ABSTRACT: Agricultural drainage ditches are a reflection of the disturbance caused by agriculture and other human perturbations associated with agricultural activities, and the density of them can be seen as an important gradient reflecting local disturbance. However, no studies to date have examined the changes in wetland communities in relation to drainage ditch densities, yet such information is urgently needed for the conservation of wetland ecosystems facing intensive cultivation. In this paper, we inventoried 67 plots at four wetland mosaics with the ditch density values ranged 0–3.6 km km⁻² in the Sanjiang Plain, and the species richness, composition and diversity were compared among those four sites. Linear regression was used to explore relationships between wetland community pattern and agricultural ditch density, and the results show that there is significant relationship ($r^2 = 0.56$, $P < 0.001$). The diversity comparisons show that there exist a clear negative relationship between ditch density and species diversity indices, and the species diversity did not differ greatly among sites, but species composition varied considerably. With increasing ditch density, an increasing loss of indigenous wetland species paralleled with an increasing incursion of upland species. Management implication from the drainage ditches is that the density of 1.2 km km⁻² be the maximum value suitable for the protection of native wetlands in Sanjiang plain.

KEY WORDS: Ditch density, wetland drainage, vegetation composition, species diversity

1. INTRODUCTION

Wetland ecosystems are a natural resource of global significance, and their high level of plant and animal diversity is perhaps the major reason why wetland protection has become a high priority worldwide (Bobbink et al. 2006). However, wetlands are degrading at ever faster rates as a result of a wide variety of human impacts. The conflict between providing for agricultural drainage and maintaining wetland areas has been intense and ongoing in many parts of the world (Walters and Shrubsole 2003). Agricultural drainage ditches have been, and are, a well-established agricultural practice, but they inevitably inflict strong and potentially devastating ecological effects on the natural wetland environments (Walters and Shrubsole 2003, Langheinrich et al. 2004). Since the 1970’s, studies on agricultural ditches have shown that drainage ditch use practices in wetland area may severely threaten the integrity and biodiversity of these vulnerable ecosystems (Hill 1976, Milsom et al. 2004). The benefit of drainage ditches often comes at the cost of functions...
that wetlands provide by decreasing total wetland area and average patch size (Spaling 1995), changing the hydrological regime (Rowell 1986, Gavin 2003, Scholz and Trepel 2004), acting as a major conduit of agricultural pollutants (Moore et al. 2001, Nguyen and Sukias 2002), fragmenting or even eliminating wildlife populations (Costanza 1997, Gibbs 2000), fostering invasive species spread (Galatowitsch et al. 2000), and increasing human use of adjacent areas (Bardecki 1988).

Agricultural drainage ditches are anthropogenic disturbances that impose distinct patterns on landscapes and influence a wide range of ecological processes. When appropriately selected, drainage ditch density gradient represents an ideal opportunity for differentiating the long-term effects of agricultural activity from other environmental patterns. Unfortunately, to date, no published studies have assessed the variation in species composition and diversity of wetland communities along the agricultural drainage ditch density gradient, yet such information is urgently needed given the conservation value of wetlands. Thus, for a better understanding of the composition and structure of wetland vegetation, and for mitigation and management purposes, it is imperative that the impact of drainage ditch density on wetlands has to be ascertained.

The goal of this paper is to assess and characterize wetland vegetation patterns under different ditch density at four sites that represent different processes of disturbance, and to elucidate their relationship to agricultural ditch density gradient. Our specific objectives were to: (1) examine whether or not there are variations with respect to species diversity and composition along the agricultural ditch density gradient; (2) determine appropriate ditch density within the study area by analyzing vegetation patterns at different sites, and thereby, make a contribution to wetland conservation in the study region.

2. STUDY AREA

Sanjiang Plain is located in the eastern part of Heilongjiang province, Northeast China and is bordered by Russia. It is well known for high concentration and diversity of wetlands, which provide crucial habitat for waterfowl and other wildlife, and play a fundamental role in maintaining the natural

Fig. 1. The study area in the Heilongjiang province, Northeast China. The main distribution pattern of the agricultural drainage ditches are indicated.
Wetland communities along ditch density gradient

functions of this region (Yang 1989). Sanjiang Plain covers an area of \(10.89 \times 10^6\) ha, which was dominated by marshes in 1893 (Zhao 1999, Liu and Ma 2000). However, since the 1950’s, this region has been fragmented by agricultural activity. The drainage of marshes for arable cultivation such as rice paddies and upland crops has resulted in the dramatic increase in cultivated land from about \(2.9 \times 10^4\) ha in 1893 to \(4.57 \times 10^6\) ha in 1994 (Liu and Ma 2000).

Our study was conducted within a 1900 km\(^2\) region (47°30’–48°00’ N, 133°20’–134°00’ E) located in Honghe and Qianfeng farmland near a native wetland natural reserve in Sanjiang Plain (Fig. 1). In the study region, the native wetland reserve is nearly natural, while the other areas have changed from wetlands to agricultural landscapes, with only a few remnant wetland mosaics present. The agricultural land use in this area is of two main types, rice paddies and upland crops. In general, as agricultural ditch density increases, the area available to farming increased parallel to the reduction of wetlands, and the area of upland corps decreases while rice paddies increase.

The average altitude above sea level in our study area is about 60 m, and the relative difference is modest (<10 m). The mean annual precipitation is around 600 mm and the mean annual temperature is 1.91°C. Water and soil in freshwater wetlands are completely frozen from October to April and begin to melt in late April, with the highest temperature occurring in July.

The type of vegetation varies from Deyeuxia angustifolia (Kom.) Y. L. Chang to Calamagrostis neglecta Ehrh. to Carex pseudocuraica Fr. Schm. as the standing water depth increases significantly in the study area.

3. MATERIAL AND METHODS

3.1. Sampling

Site selection for vegetation analysis was predetermined using the TM image of 2004 and the ditch infrastructure data set at a 1:100,000 scale for 2002. We digitized the ditches manually from image before using them in the geographic information system (GIS). Ditch density was calculated as the length of ditches for each grid cell of 600 m \(\times\) 600 m. Thus the study area is partitioned into

![Pattern of agricultural ditch density in the study area. The ditch density is calculated for each grid cell of 600 \(\times\) 600 m. Points indicate locations of four vegetation sampling transects.](image)
6 ranks with different ditch density (Fig. 2). With increasing ditch density, wetlands shifted from consisting of many clustered annular wetlands to fewer and more isolated wetland mosaics. This resulted in the proportion of wetland in the landscape shifting from more than 90% in rank of 0–0.6 km km\(^{-2}\) to less than 5% in rank of 1.8–2.4 km km\(^{-2}\) to almost extinct in rank of 3.0–3.6 km km\(^{-2}\). Only those ranks with single wetland mosaic >0.4 ha were selected for vegetation sampling, because, as noted by Gibbs (2000), 0.4 ha is the minimal sufficient area to sustain wetland biota. The ranks with ditch density larger than 2.4 km km\(^{-2}\) were not selected because the wetland mosaics in them did not meet the size criteria. Thus, only four sites were chosen for vegetation sampling (Fig. 2). Of those four sites, site A is located in the wetland reserve with nearly no disturbance, site B is bordered by agricultural fields on the north and south sides, while the other two sites are surrounded by agricultural fields.

Vegetation surveys were carried out in July of 2005, at the peak of vegetative cover and species richness. At each site, one annular wetland mosaic was selected and one transect was set from the edge to the center to include a majority of species. Homogeneous sample plots (1 m × 1 m) were set up along each transect with points at about 10 m intervals. In each plot, all species were identified and measured for coverage, total height and frequency. A total of 67 plots were sampled (Table 1).

We also scored each plot during the survey on a scale of ditch density, ranging from the minimal density to maximal density of the rank to which it belongs. Our assessment combined multiple observations on the distance to the center, water abstraction, and species construction. Thus, for example, in site B, the center plot of the wetland obtained a score of 0.6, while the nearest plot to the road obtained the score of 1.2. Those plots between them may obtain intermediate score of 0.9.

### 3.2. Data analysis

A relative importance value for each species was computed as an average of the relative cover, relative height and relative frequency. Thus, a floristic data matrix of 102 species (67 × 102) was made for diversity indices calculation and multivariate analyses.

In order to investigate the range of variation in the data set, a detrended correspondence analysis (DCA) of the species data were performed. DCA is an indirect ordination technique, which ordinates the floristic data independently from environmental data (Kent and Coker 1992). Furthermore, to detect any general patterns in species composition between plots and agricultural ditch density gradient, the sample scores of the first DCA-axis were used as a response variable in a regression analysis where ditch density was used as a predictor variable.

To clarify the similarities in species composition between sites, Sørensen's similarity index expressed in % was used (Sørensen 1948). When calculated, species richness was log transformed to compensate for between site variations in study site area (Huston 1979, Nilsson et al. 1997).

The species-diversity indices were calculated for each plot, and the average value of the plots at each site was accepted as the diversity indices of each site. The Shannon information index (Shannon and Weaver 1949) was used to measure diversity on the basis of importance value, and richness was calculated as the Margalef’s index (Margalef 1958). The Pielou’s index (Pielou 1975) for the relative species evenness and the Simpson’s index (Simpson 1949) for the concentration of species dominance were

<table>
<thead>
<tr>
<th>Attributes</th>
<th>Site A</th>
<th>Site B</th>
<th>Site C</th>
<th>Site D</th>
</tr>
</thead>
<tbody>
<tr>
<td>Patch size (ha)</td>
<td>12.8</td>
<td>8.0</td>
<td>6.1</td>
<td>4.5</td>
</tr>
<tr>
<td>Ditch density (km km(^{-2}))</td>
<td>0–0.6</td>
<td>0.6–1.2</td>
<td>1.2–1.8</td>
<td>1.8–2.4</td>
</tr>
<tr>
<td>Plot number</td>
<td>22</td>
<td>18</td>
<td>16</td>
<td>11</td>
</tr>
</tbody>
</table>
also calculated. One-way analysis of variance (ANOVA) was used to assess statistical differences in various species diversity parameters between sites. All significance levels were reported at the $P \leq 0.05$ level.

The DCA analysis was performed with the software package CANOCO 4.5 for Windows (ter Braak and Smilenaer 2002). The default settings were used except that the option of down-weighting of rare taxa was selected. Linear regression was used to explore relationships between community parameters and drainage ditch density. One-way ANOVA was also used for comparisons of parameters between the four sampling sites. In all cases, we considered $P \leq 0.001$ as statistically significant.

Fig. 3. Detrended correspondence analysis (DCA) of the 67 plots of the four sites (see Fig. 2). Axis 1 and 2, respectively, represent for first and second coordinates of sites.

Fig. 4. Linear regression analyses between the first DCA axis and the level of agricultural ditch density.
4. RESULTS

The first two axes of DCA ordination illustrated the main patterns, and using the first and second axes, all plots were arranged in two dimensions (Fig. 3). Along the first axis, plots were arranged in the following order from left to right: plots of site A, site B, site C and site D. For the second axis, there was no clear grouping trend. Moreover, the DCA ordination revealed eigenvalues for the first, second, third, and fourth axes of 0.675, 0.453, 0.307, and 0.184, respectively; the eigenvalue for the first axis was much higher than that for the other axes. In other words, the second, third, and fourth axes did not sufficiently reflect variations in species composition. So we considered only the first axis of the DCA ordination, because it is sufficient to explain the main portion of variability in the data set. Linear regression was used to explore relationships between DCA-axis1 and drainage ditch density, and the results show that there is significant positive relationship (Fig. 4, $r^2 = 0.56$, $P < 0.001$).

A total of 102 species were found in the study. On site A, B, C and D, 42, 55, 37 and 36 species were identified, respectively. The species richness was increased under the site B ditch intensity as compared to site A, and the species number in site D is the lowest. It is discernible that species richness has a tendency to decline with increasing ditch density.

4.1. Species composition

Table 2 shows a comparison of plant density for the 10 most abundant species found in the study area and illustrates that plant community types varied widely among the four wetland mosaics. It can be seen that the four sites studied represented different combinations of species with different dominants and co-dominants. For example, Carex pseu-
docuraica Fr. Schm., Carex lasiocarpa Ehrh. and Menyanthes trifoliata L. were restricted to site C and D.

Community similarity between each site pair was measured by Seren sen’s index (Table 3). The similarity indices of pairwise comparisons ranged from 26 to 55%. In general, site A and site B were similar, but they are different to the other two similar sites. The appearance of this trend was due in part to the low ditch density in site A and B which resulted in the dominance of indigenous wetland species. The lowest similarity index was found between sites A and D, while the highest was found between sites A and B.

4.2. Species diversity

Table 4 shows the comparison of alpha diversity of each site type. In general, the pattern of diversity along the ditch density gradient indicated that alpha diversity indices decreased as the level of ditch density increased (Table 4). Margalef’s index of species richness ranged from 0.81 to 1.24, highest values were recorded at site B and the lowest at site D. A similar pattern was found for Shannon-Wiener diversity index, which ranged from 1.58 to 1.94. An opposite pattern was found for dominance index. In the present study, evenness was not a sensitive indicator of disturbance as all four sites had somewhat similar values.

The significant difference in diversity indices only exits between site B and other three sites, while the diversity of the other three sites has a similar pattern (Table 4). The linear regression fitted to the diversity and ditch density data achieved a relatively poor, yet significant, correlation (Table 5). This indicated that diversity and ditch density have significantly negative relationships. That is, as the level of ditch density increased, the diversity declined.

5. DISCUSSION

Present-day biodiversity crisis urges conservation biologists to elucidate the relationships between diversity and disturbance in various ecosystems (Connell 1978, Reice...
Table 2. Mean cover (m² 100 m⁻²) for the 10 plants representing the most abundant species by cover averaged across four sites (Fig. 2)

<table>
<thead>
<tr>
<th>Species</th>
<th>Site A</th>
<th>Site B</th>
<th>Site C</th>
<th>Site D</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>x</td>
<td>SE</td>
<td>x</td>
<td>SE</td>
</tr>
<tr>
<td>Carex pseudocarica Fr. Schm.</td>
<td>28.82</td>
<td>35.07</td>
<td>5.33</td>
<td>21.16</td>
</tr>
<tr>
<td>Glyceria spiculosa (Schmidt) Roshev</td>
<td>21.59</td>
<td>31.71</td>
<td>11.94</td>
<td>28.24</td>
</tr>
<tr>
<td>Carex lasiocarpa Ehrh.</td>
<td>12.00</td>
<td>24.48</td>
<td>18.89</td>
<td>27.20</td>
</tr>
<tr>
<td>Calamagrostis neglecta Ehrh.</td>
<td>8.85</td>
<td>24.91</td>
<td>26.78</td>
<td>32.13</td>
</tr>
<tr>
<td>Menyanthes trifoliata L.</td>
<td>8.67</td>
<td>20.93</td>
<td>0.33</td>
<td>3.54</td>
</tr>
<tr>
<td>Deyeuxia angustifolia (Kom.) Y. L. Chang</td>
<td>0.33</td>
<td>1.08</td>
<td>0.83</td>
<td>3.54</td>
</tr>
<tr>
<td>Carex appendiculata (Trautv.) Kükenth</td>
<td>0.00</td>
<td>0.00</td>
<td>3.87</td>
<td>13.26</td>
</tr>
<tr>
<td>Artemisia annua L.</td>
<td>0.00</td>
<td>0.00</td>
<td>0.00</td>
<td>0.00</td>
</tr>
<tr>
<td>Sanguisorba parviflora (Maxim.) Takeda</td>
<td>0.00</td>
<td>0.00</td>
<td>0.19</td>
<td>0.54</td>
</tr>
<tr>
<td>Caltha natans Pall.</td>
<td>0.00</td>
<td>0.00</td>
<td>0.00</td>
<td>0.00</td>
</tr>
</tbody>
</table>

Table 3. Similarities of plant composition for sites (Fig. 2) using Sørensen's similarity index

<table>
<thead>
<tr>
<th></th>
<th>Site B</th>
<th>Site C</th>
<th>Site D</th>
</tr>
</thead>
<tbody>
<tr>
<td>Site A</td>
<td>54.8%</td>
<td>35.2%</td>
<td>26.3%</td>
</tr>
<tr>
<td>Site B</td>
<td>36.5%</td>
<td>34.9%</td>
<td></td>
</tr>
<tr>
<td>Site C</td>
<td>44.9%</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Table 4. Pattern of species diversity in four sites (Fig. 2) according to increasing level of ditch density. Parameters in the same column, followed by the same letter, do not differ at the 0.05 level

<table>
<thead>
<tr>
<th>Site</th>
<th>Ditch density (km km⁻²)</th>
<th>Richness</th>
<th>Diversity</th>
<th>Evenness</th>
<th>Dominance</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Margalef’s</td>
<td>Shannon-Wiener</td>
<td>Pielou</td>
<td>Simpson</td>
</tr>
<tr>
<td>A</td>
<td>0–0.6</td>
<td>0.94a</td>
<td>1.72a</td>
<td>0.90a</td>
<td>0.22ab</td>
</tr>
<tr>
<td>B</td>
<td>0.6–1.2</td>
<td>1.24b</td>
<td>1.94b</td>
<td>0.91a</td>
<td>0.18a</td>
</tr>
<tr>
<td>C</td>
<td>1.2–1.8</td>
<td>0.82a</td>
<td>1.58a</td>
<td>0.91a</td>
<td>0.24b</td>
</tr>
<tr>
<td>D</td>
<td>1.8–2.4</td>
<td>0.81a</td>
<td>1.62a</td>
<td>0.90a</td>
<td>0.24b</td>
</tr>
</tbody>
</table>

Table 5. Linear regressions of species diversity to ditch density

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Value</th>
<th>SE</th>
<th>t</th>
<th>P</th>
<th>R²</th>
</tr>
</thead>
<tbody>
<tr>
<td>Margalef’s richness</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Intercept</td>
<td>1.167</td>
<td>0.086</td>
<td>13.627</td>
<td>&lt;0.001</td>
<td>0.104</td>
</tr>
<tr>
<td>Ditch density</td>
<td>−0.191</td>
<td>0.089</td>
<td>−2.751</td>
<td>0.008</td>
<td>0.104</td>
</tr>
<tr>
<td>Shannon-Wiener diversity</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Intercept</td>
<td>1.899</td>
<td>0.069</td>
<td>27.624</td>
<td>&lt;0.001</td>
<td>0.121</td>
</tr>
<tr>
<td>Ditch density</td>
<td>−0.167</td>
<td>0.056</td>
<td>−2.898</td>
<td>0.004</td>
<td>0.121</td>
</tr>
<tr>
<td>Pielou evenness</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Intercept</td>
<td>0.896</td>
<td>0.010</td>
<td>88.329</td>
<td>&lt;0.001</td>
<td>0.139</td>
</tr>
<tr>
<td>Ditch density</td>
<td>0.013</td>
<td>0.008</td>
<td>1.618</td>
<td>0.110</td>
<td>0.039</td>
</tr>
<tr>
<td>Simpson’s dominance</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Intercept</td>
<td>0.187</td>
<td>0.014</td>
<td>13.723</td>
<td>&lt;0.001</td>
<td>0.099</td>
</tr>
<tr>
<td>Ditch density</td>
<td>0.030</td>
<td>0.011</td>
<td>2.666</td>
<td>0.010</td>
<td>0.099</td>
</tr>
</tbody>
</table>
There are, however, alarmingly few rigorous studies estimating the relationship between agricultural constructed landscapes and wetland ecosystems, and the situation is even worse for the agricultural drainage water bodies, which have traditionally received little attention from the conservation biologists. The same certainly applies to agricultural drainage ditches, and to our knowledge, no studies to date have examined the changes in wetland communities in relation to drainage ditch density gradient. Nevertheless, acquiring such information is critical for the conservation of these key wetland habitats with anthropogenic disturbance. Our study was different from previous research as drainage ditch density was chosen as the disturbance gradient. It was the response pattern rather than the absolute data that was of primary interest. The results of our study showed that there are statistically significant positive relationships between wetland community pattern and agricultural ditch density. This is, in part, as drainage ditch contributes to land use intensification, it is the indicator of farming intensity. In addition, drainage ditches normally result in discontinuous wetlands existing as biogeographical islands, which are prone to impacts from external stressors imposed by surrounding land-uses (Adamus 1992, Denny 1994).

Our study also showed that the species diversity of the wetland community had a tendency to decline with increasing ditch density, with exception to site B supported greatest diversity. The intermediate disturbance hypothesis may explain this result, which predicts that moderate levels of disturbance maximize species diversity (Connell 1978, Huston 1979, Pollock et al. 1998). In theory, strong disturbance at site D should exterminate most species and cause the diversity to decrease, whereas the species richness and diversity of site C and D are nearly the same. This probably because there is a trade-off between the exclusion of species and attraction of new disturbance-tolerant species at site D, as high fragmentation at site D reflects high habitat heterogeneity, which could attract new species (Rivard et al. 2000). In addition, at site D, nutrient input to the wetland was higher, while nutrient enrichment tends to favor general grass species (Boatman and Wilson 1988, Hobbs and Huenneke 1992).

One key finding of our research is that along the drainage ditch density gradient, species composition changed significantly while the variation in species diversity is minor. In general, the number of plant species and diversity indices did not significantly differ between sites, except for site B. However, in contrast to diversity indices, the overall species composition diverged considerably. For example, even species as Glyceria spiculosa (Schmidt) Roshev were frequent and abundant at site A and B had decreased dramatically at site C and D. Indeed, the increasing loss of indigenous wetland species in our study area paralleled the increasing incursion of upland species with the ditch density increasing. This result can be mainly explained by three ecological interpretations. Above all, it is widely recognized that hydrological conditions provide the basic control of wetland structure (Zedler 2000). The species composition change may be mainly due to the loss of suitable moist habitat through the alteration of wetland hydrology, for the specific hydrological regime had been altered through drainage. In such altered conditions, formerly subordinate species, such as Deyeuxia angustifolia (Kom.) Y. L. Chang attained greater dominance, thus indigenous wetland species were replaced by more generalist species. Furthermore, habitat fragmentation can cause a change in species composition, especially declines in area-sensitive species or those with limited dispersal capabilities (Saunders et al. 1991, Andrén 1994). Moreover, some species may be very sensitive to intensive agriculture and may be eliminated even under moderate ditch density, e.g., Carex pseudocuraica Fr. Schm., which also contributed to the difference of species composition among sites.

In our study, in order to obtain an effective description of the vegetation pattern and the ditch density gradient relations, both ordination and regression techniques were employed. Preliminary analyses were made using Detrended Correspondence Analysis (DCA) to check the magnitude of change in species composition. DCA is an indirect
Wetland communities along ditch density gradient analysis technique, which summarize species abundance data by assessing the dominant patterns of variation in species composition of sample plots (Palmer 1993). Furthermore, regression analysis was used between the first ordination axis and ditch density gradient. Both those two analysis techniques gave easily interpretable and consistent results, and demonstrate the clear relation between the distribution of the floristic patterns and the ditch density gradient. Since these methods used in our study makes simpler understanding of the complex relationship between plants and ditch density gradients, and are of high accuracy and have different abilities, they could be used for habitat analysis and determination of effective ecological factors.

The approach used in this research was not optimal. Firstly, the study may have been based on incomplete ditch data sources which substantially underestimated ditch density. Secondly, we did not take ditch classification and ditch condition into account, while different types of ditches have varying impacts on the ecological processes they affect, and the effects of ditches on vegetation in drained wetlands have been observed to vary, depending on ditch condition (Fisher et al. 1996). Thirdly, our ditch scoring system mainly described disturbance severity, while the exact nature of disturbance varied somewhat.

A better understanding of the wetland community pattern along the agricultural drainage ditch density gradient could contribute to enhancing sustainable wetland management practices, with attendant benefits for biodiversity of the region. Our study clearly illustrated that ditch density gradient is an important determinant of the vegetation structure and composition of the wetland in Sanjiang plain. The negative relationship between drainage ditch density and species diversity is key information for wetland conservation. From a wetland management perspective, increasing the ditch density would facilitate agricultural more utilization of the wetland area, and resulted in decrease of diversity. We suggest that the agricultural ditch density of 1.2 km km⁻² should be the maximum value suitable for the protection of native plants (and probably of other wildlife) in wetlands in Sanjiang plain. We also suggest that current market forces are likely to result in a rapid attrition of wetlands. The change from upland crops to paddy fields will result in increasing ditch density, and thus to a changed wetland community.

Table 2 is useful not only because it indicates the species affiliated with each type of site within our study area, but also because it lists those species that are common across each site type. Furthermore, as the species at site B are likely to be the first to respond to changes in agricultural disturbance pressure, e.g., Calamagrostis neglecta Ehrh., they represent the best candidates for monitoring and predicting future changes in wetland condition. We recommend using those both affiliated and common species for restoration efforts. While the wetlands with high ditch density, such as those at site D, tended to have more pioneer species, non-native dominants, and species with relatively lower conservation quality (e.g. Artemisia annua L.), and they are the key ecosystem of restoration.

Wetlands in Sanjiang Plain remain under threat. While progress has been made in better balancing agricultural and ecosystem values, the former continue to clearly outweigh the latter. We argue that progress should be made to incorporate environmental considerations into the drainage process. In addition, the harmful role of drainage ditches can be changed within the framework of intelligent management. For example, the function of such ditches can be changed from drainage to irrigation by limited hydraulic engineering (Langheinrich et al. 2004). By designating and managing agricultural drainage ditches, we can better integratively design drainage ditches for purposes of wetland mitigation, or offer potential best management practices (BMPs) for continued effective drainage ditch maintenance. One such BMP is to make the ditches buffers between areas of intensive agriculture land use and potentially at risk at wetland ecosystems (Moore et al. 2000, Boulton et al. 2004 a, b) This practice makes sense both from an agronomic and conservation perspective. We also argue that greater emphasis should be placed on optimal spatial aspects of drainage ditches, and spatial differences in drainage ditch density
should play a role in the selection of optimal locations for nature protection areas.

6. CONCLUSIONS

The study provides fundamental answers concerning the variation in species composition and diversity along the ditch density gradient. Our data showed that changes (which will normally be increased) in agricultural ditch density within a region will imply a higher likelihood of adverse impacts on the natural wetland; there exits a clear negative relationship between ditch density and the diversity of wetland community; species composition is altered significantly with increasing ditch density. We suggest that the relationships between wetland community and agricultural drainage ditch density should be taken into account in the design and management of drainage development programs and projects.

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