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# INFLUENCE OF CATTLE GRAZING AND PASTURE LAND USE ON MACROINVERTEBRATE COMMUNITIES IN FRESHWATER WETLANDS

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*Abstract:* Responses of wetland abiotic variables and aquatic invertebrate community structure to cattle stocking density, pasture type, and dominant vegetation were evaluated in subtropical pastures. Cattle were stocked at four treatment levels on improved (fertilized) and semi-native (unfertilized) pastures in south-central Florida, USA. Improved pasture wetlands were dominated either by *Panicum hemitomon* (maidencane) or by a mixture of *Polygonum* spp. (smartweed) and *Juncus effusus*; semi-native pasture wetlands were dominated mainly by maidencane. Cattle stocking density had few significant effects on water-column nutrient concentration or invertebrate community structure. However, water-column nutrient concentrations were significantly greater in the wetlands on improved pastures compared to semi-native pastures. Invertebrate richness and diversity were greater in wetlands on semi-native pastures than on improved pastures, despite lower nutrient concentrations in the former. Overall, the cattle stocking treatment had little impact on invertebrate community structure in these systems relative to prior pasture land use. However, vegetation type influenced invertebrate communities and explained some of the differences between pasture types. Semi-native (lower nutrient) wetland pastures dominated by maidencane had significantly greater invertebrate richness and diversity than improved (higher nutrient) wetland pastures dominated by mixed vegetation but showed no difference when compared to improved wetland pastures dominated by maidencane. Chironomids were the dominant invertebrate in wetlands of both pasture types. Correspondence analysis revealed that ostracods and Culicidae larvae might be useful as bioindicators of subtropical wetlands that are experiencing cultural eutrophication.

*Key Words:* cattle grazing, Florida wetlands, invertebrates, nutrients, Chironomids, improved pasture, semi-native pasture, land use

## INTRODUCTION

Wetlands serve many ecosystem functions, including the ability to sequester nutrients from the landscape (Cooper and Findlater 1990, Johnston 1991, Kadlec and Knight 1996, Reddy et al. 1999). The ability to serve as nutrient sinks can be particularly valuable in agricultural landscapes, where nutrient loads are often high and threaten the integrity of downstream water bodies (e.g., Carpenter et al. 1998). In many cases, wetlands are engineered specifically to receive and treat excess nutrients (Guardo et al. 1995, Reed et al. 1995, Kadlec and Knight 1996). Wetland structure and function can be impacted by agricultural activities; in addition to nutrient loads, wetlands can experience increased sedimentation associated with tillage practices, increased pesticide runoff, and altered hydrologic regimes (Euliss and Mushet 1996a,b, Steinman and Rosen 2000). Research in the prairie pothole region of North America has examined, in particular, the impact of row crop practices on wetland ecology (Euliss et al. 1999, Tangen et al. 2003). Far less attention has been paid to the effects of cattle grazing on wetland ecosystems.

Cattle grazing can impact wetlands both directly and indirectly. Direct impacts include herbivory of aquatic vegetation, nutrient inputs via urine and fecal deposition, and trampling of the sediments (Coffin and Lauenroth 1988, Archer and Smeins 1991, Collins et al. 1998). Indirect impacts include changes in macrophyte and algal species composition induced by nutrient loading and selective herbivory, which in turn can affect higher trophic levels that rely on these autotrophs for habitat, refugia, and prey (Rader and Richardson 1992, 1994, Rader 1994).

One of the most critical regions for addressing the influence of cattle grazing on wetland dynamics is south-central Florida, USA, where extensive subtropical rangelands interface with some of the most sensitive wetland areas in the United States. The watershed of Lake Okeechobee, which serves as the headwaters for the Florida Everglades, has a land use dominated by beef cattle ranching (Flaig and Havens 1995, Steinman and Rosen 2000). During the past 30 years, the coverage of wetlands in the watershed has decreased from approximately 25% to 15%, as a drainage network has been imposed for improved flood control and as more land has been converted to pasture for beef cattle production (Flaig and Havens 1995, Flaig and Reddy 1995, SFWMD 1997). As a result of these land-use changes, phosphorus concentrations and loads entering Lake Okeechobee have more than doubled over the past 30 years (James et al. 1995), resulting in a number of ecological impairments in the lake (Havens et al. 1996, Steinman et al. 1999).

In order to improve the ecological conditions of Lake Okeechobee and its watershed, different methods have been proposed to increase water storage and reduce non–point-source pollution from agricultural lands draining into the lake. One option being pursued by regulatory agencies and non-profit conservation agencies such as The Nature Conservancy is the restoration of wetlands on cattle ranches, which will result in greater retention of water and nutrients. Very little information is available on seasonal wetlands on private lands, and this information will be critical for assessing restoration success.

Analysis of aquatic invertebrate communities can be a valuable tool in this effort, as these organisms are widely used in bioassessments of aquatic ecosystems (e.g., Rosenberg and Resh 1993, Karr and Chu 1997) and serve as an important food source for higher trophic levels. Greater attention is now being placed on the ability of invertebrates to assess the ecological integrity of wetlands (Burton et al. 1999, King et al. 2000, Rader et al. 2001). In this study, we took advantage of a recently established, replicated cattle grazing experiment on an operational commercial cattle ranch, to investigate the effect of cattle grazing and pasture land use on invertebrate communities in seasonal freshwater wetlands. We predicted that cattle stocking density and historic land use would alter the composition of the invertebrate community structure and that these changes would be useful for assessing the ecological impact of ranching practices in these depressional wetlands.

## METHODS

### Site Description

The experiment took place at the MacArthur Agro-ecology Research Center (MAERC), a 4170-ha working cow-calf operation and citrus grove at Buck Island Ranch in south-central Florida. Sixteen experimental pastures, eight of improved summer pasture and eight of semi-native winter pasture, were used in this study. The 16 pastures are hydrologically separated and fully instrumented to monitor volume and chemistry of surface water draining from each pasture during periods of flow.

### Stocking Number Experiment

A cattle stocking rate optimization project was established at MAERC in 1998 to test the effects of different pasture land-use treatments and cattle stocking densities on water quality and nutrient cycling. Improved and semi-native pastures were stocked at four different levels (0, 15, 20, and 35 cow-calf units per pasture, where one cow-calf unit equals one cow and her calf), resulting in two replicates per stocking treatment. Stocking densities were lower in the semi-native winter pastures, due to their larger areas (30.4 ha), than in the improved summer pastures (20.2 ha) (Table 1). Difference in pasture size was due to the fact that herds grazed the semi-native pastures in the winter when grass productivity was lower and so a larger area was needed to support the same number of animal units that grazed the improved pastures in the summer. Cattle were 4- to 9-year-old Braford cows, selected randomly from one of the commercial herds at Buck Island Ranch. Each herd occupied the improved, summer pastures during May–October and the semi-native, winter pastures during November–April. Pasture treatments began in October 1998 when the test herds were moved into the winter pastures. Water quality and invertebrate sampling for this study was conducted from July through October 2001, corresponding to a duration of three years for the experimental treatments.

The experimental design simulated normal ranching practices in this region, where ranchers usually have two major pasture systems and include a seasonal rotation of cattle. In the summer, cattle are generally

Table 1. Details of experimental pastures. Improved pastures are 20.23 ha and semi-native pastures are 32.38 ha in size. Data from wetlands within the same pasture were composited for statistical analyses, as pasture, not wetland, served as the experimental unit.

| Pasture Type | Stocking Number (cows per pasture) | Stocking Density (cows/ha) | Pasture # | Wetland ID | Dominant Vegetation* |
|---|---|---|---|---|---|
| Improved | 0 | 0.0 | 1 | S1N | Maidencane |
| Improved | 0 | 0.0 | 1 | S1S | Maidencane |
| Improved | 0 | 0.0 | 8 | S8N | Mixed** |
| Improved | 0 | 0.0 | 8 | S8S | Mixed |
| Improved | 15 | 0.74 | 4 | S4N | Mixed |
| Improved | 15 | 0.74 | 4 | S4S | Mixed |
| Improved | 15 | 0.74 | 6 | S6 | Mixed |
| Improved | 20 | 0.99 | 2 | S2 | Mixed |
| Improved | 20 | 0.99 | 7 | S7 | Mixed |
| Improved | 35 | 1.73 | 3 | S3N | Mixed |
| Improved | 35 | 1.73 | 3 | S3S | Mixed |
| Improved | 35 | 1.73 | 5 | S5 | Mixed |
| Semi-native | 0 | 0.0 | 7 | W7N | Maidencane |
| Semi-native | 0 | 0.0 | 7 | W7S | Maidencane |
| Semi-native | 15 | 0.46 | 1 | W1N | Maidencane |
| Semi-native | 15 | 0.46 | 1 | W1S | Maidencane |
| Semi-native | 15 | 0.46 | 6 | W6N | Maidencane |
| Semi-native | 15 | 0.46 | 6 | W6S | Maidencane |
| Semi-native | 20 | 0.66 | 2 | W2N | Maidencane |
| Semi-native | 20 | 0.66 | 2 | W2S | Maidencane |
| Semi-native | 20 | 0.66 | 8 | W8S | Maidencane |
| Semi-native | 20 | 0.66 | 8 | W8S | Maidencane |
| Semi-native | 35 | 1.08 | 3 | W3 | Maidencane |
| Semi-native | 35 | 1.08 | 5 | W5 | Maidencane |

* General characterization of dominant species. Percent cover varies among wetlands.

** Mixed includes mostly *Juncus, Polygonum,* and *Sagittaria* spp.

stocked on improved pastures that are planted in *Paspalum notatum* Fluegge (Bogdan) (Bahia grass) and are fertilized annually with N (generally, ~ 50kg N/ha); hence, they are considered ''improved.'' In the winter, cattle are moved to semi-native pastures that are dominated by a mixture of bahia grass and native grasses (*Andropogon virginicus* L., *A. glomeratus* Walt., *Paspalum laeve* Michx., and *Axonopus affinis* Chase) and generally are not fertilized. In addition to N fertilizer inputs, the improved pastures used in this experiment received annual application of P fertilizer for at least 15–20 years prior to 1987, at which time P fertilizer application to Bahia grass pastures was discontinued per the recommendation of the agronomists at the University of Florida. This past fertilizer use has been linked to elevated P concentrations in surface runoff and soil in the improved pastures relative to the semi-native pastures (P. Bohlen, unpublished data). The stocking numbers for the stocking experiment were chosen in consultation with the Florida Cattlemen's Association to reflect actual grazing practices in the region most accurately and, hence, do not follow a natural geometric progression.

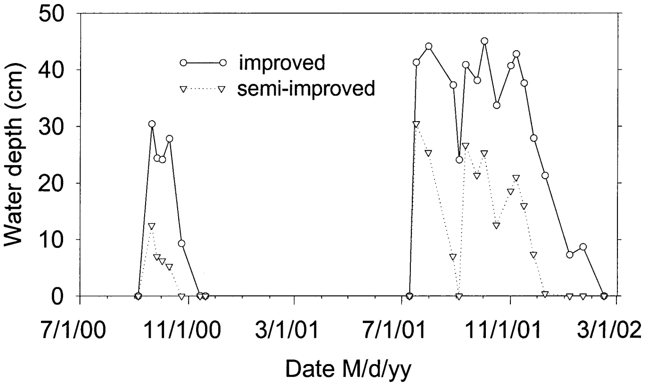

Figure 1. Mean water depth in wetlands from improved and semi-native pastures revealing the seasonal nature of these systems. Data presented in this paper correspond only to the July through October 2001 period.

## Sampling and Laboratory Analysis

Twenty-four wetlands, ranging in area from 0.5 to 2.0 ha, were selected from among the 16 pastures for use in this study (Table 1). The pasture served as the experimental unit in all statistical analyses (see below), so in those cases where there were two wetlands per pasture, the average of the two wetlands was used in the analysis.

Dominant vegetation was not consistent among wetlands; two of the 12 wetlands in the improved pastures were dominated (70–80% cover) by *Panicum hemitomon* Schultes (hereafter, referred to as maidencane). The remaining 10 improved pasture wetlands were dominated by a mixture of different plant genera, including *Polygonum*, *Juncus*, *Sagittaria*, and *Andropogon* (hereafter, referred to as mixed vegetation). The 12 wetlands in the semi-native pastures were dominated by maidencane, with some emergent macrophytes (*Sagittaria lancifolia* L. and *Pontederia cordata* L.) in the deeper portions of the wetlands. Wetlands chosen for this study were based on similarities in size, vegetation, and hydroperiod to the greatest extent possible (Table 1).

Wetlands were sampled for physical and chemical characteristics in August and September 2001, immediately after the wet season rains resulted in standing water (Figure 1). Temperature, conductivity, pH, and dissolved oxygen were measured with a Hydrolab Surveyor II datasonde. In addition, water samples were collected and analyzed according to standard USEPA

(1979) methods for nutrients, color, and total suspended solids.

Macroinvertebrate communities were sampled the first week of October 2001. Sixteen sweeps (eight with 0.5-mm mesh and eight with 1.0-mm mesh; area of net opening was 0.1 $m^2$), each 0.5 m in length, were composited to result in one sample. Three samples with each net were taken per wetland, and care was taken to avoid sweeping in areas previously sampled. Data reported here are for the 1.0-mm-aperture sweeps.

In the field, net samples were composited into a white plastic tray. After fishes, tadpoles, and large spiders were removed, the sample was transferred to a plastic bag and placed inside a cooler. Upon return to the laboratory, samples were refrigerated until the following day, when a solution of 4% formalin, Rose Bengal, and water was added to each. The samples were allowed to sit for one week. Each bag was then rinsed through two nested sieves, which allowed for the removal of large vegetation and any clearly visible large invertebrates. All invertebrates removed at this point were placed in a jar containing 70% ethanol.

The remainder of a sample, which consisted of fine organic matter, detritus, decaying vegetation, and invertebrates, was subsampled for community structure analysis. A pilot study revealed that 1/4 of each sample from the semi-native pastures and 1/32 of each sample from the improved pastures was sufficient to represent the invertebrate species abundance of the entire sample. The difference in the subsample proportions was due to the greater amount of organic matter in the improved pasture samples. Subsamples were collected from a sample-splitting device, and all invertebrates in a particular subsample were removed and placed in a vial of 70% ethanol.

Regardless of the portion of each sample split and sorted, fixed count subsamples of at least 100 individuals were removed from each vial (Plafkin et al. 1989, but see Barbour et al. 1999, King and Richardson 2002). Specimens were identified to the genus level whenever possible and to the family level at a minimum. In addition, taxa were assigned to functional feeding groups based on Merritt and Cummins (1996).

### Statistical Analysis

The effects of stocking density and pasture type on abiotic variables were analyzed with analysis of variance, whereas biotic variables were analyzed with Mann-Whitney and Kruskal-Wallis non-parametric tests, given the non-normal distribution of the data. The analyses on abiotic variables were conducted for each date separately, as data from the two sampling dates could not be considered independent of each other. For both the abiotic and biotic data sets, one treatment level (zero cattle) in the semi-native pastures could not be replicated because one of the replicate pastures did not contain any hydrated wetlands (the other replicate pasture contained two wetlands, but they represent pseudoreplicates). As a consequence, we view significant results from these tests as suggestive but do not definitively ascribe significant differences to the treatment. In addition, Mann-Whitney and Kruskal-Wallis tests were performed specifically on the relative abundances of the family Chironomidae because they have been shown previously to be good indicators of wetland condition in south Florida (King and Richardson 2002).

Principal components analysis (PCA) was performed on chemical/physical data collected during August and September separately. Water depth, temperature, DO, pH, specific conductance, color, ammonium-N, nitrate/nitrite-N, soluble reactive phosphorus (SRP), total Kjeldahl-N (TKN), and total phosphorus (TP) were included in both analyses. In order to condense this complex data set, principal components (PCs) were derived from the correlation matrix (R) for each analysis. Four sites were excluded from the August analysis and three from the September analysis due to missing data. One missing temperature datum (site S6) was replaced with the August mean of all sites sampled and used in the August analysis.

Invertebrate community structure was characterized by taxon richness, the relative abundance of individual taxa, the relative abundance of different trophic guilds, and taxon diversity (Simpson's index). We included count data of core taxa, defined as those taxa comprising at least 5% of the relative abundance of taxa in at least one pasture, in a correspondence analysis (CA) of all 24 wetlands. Pearson correlation (SYSTAT version 5.0, Evanston, Illinois) was then used to relate dimensions to nutrient levels. Significance was established a priori at $p < 0.05$. Spearman's rho also was used to relate stocking density to nutrient concentrations in the wetlands, as stocking rates were treated as rank-order data. All statistical analyses except the correlations and non-parametric tests were conducted using SAS (version 8).

## RESULTS

### Invertebrate Community Structure

Seventy-eight aquatic invertebrate taxa, representing forty-five families, twelve orders, and two classes, were identified from the wetlands. Fourteen of the 78 taxa had relative abundances exceeding 5% of the total invertebrate abundance in any one sample; of these core taxa, the invertebrate families that were found in

Table 2. Mean values of nutrient concentrations and ANOVA results testing for the effect of stocking number for improved and semi-native pastures on two sampling dates. $NO_3$ data were below detection for most samples, so data are not reported. Units for nutrient data are in mg/L. The value for the semi-native control treatment (asterisk) is the mean of two wetlands from the same pasture (pseudo-replicated; see text for explanation). P-values less than 0.10 are in bold.

| Nutrient | Treatment Level (cows per pasture) | Mean Value | F-value (p) [df] | Mean Value | F-value (p) [df] |
|---|---|---|---|---|---|
| | | August 1, 2002 | | September 24, 2001 | |
| | | Improved Pasture | | | |
| Ortho-$PO_4$ | 0 | 2.15 | 0.31 | 0.915 | 3.08 |
| | 15 | 1.59 | (p = 0.90) | 0.268 | (p = 0.15) |
| | 20 | 1.47 | [df = 3] | 0.155 | [df = 3] |
| | 35 | 0.96 | | 0.144 | |
| Total Phosphorus | 0 | 2.33 | 0.17 | 1.31 | 1.82 |
| | 15 | 1.63 | (p = 0.90) | 0.52 | (p = 0.28) |
| | 20 | 1.89 | [df = 3] | 0.42 | [df = 3] |
| | 35 | 1.51 | | 0.49 | |
| $NH_4$ | 0 | 1.45 | 0.50 | 0.09 | 0.96 |
| | 15 | 0.26 | (p = 0.70) | 0.05 | (p = 0.49) |
| | 20 | 0.28 | [df = 3] | 0.05 | [df = 3] |
| | 35 | 0.32 | | 0.05 | |
| TKN | 0 | 7.10 | 0.49 | 5.32 | 0.82 |
| | 15 | 6.05 | (p = 0.71) | 3.35 | (p = 0.82) |
| | 20 | 5.31 | [df = 3] | 3.45 | [df = 3] |
| | 35 | 8.93 | | 5.94 | |
| | | Semi-native Pastures | | | |
| Ortho-$PO_4$ | 0 | 0.25* | 7.19 | 0.085 | 5.96 |
| | 15 | 0.09 | (p = 0.12) | 0.048 | (p = 0.23) |
| | 20 | 0.08 | [df = 3] | 0.043 | [df = 3] |
| | 35 | 0.07 | | 0.038 | |
| Total Phosphorus | 0 | 0.60* | **16.12** | 0.14 | 0.21 |
| | 15 | 0.22 | **(p = 0.09)** | 0.12 | (p = 0.88) |
| | 20 | 0.27 | [df = 3] | 0.12 | [df = 3] |
| | 35 | 0.19 | | 0.09 | |
| $NH_4$ | 0 | 0.47* | **370.5** | 0.06 | **6.43** |
| | 15 | 0.07 | **(p = 0.008)** | 0.04 | **(p = 0.008)** |
| | 20 | 0.06 | [df = 3] | 0.04 | [df = 3] |
| | 35 | 0.01 | | 0.04 | |
| TKN | 0 | 6.98* | 5.13 | 3.88 | 0.40 |
| | 15 | 4.85 | (p = 0.16) | 2.54 | (p = 0.76) |
| | 20 | 6.05 | [df = 3] | 3.32 | [df = 3] |
| | 35 | 4.00 | | 2.97 | |

the most wetlands were Chironominae, Tanypodinae, and Sididae.

### Stocking Number

Nutrient concentrations were relatively high in the water column of the wetlands (Table 2). However, stocking number had a marginal or statistically significant effect in only three analyses (out of 20; TP and $NH_4$ on August 1, and $NH_4$ on September 24). All three significant effects were observed in the semi-native pastures. Contrary to our expectation, mean nutrient concentrations were almost always greater in the control pastures (zero cows) than the pastures with the highest number of cows (Table 2). The high values in the control, improved pasture were not measured in both replicates; the pasture with wetlands dominated by maidencane had much higher nutrient concentrations (except for $NO_3$) than its replicate (Table 3). No statistically significant correlations were found between stocking density and nutrient concentrations ($p > 0.05$; $n = 16$).

Stocking number had no statistically significant effect on invertebrate taxon richness or diversity, re-

Table 3. Mean nutrient concentrations and physical variables (± SD) in two different pasture types at the MacArthur Agro-ecology Research Center. Nutrient data in parentheses are specific to the wetlands in improved pasture dominated by maidencane (see Table 1); these values are included in the overall mean for improved pastures. Units for nutrient data are in mg/L. Asterisks indicate a statistically significant difference: * = $p < 0.10$; ** = $p < 0.05$; *** = $p < 0.01$ for improved vs. semi-native pasture within a sampling date.

| Variable | August 1, 2001 Improved Pasture | August 1, 2001 Semi-native Pasture | September 24, 2001 Improved Pasture | September 24, 2001 Semi-native Pasture |
|---|---|---|---|---|
| Ortho-$PO_4$ | 1.62 ± 0.85<br>(3.30 ± 1.59) | 0.11 ± 0.07*** | 0.37 ± 0.41<br>(1.33 ± 1.11) | 0.05 ± 0.02* |
| Total Phosphorus | 1.90 ± 0.85<br>(3.61 ± 1.35) | 0.28 ± 0.16*** | 0.68 ± 0.51<br>(1.88 ± 1.23) | 0.11 ± 0.04** |
| $NH_4$ | 0.61 ± 0.99<br>(2.81 ± 0.37) | 0.13 ± 0.17 | 0.06 ± 0.03<br>(0.13 ± 0.08) | 0.04 ± 0.01 |
| $NO_2$-$NO_3$ | 0.010 ± 0.009<br>(0.004 ± 0.003) | 0.008 ± 0.006 | 0.002 ± 0.002<br>(0.005 ± 0.001) | 0.002 ± 0.002 |
| TKN | 6.55 ± 2.26<br>(9.96 ± 1.43) | 5.12 ± 1.26 | 4.51 ± 1.97<br>(6.49 ± 0.36) | 3.08 ± 0.89* |

gardless of pasture type (Table 4). In the improved pastures, the highest mean values were observed in the zero and high stocking treatments; in the semi-native pastures, no trend was evident with respect to stocking number (Table 4).

### Pasture Type and Vegetation

Regardless of sampling date, mean nutrient concentrations were consistently higher in the improved vs. semi-native pastures (Table 3). This difference was particularly evident for both dissolved and total phosphorus and was demonstrated to a lesser extent with nitrogen, as well. The higher nutrient concentrations in improved pasture wetlands were consistent within stocking treatments (Table 2).

Invertebrate taxon richness and diversity were significantly greater overall in semi-native pastures—the ones with lower nutrient concentrations—than improved pastures (Table 5). However, vegetation type within a wetland influenced this outcome, as determined by three separate analyses: 1) maidencane vs. mixed vegetation in improved pastures only; 2) maidencane in improved vs. maidencane in semi-native pastures; and 3) mixed vegetation in improved vs. maidencane in semi-native pastures (Table 5). Trophic guild diversity was marginally greater in improved pasture wetlands dominated by maidencane than in those with mixed vegetation; the other indicators showed no statistically significant differences between these two improved pasture wetland vegetation types. A marginally significant difference was noted between maidencane in improved than in semi-native pastures, with taxon richness being higher in semi-native pastures (Table 5); again, the other indicators were not significantly different in this comparison. Finally, invertebrate diversity and richness were greater in the semi-native pasture with maidencane than in the pastures with wetlands dominated by a mixed vegetation (*Juncus*/*Polygonum*) community (Table 5).

Mean relative abundance of chironomids showed the opposite pattern to the invertebrate community structure indices. In improved (high nutrient) pastures, chironomid relative abundance was significantly greater in wetlands with mixed vegetation than in those dominated by maidencane (Table 5). In comparisons of improved vs. semi-native pastures, mean chironomid relative abundance was greater in the semi-native pastures, irrespective of vegetation type (Table 5).

When vegetation type was ignored, invertebrate relative abundances were relatively similar between the improved and semi-native pastures, with chironomids dominating in both wetland types (Table 6). The Chironominae accounted for 56.4% and 43.0%, and the Tanypodinae accounted for 11.0% and 5.5% of invertebrate relative abundance in the improved and semi-native pastures, respectively.

The wetlands became hydrated in July, 2001 and maintained a minimum mean water depth of 20 cm in the improved pastures throughout the sampling period, despite water losses to evapotranspiration in late August (Figure 1). However, mean water depth in the semi-native pasture wetlands did reach zero briefly in early September, but the wetlands were rehydrated soon thereafter. Overall, water depths were significantly greater in improved than semi-native pastures (Figure 1). On August 1, 2001, mean depths ranged from 17 to 62 cm in the improved pastures and from 15 to 25 cm in the semi-native pasture wetlands. On September 24, 2001, mean water depths ranged from 10 to 56 cm and from 13 to 27 cm in the improved and semi-native pasture wetlands, respectively. Mean

Table 4. Mean values of invertebrate taxon richness and Simpson's diversity index for taxa and trophic guilds and results of the Kruskal-Wallis test (H) evaluating the effect of stocking number (0, 15, 20, or 35 cows per pasture). Pasture type (improved and semi-native) were analyzed separately. The value for the semi-native control treatment (asterisk) is the mean of two wetlands from the same pasture (pseudo-replicated; see text for explanation).

| Metric | Treatment Level (cows per pasture) | Mean Value | H (p) |
|---|---|---|---|
| Improved Pasture | | | |
| Taxon Richness | 0 | 28 | 2.0 (0.57) |
| | 15 | 31 | |
| | 20 | 24.5 | |
| | 35 | 29.5 | |
| Simpson's Taxa | 0 | 3.30 | 3.50 (0.32) |
| | 15 | 2.37 | |
| | 20 | 2.89 | |
| | 35 | 4.59 | |
| Simpson's trophic guild | 0 | 2.05 | 3.50 (0.32) |
| | 15 | 1.58 | |
| | 20 | 1.83 | |
| | 35 | 2.21 | |
| Semi-native Pasture | | | |
| Taxon Richness | 0 | 33* | 0.17 (0.98) |
| | 15 | 33 | |
| | 20 | 32.5 | |
| | 35 | 32.5 | |
| Simpson's Taxa | 0 | 4.93* | 1.17 (0.7) |
| | 15 | 4.16 | |
| | 20 | 5.33 | |
| | 35 | 5.61 | |
| Simpson's trophic guild | 0 | 3.33* | 1.50 (0.68) |
| | 15 | 2.61 | |
| | 20 | 3.01 | |
| | 35 | 2.70 | |

concentrations of nutrients were consistently greater on August 1 than September 24th (Tables 2, 4). This pattern was generally consistent for all nutrients, at all stocking treatment levels, and for both pasture types (Table 2).

## Multivariate Analyses

Principal component analysis was used to examine the relationship of site distribution and abiotic variables. For the August data, PC1 and PC2 explained 39% and 18% of the total variance, respectively (Figure 2). PC1 clearly separated the improved from the semi-native pasture wetlands, with the improved pasture wetlands positively related to increasing phosphorus, depth, and conductivity. PC2 was effective at separating out the two improved, maidencane wetlands

Table 5. Mean values of invertebrate taxon richness, Simpson's diversity index for taxa and trophic guilds, and chironomid relative abundance, and results of the Mann-Whitney test evaluating the effect of pasture type (improved vs. semi-native) and vegetation type (maidencane vs. mixed vegetation). Results with a P-value of less than 0.10 are in bold. Analyses involving maidencane, semi-native pastures required use of data from two wetlands within the same pasture (see text for details).

| | | Taxon Richness | | Simpson's: Taxa | | Simpson's: Trophic Guild | | Chironomid Relative Abundance | |
|---|---|---|---|---|---|---|---|---|---|
| Contrast | N | Mean | u (p) | Mean | u (p) | Mean | u (p) | Mean | u (p) |
| Improved vs. semi-native | 8–7 | 28.31 vs. 32.56 | **10.5 (=0.049)** | 3.29 vs. 5.01 | **11.0 (=0.048)** | 1.92 vs. 2.91 | **1.0 (<0.01)** | 64.65 vs. 46.36 | **49 (0.04)** |
| Maidencane vs. mixed (improved pasture only) | 2–7 | 27.5 vs. 28.43 | 4.5 (0.46) | 4.64 vs. 3.09 | 12.0 (0.14) | 2.56 vs. 1.82 | **13.0 (0.08)** | 23.61 vs. 70.51 | **0 (0.04)** |
| Maidencane (improved vs. semi-native) | 2–7 | 27.5 vs. 23.56 | **1.0 (0.08)** | 4.64 vs. 5.01 | 5.0 (0.56) | 2.56 vs. 2.91 | 4.0 (0.38) | 23.61 vs. 46.36 | **0 (0.04)** |
| Mixed (improved vs. maidencane (semi-native) | 7–7 | 28.43 vs. 32.56 | **38.5 (0.07)** | 3.09 vs. 5.01 | **42 (0.03)** | 1.82 vs. 2.91 | **49 (0.002)** | 70.51 vs. 46.36 | **49 (<0.01)** |

Table 6. The most abundant invertebrate taxa (>5% relative abundance in any one sample) collected from wetlands in this study. Code refers to labels in Figure 4.

| Order | Family | Genus | Code | Relative Abundance (%) Improved Pastures | Relative Abundance (%) Semi-native Pastures |
|---|---|---|---|---|---|
| Coleoptera (adult) | Noteridae | *Hydrocanthus* | NA | 1.3 | 4.7 |
| Coleoptera (larvae) | Helodidae | "a" | HL | 0.2 | 4.3 |
| Coleoptera (larvae) | Noteridae | "a" | NL | 0.1 | 1.5 |
| Diptera (larvae) | Ceratopogonidae | "a" | CE | 3.1 | <0.1 |
| Diptera (larvae) | Chironomidae | "a" | CH | 56.4 | 43.0 |
| Diptera (larvae) | Culicidae | "b" | CU | 3.3 | 6.2 |
| Diptera (larvae) | Tanypodinae | "a" | TA | 11.0 | 5.5 |
| Hemiptera | Belostomatidae | *Belostoma* | BE | 1.2 | 4.1 |
| Hemiptera | Octheridae | "a" | OC | 0.1 | 1.8 |
| Ostracoda | | "a" | OS | 6.2 | 3.2 |
| Cladocera | Calanoida | "b" | CA | 0.1 | 0.1 |
| Cladocera | Sididae | | SI | 3.0 | 7.1 |
| Cladocera | Cyclopoida | | CY | 0.1 | <0.1 |

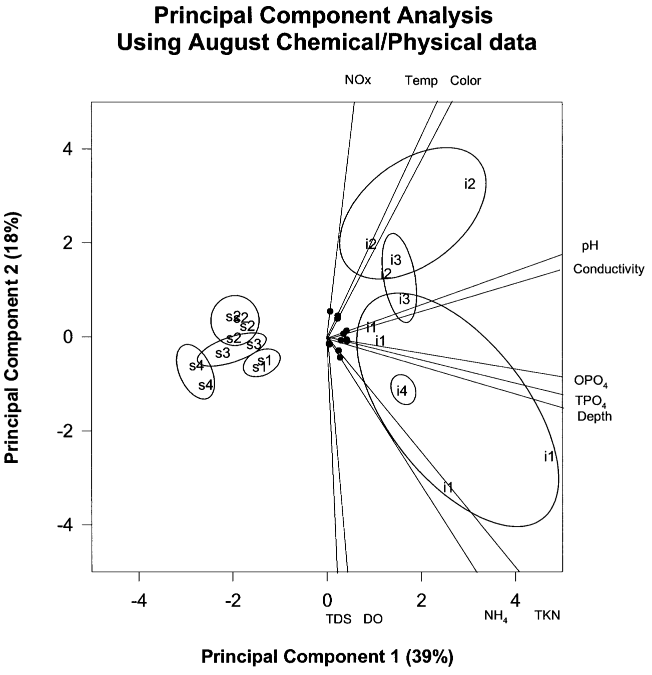

Figure 2. Biplot results of principal components analysis using the physical and chemical data collected in August from wetlands in the improved and semi-native pastures. Sites are coded by season (i = improved; s = semi-native) and stocking density (1 = 0 cow-calf pairs/pasture; 2 = 15 cow-calf pairs/pasture; 3 = 20 cow-calf pairs/pasture; 4 = 35 cow-calf pairs/pasture). Lines represent vectors associated with environmental data; they are extended to each axis for ease of interpretation. Dots represent coordinates of each vector. See text for discussion of most important parameters.

("i1"s in lower right hand corner), based largely on their relatively high $NH_4$ and TKN concentrations. PCA of the September data was not as effective at separating the improved and semi-native pasture wetlands (Figure 3), although PC1 and PC2 explained approximately the same amount of variance as in the analysis based on August data. The most distinct separations involved the improved, maidencane wetlands

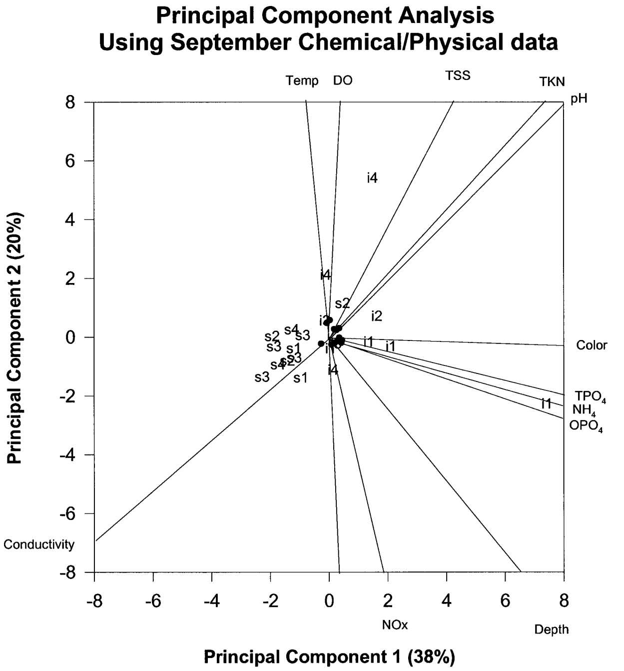

Figure 3. Biplot results of principal components analysis using the physical and chemical data collected in September from wetlands in the improved and semi-native pastures. Symbols and explanation of plot design same as in Figure 2.

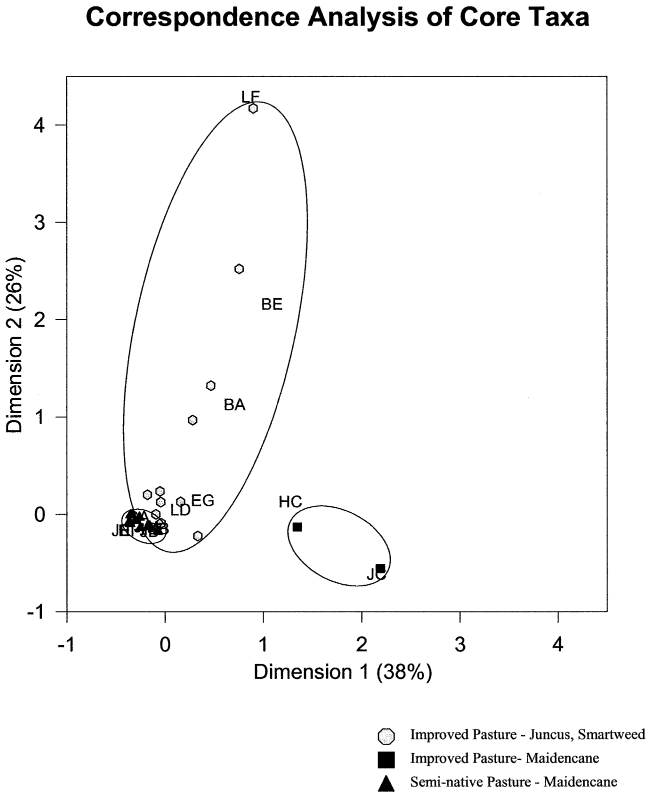

Figure 4. Biplot results of correspondence analysis using the invertebrate community structure. Invertebrate taxa are coded by letters (see Table 6). Sites are coded by symbols.

reflecting their elevated nutrient status, and the improved wetlands exposed to the highest stocking density, largely because of their elevated dissolved oxygen levels.

Correspondence analysis was used to examine the relationship between wetland type and the core invertebrate taxa (Figure 4). Dimensions 1 and 2 explained 38% and 26% of the variance, respectively. Dimension 1 reflected a nutrient gradient, as it was significantly correlated with ammonium ($p < 0.001$) and marginally correlated with SRP ($p=0.07$). Ostracods (OS) and Culicidae larvae (CU) corresponded with the high nutrient, maidencane-dominated wetlands in the improved pasture. Dimension 2 appeared to correspond with vegetation type, as the maidencane sites were found near the bottom of this dimension and the mixed (*Juncus*/*Polygonum*) vegetation sites were located near the top of the dimension.

## DISCUSSION

### Stocking Number

The lack of an effect of cattle stocking density on water quality or invertebrate community structure was surprising, given that cattle can remove substantial amounts of vegetation through herbivory and trampling and return substantial amounts of labile nutrients to the landscape in the form of urine and feces (Archer and Smeins 1991). Indeed, cattle may defecate up to 14 times in a 24-hr period (Brown and Archer 1987). Although cattle grazing does not result in a net increase of nutrients to a closed system (Scrimgeour and Kendall 2002), the grazing and defecation processes alter the forms of key nutrients, often making them more bioavailable. We anticipated that this change in the form and availability of nutrients would influence invertebrate community structure (cf. Rader and Richardson 1994, King and Brazner 1999, Del Rosario et al. 2002; but see Spieles and Mitsch 2000). The overall lack of invertebrate response to cattle grazing likely results from a combination of factors, including (1) the overriding influence of prior pasture land use (see below), (2) the environmental constraints associated with adapting to life in seasonal wetlands, as has been shown for vegetation (cf. Seabloom et al. 2001), and (3) the limited impact of cattle on wetland chemistry. The lack of a relationship between cattle number and nutrient concentration may have been because the cattle spent relatively little time in the wetland areas of each pasture, especially when the wetlands were dry. We did not quantify cattle movement within pastures. Another possibility that we were unable to test is that long-term differences in vegetation between the two pasture types may have been influenced by the timing of cattle grazing in these systems (summer in the improved pastures; winter in the semi-native pastures). If this was the case, then some of the subtle influence of wetland vegetation on invertebrate communities (see below) may have been related to long-term effects of grazing prior to establishment of the experimental treatments.

Given that some of the highest nutrient concentrations were measured in the reference wetlands where there were no cattle, it appears that pasture land use influenced nutrient concentrations more than grazing in these systems. In areas that previously experienced high stocking densities and, perhaps more importantly, that were exposed to long-term fertilizer inputs to the surrounding pastures, the cumulative impact of high nutrient loads apparently left a legacy that overrode the affect of the current cattle stocking treatments. The role of prior land use needs to be taken into account when areas are selected for restoration. For example, the Comprehensive Everglades Restoration Plan, a 25-year, $8 billion effort to restore the greater Florida Everglades (www.evergladesplan.org), calls for thousands of hectares in the Lake Okeechobee watershed to be purchased for the creation and/or restoration of wetlands. Much of the land use in this region is pasture for cattle production (Flaig and Havens 1995, Steinman and Rosen

2000). It is critical that areas selected for rehydration have suitable soil chemistries so that they serve as a phosphorus sink and not as a source (cf. Reddy et al. 1995).

### Pasture Type and Vegetation

The effect of improved vs. semi-native pasture type on invertebrates was complex and confounded by the differences in vegetation among the wetlands. The greater invertebrate taxon richness and diversity in semi-native vs. improved pastures, despite the lower nutrient concentrations, provides evidence that the type of pasture land use has a strong influence on wetland invertebrate structure in these subtropical ecosystems. Vegetation type also influenced invertebrate community structure, however. The most distinct difference occurred when comparisons included contrasting vegetation type and pasture land use; we measured greater invertebrate richness and diversity in maidencane wetlands on semi-native pastures compared to mixed vegetation wetlands on improved pastures (Table 5). Increased productivity associated with greater nutrients in the improved wetlands may have resulted in competitive exclusion of some taxa (cf. Huston 1994), resulting in reduced diversity. However, when contrasts included (1) different vegetation type but pastures of the same land use or (2) the same vegetation type but pastures of different land use, only marginal or non-significant differences were detected. This suggests that the influence of land use is muted to some degree by vegetation type and vice versa.

Studies in other wetlands have shown that vegetation type and morphology alone can influence the growth and distribution of invertebrates (Hargeby 1990, McLaughlin and Harris 1990, Streever et al. 1995, Burton et al. 2002). Wetland sediments dominated by maidencane usually contain a thick layer of roots and organic matter (Swarzenski et al. 1991), which likely presents a habitat very different in terms of physical and chemical structure compared to wetlands dominated by *Juncus* and *Polygonum*. However, our contrasts using maidencane wetlands from improved pastures must be qualified because of the lack of replicates and potential sampling bias; the thick layer of roots in the maidencane wetlands impacted sampling efficacy near the sediments compared with the *Juncus*/*Polygonum* wetlands. This may have resulted in a greater number of individuals of Chironominae and Tanypodinae sampled from the *Juncus*/*Polygonum* wetlands, where sweeps were less impeded by the standing vegetation.

The multivariate analyses showed that the improved pasture wetlands were much more loosely grouped than the semi-native pasture wetlands. It is likely that the greater diversity of habitat type and wider range of physical and chemical conditions in the improved pasture wetlands contributed to the looser aggregation of sites based on the invertebrate core taxa (Figure 4). In addition, increased disturbance due to more intensive ranching practices in improved pasture wetlands and possibly also the presence of cattle during the wet season may have resulted in greater variability when compared to the less disturbed, more homogenous wetland sites in the semi-native pastures.

### Bioassessment Value of Invertebrate Data

The correspondence analysis revealed clear associations between invertebrate distribution and environmental conditions and resulted in a distinct separation between the different types of wetlands. In particular, the presence of ostracods, copepods, and Culicidae larvae helped discriminate the improved, maidencane wetlands from both the improved, *Juncus*/*Polygonum* wetlands and the semi-native, maidencane wetlands. The improved, maidencane wetlands had higher nutrient ($NH_4$ and TP, especially) concentrations than the other wetlands, suggesting ostracods and Culicidae larvae may be useful as indicators of eutrophic, subtropical wetlands. Rader and Richardson (1994) also found that ostracods were much more abundant in enriched than unenriched regions of the Florida Everglades.

King and Richardson (2002) presented compelling evidence on the utility of species-level Chironomidae data for bioassessment in wetlands. Although they found genus and species-level information on chironomids to be the most useful, our data suggest that family-level information can be of value. The relative abundance of Chironomidae was greater in the improved, *Juncus*/*Polygonum* wetlands compared to other wetland types—this pattern did not emerge when the analyses included either the entire invertebrate community or just the most abundant invertebrate taxa. Clearly, taxonomic resolution should be conducted to the finest level of resolution practicable (Hawkins et al. 2000, Lenat and Resh 2001, King and Richardson 2002), but our data indicate that, in some circumstances, family-level information may be sufficient to discriminate among impacts or wetlands.

### Role of Other Factors

Hydrology, and hydroperiod in particular, plays a critical role in the ecology of wetlands (Wilcox 1993, 1995, Keough et al. 1999, Brooks 2000, Wilcox et al. 2002), as hydrology influences the habitat provided by plants, which in turn has an impact on invertebrates and higher trophic levels (Rader and Richardson 1992, Dahm et al. 1995). The wetlands sampled in the present study did not fill with water until late July, and water quality sampling began the following week. It is likely that the higher mean nutrient concentrations in August (one week after the wetlands became hydrated), compared to late September, represents a ''flushing'' of nutrients from the exposed soils to the overlying water column (Qiu and McComb 1994).

Hydrology also will influence aquatic invertebrate communities, which depend upon flooded conditions for habitat growth and reproduction. Seasonally flooded systems are extremely dynamic, and length of hydroperiod, date of first flooding, and timing of drawdown can influence invertebrate communities. Longer periods of flooding lead to the development of different communities than those present in systems with short periods of inundation (Murkin and Ross 2000). In south Florida, there is large interannual variability in rainfall and thus, hydroperiod, in seasonal ponds. The hydroperiods on our study wetlands varied greatly between 2000, which was the driest year on record, and 2001, which was a normal rainfall year (Figure 1). The potential effect of hydroperiod on invertebrate ecology needs to be considered when evaluating their community structure, and this is especially challenging in temporally variable, seasonally flooded systems.

Invertebrate community structure will be, in part, a function of its food source. One of the most direct impacts of nutrient additions to wetlands is a change in algal community structure (Rader and Richardson 1992, McCormick and O'Dell 1996, McDougal et al. 1997, Havens et al. 1999). Nutrient addition also can impact the quality of detritus (Rosemond et al. 2001). Algal data are not yet quantified, but it is possible that the differences in invertebrate structure were related to the availability and quality of the algal and detrital food resources (Steinman 1996, Murkin and Ross 2000).

In summary, the results from this study indicate that prior pasture land use had a large impact on nutrient conditions and invertebrate communities in wetlands of subtropical pastures, and this effect likely overwhelmed the effect of more recently imposed cattle stocking density treatments in these pastures. This finding has implications for future restoration efforts in this region, as areas heavily loaded with phosphorus from past land practices would make poor candidates for stormwater treatment areas or detention basins. Future studies using grazer exclosures, manipulation of vegetation type and hydroperiod, and the use of a broader range of stocking densities should provide better information on the potential influence of cattle grazing on invertebrates and nutrient status in these wetlands.

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