

Ecological Assessment of Plant Succession and Water Quality in Abandoned Rice Fields

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ABSTRACT: The increasing area of abandoned rice fields could provide new opportunities for wetland restoration in Asia. However, it is unknown how quickly or completely abandoned rice fields will recover from agricultural disturbances. We assessed water quality and plant community succession in abandoned rice fields with different hydrology in a mountain valley to understand the effects of hydrological regime on recovery. Water level, soil redox potential, water quality, plant composition, and primary production were measured. The sites, coded as D6, N13, and N16, had been recovering for 6, 13, and 16 years by 2006. N13 and N16 have been recovering naturally whereas D6 has been drained with a nearby dike and was tilled in 2001. The typical hydroperiods of D6, N13, and N16 were no surface water, permanently flooded, and seasonally flooded, respectively. The major change in vegetation structure of both D6 and N13 was the replacement of herbaceous species by woody species. Drawdown accelerated this change because *Salix koreensis* grew better in damp conditions than in flooded conditions. *Phragmites japonica* reduced plot-level plant species richness. The removal efficiency of NH₄-N, NO₃-N, and PO₄-P from water varied seasonally, ranging between -78.8 to 44.3%, 0 to 97.5%, and -26.0 to 44.4%, respectively. In summary, abandoned rice fields quickly became suitable habitat for native wetland plant species and improved regional water quality. Variation among our sites indicates that it is likely possible to manage abandoned rice fields, mostly through controlling hydrology, to achieve site-specific restoration goals.

Key words: Abandoned rice field, Ecological assessment, Nutrient removal efficiency, Plant succession, Secondary succession, Water quality, Wetland restoration

INTRODUCTION

Around half of all wetlands worldwide have been destroyed (Streever 1999). Conservation of the remaining areas is critical, but wetland creation and restoration must be considered as part of the solution to wetland habitat loss. Historically, many wetlands were drained to provide land for cultivation. Conversion of agricultural fields back to wetlands is gaining popularity; the National Research Council plans for the conversion of four million hectares of fields into wetlands by 2010 in the United States (National Research Council 1992). Abandoned agricultural fields also serve as buffer zones that protect sensitive biotopes in Europe (Hansson and Fogelfors 1998).

Rice fields are not only the most typical agricultural systems in Asia but are also the largest man-made wetlands in the world (Matthews and Fung 1987, Aselmann and Crutzen 1989). Many rice fields are now being abandoned because of low fertility, low economic benefit, rural exodus of farmers, and government agricul-

tural policies aiming to reduce overproduction. In Korea, 160,000-ha of rice fields were abandoned from 1975 to 2005 (NAPQMS 2005). In Japan, 207,000-ha of rice fields (including 54,000-ha of abandoned rice fields) were not planted since the 1960s because of government "set-aside" programs to reduce overproduction (Fujioka et al. 2001). China has 34 million ha of rice fields (Matthews 1991), and thus even a small percentage being abandoned would represent a great opportunity for wetland restoration.

The feasibility of restoring ARFs to more naturally functioning wetlands should be greater than that of other agricultural systems simply because the major components of a wetland (i.e. the biota, hydrology, hydric soils) already exist there. Many wetland plant species, considered weeds in the agricultural setting, already persist in rice fields, depending on the type of agricultural practices (Kosaka et al. 2006). In addition, the rehabilitation of ARFs is not completely dependent on dispersal of seeds from the surrounding landscape because rice fields typically have diverse and abundant propagules in their seed bank (Brown and Bedford 1997). Thus, restoration of ARFs may be an economical and convenient way to

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increase the area of wetland habitats.

However, it is uncertain whether (or how quickly) purely natural recovery processes will revert ARFs to a desirable state. Rice fields are characterized by extreme physical instability resulting from repeated human disturbance such as flooding, plowing, weeding, and repairing paddy levees. This physical instability can impair the successive development of biotic communities towards that observed in natural wetlands (Watanabe and Roger 1984, Lee et al. 2002). Rice fields also typically have significant nutrient or herbicide pollution which may impact succession. Clearly, the functional states of ARFs should be assessed in order to determine the potential of reclaiming wetland habitat lost to rice agriculture.

Lee et al. (2002) introduced five typical seres of secondary succession in ARFs. However, rice field succession is responsive to specific factors such as hydrological regime. For example, perennial plant species became dominant within 7 years in dry conditions, but only after 11 years in flooded conditions (Kang et al. 2002). Additionally, management of surrounding disturbance regimes, hydrology, soil texture, and fertilization in ARFs can lead to the establishment of significantly different vegetation. (Schmidt 1988, Mesléard et al. 1999). Thus, it may be possible to control the direction of vegetation development in ARFs to meet site-specific objectives by altering key environmental conditions.

Comin et al. (2001) found that rice fields one to six years after abandonment remove nutrients efficiently, provide avifauna community habitat, and increase landscape diversity. While encouraging, these results were from recently abandoned sites in a lowland region. It would be ideal to judge the success of habitat restoration after a longer period, perhaps 15 to 20 years (Mitsch and Wilson 1996). Moreover, the recovery of terraced rice field systems in mountainous valleys needs to be assessed as well because they are more likely abandoned than lowland fields as the terrain limits mechanized cultivation. Valley rice fields might be less amenable to wetland restoration because of their high drainage. For these reasons, we investigated three ARFs that were part of terraced agricultural systems in mountainous valleys.

Restoration projects often aim to intervene in natural succession processes because the severe disturbances in the past might impede vegetation development (Robinson and Handel 2000). Most often, increasing the rate of change towards a "natural" community is a goal. We hypothesized that hydrologic regime is one of the key factors in wetland plant succession, based on previous studies (e.g., Kang et al. 2002). Thus, we studied three typical ARFs with different hydrological regimes (e.g., no surface water, permanently flooded, and seasonally flooded) in order to 1) understand the effects of hydrological regime on secondary succession, 2) assess the state of mountain valley ARFs in terms of water quality and plant habitat

quality, and 3) start collecting information necessary to develop effective restoration practices for this habitat type.

MATERIALS AND METHODS

Study Site

The study site is located in a valley of Mt. Bukhan, northwest of Seoul, Korea ($37^{\circ} 39' N$, $126^{\circ} 56' E$) (Fig. 1a). The former rice paddies consisted of terrace systems with successive levees. The sites were categorized and coded based on recovery years until 2006 and historical management status (D: disturbed or N: naturally recovered). The history of each site was determined mainly by fourteen successive aerial photos taken from 1980 to 2004. The extent of rice fields was easily identified in the 1980 photo because rice monocultures create a clear visual boundary. We then used the remaining photos to trace the different patterns of secondary succession at each site. In addition, we interviewed neighbors and landowners for additional information on historical management practices. Each zone, named as D6, N13, and N16 was first abandoned in 1996, 1994, and 1991. N13 and N16 have been abandoned without any management, while D6 has been drained with a nearby dike, tilled in 2001, and abandoned again since then. The highest site (N16) in elevation was abandoned first and the lowest site was abandoned last (D6). These landscape patterns are typical because higher elevation sites are harder to access and cultivate. D6, N13, and N16 cover about 0.4 ha, 0.7 ha, and 0.3 ha. D6 and N13 are adjacent to each other and the watersheds of D6 and N13 both include drain-

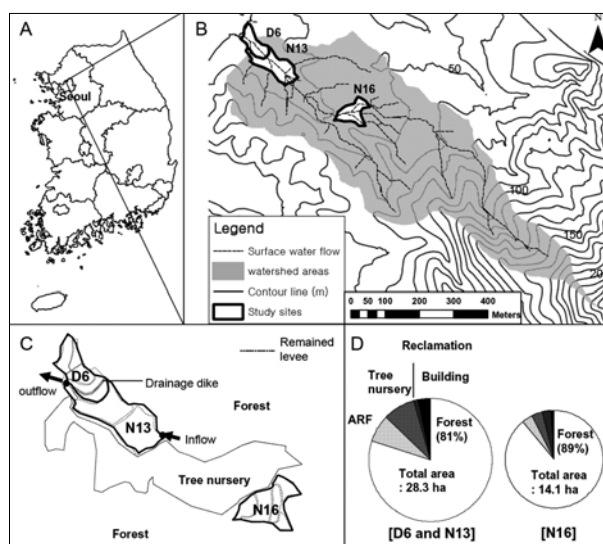


Fig. 1. Location of study sites: (a) location in Korea, (b) topology of site, watershed, and surface water flow, (c) site and its surrounding land use, and (d) types of land use within watersheds of each site.

age from N16 (Fig. 1b). The area of watersheds for D6 and N13 is 28.3 ha, while the area of watershed for N16 is 14.1 ha. The type of land use in each watershed was categorized as rice field, ARF, tree nursery, reclamation, building, or forest, with both watersheds being more than 80% forest (Fig. 1d). The study sites are included in a palustrine system, and following Cowardin et al. (1979) would be considered persistent emergent wetlands.

Methods

Water level was monitored monthly at six measurement points within D6 and N13 from May 2003 to November 2006. The measurements were taken by using 5-cm diameter PVC pipes with several 2-mm diameter holes on its side to permit water influx. Pipes were installed in the places that were expected to have different hydroperiods (i.e. the middle and edge of the paddies). The surface water regimes of the wetland were classified following Cowardin et al. (1979). The boundary of the watershed and surface water flow of the wetland were analyzed based on a topographic map by using a GIS application (ArcGIS ver 9.0, ESRI, U.S.A.).

Water quality variables were measured at three locations (i.e. inflow, interior of the field or the "inside", outflow) in N13 periodically from 2003 to 2006. Because standing water was irregularly present at sites D6 and N16, water quality variables were not measured. The concentration of dissolved oxygen (DO), specific conductivity, and pH of the water were measured three times per year (May, August, and October) from 2003 to 2005 and every month in 2006 using a meter (Corning, Model 311, U.S.A.). During 2003~2006 water samples were filtered (Whatman cellulose nitrate membrane filter, pore size 0.45 μm) and concentrations of $\text{NH}_4\text{-N}$, $\text{NO}_3\text{-N}$, and $\text{PO}_4\text{-P}$ were analyzed colorimetrically using the indophenol, hydrazine, and ascorbic acid reduction methods, respectively (Murphy and Riley 1962, Kamphake et al. 1967, Solorzano 1969). The removal efficiency was calculated by the summation of load method, in which all nutrient concentrations of inflow and outflow were summed and compared (US EPA 1986) according to the following equation: $(C_i - C_o)/C_i$ where C_i is the summation of inflow concentration and C_o is the summation of outflow concentration.

We constructed and calibrated platinum redox electrodes following established methods (Faulkner et al. 1989, Patrick 1996, Swerhone et al. 1999). Four electrode replicates were installed at each of five different soil depths (i.e. 5 cm to 25 cm) within N13 (total of 20 electrodes). Additionally, four electrode replicates were installed 15 cm deep at each of four locations (total of 16 electrodes). The sampling locations were randomly distributed across N13 in order to detect any spatial variation within the site. The *in situ* redox potential (Eh) was measured monthly from May 2003 to

November 2006 as an indicator of biogeochemical transformation in the sediment of N13, using our platinum redox electrodes and a portable pH/mV meter (Fisher, Model AP63, U.S.A.) with a saturated calomel reference electrode (Fisher, Cat. No.13-620-259 U.S.A.). Raw Eh values were corrected for temperature by adding a correction factor (Vepraskas and Faulkner 2001).

Most plants were identified using the taxonomic nomenclature designated by Lee (2003) while *Typha* spp. (cattails) were identified using Kim et al. (2003). We distinguished seven distinct plant communities based on the most dominant plant taxa. The area covered by each community was calculated from vegetation maps constructed by field sketching and GPS mapping every August from 2003 to 2006. The maps were scanned and digitized to calculate the area of each plant community using GIS. In addition, three fixed line transects (50 m, 50 m, and 100 m long, crossed within N13) were established at random spatial locations, and the relative coverage of each species on each transect was measured using the line intercept method (Canfield 1941).

One plot within each of seven plant communities was established to directly observe the change of species composition. Following the method of Greig-Smith (1964), the plot size was set at least twice as large as the average canopy spread of the largest species. Two different plot sizes were formed: a 5 \times 5-m plot for *Salix koreensis* (Korean willow) and 1 \times 1-m plots for the other herb communities. The coverage and number of all plants were measured at each plot every August from 2003 to 2006. The Shannon-Wiener diversity index (H') was used to describe species diversity in each plot.

The DBH (diameter at breast height) of *S. koreensis* was measured on November 2006 at two additional plots (5 \times 5-m), where *S. koreensis* were dominated, in the middle and edge of N13. Within these plots, leaf litters falling from *S. koreensis* were captured on fixed, smaller plots (0.5 \times 0.5-m), dried at 80 $^\circ\text{C}$ for 48 hours, then weighed. Average tree age was estimated during November 2006 at each site by using an increment bore at breast height on randomly selected trees, then counting the number of tree rings of each core sample under a dissecting microscope. The leaf area index (LAI), an indicator of foliage density, was measured at each site during September 2006 using a plant canopy analyzer (LI-COR, LAI-2000, U.S.A.).

We used the area harvest method (Shaver and Chapin III 1991) to estimate the net primary production of vegetation by cutting all aboveground biomass in a single plot in the five major communities every October from 2003 to 2006. After drying at 80 $^\circ\text{C}$ for 48h, the biomass was weighed.

Repeated measures ANOVA was used to test for differences between sampling locations (within or across sites) in response va-

riables (e.g., water quality variables, redox potential) measured several times. For the other response variables (e.g., the age, LAI, DBH, and leaf litter production of trees), one-way analysis of variance (ANOVA) was used to test for differences among factors. Duncan's multiple range test was used for multiple comparisons. All statistical analyses were conducted using SAS software (version 9.1, SAS Institute, U.S.A.).

RESULTS

Hydroperiod and Vegetation Succession

The hydroperiod of D6 was classified as 'flooded intermittently' in the middle of the paddies and 'no surface water' at the edge of the paddies (Fig. 2a). The inundation occurred mostly during the rainy season. The hydroperiod of N13 was classified as 'flooded permanently' in the middle of the paddies and 'exposed intermittently' at the edge of the paddies (Fig. 2b). The hydroperiod of N16 was classified as 'seasonally flooded' (flooded during the rainy season) on the basis of an interview with neighbors.

The patterns of plant succession differed among sites (Fig. 3). Plant diversity was highest in flooded condition (overall plant species richness of D6, N13, and N16 was 7, 34, and 2, respectively). *Salix koreensis* extended throughout site D6 after six years, but not until 13 years post-abandonment in N13 (Fig. 4). The most abundant herbaceous species for D6, N13, and N16 were *Phragmites communis*, *Phragmites japonica*, and *Persicaria thunbergii* (Polygonaceae), respectively (Fig. 4). *Phragmites japonica* seems to be superior to *Phragmites communis* in flooded condition but inferior in drained conditions.

Plot sampling revealed that while the plant community in some areas changed little during the study period, other areas were very dynamic. Where initially abundant, *Phragmites japonica* maintained overwhelming dominance throughout the study years in plot 4 (Ta-

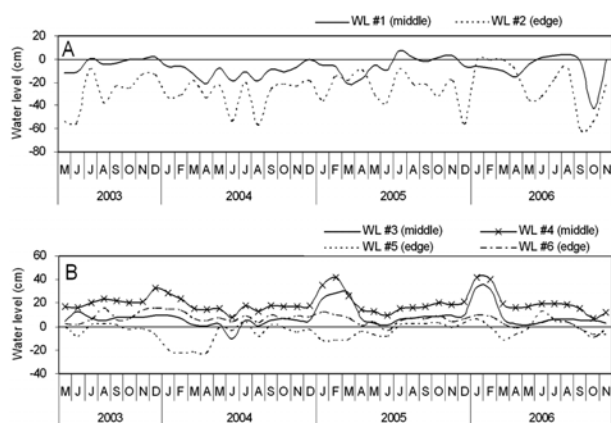


Fig. 2. Water level of sites: (a) D6 and (b) N13.

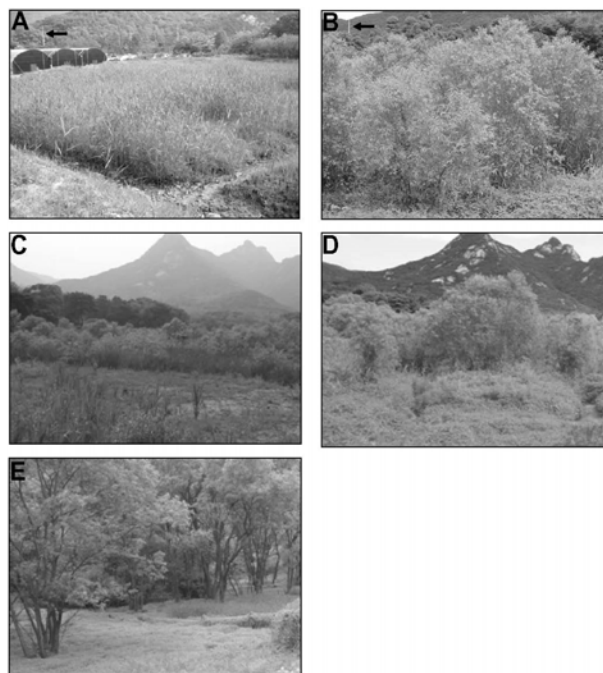


Fig. 3. Photographs of study site (Photo by C. Byun). Arrow bar indicates utility pole as perspective standard. (a) D6 in 2003 (b) D6 in 2006 (c) N13 in 2003 (d) N13 in 2006 (e) N16 in 2006.

ble 1). It also invaded into other plant communities; *Typha* spp. and *Persicaria thunbergii* were replaced as dominants by *P. communis* in plot 3, and *P. japonica* in plot 5 (Table 1). The diversity index (H') of these two plots (3 and 5) decreased as well. The aboveground biomass of *P. japonica*-dominated plot (4) also was substantially greater than other plots (Table 1). Thus, *Phragmites* spp. showed invasive features in N13. The *Typha* spp. dominated community showed relatively high diversity as new taxa established in plot 2 (Table 1). The reeds common at other sites were absent at N16. Instead *Persicaria thunbergii* was the only plant to persist in the dim light characteristic under these willow stands. The dense foliage (~ 2.15 LAI) appeared to retard reed species (Table 2).

Salix koreensis sprouted at the sites mostly between 3~6 years after abandonment in N13 and N16, while it sprouted promptly in the drained condition of D6 (Table 2). Tree age, DBH, and LAI (index of leaf foliage density) increased as the site aged. Tree density at the oldest site was much less than the other sites (Table 2). The DBH of *S. koreensis* were significantly greater at low in the edge of paddies (8.35 ± 3.35 cm, $N = 70$) than in the middle of paddies (4.48 ± 0.96 cm, $N = 70$) in N13 (ANOVA, $F_1 = 76.18$, $p < 0.0001$). The leaf fall of *S. koreensis* were also greater in the edge of paddies (0.16 ± 0.02 kg/m²) than (0.08 ± 0.02 kg/m²) middle of paddies in N13 (ANOVA, $F_1 = 27.49$, $P = 0.0019$).

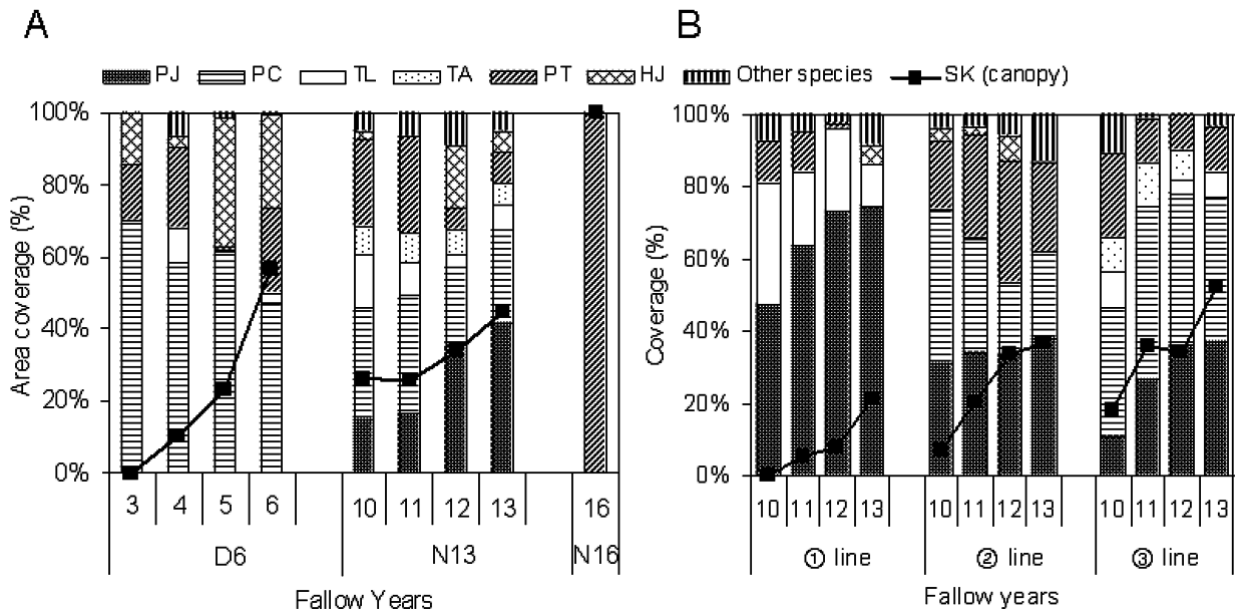


Fig. 4. (a) Relative area determined by yearly mapping of plant communities in D6, N13, and N16. (b) Relative coverage across three line transects in N13. Species abbreviations: SK *Salix koreensis*, PJ *Phragmites japonica*, PC *Phragmites communis*, TL *Typha latifolia*, TA *Typha angustifolia*, PT *Persicaria thunbergii*, HJ *Humulus japonicus*.

Soil Redox Potential and Water Quality

Soil redox potential and water quality was investigated in N13 because it is a typical ARF with permanent surface water. Soil Eh at depths below 15 cm was -166.28 ± 20.54 mV during the growing seasons (from March to October) and -128.13 ± 22.49 mV during the winter (from November to February) (Fig. 5). However, Soil Eh at above 10cm was -74.61 ± 109.84 mV during the growing seasons and 25.35 ± 90.67 mV during the winter (Fig. 5), significantly greater than redox potential in deeper soil (repeated measures ANOVA, $F_4 = 70.52, p < 0.0001$). Horizontal spatial differences of Eh at the same depth were relatively slight (repeated measures ANOVA, $F_4 = 4.31, p < 0.0018$), for their averages ranged from -143.50 to -163.41 mV.

Water quality variables of the inflow, inside, and outflow of N13 are shown in Fig. 6. The pH of outflow was significantly greater than that of the inflow and inside. Dissolved oxygen concentration of water inside the ARF was marginally less than that of the inflow and outflow, particularly in August and October. The overall removal efficiencies of $\text{NH}_4\text{-N}$, $\text{NO}_3\text{-N}$, and $\text{PO}_4\text{-P}$ in water were -11.1% (-78.8 to 44.3%), 50.2% (0 to 97.5%), and 14.3% (-26.0 to 44.4%), respectively. The concentration of $\text{NO}_3\text{-N}$ decreased significantly as water passed through N13 (repeated measures ANOVA, $F_2 = 4.99, P = 0.0233$). However, the concentrations of $\text{PO}_4\text{-P}$ and $\text{NH}_4\text{-N}$ was not different among sampling locations (repeated measures ANOVA, $F_2 = 0.46$ and $1.37, P = 0.5177$ and 0.7212 , respectively).

The seasonal variation of removal efficiencies can be visualized by comparing the inflow and outflow (Fig. 6; dashed and solid lines) concentrations at various times. The concentration of $\text{NO}_3\text{-N}$ in the inflow and the removal efficiency of $\text{NO}_3\text{-N}$ decreased in August and October (but not significantly so, time effect, Table 3) corresponding to periods of flooding. We detected little seasonal changes in $\text{NH}_4\text{-N}$ and $\text{PO}_4\text{-P}$ inputs.

DISCUSSION

Hydroperiod and Vegetation Change

Features of the watershed and its position likely determine whether terraced ARFs maintain flooded conditions. Generally, ARFs at low elevation get more water from their watersheds and are thus flooded more than ARFs at high elevation. On the other hand ARFs in mountain valleys often have greater drainage than those located on the lowland because levees on the slopes soon collapse without maintenance. Moreover, soil moisture of ARFs gradually diminishes as drainage in recovering wetlands returns to more natural states, and as colonizing shrubs and trees increase transpiration rates (Lee et al. 2002).

N13 remained flooded during our study period. The relatively stable hydrological regime of N13 may reflect its large (28.3 ha), mostly forested watershed (Fig. 1d). Forests can store intense rainfall in the wet season and release it throughout the year. On the other hand, N16 was flooded only during the rainy season because

Table 1. Species diversity, aboveground biomass, and species composition of the plant communities in seven plots at different times since abandonment

Plot	Site	RY ^a	Relative coverage (%)							Species richness	Diversity (H')	Biomass (kg/m ²)
			SK	PJ	PC	TL	TA	PT	etc			
1	D6	3	0	0	80	0	0	1	5	7	0.16	0.70
		4	0	0	95	0	0	70	15	7	0.44	0.66
		5	0	0	75	0	0	70	5	5	0.37	0.39
		6	0	0	80	0	0	65	15	5	0.43	1.02
2	N13	10	0	0	0	45	0	0	1	2	0.05	0.84
		11	0	0	0	15	0	45	40	3	0.44	1.13
		12	0	0	0	40	0	80	65	5	0.58	0.65
		13	0	0	0	90	0	20	63	6	0.59	0.78
3	N13	10	0	0	5	40	15	60	7	7	0.57	0.84
		11	0	0	90	1	5	20	2	6	0.33	1.13
		12	0	0	70	1	1	90	3	7	0.37	0.65
		13	0	0	90	1	1	30	25	8	0.45	0.78
4	N13	10	0	100	1	1	0	0	1	4	0.07	1.98
		11	0	95	0	0	0	0	0	1	0.00	1.77
		12	0	100	0	0	0	0	0	1	0.00	1.69
		13	0	100	0	0	0	0	0	1	0.00	1.19
5	N13	10	0	1	0	70	0	80	1	4	0.33	–
		11	0	40	0	45	0	1	0	3	0.32	–
		12	0	100	0	7	0	0	0	2	0.10	–
		13	0	100	0	0	0	15	0	2	0.17	–
6	N13	10	63	3	0	0	0	88	37	13	0.60	0.41
		11	70	63	0	1	0	38	25	13	0.66	0.43
		12	70	70	1	1	0	38	10	14	0.59	0.54
		13	80	80	0	1	0	30	4	14	0.50	0.51
7	N16	16	100	0	0	0	0	100	0	2	0.30	–

^a RY means recovered years of the site. Species abbreviations: SK *Salix koreensis*, PJ *Phragmites japonica*, PC *Phragmites communis*, TL *Typha latifolia*, TA *Typha angustifolia*, PT *Panicum thunbergii*, etc all other species.

it has a smaller watershed (14.1 ha) and its levees had already collapsed.

The major change in vegetation composition was the replacement of herbaceous species by woody species. *Salix koreensis* became the dominant woody species within 6 years after abandonment in drained condition of D6 (Figs. 4a and 4b). However *S. koreensis* took 13 years to dominate in the flooded conditions of N13, and after 16 years completely covered the seasonally flooded site N16 (Figs. 4a and 4b). This is consistent with Lee et al.'s (2002) finding that *Salix* spp. became dominant in lowland ARFs. However, Lee et al. (2002) reported that *S. koreensis* was replaced by *Alnus japonica* (alder) after 10 fallow years, which was not the case in our

study. These different patterns of species recruitment could be due to site specific differences in hydrological regime, nutrient balance, topology, and/or surrounding species (i.e. availability of seeds / pollen).

Our results suggest that hydrological regime has a significant role in succession of *S. koreensis* in ARFs. Given that the seeds of *S. koreensis* are densely and widely distributed (Gage and Cooper 2005), its presence in the plant community depends largely on proper environmental conditions. Seedlings of a species in the same genus, *Salix cinerea*, need good light and damp, but not flooded conditions (Haslam 2003). Correspondingly, in this study the results from the tree rings of *S. koreensis* indicated that they sprouted from

three to six years after abandonment in N13 and N16 but almost immediately in D6 (Table 2). Trees also grew faster in the damp conditions of D6 than the inundated conditions of N13 (Fig. 4a).

Spatial patterns of the recruitment of *S. koreensis* can be inferred

from past aerial photography; *Salix koreensis* was already present at the levee in N16 in 1994 and extended into the middle of the paddies in N16 in 1998. Leaf fall and DBH of trees at the edge of the paddies was also greater than that in the middle of the paddies, indicating it established at the edge first. In sum, it appears

Table 2. Characteristics of *S. koreensis* in each site during 2006. Values are average \pm 1 SD (For specific detail of ANOVA result, see Table 3).

Variables	D6 ^c	N13 ^c	N16 ^c	<i>p</i>
Tree age(Years)	6.1 (1.1)	9.3 (1.0)	11.2 (1.7)	<0.0001
Years to sprout ^a	0.1 (1.1)	3.7 (1.0)	4.8 (1.7)	<0.0001
DBH ^b (cm)	4.59 (1.1)	6.47 (3.1)	12.80 (2.7)	<0.0001
LAI(leaf area index)	1.61 (0.17)	1.99 (0.08)	2.15 (0.02)	0.0006
Density(number/m ²)	0.34	0.32	0.12	-

^a 'Years to sprout' is the difference between the number of years the site had been abandoned and the average tree age.

^b DBH is diameter at breast height.

^c Values in parenthesis are the standard deviation of the mean.

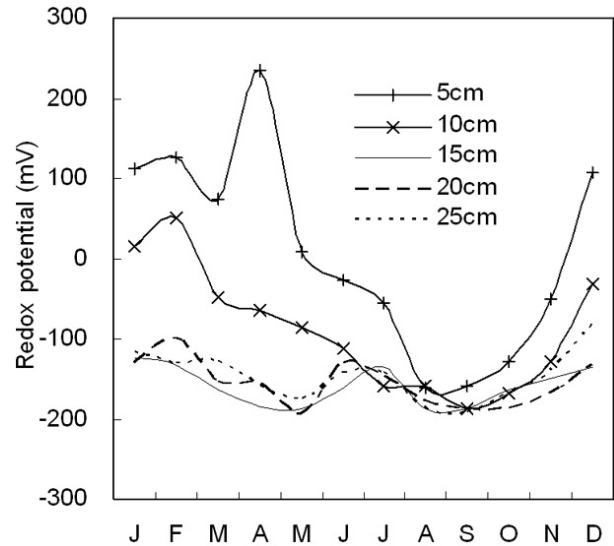


Fig. 5. Redox potential by soil depth in N13. All values are averages.

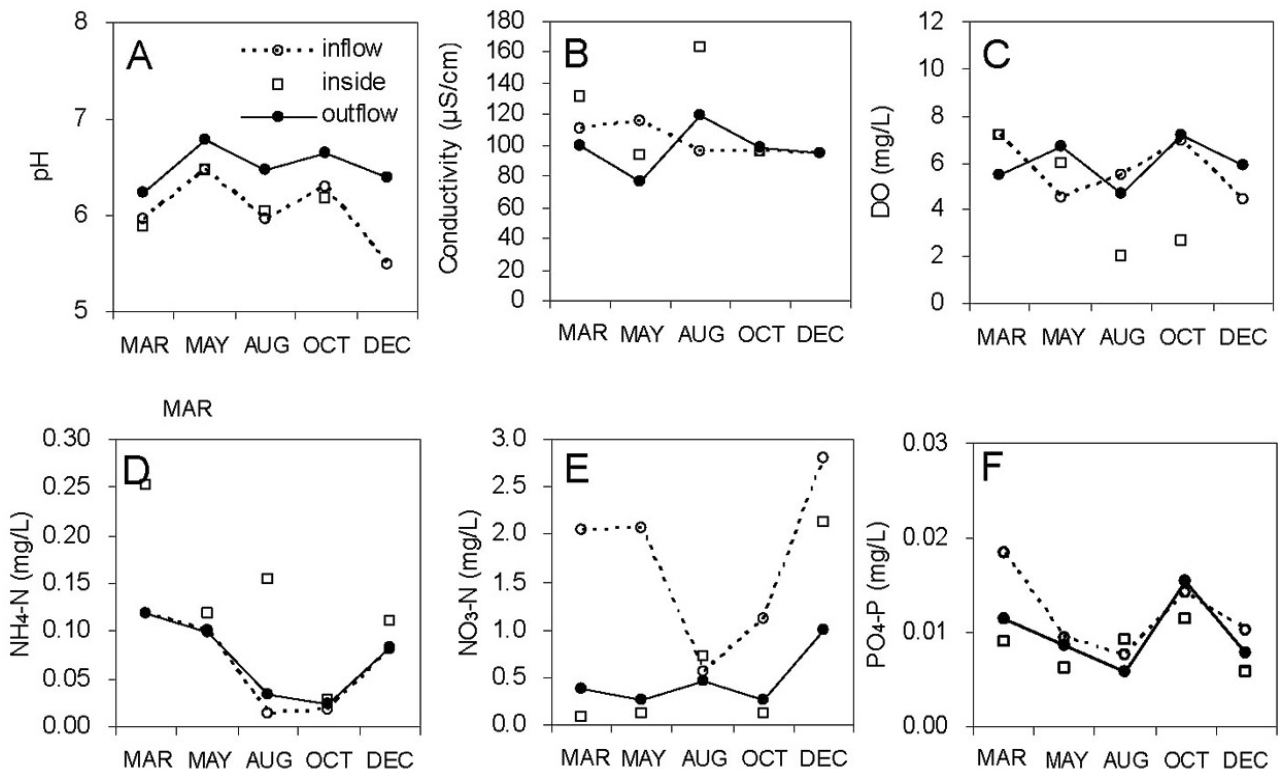


Fig. 6. Water variables of N13: (a) pH, (b) conductivity, (c) dissolved oxygen (DO), (d) NH₄-N, (e) NO₃-N, and (f) PO₄-P.

Table 3. Summary of ANOVA test results

Test	Variables	Independent factors	F	P	df	N
One way ANOVA	Tree age	- Sampling sites (D6, N13, and N16)	35.43	< 0.0001	2	31
	LAI	- Sampling sites (D6, N13, and N16)	18.99	0.0006	2	12
	DBH	- Sampling sites (D6, N13, and N16)	40.73	< 0.0001	2	54
	DBH	- Sampling location (middle and edge)	76.18	< 0.0001	1	139
	Leaf fall	- Sampling location (middle and edge)	27.49	0.0019	1	8
Repeated measures ANOVA	NO ₃ -N	- Sampling location (inflow, inside, outflow)	4.99	0.0233	2	33
		- time (5 months)	2.63	0.0868	4	33
		- Sampling location × Time	0.74	0.6564	8	33
	NH ₄ -N	- Sampling location	1.37	0.5177	2	33
		- time	1.27	0.3187	4	33
		- Sampling location × Time	0.30	0.9573	8	33
	PO ₄ -P	- Sampling location	0.46	0.7212	2	33
		- time	0.77	0.5565	4	33
		- Sampling location × Time	0.15	0.9948	8	33
	Soil Eh	- Soil depth (5, 10, 15, 20 and 25 cm)	70.52	< 0.0001	4	1,256
		- Time (Monthly)	11.82	< 0.0001	45	1,256
		- Soil depth × Time	180	< 0.0001	180	1,256
		- Sampling location (5 points within N13)	4.31	0.0018	4	1,148
		- Time	1.73	< 0.0001	45	1,148
	- Sampling location × Time	10.18	< 0.0001	180	1,148	

that inundation retards colonization by *S. koreensis* and slows its growth once established. In contrast, drawdown triggered establishment of *S. koreensis* and accelerated succession towards woody vegetation. These results emphasize hydrology as a key factor in the natural recovery process of ARFs, and perhaps that management strategies could be employed to “push” sites towards desired endpoints (e.g., tree dominated, reed dominated).

Phragmites japonica showed features of an invasive plant even though it is a native in Korea; it produced the most biomass, changed the composition of the remaining species, and decreased plant species diversity in N13 (Table 1). The dominance of *Phragmites* spp. in created emergent wetlands has been commonly reported (Havens et al. 1997, Moore et al. 1999), but the aboveground biomass of *P. japonica* in our sites (Table 1) was far greater than that of other reed dominated wetlands (Mason and Bryant 1975, Kim et al. 2000). This strong dominance may be a major concern we can encounter in biodiversity restoration projects.

The overwhelming abundance of *P. japonica* might have two

explanations. First, ARFs may have altered properties (i.e. hydrology, nutrient balance, soils) that are more favorable for this species than others. Second, the invasiveness of *P. japonica* may be related to its growth form; *Phragmites* spp. can invade into monospecific stands of other species, using a localized proliferation of dense roots (Farnsworth and Meyerson 1999). It also accumulates a dense litter layer due to its high production. As a result, it can change a flat surface into a mound-shaped one through its accumulation of litter around the roots (C. Byun, *personal observation*). The cues other wetland plant species use to germinate such as moisture, temperature, and light fluctuations can be modified under intensive litter layer (Carson and Peterson 1990). Thus, *Phragmites* physical modification of its microenvironment may affect other species reproductive success.

Typha spp. persisted in N13 mainly in areas of relatively high water (more than about 50 cm), near the middle of paddies and in the waterways between paddies (C. Byun, *personal observation*). Unlike *P. japonica*, *Typha* spp. can grow well in deep water (New-

man et al. 1996).

Abandoned Rice Fields as Plant Habitat

The plant diversity (H') of the sites (0.30 to 0.35) was not as high as natural wetlands (1.29) and mitigated palustrine wetlands (1.83) in the U.S. (Balcombe et al. 2005), or of a natural riverine (1.25) and a natural lacustrine (1.08) wetland in Korea (Kwon et al. 2006). However, the species richness of hydrophytes at site N13 (34) was considerably higher than average species richness of natural wetlands (8) and mitigated palustrine wetlands (13) in the U.S. (Balcombe et al. 2005) and species richness of a natural riverine (12) and a natural lacustrine (17) wetland in Korea (Kwon et al. 2006).

Low plant diversity and high dominance seems mostly due to strong competition among species. Invasive plants like *P. japonica* simplified the composition of plant communities in ARFs. *Salix* spp. also allow few species to exist under the canopy (Schmidt 1988), as we found only one species (*P. thunbergii*) under *S. koreensis* in N16. In addition, the physical structure of successive levees in the terraced system limits the amplitude of water level fluctuations in ARFs. Low water level fluctuation provides only a narrow range of hydrologic habitats (Cronk and Fennessy 2001), and these homogeneous conditions may limit plant species richness. In continuously flooded conditions, vegetation usually has low species diversity, high density of obligate wetland plants, and consists of perennials with no annuals (Mitsch and Gosselink 2000). As mentioned above, *Typha* spp. seemed to find a refuge from *Phragmites* competition in small pockets of deeper water in the rice fields. Restoration plans for ARFs should include provisions to maintain areas with heterogeneous water levels to promote higher site-wide species diversity. Lastly, low availability of plant seeds may have contributed to the low plant species richness at our sites. ARFs usually are supplied with plant propagules from surrounding wetland habitats and the seed bank. ARFs located in a valley usually have small streams nearby that may transport some plant seeds, but typically the seeds are of terrestrial plant species from the surrounding forested landscape. Species richness may continue to increase with more time since abandonment as plant seeds find their way to the sites (Lee et al. 2002). Active restoration efforts might include reseeding ARFs with native wetland plant seeds to speed the establishment of wetland plant species.

Abandoned Rice Fields as Water Purification Systems

N, Mg, Fe, S, and C were expected to be reduced in most seasons given the observed range of redox potential at site N13 (Reddy and D' Angelo 1994, Mitsch and Gosselink 2000). Denitrification is active in the observed range of redox potential (Fig. 5), thereby

rendering the conditions suitable for water purification (Reddy and D' Angelo 1994). In addition, high productivity of monospecific stands of plants such as *Phragmites* are known to support high nutrient uptake (Kang et al. Nitrogen removal from a riverine wetland: a field survey and simulation study on *Phragmites japonica* 2002).

Overall removal efficiencies of the sites for $\text{NO}_3\text{-N}$ (0 to 97.5%, average: 50.2%) were within the ranges observed in natural treatment surface flow wetlands (average: 77.5%) and a constructed one (average, 44.4 %) in the United States (Knight et al. 1993), suggesting ARFs may improve local water quality with some aspect. In contrast, the overall removal efficiency for $\text{NH}_4\text{-N}$ (-78.8 to 44.3%, average: -11.1%) was far lower than that in other natural wetlands (average: 71.5%) and a constructed one (average: 33.9%) in U.S. (Knight et al. 1993). However, It is important to note that the concentrations of $\text{NO}_3\text{-N}$ were far greater than $\text{NH}_4\text{-N}$ (Fig. 6d & 7e), and thus the ARFs were still effective at sequestering the predominant available nitrogen. The overall removal efficiency for $\text{PO}_4\text{-P}$ was not significantly greater than zero (-26.0 to 44.4%, average: 14.3%) but were within the ranges observed in 1 to 6-years-old ARFs (-254 to 98%) in Spain (Comin et al. 2001).

Removal efficiencies for $\text{NH}_4\text{-N}$ and $\text{NO}_3\text{-N}$ were especially low in late summer (Fig. 6), for the oxidized soil layer of the sites diminished in August (Fig. 5) because of flooding during the monsoon season in Korea. Eh decreased with increasing depth because the diffusion of dissolved oxygen into the sediment was limited (Patrick and DeLaune 1972). The oxidized soil layer (upper soil layer with higher Eh) plays a role in a part of the denitrification process because ammonium diffusion from sediment to the oxidized soil layer limits the entire denitrification process (Reddy and Patrick Jr 1984, Reddy and Graetz 1988). Without the layer, some $\text{NH}_4\text{-N}$, which is produced by ammonification of organic nitrogen, is expected to leach before entering the nitrification process. Thus, active restoration efforts to aerate the water will be needed in order to overcome these limitations.

Additionally, the water quality in the sites was good enough to support many aquatic organisms. However there was no fish community except *Misgurnus anguillicaudatus* (oriental weatherfish) in the site (C. Byun, *personal observation*). The concentration of DO inside site N13 was significant less than that of the inflow and outflow water, and was low enough to impair fish populations (USEPA 1976); anoxia was most common from August to October (Fig. 6c). Therefore, given our data, using ARFs in terraced systems for fish reserves is not recommended.

CONCLUSION

Our results suggest ARFs may be responsive to active manage-

ment, and we offer four restoration implications. First, the succession pattern in valley ARFs can likely be manipulated by hydrological management. Natural succession led abandoned rice fields in a valley to become a reed-dominant wetland in 10 years and willow-dominant wetland within 13 years. Augmenting drainage led an abandoned rice field to be a willow-dominant wetland within 6 years. In order to delay succession and sustain herbaceous wetland vegetation, flooding could be maintained by reinforcing levees. On the other hand, drawdown by enhancing drainage will accelerate succession toward woody habitat. Second, controlling the dominance of key species across a site may be possible by maintaining spatially or temporally variable hydrologic conditions. More plant species could survive in intentionally managed pockets of deeper water in rice fields, for the dominance of key species such as *P. japonica* and *S. koreensis* can be controlled in deeper water (> 50 cm). Third, restoration efforts to manage areas with heterogeneous hydrological regimes across the landscape may increase regional species diversity. It may be possible to establish and maintain different wetland plant communities across an elevation gradient because of the natural hydrological variation. Abandoned rice fields may become cattail-dominated in the high water levels typical at low elevations, reed-dominated in the shallow water levels at middle elevations, and tree-dominated in the seasonally or intermittently flooded conditions on mountaintops. Lastly, our one study site removed about 50.2% of NO₃-N but had negligible effects on NH₄-N and PO₄-P. Low performance during the rainy season was partly due to the highly reduced upper soil layer. Thus, active restoration efforts may be needed in order to overcome these limitations. Recreating a Floodplain to slow hydrologic residence time and to decrease water depth during the rainy season could improve nutrient removal efficiency.

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LITERATURE CITED

- Aselmann I, Crutzen PJ. 1989. Global distribution of natural freshwater wetlands and rice paddies, their net primary productivity, seasonality and possible methane emissions. *J Atmos Chem* 8: 307-358.
- Brown C, Bedford BL. 1997. Restoration of wetland vegetation with transplanted wetland soil: an experimental study. *Wetlands* 17: 427-437.
- Canfield RH. 1941. Application of the line intercept method in sampling range vegetation. *J Forest* 39: 388-394.
- Carson WP, Peterson CJ. 1990. The role of litter in an old-field community: impact of litter quantity in different seasons on plant species richness and abundance. *Oecologia* 85: 8-13.
- Comin FA, Romero JA, Hernandez O, Menendez M. 2001. Restoration of wetland from abandoned rice fields for nutrient removal, biological community and landscape diversity. *Restor Ecol* 9: 201-208.
- Cowardin LM, Carter V, Golet FC, LaRoe ET. 1979. Classification of wetlands and deepwater habitats of the United States. Washington, DC.
- Cronk JK, Fennessy MS. 2001. *Wetland Plants: Biology and Ecology*. CRC press, Boca Raton, Florida.
- Farnsworth EJ, Jr., Meyerson LA. 1999. Species composition and inter-annual dynamics of a freshwater tidal plant community following removal of the invasive grass, *Phragmites australis*. *Biol Invasions* 1: 115-127.
- Fujioka M, Armacost JW, Jr., Yoshida H, Maeda T. 2001. Value of fallow farmlands as summer habitats for waterbirds in a Japanese rural area. *Ecol Res* 16: 555-567.
- Gage EA, Cooper DJ. 2005. Patterns of willow seed dispersal, seed entrapment, and seedling establishment in a heavily browsed montane riparian ecosystem. *Can J Botany* 83: 678-687.
- Greig-Smith P. 1964. *Quantitative Plant Ecology*, 2nd Ed. Butterworths, London.
- Hansson M, Fogelfors H. 1998. Management of permanent set-aside on arable land in Sweden. *J Appl Ecol* 35: 758-771.
- Haslam SM. 2003. *Understanding Wetlands*. CRC Press, New York.
- Havens KJ, Priest WI, Berquist H. 1997. Investigation and long-term monitoring of *Phragmites australis* within Virginia's constructed wetland sites. *Environ Manage* 21: 599-605.
- Kamphake LJ, Hannah SA, Cohen JM. 1967. Automated analysis for nitrate by hydrazine reduction. *Water Res* 1: 205-216.
- Kang BH, Shim SI, Ma KH. 2002. Floristic composition of plant community in set-aside fields with regard to seral stages. *Korean J Environ Agr* 22: 53-59.
- Kang S, Kang H, Ko D, Lee D. 2002. Nitrogen removal from a riverine wetland: a field survey and simulation study on *Phragmites japonica*. *Ecol Eng* 18: 467-475.
- Kim C, Shin H, Choi HK. 2003. A phenetic analysis of *Typha* in Korea and far east Russia. *Aquat Bot* 75: 33-43.
- Kim CS, Son SG, Lee JH, Oh KH. 2000. Community structure, productivity, and nutrient uptake of the vascular plants in the wetlands of the Asan-Lake. *Korean J Ecol* 23: 201-209.
- Kosaka Y, Takeda S, Sithirajvongsa S, Xaydala K. 2006. Plant diversity in paddy fields in relation to agricultural practices in Savanakheth Province, Laos. *Econ Bot* 60: 49-61.
- Lee CS, You YH, Robinson GR. 2002. Secondary succession and natural habitat restoration in abandoned rice fields of central Ko-

- rea. *Restor Ecol* 10: 306-314.
- Mason CF, Bryant RJ. 1975. Production, nutrient content and decomposition of *Phragmites communis* Trin. and *Typha angustifolia* L. *J Ecol* 63: 71-95.
- Matthews E, Fung I. 1987. Methane emissions from natural wetlands: global distribution, area, and environmental characteristics of sources. *Global Biogeochem Cy* 1: 61-86.
- Mesléard F, Lepart J, Grillas P, Mauchamp A. 1999. Effects of seasonal flooding and grazing on the vegetation of former ricefields in the Rhône delta (Southern France). *Plant Ecol* 145: 101-114.
- Mitsch WJ, Gosselink JG. 2000. *Wetlands*, 3rd Ed. John Wiley & Sons, New York.
- Mitsch WJ, Wilson FF. 1996. Improving the success of wetland creation and restoration with know-how, time, and self-design. *Ecol Appl* 6: 77-83.
- Moore HH, Niering WA, Marsicano LJ, Dowdell M. 1999. Vegetation change in created emergent wetlands (1988-1996) in Connecticut (USA). *Wetlands Ecol Manage* 7: 177-191.
- Murphy J, Riley JP. 1962. A modified single-solution method for the determination of phosphate in natural waters. *Anal Chem Acta* 27: 31-36.
- National Research Council. 1992. *Restoration of Aquatic Ecosystems*. National Academy Press, Washington, DC.
- Newman S, Grace JB, Koebel JW. 1996. Effects of nutrients and hydroperiod on *Typha*, *Cladium* and *Eleocharis*: implications for Everglades restoration. *Ecol Appl* 6: 774-783.
- Patrick WH, DeLaune RD. 1972. Characterization of the oxidized and reduced zones in flooded soil. *Soil Sci Soc Am J* 36: 573-576.
- Reddy KR, D' Angelo EM. 1994. *Soil processes regulating water quality in wetlands*. Elsevier Science, Amsterdam.
- Reddy KR, Graetz DA. 1988. The ecology and management of wetlands. In: *The ecology and management of wetlands* (Hook DD, McKee Jr. WH, Smith HK, Gregory J, Burrell VG, DeVoe MR, Sojka RE, Gilbert S, Banks R, Stolzy LG, Brooks C, Matthews TD and Shear TH, eds). Timber Press, Portland, pp 307-319.
- Reddy KR, Patrick Jr. WH. 1984. Nitrogen transformations and loss in flooded soils and sediments. *Crit Rev Env Contr* 13: 273-309.
- Robinson GR, Handel SN. 2000. Directing spatial patterns of recruitment during an experimental urban woodland reclamation. *Ecol Appl* 10: 174-188.
- Schmidt W. 1988. An experimental study of old-field succession in relation to different environmental factors. *Vegatio* 77: 103-114.
- Shaver GR, Chapin III FS. 1991. Production: biomass relationships and element cycling in contrasting arctic vegetation types. *Ecol Monogr* 61: 1-31.
- Solorzano L. 1969. Determination of ammonia in natural waters by the phenylhypochlorite method. *Limnol Oceanogr* 14: 799-801.
- Streever B. 1999. *Bring Back the Wetlands*. Sainty & Associates Pty Ltd, Potts Point.
- Vepraskas MJ, Faulkner SP. 2001. *Redox Chemistry of Hydric Soils*. Lewis Publishers, New York.
- Watanabe I, Roger PA. 1984. Ecology of flooded rice fields. In: *Wetland Soil* (Swaminathan MS, ed). International Rice Research Institute, pp 229-243.

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