The potential to improve water quality in the middle Rio Grande through effective wetland restoration
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ABSTRACT
The Rio Grande, which forms the United States—Mexico border for much of its course, receives diverse pollutants from both urban and agricultural areas, most notably in the sister cities of El Paso (TX, USA)—Ciudad Juárez (CHI, Mexico). This study aimed to describe regional trends in water quality in waters near the El Paso–Ciudad Juárez metroplex and to examine the potential for water quality improvement through the use of a created wetland. Very few differences in nutrient chemistry were found among drains, canals and the Rio Grande, with the exception of elevated chloride and lower phosphorus levels found in the drains. Overall, chloride concentrations increased with distance downstream, likely due to concentration of salts via evaporation from irrigated agriculture. A wastewater treatment plant (WWTP) contributed substantially to total phosphorus and nitrate levels, which, together with ammonia, tended to exceed state criteria for water quality downstream of the WWTP outflow. The created Rio Bosque wetlands reduced nitrate concentrations in the water, possibly via denitrification enhanced by algae; algae increased in biomass as water flowed through the wetlands. However, the diversion of water for irrigated agriculture, resulting in the absence of water, and thus aquatic plants, in the wetland in the summer has limited the ability of this wetland to improve regional water quality.

Key words | algae, Rio Grande, water quality, wetland

INTRODUCTION
The Rio Grande forms the international border between the United State and Mexico from the sister cities of El Paso (TX, USA)—Ciudad Juárez (CHI, Mexico) to the Gulf of Mexico. Both countries depend on the river for irrigated agriculture and, to a lesser extent, for municipal water supply; however, recreational and aquatic life uses are severely limited in the river. The Rio Grande is contaminated by heavy metals (Rios-Arana et al. 2003) and exceeds state standards for nutrient and bacterial pollution, and total dissolved solids (IBWC 2008). The identification of sources of nutrient pollution and methods to improve water quality would increase the value of this international water body for both nations.

During the process of canalization of the Rio Grande between El Paso and Juárez in the 1930s, many naturally occurring riparian wetlands were destroyed (Watts et al. 2002). Some have suggested that the absence of wetland and riparian vegetation may be partly responsible for elevated nutrient levels throughout the middle Rio Grande (Oelsner et al. 2007). Wetlands and riparian areas can provide many ecosystem services such as nutrient retention, sediment attenuation, increased biodiversity, and flood abatement (Zedler 2003; Fisher & Acreman 2004). In particular, a functioning wetland can reduce the levels of many pollutants, including nutrients, bacteria, and heavy metals (e.g. Quinonez-Diaz et al. 2001; Mitsch et al. 2005;
Verhoeven et al. 2006; Yang et al. 2006). Created wetlands have been shown to improve the water quality of WWTP effluent (e.g. Smith et al. 2000) and of rivers impacted by non-point source agriculture (e.g. Beutel et al. 2009) and urban/agricultural land use (e.g. Mitsch et al. 2005).

With the degradation of water quality in the Rio Grande and the loss of most of the riparian wetland habitats in the region, it has become imperative to understand and rehabilitate our regional water resources. Protection and enhancement of functioning wetlands has the potential to improve water quality and aid in the attainment of state-established water quality standards mandated by the US Clean Water Act. However, the scarcity of studies examining water quality in the region, both in the Rio Grande and in associated wetlands, limits our ability to determine if this would be an effective strategy for ameliorating water quality in the region. The goal of this study was to describe regional trends in water quality and examine the potential for water quality improvement through the use of river side wetlands.

Study Sites

The Rio Grande was rectified starting in the 1930s to stabilize the international border and provide flood protection, reducing the total length of the river in the region from 250 to 140 km (Reinhardt 1937). Agricultural canals and drains have, however, substantially increased the total length of waterways in the border region. For example, in El Paso and Hudspeth Counties (TX), there are more than 600 km of canals and more than 400 km of drains (Figure 1).

For irrigation in the middle Rio Grande valley, water is diverted from the Rio Grande into agricultural canals that deliver the water to farms; water then percolates into a subsurface drain system that takes the water to a surface drain, which eventually returns the flow to the river. In particular, at the point where the Rio Grande becomes the international border, water is diverted by the American Dam into the US (American Canal and Franklin Canal) and Mexican canal systems, while the American Canal extension delivers water to the Riverside Canal. Any water not utilized in El Paso County is carried by the Riverside Canal into the Tornillo Canal in Hudspeth County. For this study, we visited 8 canal, 9 drain and 6 Rio Grande sites starting 12 km upstream of the international border and continuing for an additional 130 km downstream.

Before the canalization of the Rio Grande, the river flowed through what is the present day site of the Rio Bosque Wetland Park (RBWP), a 150 ha park designed to recreate a mosaic of upland and riparian habitats. However, since being isolated from the main river channel, the park area has been without a perennial water source. In 1997, the park landscape was cleared and grated to build a wetland complex and water-delivery system as part of the mitigation for the American Canal extension. Approximately one-quarter of the park was dedicated to re-establishing riparian and wetland communities; two wetland cells total nearly 30 ha in size when full. Since 1998, secondary treated wastewater has been delivered to the wetland cells of the RBWP from the Roberto Bustamante Wastewater Treatment Plant (WWTP) during the months of October through February, when water is not being used for irrigated agriculture. Water entering the park at the inflow (Site A) is delivered through water delivery gates into two wetlands cells (Cells 1 and 2) (Figure 2). On average, three times as much water is delivered into Cell 2 (0.30 m$^3$/s) than Cell 1 (0.09 m$^3$/s). Water exits Cell 2 (and the park) at another water delivery gate, whereas Cell 1...
water level is controlled through a water delivery gate that diverts water into Cell 2. The residence time of water in the ponds is relatively short, with Cell 1 draining of water within 2 days after inputs are halted and Cell 2 draining within 5 days. During the irrigation season, water is diverted into the Riverside Canal and the Rio Bosque is largely dry. The University of Texas at El Paso (UTEP) and its partners are working to guide and shape the recovery of the wetland park to promote native river-valley plant communities (Watts et al. 2002).

**METHODS**

To determine regional trends in water quality in the Rio Grande and agricultural water delivery channels, we visited 8 Canal (3 main, 5 lateral), 9 Drain and 6 Rio Grande sites 4–5 times during the irrigation season (March to October) 2007 and collected samples for nutrient chemistry (as described below). Sites ranged over more than 140 km of river from the Texas-New Mexico border through El Paso and Hudspeth Counties south to Fort Quitman, TX (Figure 1). To our knowledge, none of these 23 sites were located immediately downstream of point sources; however, one of our sites on the Riverside Canal was located immediately upstream of the outflow from the Bustamante WWTP.

To examine the impacts of wastewater inflow on water quality, we also sampled one additional site on the Riverside Canal, immediately downstream of the WWTP outflow. Data from the Riverside Canal (downstream of the WWTP) was not included in the overall analysis of water quality in the region.

During the months of October to February 2005–2009, the water delivery channels and wetlands cells of the RBWP were sampled for nutrient chemistry, phytoplankton and attached algae (periphyton). Four sites were sampled: the inflow from the Bustamante WWTP (Inflow), wetland Cell 1, wetland Cell 2, and the water outflow from Cell 2 (Outflow). For the first two years of the study, all sites were sampled every 2–4 weeks. In the winter 2007–2008, sites were sampled bi-weekly until mid-December when water deliveries ended. In the winter of 2008–2009, sampling of the water continued on a monthly basis.

For determination of nutrient chemistry, water was collected from each site at an open water area in 125 mL acid-washed bottles. Total phosphorus (TP) was determined using the ascorbic acid method following persulfate digestion (APHA 1998); nitrate (NO₃-N), ammonia (NH₃-N) and chloride (Cl⁻) were analyzed according to Hach protocols and standard methods (APHA 1998). All analyses were performed on a Genesys 10 UV Spectrophotometer. Conductivity, temperature, pH, salinity, redox potential and dissolved oxygen were measured in the field using an YSI® 556 multiprobe (YSI Incorporated, Yellow Springs, OH, USA), which was calibrated prior to each trip.

At the RBWP, chlorophyll-a concentration was used as a surrogate for algal biomass. For determination of phytoplankton chlorophyll, one liter of water was collected in opaque 1-L bottles and filtered immediately on Whatman GF/C filters, wrapped in foil and frozen until analysis. Sediment periphyton was collected from 5 separate locations at each sampling site using an inverted Petri dish and spatula. All 5 periphyton samples from the site were combined into one composite sample. Algal chlorophyll was separated from the sediment by vigorously mixing the sediment with distilled water and pouring off the surface liquid. This was repeated 10 times, until the water was relatively clear, to get the final periphyton sample. The liquid sample was stored in a test tube, wrapped in foil and frozen until analysis. Chlorophyll was extracted from both

![Figure 2](attachment://image.png)
phytoplankton and periphyton using 90% acetone and a 1-hour (phytoplankton) to 24-hour (periphyton) extraction period in the freezer. Results were corrected for phaeo-pigments by acidification (Wetzel & Likens 2000). Absorbances were determined on a Genesys 10 UV Spectrophotometer.

We used paired t-tests to compare whether there were differences among nutrient concentrations at the RBWP sites sampled on the same dates. Simple linear regressions were used to relate algal biomass to the measured environmental variables. Differences in water quality among drains, canals and the Rio Grande were determined using a repeated measures ANOVA, followed by Tukey HSD test. Data were log-transformed, as required, to normalize the data. All statistical analyses were performed using JMP software (Version 4.0, SAS Institute Inc., Cary, N.C.).

RESULTS

Repeated measure ANOVA indicated that there were no significant interactions of site type and time on the concentrations of nutrients in the water bodies in the region. However, there was a significant effect of site type alone on Cl \( (F_{2,10} = 6.05, p = 0.0189) \) and TP \( (F_{2,13} = 5.7, p = 0.0166) \) concentrations. The agricultural return flow drains had the highest concentrations of Cl \( (p = 0.0094; \text{Tukey HSD}) \), while drains tended to have lower levels of TP than canal sites \( (p = 0.0305; \text{Tukey HSD}) \); otherwise, there were no significant differences in nutrient concentrations among site type (Figure 3). There was also a significant effect of time on NO\(_3\)-N \( (F_{1.7,21.5} = 21.96, p < 0.0001) \) and NH\(_3\)-N \( (F_{2.5,31.9} = 3.6, p = 0.0306) \) concentrations. Nitrate and ammonia concentrations were lowest in July, compared to all other months \( (p < 0.05; \text{Tukey, HSD}) \). On average, state nutrient criteria for Cl were exceeded at the drain sites, while TP criteria were exceeded in the Rio Grande. Overall, total phosphorus criteria levels were exceeded on 14% of sampling occasions, nitrate criteria were exceeded 2% of the time, and ammonia and chloride criteria were exceeded greater than 30% of the time.

Downstream of the WWTP discharge into the Riverside Canal, TP and NO\(_3\)-N were significantly elevated relative to upstream levels \( (t\text{-test}, p < 0.05) \), and these concentrations were above state screening levels (Figure 4). Chloride levels were significantly reduced below the outfall. NH\(_3\)-N was above state screening levels at both upstream and downstream stations.

Comparisons of trends in nutrient levels with distance downstream from the most upstream point indicated that chloride levels increased with distance downstream for canals \( (v = 1.64 + 0.008; r^2 = 0.89, p = 0.0013) \), drains \( (v = 2.17 + 0.004x; r^2 = 0.62, p = 0.0102) \) and the Rio
Grande ($y = 1.93 + 0.005x; r^2 = 0.92, p = 0.0023$). None of the other measured nutrients showed similar trends. Although the slope of the line for the canal sites is much steeper than the other 2 site types, this is likely because the first canal sites were not sampled until 40 km downstream; CI concentrations in the canals approached the other site types further downstream (Figure 5).

Water quality varied throughout RBWP; however, very few significant differences were observed in water quality after the water flowed through the wetland. Nitrates were significantly higher in the inflow compared to both Cell 2 (paired t-test; $p = 0.0264$) and the outflow (paired t-test; $p = 0.0392$) (Figure 6). The water at the inflow was significantly warmer than all others sites ($p < 0.001$). In many cases, nutrient levels in the RBWP were an order of magnitude higher than those observed in the other regional water bodies; all nutrient levels exceeded state screening levels for the Rio Grande.

Phytoplankton biomass was significantly lower at the inflow, compared to all other sites in the RBWP (Figure 7; $p < 0.005$). Simple linear regression (not shown) indicated that phytoplankton biomass increased with distance from the inflow ($r^2 = 0.36; p < 0.0001$). Similarly, attached algae increased in biomass as water flowed through the park, although only cell 1 was significantly different from the inflow ($p = 0.0368$). Simple linear regression (not shown) indicated that, on average, periphyton biomass increased with distance from the inflow ($r^2 = 0.98; p = 0.0095$).

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**Figure 5 |** Relationship between log-transformed Cl levels and river distance downstream (from upstream sites).

**Figure 6 |** Comparison of mean nutrient concentrations at four sites in the Rio Bosque Wetland Park. Different letters show significant differences (paired t-test, $p < 0.05$).

**Figure 7 |** Comparison of mean algal concentrations at four sites in the Rio Bosque Wetland Park. Different letters show significant differences (paired t-test, $p < 0.05$).
DISCUSSION

During the irrigation season of 2007, water quality in the Rio Grande exceeded state water quality standards for ammonia and chloride levels on a regular basis (>30% of samples), and exceeded total phosphorus criteria on 14% of samples. These results agree with state documents that list these sections of the river at concern for exceeding screening levels for nutrients, as well as bacteria and total dissolved solids (IBWC 2008). As in other studies in the Rio Grande (Oelsner et al. 2007), we found that wastewater was likely a large contributor to nutrient loads in the region; in particular, criteria for TP, NO$_3$-N and NH$_3$-N were exceeded downstream of the Bustamante WWTP. At this international boundary, the Rio Grande receives inputs from largely secondary treatment plants on the US side of the border, and primary treated water from Ciudad Juarez, Mexico. Further downstream of the 2 cities, inflowing water comes primarily from agricultural return flows. It is likely that the agricultural fields through which the water has recently passed are also a non-point source of pollution to regional waters (Oelsner et al. 2007); however, we saw no significant difference among agricultural drains and canals with respect to most nutrient levels. There were, however, significantly high levels of Cl in the agricultural drains and chloride increased with distance downstream. Flushing of salts concentrated by evaporation in agricultural fields is likely a large source of chloride to the Rio Grande. This effect would increase downstream as Hudspeth County receives water that has already passed through El Paso County’s irrigation channels.

Seasonally, nitrate and ammonia levels were highest in the spring and fall. Similar seasonal trends have been reported in upstream reaches of the Rio Grande (Oelsner et al. 2007) and in created wetlands (Beutel et al. 2009; Maltais-Landry et al. 2009), due to increase biological demand in mid-summer. Given nutrient criteria exceedances in the region, methods that would reduce nitrogen and phosphorus concentrations in the regional waterways year-round, without enhancing salinity, are required to aid in the attainment of nutrient criteria in this international waterway.

The absence of wetland and riparian vegetation may be partly responsible for the lack of nutrient uptake capabilities throughout the middle Rio Grande (Oelsner et al. 2007). A functioning wetland designed to clean agricultural return flow (e.g. Beutel et al. 2009) or WWTP effluent (e.g. Smith et al. 2000) could act to reduce nutrient levels in these waters and ultimately aid in the attainment of nutrient criteria in the region. The Rio Bosque wetlands currently receive secondary treated effluent directly from the Bustamante Wastewater Treatment Plant (WWTP) during the non-growing season and are ideally situated adjacent to drains, canals, the Rio Grande and the international border to act as treatment wetlands beneficial to both countries. However, competing demands for water along this international waterway are among the many challenges facing the Rio Bosque.

In functioning wetlands, phosphorus retention occurs via plant uptake, microbial degradation and sedimentation (Reddy et al. 1999; Fisher & Acreman 2004; Gu & Dreshel 2008); however, not all wetlands are efficient at removing nutrients (Fisher & Acreman 2004). In the RBWP, total phosphorus (TP) levels did not vary throughout the wetland. When water enters a wetland during the growing season, phosphorus is often taken up by plants; however, in the Rio Bosque, there were no plants growing during the study period, as water is only delivered to the park during the winter months. While there was no active plant growth during the winter months, the cells were covered with a variety of wetland and non-wetland plant species in the summer months. Thus, instead of taking up nutrients, the senescence of these plants after flooding in the fall may have released phosphorus into the water column during autumn and winter months (Krøger et al. 2007). Phosphorus can also be easily adsorbed by soils (Reddy et al. 1999). In particular, increased phosphorus retention capacity can be seen in clay sediments, in sediments with high organic matter (Reddy et al. 1999; Novak & Watts 2006) and under reducing conditions (Fisher & Acreman 2004). While soils in the RBWP wetlands cells tend to be more organic than soils in more upland areas of the park, these levels are still relatively low (<5% organic matter; unpubl. data). Furthermore, we do not believe that phosphorus release from the sediment was contributing to consistently high P levels in the wetland as preliminary results indicate that release rates of phosphorus from the sediment (5.2 mg m$^{-2}$ day$^{-1}$; unpubl. data) were relatively low compared to published...
values, which can range from 0.250 to 51.50 mg m\(^{-2}\) day\(^{-1}\) (Nurnberg 1988). Similarly, total phosphorus concentration in the sediments was relatively low (11.21 µg g\(^{-1}\); unpubl. data). Phosphorus release and retention is also affected by the amount of oxygen in the sediment and water (Carlton & Wetzel 1988; Reddy et al. 1999; Dodds 2003; Fisher & Acreman 2004). Phosphorus tends to be released from soils under anoxic situations (Carlton & Wetzel 1988; Nurnberg 1988; Dodds 2003). Dissolved oxygen (DO) levels were relatively high during the daylight hours at the Rio Bosque (Inflow: 6.2 ± 0.43 mg/L; Outflow: 8.7 ± 0.98 mg/L); however, it is possible that lower nighttime DO may have encouraged phosphorus release.

Nitrates levels were significantly reduced at the outflow of the RBWP, while ammonia concentrations remained unchanged. Nitrates are reduced as water flows through wetlands (e.g. Fleming-Singer & Horne 2006; Maltais-Landry et al. 2009) primarily via denitrification (Mitsch & Gosselink 2007; Maltais-Landry et al. 2009), but it can also be taken up by plants and adsorbed to sediment (Vymazal 2007). Denitrification is the process whereby bacteria convert nitrates and nitrites into atmospheric nitrogen; rates of denitrification are affected by dissolved oxygen levels, organic carbon concentration, temperature and pH (Vymazal 2007). For example, nitrate removal tends to be greatest in the summer months when temperatures are relatively high (Beutel et al. 2009; Maltais-Landry et al. 2009). Despite receiving WWTP effluent only in the winter months, water temperatures in the RBWP were within the range reported for maximum denitrification rates (Vymazal 2007) (Inflow: 23.55 ± 0.60°C; Outflow: 15.11 ± 0.82°C) indicating that denitrification could be acting to reduce nitrate levels in the wetland. Nitrate removal can also be related to organic matter accumulation, (Craft 1997), which is greatest in planted wetland cells (Lin et al. 2007), and leads to both organic N accumulation and denitrification. However, organic matter content of these desert soils is relatively low (<5%) and the absence of plant growth during the current period of water availability would reduce this effect. Finally, while denitrification tends to occur under anoxic conditions, it can occur in the presence of oxygen (Vymazal 2007). On 20 percent of sampling occasions, ORP was suitable anoxic (<100 mV) to promote denitrification. Even lower dissolved oxygen levels during the night time may have enhanced denitrification; however, diurnal oxygen or ORP measurements were not taken. Future research will include examination of diurnal dissolved oxygen profiles in the wetland cells to determine if night time oxygen levels are low enough to promote phosphorus release or denitrification.

Algae may also play an important role in the uptake of nutrients from polluted waters (Vymazal 1988; Wu & Mitsch 1998; Dodds 2003; Gu & Dreshel 2008). The increase in both periphyton and phytoplankton biomass after the inflow into the RBWP indicated that algae may have played a role in reducing nitrogen concentrations in water flowing through the park. Other studies have found an increase in algal biomass near the outflow of wetlands receiving nitrate-rich water (Fleming-Singer & Horne 2006; Lin et al. 2007), similar to the concentrations observed in this study. Furthermore, several studies have found that wetland cells dominated by algae can have similar nutrient removal rates as vegetated cells (Lin et al. 2007; Gu & Dreshel 2008). Algae can reduce nutrient concentrations by direct uptake (Dodds 2003) or by promoting denitrification (Toet et al. 2003). Furthermore, high periphyton cover may reduce phosphorus release from sediment by increasing oxygen level at the sediment-water interface (Carlton & Wetzel 1988). While metabolic uptake of nutrients by periphyton is limited by lower temperature (Dodds 2003), which may occur during winter months in many temperate regions, water temperatures in this desert wetland were generally greater than 15°C, indicating that algae may have been capable of contributing substantially to nutrient uptake in the RBWP. Experiments examining the role of benthic algae in nutrient release and retention will be completed to clarify these trends.

Hydrophytic vegetation is key to providing the nutrient retaining capacities of wetlands, either through the uptake of nutrients directly (Kao et al. 2003; Brisson & Chazarenc 2009), or by indirectly affecting nutrient concentrations through alteration of the redox potential, microbial community, and soil characteristics (Williams 1985; Brix 1997). In particular, depending on ambient conditions, different species of plants are more efficient at the uptake of nutrients than others (Kao et al. 2003; Brisson & Chazarenc 2009). Preliminary results indicate that the ponds in the Rio Bosque have developed some characteristics of a wetland plant
community since the park was established more than 10 years ago (unpubl. data); however, the absence of water during the growing season has likely substantially slowed this process.

The seasonal delivery of water only during winter, when it is not needed for irrigated agriculture, limits not only development of wetland plant communities in the RBWP, but also the development of valuable wetland functions. Conditions during our relatively mild winters likely promoted nitrate uptake via denitrification and algal growth; however, if water were diverted into the park during the summer, thereby promoting hydrophytic plant growth and other wetland functions, this could be a first step in considerably improving regional water quality. In particular, elevated nutrient concentrations emitted from wastewater treatment plants could be ameliorated. While elevated salinity is a concern for regional waters, chloride concentrations do not appear to increase after passing through the RBWP; therefore, it appears that use of the Rio Bosque as a treatment wetland in the summer months could only benefit regional water quality. Unfortunately for the Rio Bosque, all surface waters in the region are allocated for use by a variety of landowners, primarily for agricultural irrigation, and any new needs must be met by transferring water rights among these users. In the El Paso region, however, there is no legal framework in place to easily allow the water rights owner to transfer their water to environmental users. The lack of water for environmental purposes is a primary barrier preventing aquatic and riparian habitat restoration in the region (King & Maitland 2003).

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