Urban Ecosystems, 7: 107–124, 2004

Restoration of urban salt marshes: Lessons from southern California

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Abstract. Extensive restoration efforts in southern California coastal wetlands highlight several challenges for urban salt marsh restoration, including: habitat isolation and fragmentation, impacts from exotic species, the loss of transitional upland habitats, and other alterations to hydrologic and sediment dynamics. Habitat isolation impairs colonization by dispersal-limited plants, so planting becomes essential to achieve diverse salt marshes. Low species richness slows the development of salt marsh functions (e.g., biomass and nitrogen accumulation) in southern California. A variety of exotic species have invaded the upper reaches of salt marshes in southern California, most commonly in marshes with hydrological modifications. The replacement of gradual slopes between wetlands and uplands by sharp transitions abutting urban development limits our ability to restore rare plant and animal populations. Where hydrologic connections are impaired by roads and other structures, the natural migration of channels is constrained, and sediment dynamics often lead to lagoon mouth closure.

A case study from Tijuana River National Estuarine Research Reserve (hereafter, Tijuana Estuary) further illustrates a specific lesson for urban salt marsh restoration concerning watershed issues and sediment dynamics. In the south arm of Tijuana Estuary, watershed urbanization, along with local climate, topography, and soils, has resulted in extreme rates of sediment accumulation. Sedimentation rates in a salt marsh in the south arm of the estuary ranged from 10 to 30 cm over a single winter (1994–95), substantially greater than historic sedimentation rates in the estuary or rates from other coastal wetlands with storm sedimentation. Sediment buried salt marsh vegetation in place and is converting intertidal salt marsh to uplands. These impacts illustrate the need to consider watershed issues and sediment control in managing and restoring urban salt marshes.

Keywords: sediment accretion, watershed, salt marsh, *Salicornia virginica*, restoration

Introduction

Around the world, urban populations are growing at rapid rates. The United Nations projects that global population will grow to 7.8 billion by 2025, and 90% of this increase will occur in urban areas of developing countries (Brockerhoff, 2000; United Nations, 2000). Within the United States, coastal counties within 80 km of the coast make up only 13% of the land area of the continental US, but this area encompasses 51% of the US population and 57% of civilian income, as of 2000 (Rappaport and Sachs, 2003). The pressures of increasing human populations and changing land uses in urban coastal regions ensure that there will be on-going impacts to urban coastal wetlands, especially salt marshes. Coastal areas will also bear the brunt of climate warming, as sea levels rise and inundate the narrow strip of land between the ocean and urban development. An ever-decreasing area of coastal wetland is anticipated, along with the ever-decreasing quality of the remaining habitat. Along with increased impacts, we anticipate growing interest in the restoration of coastal wetlands to offset further degradation of habitat.

The restoration and management of salt marshes in existing urban locations and in newly urbanizing regions can benefit from lessons learned from locations that already have significant urban impacts and substantial restoration efforts. In southern California, urbanization to support ~20 million people has pushed the limits of the landscape to provide both housing and open space. While many coastal wetlands have been protected as nature reserves, their public status does not exempt them from highway widening or installation of other features, such as sewer pumping stations—the former because major highways were built in the low, flat lands, and the latter because urban wastewater is easiest to collect in the lowest part of the landscape. Nor has the establishment of a National Estuarine Research Reserve at Tijuana Estuary prevented tons of sediment from flowing into the Tijuana Estuary, which has the additional problem of being located downstream of a large Mexican watershed, where erosion is beyond US control. Repeated sedimentation events have led to sediment removal in order to restore salt marshes for endangered species, but continued sedimentation also slows restoration efforts.

Given the history of urban impacts and the interest in restoring local wetlands, experiences gained in restoring southern California urban wetlands could benefit habitat restoration efforts elsewhere. The following review of opportunities and challenges for urban salt marsh restoration focuses on the lessons that can be learned specifically from Tijuana Estuary and other southern California salt marshes. Our case study highlights the challenge of managing sediment inputs from an urbanized watershed, as well as the impacts of sedimentation on vegetation in the south arm of Tijuana Estuary.

Opportunities for urban salt marsh restoration

Urban wetlands offer opportunities to capitalize on the social benefits of restored urban ecosystems (Ehrenfeld, 2000). For example, in San Diego Bay, close to 90% of wetland acreage has been impacted (MacDonald, 1990). In urban areas with only a small amount of remaining wetland habitat, each restoration project provides a substantial increase in area. Efforts are underway to evaluate a series of alternatives for the restoration and management of over 400 ha of salt ponds in south San Diego Bay (see http://sandiegorefuges.fws.gov/new/ccp/CCP.htm). An even larger-scale effort to restore more than 6000 ha of salt ponds is also being initiated in south San Francisco Bay (see http://www.southbayrestoration.org/ or http://www.coastalconservancy.ca.gov/). The expansion of the area of tidal influence at Ballona Wetland, immediately north of the Los Angeles Airport could double the area of salt marsh in the entire county. The restoration of urban wetland habitats reduces habitat loss rates and can result in large net gains in many cases.

The restoration of urban wetlands also affords outstanding opportunities to provide educational benefits and improve the public's understanding and concern for natural areas (Ehrenfeld, 2000). Because of their location in the midst of large population centers, urban wetlands are likely to receive more visitors than remote wetlands. Cities offer large audiences for education and public involvement. Urban residents are often eager to become involved in restoration activities, especially if the wetland is near home, and they see it often enough to track the results of their efforts. An excellent example is Famosa Slough in San Diego, which attracted an active "Friends" group that worked for over a decade to prevent the land from being developed and eventually convinced the City to buy the 5-ha site. Once the land was acquired for public enjoyment, the Friends, under the leadership of Jim and Barbara Peugh (local residents) worked to eradicate exotic plants, remove trash, and replant native vegetation. Next, they helped develop grant proposals to obtain funding to reduce the impacts of stormwater inflows. The Friends produce a newsletter and hold frequent bird walks and clean-up events, as well as native-species planting efforts. It is unlikely that any of this work would have been accomplished if left up to City staff (J. Zedler, pers. obs.). Casagrande (1997) identified a number of social benefits of involving the public in restoration projects and recommended that the public be involved in all phases of projects from planning to implementation and monitoring in order to maximize social benefits.

Challenges for urban salt marsh restoration

Habitat isolation. Increasing degrees of urbanization involving roads and other development cause habitat fragmentation (Forman and Alexander, 1998). Isolation and fragmentation lead to several concerns for restoration. First, urban restoration sites are highly likely to be far from natural wetlands that could supply propagules of plants and animals. Organisms that are dispersal limited (those whose propagules are not easily spread over long distances) are unlikely to reach small, isolated sites. Even if propagules arrive, the species might not establish, due to the low probability that environmental conditions would be suitable for growth while the propagules are viable. Thus, the fragmentation tends to reduce diversity and/or limit the species present to those that are weedy and widely dispersed. Dispersal-limited species will need to be planted or introduced at newly restored sites (Lindig-Cisneros and Zedler, 2002). Fragmented habitats might also constrain genetic movement between populations, increasing the likelihood of extirpation of local populations and leading to a reduction in regional biodiversity, although Helenurm and Parsons (1997) did not find evidence of this for an endangered plant reintroduced to a salt marsh along San Diego Bay.

A reduction in species diversity is obviously important from the perspective of biodiversity conservation; recent research shows that low species richness can also cause a reduction in ecosystem function. At Tijuana Estuary, we evaluated the link between plant species diversity and three functions of a restored salt marsh. Increased plant species diversity led to a more complex canopy (Keer and Zedler, 2002), reduced recruitment/invasion (Lindig-Cisneros and Zedler, 2002), greater primary productivity and a higher rate of nitrogen accumulation (Zedler *et al.*, 2001; Callaway *et al.*, 2003).

Impacts from exotic species. Urban salt marshes are particularly vulnerable to invasion by exotic species because of on-going anthropogenic disturbances, a high level of habitat fragmentation, and abundant propagules. San Diego's most widely publicized episode of exotic species invasion concerned New Zealand's white mangrove (*Avicennia marina* (Forsk.) Vierh.), which was deliberately planted in Mission Bay Marsh by a physiologist who wanted to grow his own plant material for study. The population grew rapidly and wildlife biologists feared that the 2-m tall mangroves would attract raptors, which would then feed on the chicks of an endangered clapper rail. Once alarmed, they mounted an annual effort to remove all individuals. This effort took several years, because the smallest seedlings were hard to see within the evergreen salt marsh canopy, but the population was eventually eradicated (J. Zedler, pers. obs.).

The number of exotic species propagules is likely to be high in urban areas because of high levels of trade and travel, and international ports have extremely high levels of occurrences for introduced species (Nichols *et al.*, 1986; Ruiz *et al.*, 2000). Newly restored salt marshes are particularly susceptible to exotic species invasions when their plant cover is still low and large bare areas entrain wrack that contains seeds from exotic species (Callaway, in press).

While invasive *Spartina* species in the low marsh have been the focus of concern in many areas (Gray *et al.*, 1991; Adam, 2002), invasive exotic plant species are also problematic in the high marsh/upland transition zone in many mediterranean salt marshes. Annual grasses are especially common invaders of southern California high marsh habitats. Through most of the year, arid conditions predominate, and soils are saline, but during the rainy season, salts are leached from the surface soil, seed banks germinate, and life cycles are completed within a few weeks or months. Noe and Zedler (2001) found that the high marsh seed bank was 99% exotics in one area of Tijuana Estuary where an endangered annual plant has persisted in small numbers. The common exotic species in southern California are weedy freshwater and brackish marsh species, as well as facultative wetland species, such as *Polypogon monspeliensis* (L.) Desf. and *Parapholis incurva* (L.) C.E. Hubb. (Kuhn and Zedler, 1997; Callaway and Zedler, 1998; Noe and Zedler, 2001; Noe, 2002). These species are most abundant in estuaries with hydrological modifications, i.e., reduced tidal inflow and excessive freshwater input from the use of imported water for urban or agricultural uses. Zedler *et al.* (1990) projected a similar pattern for the invasive *Typha orientalis* C. Presl. in southwestern Australia. Stormwater runoff from streets lowered the salinity, and the drains eroded channels through the native *Juncus kraussii* Hochst. marsh, opening the canopy so that seedlings could establish. Once established, *Typha* readily expanded vegetatively (J. Zedler, pers. obs.).

Loss of transitional upland habitats. Of all the habitats lost in urban wetlands, the transition from salt marsh to upland is the most diminished and modified. These transitional areas are especially vulnerable to development, because they are not necessarily recognizable as wetlands under the U.S. Clean Water Act (Ryan, 2003). Transitional habitats are important for those species that use upland areas for some part of their lifecycle or that move across upland wetland boundaries for other reasons, e.g., foraging, pollination, etc. Parsons and Zedler (1997) showed that a rare plant in southern California (*Cordylanthus maritimus* subsp. *palustris* (Behr) T.I. Chuang & Heckard) is pollinated by solitary bees that nest in the ground above the tide line; without a transition zone, pollination could be limited by the availability of appropriate pollinators. James and Zedler (2000) studied a rare shrub that characterizes the southern California transition to upland and noted that the spiny plant (*Lycium californicum* Torr. & A. Gray) served many wildlife functions, including providing bird perches, animal cover, nesting habitat, and small-rodent runways. More research is need on these transitional upland habitats in order to identify species that use these habitats, to understand the functions that these habitats provides, and to evaluate the amount and type of transitional habitats that might be most appropriate for restoration projects. Further consideration is also needed to identify how these areas might be protected.

Alterations to hydrologic and sediment dynamics. Finally, urbanization can substantially affect hydrological conditions, including sediment transport and shifts in seasonal or annual hydrologic regimes. The effect of hydrological impacts varies by region, with substantially different impacts in wet vs. arid regions. In arid areas, inputs of freshwater and sediment are likely to be substantially increased, leading to reductions in salinity and increases in rates of estuarine sedimentation. Greer and Stowe (2003) documented substantial changes in the Los Peñasquitos watershed (San Diego County) with increased freshwater input being a result of watershed urbanization. In addition, urbanization within the watershed can increase the amount of available sediment, through soil disturbances that are associated with development. Within wetter regions, the annual change in hydrological conditions may not be so substantial; however, there may be significant seasonal shifts in the hydrograph, as well as changes in available sediment.

Restoration becomes a major challenge where urbanization has imposed hydrological constraints on a coastal wetland. For example, many estuaries in southern California are dissected by roads, railroads, and other impacts that restrict the flow of water within the estuary. Restrictions at the mouth of the estuary often impair tidal flow, increase sedimentation, and set up a feedback system that reduces the tidal prism and further reduces tidal flow. This positive feedback loop reaches its maximum influence with closure of the tidal inlet (e.g., Los Peñasquitos Lagoon, and other southern California coastal wetlands). Water salinities, temperatures, and oxygen concentrations then undergo amplified variations, and the extremes cause fish and invertebrate kills (Zedler, 2001).

A case of extreme sedimentation at Tijuana Estuary

The lessons from southern California highlight the range of challenges that exist for the restoration of urban salt marshes. As an illustration of the specific challenges of watershed urbanization and sediment impacts on coastal salt marshes, we report on a short-term, catastrophic sedimentation event at Tijuana Estuary. Field observations of up to 30 cm of sediment deposition along a salt marsh creek led us to test the effects of low-to-high sedimentation rates on extant salt marsh vegetation dominated by *Salicornia virginica* L. In particular, we asked: (1) How much sediment was deposited in the 1994–95 rainfall year? (2) How much sediment could the salt marsh vegetation survive? (3) Could the vegetation recover completely from the sedimentation event? (4) What other impacts might interfere with recovery?

Methods

Study site

Tijuana Estuary (figure 1) is located in the extreme southwest corner of the continental United States. Approximately 75% of the estuary's 4,400 km^2 watershed is in Mexico, with the remaining area in the U.S. (Zedler *et al.*, 1992). The watershed consists primarily of natural and agricultural lands; however, a significant part of the immediate area adjacent to the estuary is impacted by urban land use, both from the city of Tijuana in Mexico, as well as Imperial Beach in the U.S. The urban area within the watershed grew from 105 km^2 in the early 1970s to 308 km^2 in 1994 (Ojeda, 2001). The south arm of the estuary is directly impacted by a number of small, local sub-watersheds with steep drainages, sandy and easily erodible soils, and a high degree of urbanization. The local sub-watershed that directly impacts the study site for this research is Goat Canyon, or Cañon de los Laureles. The sub-watershed is approximately 12 km^2, and 90% of this area is within Mexico, with extensive recent development ranging from squatter settlements to large residential projects.

Much of the south arm of Tijuana Estuary (figure 1) was historically tidal salt marsh, but substantial wetlands in this area have been converted to uplands, in part due to sedimentation impacts over the last century (Williams and Swanson, 1987; Zedler *et al.*, 1992). Former wetlands were used by the military during World War II, and some areas were farmed during wet years. The entire estuary is now managed as a National Estuarine Research Reserve, with land uses ranging from research and endangered species management to

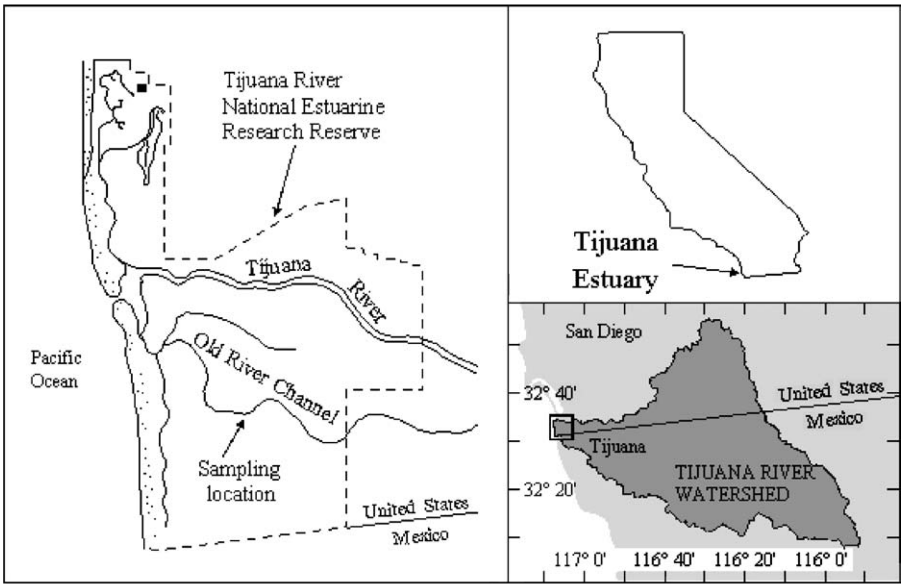

Figure 1. Tijuana Estuary, including the old river channel and the sampling location in the south arm of the Estuary.

horseback riding, passive recreation, and border patrolling. Trespassing was very common until the late 1990s, due to immigrant travel across the adjacent international border. The "old river channel" is a minor barrier to foot traffic moving north from Mexico. Based on historical maps and aerial photographs, the Tijuana River moved north sometime between 1904 and 1928, leaving a broad channel that has accreted sediments that supported a tidal salt marsh plain (Zedler *et al.*, 1992). Tidal influence is minor due to restrictions on the channel closer to the estuary mouth. Tidal amplitude at the study site is generally less than 0.25 m. The vegetation adjacent to the channel includes *S. virginica*, *Frankenia salina* (Molina) I.M. Johnst., *Limonium californicum* (Boiss.) A. Heller, *Juncus acutus* L., and *Salicornia subterminalis* Parish.

During severe storms, sediments flow down Goat Canyon and other nearby canyons. The flow often is captured by local dirt roads and connects up with the old river channel approximately 50 m east of our sampling location. Prior to winter 1994–95 the area adjacent to the old river channel had dense cover of *S. virginica* and other halophytes (J. Zedler, pers. obs.). Immediately after these winter storms (March, 1995), the marsh area was blanketed in sandy sediment, with none of the vegetation protruding (P. Fong, UCLA, pers. comm.).

Depth of sedimentation

To document sediment deposition, we selected a 75-m-long area parallel to the old river channel and starting at the source of sediment input into the old river channel. We sampled along the 75-m transect using a stratified random design, dividing the transect into upstream (0–25 m), mid (26–50 m), and downstream (51–75 m) sections, and we collected 24 cores in June 1995 to measure the depth of newly deposited sediment. Four pairs of cores were taken from each section from (a) randomly located areas where *S. virginica* had re-established over at least 1 m^2(vegetated) and (b) the closest bare areas of at least 1 m^2 where *S. virginica* had not re-established as of June 1995 (unvegetated). Paired cores were within 2 m of each other in all cases. Core locations were paired in order to compare the depth of sedimentation in areas where *S. virginica* was able to recover within the first year of sampling to those areas where recovery had not occurred.

Cores were collected with a 10-cm diameter coring tube with a razor-sharp cutting head (Hargis and Twilley, 1994). Samples were extruded in the field. There were obvious differences between the storm-deposited sediment (light colored, sandy) and pre-storm sediment (dark, fine texture), and the transition in sediment characteristics was always very sharp. Hence, we were able to measure the depth of the 1995 sediments in the field (to the nearest 1 cm).

Sediment characteristics by depth

In order to quantify differences in storm-deposited versus pre-storm sediments, we kept a subset of the cores for laboratory analyses (a pair of cores from randomly selected vegetated and unvegetated areas in each of the 25-m sections). These sediment cores were sectioned into 10-cm increments, except at the "transition depths," (between storm-deposited and non-storm-deposited sediments) around which we used 5-cm increments in order to evaluate the transition in sediment characteristics more precisely. We analyzed each section for sediment texture using the hydrometer method (Gee and Bauder, 1986); salinity of the porewater by extraction and measurement with a temperature-compensated refractometer; and the amount of buried plant material by rinsing the cores over a 0.45-mm screen, collecting aboveground parts of *S. virginica*, oven drying at 80°C, and weighing the biomass. Depth profiles were prepared for each sediment character from each core.

Vegetation recovery

Vegetation recovery was measured using the 75-m transect that was parallel to the channel as well as 15 transects that were perpendicular to the channel (one parallel transect at a randomly chosen intersection within each 5-m interval of the 75-m transect). Vegetation transects were marked permanently and measured in June 1995, June 1996, and July 1997. We sampled vegetation with the line-intercept method, recording cover in 10-cm increments. Perpendicular transects ran from the edge of the creek to the base of the old river terrace. In addition to assessing cover, we measured the tallest plant per each meter increment for the perpendicular transects. Data from the parallel and perpendicular transects were divided into the same three 25-m sections as in the sediment study.

Extent of trampling

We resampled the area in June 1996, after noting extensive human disturbance, evidenced by abundant footprints and plants that had been trampled into the sediment. In order to quantify the extent of trampling on the June 1996 sampling date, we recorded the presence of footprints within a 0.25-m band adjacent to the transect tape at 1-m intervals along the 75-m-long parallel transect.

Soil salinity

In addition to sampling soil salinity in the cores collected in 1995, we measured soil salinity at a subsample of the intersections of the perpendicular transects and the 75-m transect in July 1997. Soil was collected from 0–10-cm depth, and interstitial water (at field levels) was expressed onto a temperature-compensated refractometer.

Data analysis

Differences in the depth of sedimentation between areas with recovery of *S. virginica* and those without recovery were evaluated using analysis of co-variance (ANCOVA). Regression analysis was used to evaluate trends along the 75-m transect for both the depth of sedimentation and vegetation recovery. Differences in vegetation recovery along the transect were also evaluated by comparing recovery in the three sections of the transect with one-way ANOVA.

Results

Depth of sedimentation

The greatest depth of newly deposited sediment was 30 cm at a sample point that was unvegetated in June 1995. Where *S. virginica* was present, the maximum depth of new sediment was 24.5 cm. All other vegetated sampling points had less than 20 cm of new sediment (figure 2(a)). As expected, the depth of newly deposited sediments was greatest at the upstream portion of the channel (closest to the source of sediment input). Sediment deposition declined significantly with distance along the transect for both vegetated (sediment depth

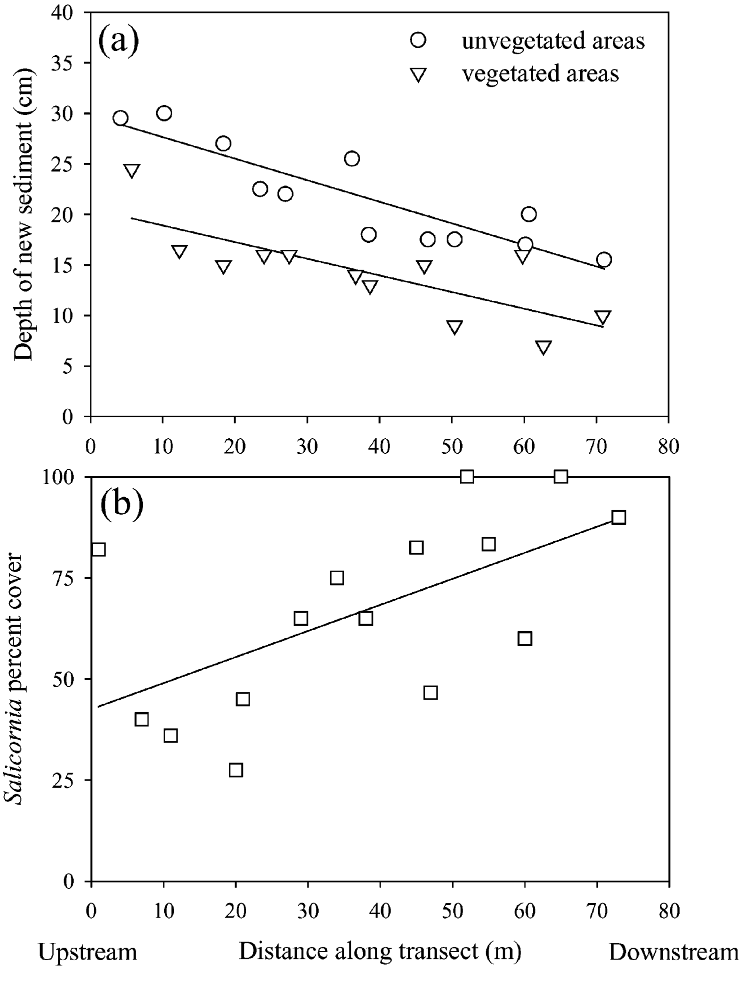

Figure 2. (a) Depth of sediment deposited during the winter of 1994–1995 along 75-m transect parallel to the old river channel in the south of Tijuana Estuary. (b) Recovery of *Salicornia virginica* along the same transect in 1995. Lines indicate linear regressions fitted to the data; regression equations are given in the text.

(cm) = 20.5 − 0.16 ∗ distance (m), $r^2 = 0.59$, $p = 0.004$) and unvegetated areas (sediment depth = 29.7 − 0.21 ∗ distance, $r^2 = 0.78$, $p < 0.001$; figure 2(a)). By the time of our first sampling in June 1995, *S. virginica* had recovered from the sediment deposition that had occurred in the winter of 1994–95, and areas of *S. virginica* recovery had significantly less sediment than areas without recovery (proc. GLM; $p < 0.001$).

Sediment characteristics by depth

Surface sediments deposited by storm activity were uniformly sandy (averaging 55 percent sand, and there was a sharp transition in characteristics below the sediments deposited in winter 1994–95 (figure 3). Below the storm-deposited material, sediments were predominantly clay and silt, with only 18% sand on average. Soil salinities were highest in the surface sandy sediments, averaging 81 ppt, dropping to 31 ppt in the finer, pre-storm sediments. The accumulation of salts in surface sediments is typical in southern California salt marsh soils, but evaporation and concentration were likely accelerated by the exposed surface (little shading from vegetation) and coarse texture.

Plant biomass in the sediment cores was highest at the transition from pre-storm to post-storm-deposits (figure 4). Maximum biomass in the cores was 15–25 mg/cm^3, and inspection of the plant material indicated that it was aerial stem tissue, rather than rhizomes or roots. Thus, aboveground plant material had been buried in place, where it remained in the sediment. The aboveground biomass of *S. virginica* was buried to greater depths in unvegetated

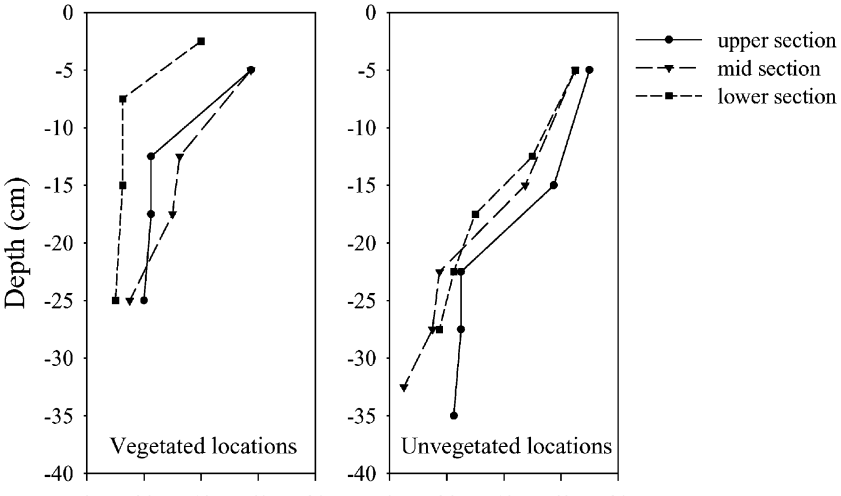

Figure 3. Depth profiles of sediment sand content from cores along the upper-, mid-, and lower-sections of the 75-m transect in the south of Tijuana Estuary. Cores were collected in areas where vegetation had recovered and where no recovery had occurred in 1995. Sharp transition in sand content indicates the change from historic marsh sediments below to storm-related deposited sandy material above.

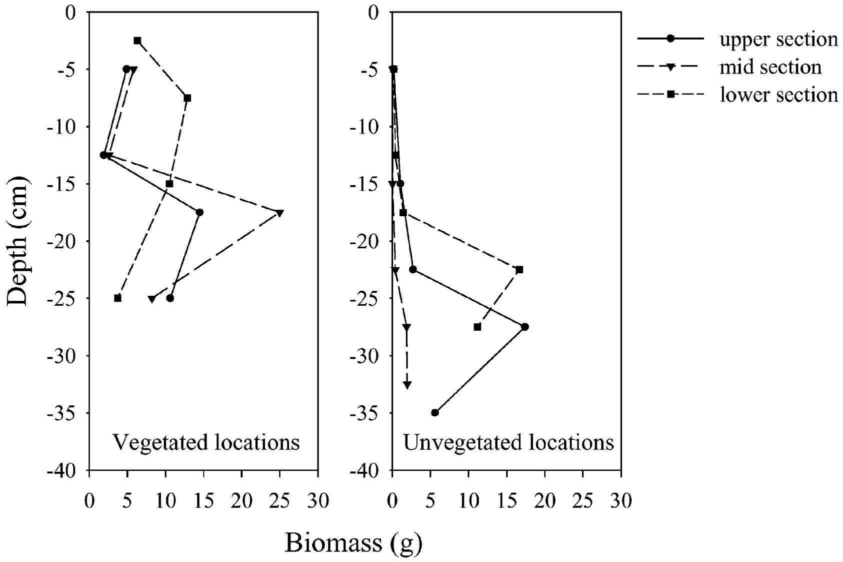

Figure 4. Depth profiles of belowground biomass from cores along the upper-, mid-, and lower-sections of the 75-m transect in the south of Tijuana Estuary. Cores were collected in areas where vegetation had recovered and where no recovery had occurred in 1995. Peaks in biomass indicate aboveground biomass that was buried in place by storm sedimentation.

areas; however, there was no difference in the cumulative amount of biomass in vegetated vs. unvegetated cores ($p = 0.15$) confirming that vegetation was present throughout the area and apparently buried too deeply in some places for the plants to recover.

Vegetation recovery and subsequent trampling impacts

In June 1995, recovery was lowest in the upstream section of the parallel transect (17% cover; figure 5(a)) where the greatest amount of sediment was deposited (figure 2(b)). Plant cover on the parallel transect increased towards the mouth of the channel, with 47% cover in the mid section and 78% cover downstream (figure 5(a); ANOVA, $p < 0.001$). This trend was also evident in the cover along individual perpendicular transects, which increased significantly with distance along the transect (cover $= 42.6 + 0.65 *$ distance (m), $r^2 = 0.37$, $p = 0.017$; without the 1st point outlier: cover $= 28.7 + 0.92 *$ distance (m), $r^2 = 0.62$, $p = 0.001$). Plant cover was slightly higher in the downstream section, with 88% cover in 1996 and 91% cover in 1997. However, in the mid and upstream sections, vegetation cover dropped in 1996 and 1997, likely due to trampling.

Perpendicular transects showed trends similar to the parallel transect in 1995, with the lowest recovery where sediment deposition was greatest (figure 5(b)). Changes in cover along the perpendicular transects were minimal from 1995 to 1997. Recovery of *S. virginica* in the mid and upstream sections was greatest adjacent to the channel and near the old river terrace. The low recovery in the center of the marsh plain coincided with the area of greatest evidence of trampling. The perpendicular transects encompassed these different

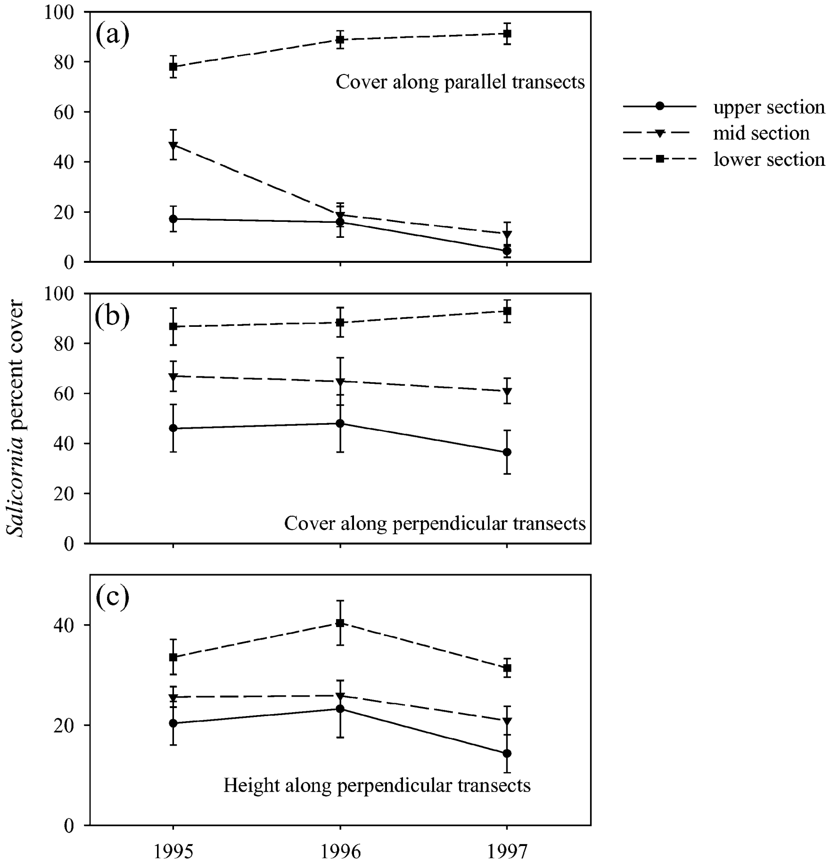

Figure 5. Vegetation recovery in the summers of 1995, 1996, 1997 along the upper-, mid-, and lower-sections of the 75-m transect in the south of Tijuana Estuary, including: (a) plant cover on parallel transects, (b) plant cover on perpendicular transects, and (c) plant height along perpendicular transects.

effects; whereas the parallel transect was mostly bare, as it coincided with places where trampling evidence was greatest.

Mean canopy height was greatest in the perpendicular transects of the downstream section, averaging 34 cm. Heights were significantly lower in the mid (26 cm) and upstream sections (20 cm) of the transect (figure 5(c)). The trends among sections were similar in 1996 and 1997, with slight increases in height in 1996, but decreases in 1997. Decreases in 1997 were likely due to either below-average rainfall during that year or to trampling damage.

Trampling evidence was greatest in the upstream section, where footprints were present in 100% of the 0.25-m-wide belt transect. In the mid and downstream sections, footprints were found in 92 and 32% of the belt transect length, respectively. Trampling was most evident in the middle of the marsh plain. Evidence of trampling was only evaluated on one date and does not account for any potential temporal variation in trampling impacts.

Soil salinity

Surface soil salinities in June 1995 ranged from 50–100 ppt. In June 1997, soil salinities were uniformly high along the parallel transect (mostly over 100 ppt). Any seed germination would be highly unlikely at this range of salinities (Kuhn and Zedler, 1997; Callaway and Zedler, 1998), and all recovery likely was due to resprouting of *S. virginica* rather than the establishment of new seedlings.

Discussion

Sedimentation impacts in urban-impacted salt marshes

The magnitude of sediment accumulation in the salt marsh in the south arm of Tijuana Estuary due to the 1994–95 winter storms was large, with 10–30 cm of sediment accumulating over this period. While this level of sediment accumulation was localized to an area adjacent to the old river channel, it is at least an order of magnitude greater than the longer-term rates of sediment accumulation that have been measured in Tijuana Estuary. Weis *et al.* (2001) measured 0.71 to 0.94 cm/yr of sediment accumulation in *Salicornia*-dominated areas since 1963 based on the ^{137}Cs peak. Their estimates for the *Spartina*-dominated areas ranged from 1.06 to $\geq$1.23 cm/yr. Higher rates are to be expected in the low marsh due to longer periods of tidal inundation (Letzsch and Frey, 1980; Pethick, 1981; Bricker-Urso *et al.*, 1989; Callaway *et al.*, 1997). The only other estimate of long-term sediment accumulation in southern California salt marshes was 0.5 cm/yr over the last century and 0.1 cm/yr during pre-European settlement, based on pollen analysis for cores from Mission Bay and Los Peñasquitos Lagoon (Mudie and Byrne, 1980). Accretion rates based on ^{137}Cs in Tijuana Estuary are at the high end of long-term rates compared to other estuaries, where accretion rates are typically 1–5 mm/yr (Stevenson *et al.*, 1986), except in areas of high subsidence where rates of 1 cm/yr or greater are common (DeLaune *et al.*, 1978; Hatton *et al.*, 1983; Stevenson *et al.*, 1986; Nyman *et al.*, 1993). The high rates of sediment accretion in Tijuana Estuary are not caused by subsidence but are likely due to storm impacts that occur periodically and drive up the average long-term rates.

The rates of storm-related sediment accumulation in the south arm of Tijuana Estuary are also greater than or equal to recently measured rates caused by storm events in the north arm of the estuary (Table 1), especially considering that this sediment accumulation occurred on the marsh plain (dominated by *Salicornia virginica*); whereas, most other measurements come from the low marsh zone (dominated by *Spartina* spp.), where inputs of sediment are likely to be greater. Cahoon *et al.* (1996) measured 1.9 to 8.5 cm of sediment accretion in *Spartina*-dominated areas during storms in 1992–93, but less than 0.2 cm accumulated in *Salicornia*-dominated areas. Ward *et al.* (2003) measured 4–12.7 cm/yr in *Spartina*-dominated areas and mudflats in the north arm of Tijuana Estuary due to storms in the winter of 1997–1998. While local areas adjacent to both the north and south arm are urbanized, it is likely that the increased rates of sediment accumulation in the south arm are due to the steep topography within the watersheds immediately adjacent to the south arm. The sediments that accumulated on the marsh plain in the south arm of the estuary

Table 1. Vertical sediment accretion rates in coastal salt marshes and other wetlands caused by storm-related events.

Location	Year	Event	Vertical sediment accretion (cm/event or storm season)	Reference
Tijuana Estuary (north arm), CA	1992–93	Winter storms	1.9–8.5	Cahoon *et al.*, 1996
Tijuana Estuary (north arm), CA	1997–98	Winter storms	4–12.7	Ward *et al.*, 2003
Tijuana Estuary (south arm), CA	1995–96	Winter storms	10–30	This study
Barataria Bay, LA	1975–79	Hurricane Bob and other storms	0.9–1.5 cm/yr[a]	Baumann, 1984
Atchafalaya Bay, LA	1985	Hurricane Danny	<0.2–2.9	Rejmánek *et al.*, 1988
Terrebonne Bay, LA	1986–87	Winter storms	na[b]	Reed, 1989
Terrebonne Bay, LA	1992	Hurricane Andrew	3–9	Nyman *et al.*, 1995
Florida	1993	Storm of the Century	<1–2[c]	Goodbred and Hines, 1995
Nauset Marsh, MA	1991–93	Multiple storms	<1–2.4 cm/yr[a]	Roman *et al.*, 1997

[a] Measured over multi-year interval with storm events.

[b] Measured mass-based sedimentation over series of small storm, rather than vertical accretion.

[c] Up to 12 cm of sediment deposited on river levees.

in 1994–95 were very coarse (figure 3) and likely came directly from erosion of urbanized hillsides in Goat Canyon; whereas, it is more likely that sediment that has accumulated in the North arm is a mix of reworked estuarine sediments and inputs from the main stem of the Tijuana River. The sediments that accumulated in the north arm during storm events in 1992–93 and 1997–98 were relatively fine and not easily distinguished from other estuarine sediments (Cahoon *et al.*, 1996; Ward *et al.*, 2003). Cahoon *et al.* (1996) noted that some coarse material was deposited on the marsh surface, due to 1983 or 1988 storms; however, this was likely dune overwash, not sediment directly from the watershed.

Not only are the rates of sediment accumulation from the south arm of Tijuana Estuary greater than those from other storm events in Tijuana Estuary, but they are also greater than studies of storm sediment impacts from other estuarine systems around the world (Table 1). Measured rates of sediment accretion associated with hurricanes and other storm events in Louisiana, Florida and Massachusetts generally have ranged from 1–3 cm, with a maximum of 9 cm of sediment accumulation associated with Hurricane Andrew in Louisiana (Table 1). It is difficult to directly explain differences in storm sedimentation rates across these wetland areas because of large differences in many parameters; however, it is likely that climate, topography, soil conditions, watershed size, and the degree of adjacent urbanization all contribute to storm sediment impacts.

Furthermore, the impacts of storm sedimentation in the south arm of Tijuana Estuary have continued to date with massive sedimentation occurring through 2003, including a substantial increase in the elevation of the entire marsh plain around the old river channel and further reduction in the tidal amplitude. Roads just upstream of the marsh area have been buried under more than 50 cm of sediment on numerous occasions over the last decade. Efforts are currently underway to reduce these impacts through both watershed management on the Mexican side of the watershed, and through the construction of detention ponds on the U.S. side of the border (Jim King, California State Coastal Conservancy, pers. comm.).

The levels of sediment accumulation and the resulting shifts in elevation that were caused by storms in 1994–95 and earlier events were enough to cause impacts to vegetation in the south arm of Tijuana Estuary. While, *S. virginica* was able to recover in some areas from this impact, recovery was poor in other areas (in part due to secondary impacts from trampling and accumulation of salts). If storm inputs continue into the future, additional salt marsh areas in the south arm of the estuary will increase in elevation, with the eventual conversion to upland vegetation. In fact, much of the historical salt marsh in the south arm of the estuary already has been converted to upland vegetation (Zedler *et al.*, 1992). This shift is unusual for salt marshes for two reasons: the rapid nature of the change in elevation and the conversion of wetland to upland conditions. While storm sedimentation is an important component of marsh development in many areas (Chmura and Kosters, 1994; Goodbred and Hine, 1995; Leonard *et al.*, 1995; Nyman *et al.*, 1995; Roman *et al.*, 1997; Courtemanche *et al.*, 1999; Hensel *et al.*, 1999), salt marshes are typically affected by very gradual shifts in elevation and vegetation (Redfield, 1972; Mitsch and Gosselink, 2000). The shift to upland vegetation is further enhanced by direct input of storm-related sediment from the watershed, rather than by tidally-driven sedimentation. There is a strong negative feedback loop between elevation and tidal sediment inputs: as elevation increases, the period of tidal inundation and hence the delivery of tidal sediment decreases (Letzsch and Frey, 1980; Pethick, 1981; Bricker-Urso *et al.*, 1989; Callaway *et al.*, 1997). However, non-tidal, storm-related sediments from the watershed could have a much greater impact in building elevations out of the tidal range because they would not be directly affected by this negative feedback with tidal range.

While the Tijuana Estuary case might seem unusual, other salt marsh systems in urbanized settings have had similar kinds of impacts. Greer and Stowe (2003) documented long-term shifts in vegetation in nearby Los Peñasquitos Lagoon. In this case the vegetation converted from salt marsh to brackish and freshwater marsh and was caused by changes in both salinity and elevation as a result of urbanization in the watershed (Greer and Stow, 2003). As the Lagoon filled with watershed sediments, the tidal prism was substantially reduced, leading to on-going problems with maintaining the tidal inlet. Following several years of continuous closure in the 1950s, the Lagoon is now managed by opening the mouth soon after it closes, based on monitoring of dissolved oxygen concentrations in the water column. In San Francisco Bay, similar hydrological problems have occurred at Crissy Field, a restored salt marsh. Design constraints forced a reduction in the size of the restored marsh, and the reduced tidal prism of the restored marsh has made it difficult to maintain an open mouth for the Lagoon (Kristen Ward, National Park Service, pers. comm.).

Restoration challenges for urban salt marshes

Our case study highlights the need to consider both upstream and adjacent land use when evaluating restoration opportunities and alternatives. Urban wetlands are particularly susceptible to impacts from the watershed and adjacent land use because the intensity of land-use impacts in urban areas is high. Local environmental conditions can also have a significant “multiplier effect” in combination with land-use issues. In our case, the watershed impacts of sedimentation were exacerbated by three local conditions: very steep watershed topography, easily erodible soils in the watershed, and very flashy/concentrated precipitation events.

A final consideration that our sedimentation case study highlights is uncertainty. Human impacts are likely to be highly variable, and outcomes will be difficult to predict in urban settings. In this case, sediment deposition attracted immigrants who trampled the vegetation that grew through the sandy deposits. Together, burial and trampling caused soil salts to build up, making vegetation recovery even less likely. In other urban settings, similar interactions between physical and human impacts should be expected, but it will be difficult to predict exact interactions and their outcomes. Because of the impacts from adjacent land use as well as the potential for uncertainty, one-time restoration efforts are unlikely to be sufficient in urban setting. Rather, on-going management of restored urban salt marshes will be necessary.

In summary, the potential for negative impacts on salt marsh restoration is not only greater in urban areas, but the specific outcomes are more uncertain due to the number of interacting factors and the difficulty of predicting human behavior. While urban areas offer many significant restoration opportunities and benefits, including education and public involvement, the challenges for urban salt marsh restoration are substantial.

Acknowledgments

This research was supported by funding from the Earth Island Institute and the National Science Foundation (DEB 0212005 to Zedler, Callaway and Madon). We thank the U.S. Fish and Wildlife Service (Tijuana Slough National Wildlife Refuge), the Tijuana River National Estuarine Research Reserve, and the California State Coastal Conservancy for their support of this research. The research was initiated when both authors were at the Pacific Estuarine Research Laboratory (PERL) at San Diego State University, and we thank the following individuals from PERL who assisted with the project: Mike Voss (field help and lab analysis of sediment samples), Jennifer Leeds and Kristen Ward (field help), and Peggy Fong (who alerted us to the sedimentation event along the old river channel). Special thanks to Steve Faulkner for inviting the manuscript.

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