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Storms do not alter long-term watershed development influences on coastal water quality

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ABSTRACT

A twelve year (2000 – 2011) study of three coastal lagoons in the Gulf of Mexico was conducted to assess the impacts of local watershed development and tropical storms on water quality. The lagoons have similar physical and hydrological characteristics, but differ substantially in the degree of watershed urban development and nutrient loading rates. In total the lagoons experienced 22 storm events during the period studied. Specifically, we examine (1) whether there are influences on water quality in the lagoons from watershed development, (2) whether there are influences on water quality in the lagoons from storm activity, and (3) whether water quality is affected to a greater degree by watershed development versus storm activity. The two urbanized lagoons typically showed higher water-column nitrate, dissolved organic nitrogen, and phosphate compared with the non-urbanized lagoon. One of the urbanized lagoons had higher water-column chlorophyll *a* concentrations than the other two lagoons on most sampling dates, and higher light extinction coefficients on some sampling dates. The non-urbanized lagoon had higher water-column dissolved oxygen concentrations than other lagoons on many sampling dates. Our results suggest long-term influences of watershed development on coastal water quality. We also found some evidence of significant storm effects on water quality, such as increased nitrate, phosphate, and dissolved oxygen, and decreased salinity and water temperature. However, the influences of watershed development on water quality were greater. These results suggest that changes in water quality induced by human watershed development pervade despite the storm effects. These findings may be useful for environmental management since they suggest that storms do not profoundly alter long-term changes in water quality that resulted from human development of watersheds.

1. Introduction

Human development of coastal watersheds leads to increased nutrient (i.e., nitrogen and phosphorus) inputs into receiving coastal waters, a process known as anthropogenic eutrophication (NRC, 2000; Smith et al., 2006). Anthropogenic nutrient inputs into coastal waters usually come from agriculture, urban development, and other human related activities in watersheds (Correll et al., 1992; Vitousek et al., 1997; Carpenter et al., 1998). In shallow coastal ecosystems, anthropogenic eutrophication may contribute to ecological transitions from seagrass-dominated to macroalgae-dominated, and then to phytoplankton-dominated systems (Lee and Olsen, 1985; Wazniak et al., 2007; Cebrian et al., 2014) or from canopy-forming algae to turf-forming algae habitats (Gorgula and Connell, 2004; Gorman et al., 2009). Algal blooms may occur as a result of increased nutrient delivery, which in turn may negatively affect communities of seagrasses, macroinvertebrates, and fishes through oxygen depletion and drastic reductions in water transparency (Duarte, 1995; Lerberg et al., 2000; Breitburg, 2002; Armitage et al., 2005). Along with increased nutrient delivery, urbanized coastal watersheds impose other kinds of stress on the biota of bays and estuaries, such as chemical pollution and increased frequency of dredging and fishing (Short and Burdick, 1996; Morrisey et al., 2003; Cebrian et al., 2009).

Storms can also affect the water quality and biotic components (i.e., algae, seagrass, marshes, macroinvertebrates, and fishes) of coastal ecosystems through physical disturbance (Valiela et al., 1998; Paerl et al., 2006; Hagy et al., 2006) and timing and delivery of freshwater inflow (Borsuk et al., 2004; Murrell et al., 2007). Urban development-induced stressors may interact with storms to synergistically affect coastal ecosystems. For example, rainfall events increase run-off into

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coastal systems, which may alter the residence time of freshwater in these systems. This in turn, affects nutrient availability and algal biomass, potentially cascading impacts on the benthic biota (Valiela et al., 1998; Paerl et al., 2001). In addition, sustained storminess may cause substantial physical damage on benthic organisms, thereby exacerbating losses induced by urban related development processes (Paerl et al., 2006; Cebrian et al., 2008; Anton et al., 2009).

Those previous studies were primarily either focusing on effects from coastal watershed development or from storms. However, few studies have examined both the effects of watershed development and storms on the structure and function of coastal ecosystems, particularly over time scales that span many years. Long-term monitoring data sets capturing storm events and gradients in urban development provide a powerful tool to fill this gap. And results from the current study can have some broader applications beyond the systems studied as many other coastal systems are experiencing similar multiple stressors.

We have been collecting water quality data in three coastal lagoons located in Big Lagoon Sound, Perdido Bay (Florida, USA) since 2000. The lagoons are shallow (< 1.0 m in depth) and have similar conditions of water-column salinity and temperature. Conversely, the three lagoons feature contrasting degrees of watershed development, from relatively non-urbanized to urbanized (Stutes et al., 2007; Cebrian et al., 2009; Ferrero-Vicente et al., 2011). Here we describe metrics of water quality in the lagoons from 2000 to 2011. During that period, a total of 22 major storms made landfall, thereby potentially impacting the lagoons.

Specifically, our goals were to assess: (1) whether there are influences on water quality (from 2000 to 2011) in the lagoons from watershed development, (2) whether there are influences on water quality (from 2000 to 2011) in the lagoons from storm activity, and (3) whether water quality is affected to a greater degree by watershed development versus storm activity.

2. Methods

2.1. Study site

We studied three shallow lagoons in the Perdido Bay system of the northern Gulf of Mexico. Mean depths (± SD) for the period examined were 0.54 ± 0.22 m for State Park (SP), 0.61 ± 0.20 m for Gongora (G), and 0.74 ± 0.21 m for Kee's Bayou (KB). The lagoons feature different degrees of watershed urban development. State Park is located in Big Lagoon State Park (Florida), has almost no urban development in its watershed and is completely surrounded by marsh and maritime forest. Gongora is fringed by houses on the northeast side, a road on the northwest tip of the lagoon, and marsh on the southeast side. The lagoon is also connected to an 18-hole golf course through a culvert that runs underneath the road. Kee's Bayou is fringed by houses (several of them with fertilized lawns) on the east side, a road on the north side, and marshes and maritime forest on the south and west sides. A map and additional description of the lagoons is provided in Stutes et al. (2007) and Cebrian et al. (2009). The areas of upland watershed, marsh and open water in SP were 629 m^2 (3%), 1523 m^2 (6%), and 22,659 m^2 (91%), respectively. The areas of upland watershed, marsh and open water in KB were 26,570 m^2 (40%), 9736 m^2 (15%), and 30,208 m^2 (45%), respectively. The areas of upland watershed, marsh and open water in G were 14,443 m^2 (41%), 10,800 m^2 (31%), and 9841 m^2 (28%), respectively. Due to these differences in watershed development, the lagoons also receive different nutrient loading rates (Table 1).

2.2. Field monitoring and laboratory analyses

For the current study, we tried to sample about 6 times a year from 2000 to 2011. However, data gaps exist during this long-term monitoring period. Hydrographic, nutrient, and chlorophyll *a* samples were collected during each sampling event.

Table 1
Loading rates to the lagoons studied. DIN = dissolved inorganic nitrogen; DON = dissolved organic nitrogen; PS = point source (only for Gongora); P = precipitation; and GW = groundwater. The lagoons are SP = State Park; G = Gongora; and KB = Kee's Bayou. All units are in mmol m^{-2} $year^{-1}$. Loading rates were calculated for the period July 1, 2003 to June 30, 2004. Detailed explanations and calculation of loading rates are provided in Lehrter and Cebrian (2010).

	DIN			DON			Phosphate		
Lagoon	PS	P	GW	PS	P	GW	PS	P	GW
SP	–	11.7	0.31	–	27.4	0.95	–	0.55	0.01
Total			12.01			28.35			0.56
G	0.21	34.4	13.6	0.90	25.2	41.4	0.01	0.31	0.47
Total			48.41			67.5			0.79
KB	–	62.7	10.6	–	26.5	32.2	–	2.32	0.37
Total			73.3			58.7			2.69

Significant P values ($P < 0.05$) are highlighted in italics and bold.

2.3. Hydrographic measurements

We used a YSI 85 multiprobe meter to measure in situ water-column dissolved oxygen (DO, surface and bottom, mg/L), salinity (surface and bottom, ppt), and temperature (surface and bottom, °C). We used a pair of Li-Cor 193sa spherical underwater quantum sensors and a data logger to measure photosynthetically active radiation (PAR) (Stutes et al., 2007). The light attenuation coefficient (k, m^{-1}) was calculated from surface and bottom light intensity and water depth according to the Beer-Lambert equation (McPherson and Miller, 1987). These measurements were taken at four fixed locations of each lagoon during each sampling visit.

2.4. Nutrients and chlorophyll *a*

Water column samples for nutrients and chlorophyll *a* were collected at the same locations where hydrographic measurements were taken, and additionally at two extra random locations in each lagoon. Samples were collected from the mid water-column. The samples were transported to the laboratory on ice and filtered through 47 mm diameter glass microfiber filters (Whatman GF/F, pore size of 0.45 μm) using vacuum filtration. Filters for chlorophyll *a* analysis and filtrate were frozen at − 20 °C until processed. The filtrate was analyzed for nutrient species such as nitrate (μM) and phosphate (μM) using colorimetric techniques on a Skalar auto analyzer (Pennock and Cowan, 2001). Then, dissolved organic nitrogen (DON, μM) was calculated. Two different methods were used to assay chlorophyll *a*. From 2000 to 2009, chlorophyll *a* were extracted for 24 h with 90% acetone, and its concentration (μg/L) measured fluorometrically with a Turner Designs Model TD-700 fluorometer using acidification (1 N HCl) (Strickland and Parsons, 1972). From 2009 to 2011, we used a non-acidification protocol thereafter (Welschmeyer, 1994). We compared measurements obtained with both protocols to ensure that they yielded no significant difference (i.e., paired *t*-test: $n = 13$, $t = -1.23$, $P = 0.243$).

2.5. Major storms covered

We considered all tropical storms or hurricanes that made landfall along the coast of the northern Gulf of Mexico during 2000–2011. This consideration was to make our analysis as inclusive as possible, which included all the storms that could potentially impact the studied lagoons. Thus, during the monitoring period, a total of 22 storms were analyzed (Table 2). Some of these storms were originally hurricanes but weakened to tropical storms at landfall. The 22 storms were: Tropical Storms Gordon and Helene in 2000; Tropical Storms Allison and Barry in 2001; Tropical Storms Hanna and Isidore, and Hurricane Lili (category 1) in 2002; Tropical Storm Bill in 2003; Tropical Storms Bonnie and Frances, and Hurricanes Ivan (category 3) and Jeanne (category 3)