Bird Conservation Initiative Canada 2019; North American Bird Conservation Initiative 2020).

The main objective of our study was to identify salt marsh characteristics affecting the occurrence of southbound Least and Pectoral Sandpipers stopping along the St. Lawrence River Estuary during fall migration. We hypothesized that the presence of these two species is associated with salt marsh width. More specifically, we addressed this hypothesis by testing the following predictions. Because width of another type of relatively narrow habitat (beaches) has been associated with shorebird use during migration (Murchison *et al*. 2016), we predicted that their occurrence at the survey site scale would increase with the size and therefore the width of the Smooth Cordgrass (*Sporobolus alterniflorus* (Loiseleur-Deslongchamps) P.M. Peterson & Saarela) lower marsh covered, at least partially, twice daily by tides. We also predicted that their occurrence at the survey site scale would increase with the size of the more diversified upper (or higher) marsh, flooded only during the highest tides, hence providing refuge to these species in most tidal conditions.

Study Area

This study was conducted on the south shore of the St. Lawrence River Estuary, along a 130 km stretch of shoreline between St-Roch-des-Aulnaies (47.311°N, 70.177°W) and St-Simon-sur-Mer (48.205°N, 69.082° W), Quebec, Canada (Figure 1a). Within the study area, water circulation is dominated by semidiurnal

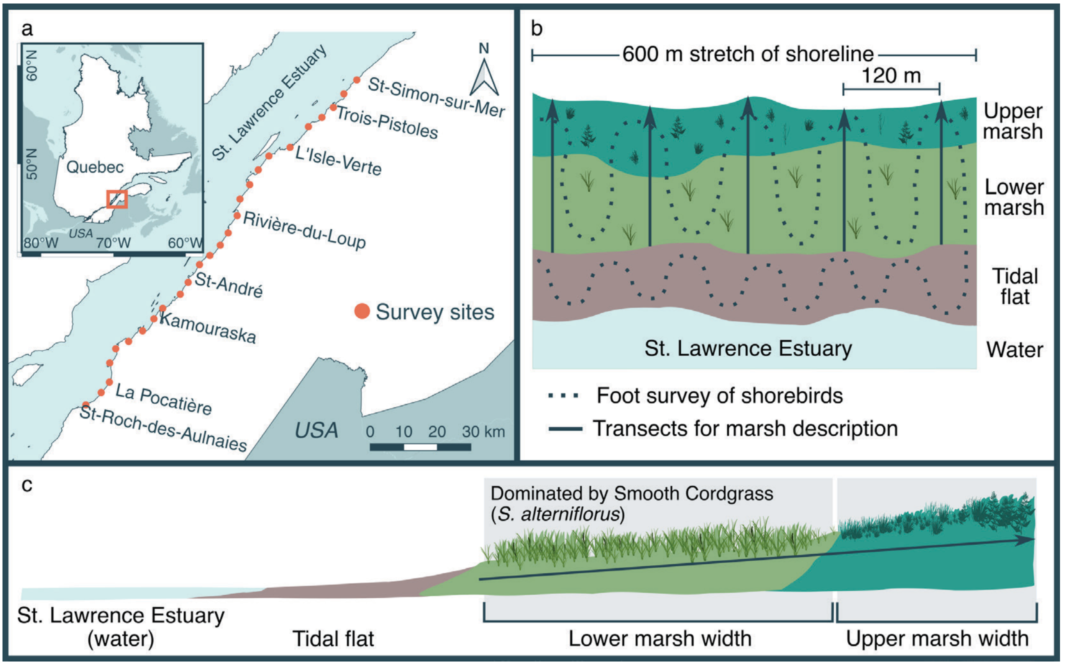

FIGURE 1. Study site locations and physical characteristics of areas surveyed. a. Location of survey sites on the south shore of the St. Lawrence River Estuary, Quebec, Canada; b. A typical foot survey of shorebirds and transects for marsh description; c. A transect for marsh description and zonation of salt marsh vegetation.

tides that can reach over 5 m in height (Fisheries and Oceans Canada 2011–2012). Intertidal substrates are highly variable, ranging from boulders, bare rock, and beaches at exposed sites to mudflats adjacent to salt marshes on more protected shorelines. Additional details on the study area can be found in Turcotte *et al*. (2017).

Salt marshes in the St. Lawrence River Estuary show the typical plant zonation resulting from tidal flooding and reported in other locations along the Atlantic Coast (Bertness and Ellison 1987; Kunza and Pennings 2008; Smith 2015). The lower marsh is covered, at least partially, twice daily by tides and is almost exclusively occupied by the native Smooth Cordgrass. Above the lower marsh, the upper marsh's seaward edge is generally dominated by Saltmeadow Cordgrass (*Sporobolus pumilus* (Roth) P.M. Peterson & Saarela). Upslope from Saltmeadow Cordgrass zone, the upper marsh vegetation becomes highly diversified and includes species such as Prairie Cordgrass (*Sporobolus michauxianus* (Hitchcock) P.M. Peterson & Saarela), Saltmarsh Bulrush (*Bolboschoenus maritimus* (L.) Palla), Seaside Plantain (*Plantago maritima* L.), and Virginia Glasswort (*Salicornia depressa* Standley; Dionne 1989; Coulombier *et al*. 2012).

Methods

Shorebird surveys

We established 26 survey sites 5 km apart along the shoreline (Figure 1a). Each survey site corresponded to a 600 m stretch of shoreline (Figure 1b), the length of which was measured with a handheld global positioning system unit at mean high tide level. Mean high tide level coincides with the upper limit on the shore of Smooth Cordgrass (Gauthier 1982; Smith 2015). Survey sites included all shorebird habitats above and below the shoreline (tidal flats, marshes, beaches, rocky shores). A first survey site was randomly selected to the nearest metre along a longitudinal axis within the study area. The other sites were thereafter positioned progressively every 5 km along the shoreline (systematic random sampling). In some cases, survey sites were relocated in similar habitat type, as close as possible from the selected site when, chiefly due to duck hunting activity, observer safety was compromised.

Surveys were conducted in 2011 and 2012 from early July through late November, corresponding to the migration period of all shorebird species observed annually in the study area. Surveys were conducted every week in 2011 (21 survey weeks) and every other week in 2012 (11 survey weeks). During precisely 30 min, each 600 m survey site was walked

(Figure 1b) by one or two nearby comoving observers (same observers in both years) to maximize visual coverage and induce flight of birds hidden in vegetation otherwise difficult to detect (Farmer and Durbian 2006; Andres *et al*. 2012). Shorebirds were identified with 60× spotting scopes or 8× binoculars when on the ground, or by their calls when in flight. Although each survey site corresponded to a 600 m stretch of shoreline, their widths were highly variable. It follows that in extensive marshes, complete surface coverage was challenging, and some birds likely remained undetected (see *Statistical analyses*). Sites were surveyed in different tidal conditions during consecutive weekly (2011) or biweekly surveys (2012). Tidal conditions may constrain habitat availability for birds feeding in intertidal habitats (Calle *et al*. 2018; Horn *et al*. 2020). Thus, we determined the relative water level at the time of the survey for each site, using predicted hourly heights for the nearest water level station located along the coast (Fisheries and Oceans Canada 2011–2012; distance to survey site: mean 9.5 km, SD 6.6 km). We defined the relative water level as the difference (m) either above or below the mean high tide level established for the nearest water level station. We used relative water level rather than absolute water level because the half funnel shape of the St. Lawrence River Estuary resulted in the mean high tide level increasing progressively going upriver in our study area (1.1 m difference).

Survey site descriptions

We selected the two most likely salt marsh characteristics based on species' natural history (Bent 1962; Farmer *et al*. 2020; Nebel and Cooper 2020), to explain occurrence at the survey site scale. We measured to the nearest metre with a measuring tape the width of each vegetation zone (lower marsh and upper marsh) from its lower edge to its upper edge along five evenly spaced (120 m) transects (two or three per year; Figure 1b). We did not notice perceivable habitat change between years. A first transect location was randomly selected to the nearest metre along a longitudinal axis within each survey site. The other transects were thereafter positioned every 120 m along the shoreline. Thus, the lower marsh width was measured from Smooth Cordgrass's appearance on the mudflat to the upper marsh's seaward edge (Figure 1c), typically occupied by Saltmeadow Cordgrass. The upper marsh width was measured from the lower marsh's upper edge to halophyte vegetation's disappearance. A mean lower marsh width and a mean upper marsh width were averaged from the five transects measures for each survey site.

Statistical analyses

There can be no certainty that all birds hidden in vegetation, along tidal pools, or drainage channels, were detected during surveys, particularly in extensive marshes. Therefore, for each species, we analyzed presence/not detected data rather than abundance. We acknowledge that even with equal effort, there would be a greater chance of missing birds on wider marshes. Also, we are unaware of a procedure to detect hidden birds other than using well-trained dogs or several people walking side-by-side; therefore we concentrated on presence/not detected.

Data collected at the same survey site over several weeks were not independent (repeated measurements design). Thus, we used generalized estimating equations (GEE) to test the predictions that occurrences of Least Sandpipers and Pectoral Sandpipers increase with: 1) the width of the lower marsh and, independently, 2) the width of the upper marsh. We looked at residual autocorrelograms to identify, for each species/year combination, the most suitable correlation structure to include in our models. In each of these four combinations, the correlation between any pair of observations declined with survey week. Thus, we used autoregressive models of a 1st order correlation structure (AR1). Because we had a small number (<30) of survey sites, we used the jackknife variance estimator. We ran separate analyses for each species.

During fall migration, Least Sandpipers are present in the study area from early July through early October (peak abundance around mid-August), while Pectoral Sandpipers were present from late August through late October (peak abundance around late September; Turcotte *et al*. 2017). Hence, we considered only survey weeks for each species during which at least one individual was detected in one of the 26 survey sites (as a result, one week was not considered in the analysis of Least Sandpiper occurrence in 2011). Preliminary models included one categorical predictor variable (study year), three continuous predictor variables (mean lower marsh width, mean upper marsh width, relative water level during the survey), and Least Sandpiper or Pectoral Sandpiper occurrence as response variables. Final models included predictor variables for which $P < 0.05$ in the preliminary model. All statistical analyses were carried out using package "geepack" (Halekoh *et al*. 2006) with R version 4.0.2 (R Core Team 2020).

Results

Shorebird surveys

Out of the 26 survey sites, Least Sandpipers were present at 23 survey sites in 2011 (from 4 July to 9 October, for a total of 963 birds detected) and 16 survey sites in 2012 (from 3 July to 22 September, for a total of 512 birds detected). Pectoral Sandpipers were present at 10 survey sites in 2011 (from 2 September to 23 October, for a total of 68 birds detected) and nine survey sites in 2012 (from 25 August to 22 October, for a total of 108 birds detected). Least and Pectoral Sandpipers were never detected at three and 13 survey sites, respectively. Total number of birds detected per survey site in 2011 and 2012 were positively correlated in both species (Least Sandpiper: Pearson $r = 0.69$, $t_{24} = 4.69$, $P < 0.0001$, $n = 26$; Pectoral Sandpiper: Pearson $r = 0.97$, $t_{24} = 18.97$, $P < 0.0001$, $n = 26$).

Survey site descriptions

Among the 26 survey sites, 14 included both a lower and an upper marsh, five contained only a lower marsh adjacent to a beach, and one included only an upper marsh because the lower marsh was completely eroded. Six survey sites were devoid of salt marsh vegetation. Lower marsh width (average of the five transects) ranged from 1.0 to 318.8 m (mean 93.8, SD 86.9, $n = 19$) while upper marsh width (average of the five transects) ranged from 4.4 to 357.8 m (mean 91.4, SD 107.8, $n = 15$). Lower marsh width and upper marsh width were not correlated in the 14 marshes where both types were present (Pearson $r = -0.27$, $t_{12} = -0.97$, $P = 0.35$, $n = 14$).

Survey site occupancy by Least and Pectoral Sandpipers

Neither year nor relative water level was associated with occurrence in either shorebird species (Table 1). However, lower marsh width and upper marsh width were both positively associated with Least and Pectoral Sandpipers' occurrence (Table 1). Six of the 26 survey sites were devoid of salt marsh vegetation. Hence, these results may be driven by the simple absence of marsh vegetation at these sites (Least Sandpiper: never detected at two of these six survey sites; Pectoral Sandpiper: never detected at any of the six survey sites). Therefore, to test if marsh width *per se* affected occurrence, we reanalyzed the data considering only the 20 sites where marsh vegetation was present. Again, upper marsh width was positively associated with Least and Pectoral Sandpipers' occurrence (Table 2). However, lower marsh width was no longer associated with Least and Pectoral Sandpipers' occurrence using the conventional criterion of $P < 0.05$ (Table 2). Nevertheless, we cannot rule out the possibility that this last result could be due to low sample size and associated reduced statistical power. Thus, it may represent a Type II error (incorrectly failing to reject a false null hypothesis). If we apply the precautionary principle of environmental decision-making, we should not readily conclude that lower marsh width has no effect on occurrence of these species.

Indeed, Figure 2 suggests that both species used salt marshes as long as a minimum width of either the lower marsh, the upper marsh, or both, was available. At survey sites where these species were observed during both study years, lower and upper marsh's total width reached at least 39 m and 106 m for Least and Pectoral Sandpipers, respectively (Figure 2). Based on CIs, mean total width was greater at those sites with our focal species than at survey sites where these species were never detected (Figure 3). Correspondingly, although we did not analyze abundance data due to possible detection issues, we found a positive trend between the maximum number of birds detected during a survey per survey site and salt marsh total width for both years and species (Figure 4).

Discussion

Our study presents, to our knowledge, the first investigation of habitat requirements for two south-

TABLE 1. Relationships between salt marsh characteristics, relative water level, year, and occurrence of Least (*Calidris minutilla*) and Pectoral (*Calidris melanotos*) Sandpipers on the south shore of the St. Lawrence River Estuary, Quebec, Canada, during fall migration, 2011 and 2012. Preliminary models include lower marsh width, upper marsh width, relative water level, and year as predictor variables. Final models include only lower marsh width and upper marsh width.

Species	Predictor variable	Preliminary model				Final model			
		β	SE	Wald statistic	P	β	SE	Wald statistic	P
Least Sandpiper	Lower marsh width (m)	0.005	0.001	12.97	0.0003	0.005	0.001	11.50	0.0007
	Upper marsh width (m)	0.008	0.001	46.92	<0.0001	0.008	0.001	50.70	<0.0001
	Relative water level (m)	−0.032	0.111	0.08	0.7755				
	Year	−0.192	0.294	0.43	0.5134				
Pectoral Sandpiper	Lower marsh width (m)	0.006	0.002	11.13	0.0009	0.007	0.002	9.86	0.0017
	Upper marsh width (m)	0.008	0.003	5.80	0.0160	0.008	0.003	6.47	0.0110
	Relative water level (m)	0.226	0.230	0.97	0.3242				
	Year	0.263	0.647	0.17	0.6841				

TABLE 2. Relationships between salt marsh characteristics, relative water level, year, and occurrence of Least (*Calidris minutilla*) and Pectoral (*Calidris melanotos*) Sandpipers on the south shore of the St. Lawrence River Estuary, Quebec, Canada, during fall migration, 2011 and 2012. Data only include the 20 survey sites where marsh vegetation was present. Preliminary models include lower marsh width, upper marsh width, relative water level, and year as predictor variables. Final models include only upper marsh width (Least Sandpiper) or lower marsh width and upper marsh width (Pectoral Sandpiper).

Species	Predictor variable	Preliminary model				Final model			
		β	SE	Wald statistic	*P*	β	SE	Wald statistic	*P*
Least Sandpiper	Lower marsh width (m)	0.002	0.001	2.64	0.1043				
	Upper marsh width (m)	0.006	0.001	29.64	<0.0001	0.006	0.001	30.04	<0.0001
	Relative water level (m)	0.001	0.118	0.00	0.9961				
	Year	−0.104	0.278	0.14	0.7079				
Pectoral Sandpiper	Lower marsh width (m)	0.005	0.002	4.33	0.0375	0.005	0.002	3.84	0.0500
	Upper marsh width (m)	0.007	0.003	4.20	0.0404	0.006	0.003	4.47	0.0350
	Relative water level (m)	0.253	0.231	1.21	0.2720				
	Year	0.273	0.621	0.19	0.6608				

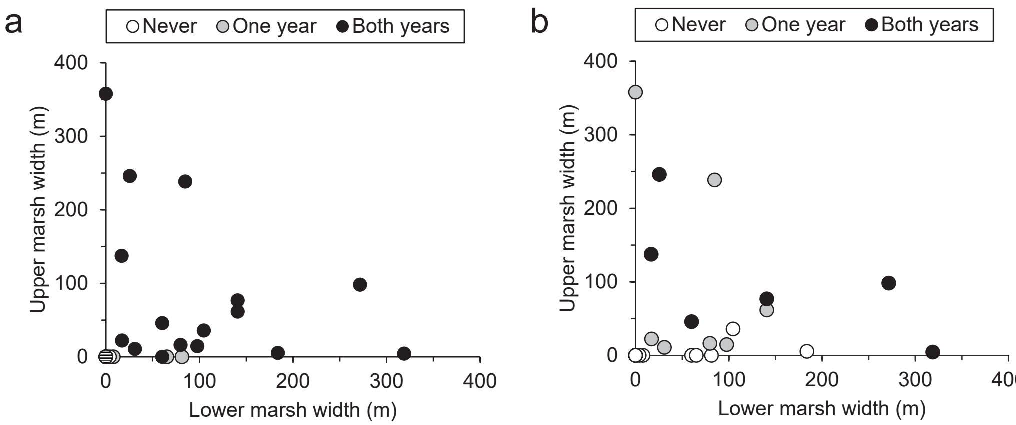

FIGURE 2. Salt marsh characteristics and occurrence of a. Least (*Calidris minutilla*) and b. Pectoral (*Calidris melanotos*) Sandpipers on the south shore of the St. Lawrence River Estuary, Quebec, Canada, during fall migration. Black circles are survey sites where species were detected during both years of the study, grey circles are survey sites where species were detected one year only, and white circles are survey sites where species were never detected. The cross-hatched circle represents six survey sites where Least Sandpipers were either detected one year only (four sites) or never detected (two sites).

bound shorebird species using salt marshes as stopover sites along the Atlantic seaboard. The fact that total number of birds detected per survey site in 2011 and 2012 was positively correlated in both species suggests that these birds' stopover site selection was not random but rather driven by habitat requirements. Indeed, we found that their occurrence along the St. Lawrence River Estuary was associated with marsh width, in particular upper marsh width. Because the upper marsh floods only during the highest tides, it provides refuge to these species in most tidal conditions. Our results highlight the importance of protecting the integrity of salt marshes for these two species. These marshes are likely highly relevant for migrating juveniles that far outnumber adults in our study area for both species (Turcotte *et al*. 2017).

Salt marshes are relatively narrow habitats that, in many places, became narrower due to the development of dikes, roads, and other civil development. Indeed, in our study area, in addition to road construction at the edge of salt marshes, extensive salt marsh diking was initiated in the mid-19th century to expand arable land (Hatvany 2002). In all of these locations, salt marshes are threatened by sea-level rise and storm-induced surges because, where landward migration becomes impossible, a coastal squeeze can occur (Torio and Chmura 2013; Watson *et al*. 2017; Mitchell and Bilkovic 2019). The upper marsh is especially at risk as it is expected to decline faster than the lower marsh (Valiela *et al*. 2018).

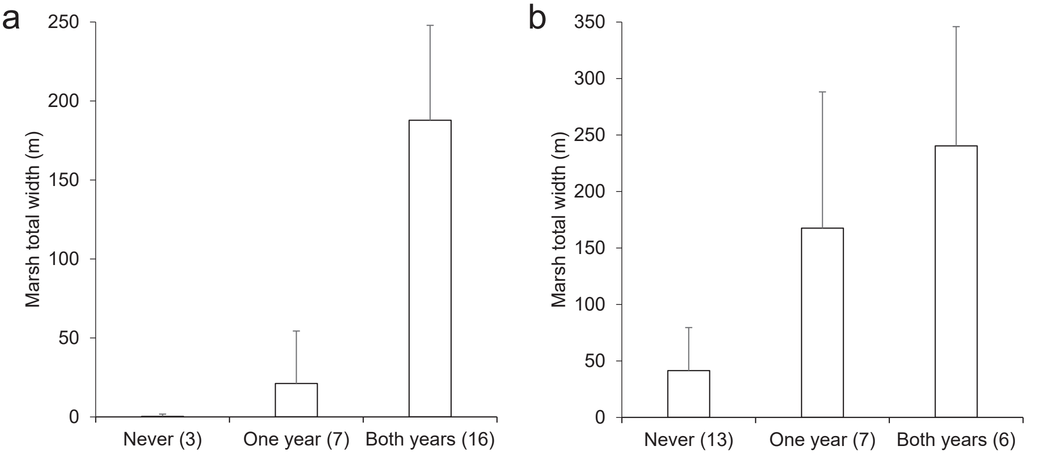

FIGURE 3. Mean salt marsh total width (+ 95% CI) in survey sites where a. Least (*Calidris minutilla*) and b. Pectoral (*Calidris melanotos*) Sandpipers were detected during both years of the study, detected one year only, or never detected on the south shore of the St. Lawrence River Estuary, Quebec, Canada, during fall migration.

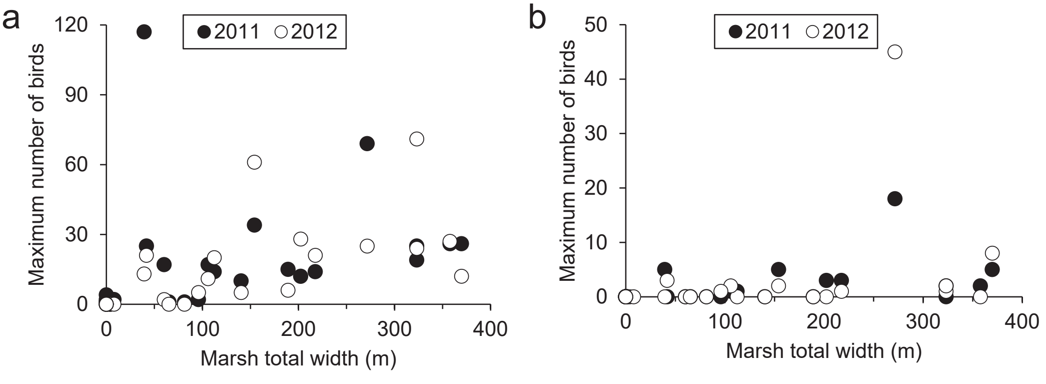

FIGURE 4. Relationship between the maximum number of a. Least (*Calidris minutilla*) and b. Pectoral (*Calidris melanotos*) Sandpipers detected per survey site per year, and salt marsh total width on the south shore of the St. Lawrence River Estuary, Quebec, Canada, during fall migration.

Degradation of stopover habitat has been linked to declines in migrating shorebird populations (Studds *et al*. 2017). However, effective management of shorelines such as realignment (e.g., moving of a coastal defense line inland to allow the re-inundation and development of an intertidal habitat [Shepard *et al*. 2011]) or climate-resilient, living shoreline design (e.g., use of stabilizing structures to protect the shoreline and enhance marsh establishment [Mitchell and Bilkovik 2019]) could increase salt marsh resilience through wave attenuation and higher accretion rate (Möller *et al*. 2014; Zedler 2017; Cahoon *et al*. 2019). We need such climate-resilient adaptation strategies. It is especially true where, due to glacial isostatic rebound (e.g., Magdalen Islands in the Gulf of St. Lawrence [Rémillard *et al*. 2016]) or compaction of Holocene strata (e.g., Mississippi Delta [Törnqvist *et al*. 2008]), subsidence exacerbates the effects of sea-level rise (Koohzare *et al*. 2008; Kirwan and Megonigal 2013; Piecuch *et al*. 2018). In areas such as the St. Lawrence River Estuary, where landward salt marsh migration is no longer possible, effective management is needed. Otherwise, salt marshes and associated salt marsh shorebirds could be locally jeopardized.

Author Contributions

Conceptualization: Y.T.; Methodology: Y.T.; Investigation: Y.T. and J.-F.L.; Data Curation: Y.T.; Formal Analysis: Y.T.; Writing – Original Draft: Y.T.; Visualization: Y.T. and É.D.; Writing – Review & Editing: Y.T., J.-F.L., É.D., and J.B.; Funding Acquisition: Y.T. and J.B.

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