Wetlands Ecology and Management (2005) 13: 469–478
DOI 10.1007/s11273-004-2000-y

Vegetation communities of 20-year-old created depressional wetlands

Robert B. Atkinson[1,3,*], James E. Perry[2] and John Cairns Jr.[1]
*[1]Department of Biology, Virginia Polytechnic Institute and State University, Blacksburg, VA 24061, USA; [2]Virginia Institute of Marine Science, Gloucester Point, VA 23062, USA; *Author for correspondence: Present address: Department of Biology, Chemistry and Environmental Science, Christopher Newport University, Newport News, VA 23606-2998, USA (e-mail: atkinson@cnu.edu; phone: +1-757-594-7619)*

Received 22 October 2001; accepted in revised form 8 August 2004

Key words: created wetlands, created wetland succession, species richness, wetland vegetation dynamics

Abstract

Many studies have chronicled the early development of vegetation in wetlands created as mitigation for wetland impacts; however, very few studies have followed the floristics of wetlands that are more than 10 years post-creation. This article reports the results of vegetation composition and structural analysis within eleven 20-yr-old created non-tidal, emergent wetlands. Vegetation and inundation were sampled in 173 plots within 11 wetlands during the 1992 and 1994 growing seasons. A drought occurred in 1993, thus analyses characterized vegetative response and included weighted average (weighted by the tolerance of the species to excess soil moisture), species richness, species composition, and life history strategy. Weighted average and species richness increased in 7 and 10 of the 11 sites, respectively. There was little change among most species including *Typha latifolia* and *Scirpus cyperinus*, the two species with highest importance values (IV). However, among the top 10 species ranked by IV, two aquatic species decreased and a facultative species increased. Only one of the 10 most important species, *Eleocharis obtusa*, was an annual and only one, *Salix nigra*, was a woody perennial and the IV of both species declined during the study. After 20 years, a transition from annual to perennial graminoid life histories is suggested; however, succession from emergent to shrub–scrub or forested wetland is not indicated.

Introduction

Wetland creation has often been required as compensatory mitigation under Section 404 of the Clean Water Act, but most studies focus on early plant composition (Kusler and Kentula 1990). The US Army Corps of Engineers and other regulatory agencies generally require a monitoring duration of 5 years or less (Mitsch and Wilson 1996). Monitoring efforts have often been limited to a few parameters, and most compensation wetlands have been created in the past 10 years. Evaluative studies typically exclude indicators of hydric soil since soil transformations are thought to be slow (Atkinson and Cairns 1994). Another frequently cited limitation of monitoring schemes is the inadequate or inherently variable hydrologic information. Thus, created wetland monitoring has tended to emphasize characteristics of colonizing vegetation.

Compensatory mitigation monitoring has not focused on predicting long-term success. In their Executive Summary, Kusler and Kentula (1990) state that revegetation of a restored or created wetland over a short period of time is of no guarantee that the wetland will continue to function over time. Race and Fonseca (1996) acknowledge a widespread lack of follow-up or monitoring and conclude that most monitoring efforts are conducted 'too early in the developmental stage to demonstrate success.' Bedford (1996) theorizes that development and persistence of wetland ecosystems are functions of longer-term hydrologic patterns. Zedler and Weller (1990) authored the overview of a large compilation on mitigation studies and state that the potential for persistence (resistance) of restored and constructed wetlands is one of the three greatest information needs in order to improve mitigation success.

Most authors agree that researchers still lack predictive capabilities for vegetative dynamics, even though revegetation is perhaps the most frequently monitored process. Created compensation sites have often received plantings of woody and/or herbaceous species that may have influenced diversity (Reinartz and Warne 1993). But colonizing herbaceous species have typically dominated early vegetative communities regardless of the targeted wetland type (several papers in Kusler and Kentula 1990; Atkinson et al. 1993; Mitsch and Wilson 1996; Vivian-Smith and Handel 1996).

Niering (1990) reviewed natural wetland vegetation dynamics and warned that short-term observations may overlook significant allogenic factors, including drought, fire, pollutants, or the introduction of aggressive invasive species. Reinartz and Warne (1993) state, however, that 'knowledge of vegetation development in created, isolated depressional wetlands is practically nonexistent.' Without identifying any particular data set, Mitsch and Wilson (1996) suggest that 15–20 years might be required for emergent wetlands to reach equilibrium and to judge success. However, predictions regarding development of community composition and equilibrium have not been validated for wetlands in the 20-year age group. The purpose of this manuscript is to describe the plant community composition and short-term changes (over 3 growing seasons) of eleven 20-yr-old created depressional wetlands in western Virginia, USA.

Methods

Site description

The study sites were located in the Appalachian plateau province in Wise County, Virginia, USA. Contour surface mining for coal disturbed over 385,000 ha in the Appalachian Mountains prior to the enactment of the Surface Mining Control and Reclamation Act of 1977 (SMCRA). Relict topographic features resulting from these pre-SMCRA mining operations include nearly vertical 'high walls' and fairly flat, but topographically irregular, 'benches' with severely compacted spoil where bulk density averaged 1.7 g cm^{-3} (Daniels and Amos 1982). While each bench was mostly well drained, several small (approximately 40 m diameter) depressions were accidentally created. The depressions in this study were formed from 1970 to 1974 and were all the result of surface mining; however, the exact year that wetland conditions became established could not be determined. Though artifacts of mining, these wetlands mimic a commonly used wetland creation technique in which surface horizon(s) are removed by heavy equipment and may be left without soil amendments (Hollands 1990).

The study sites were classified either as meadow or sunken-convex (Gosselink and Turner 1978), palustrine emergent persistent (Cowardin et al. 1979), or depressional wetlands (Brinson 1993). Each wetland was assigned a water regime modifier based on the classification scheme set forth in Cowardin et al. (1979). Site water regime modifiers included seasonally flooded (4 sites), semipermanently flooded (3 sites), and intermittently exposed (4 sites) (Table 1). The duration criterion of the Corps of Engineers for wetland hydrology (i.e., saturation or inundation within 25 cm of the soil surface for at least 5% of the growing season) (USACOE 1987) was exhibited by all plots in the 11 depressional wetland study sites.

Revegetation efforts were limited during the pre-SMCRA era, and alluvium was deposited over compacted mine spoil in small depressions (Herricks et al. 1974). Infiltration became restricted and hydroperiod lengthened as a result. Over time, hydrophytic vegetation colonized the depressions. Redoximorphic features were present in all 11 depressional wetland sites. Low mean chroma matrices (approximately 1.0 on the Munsell Soil Chart [Munsell Soil Color Charts, Kollmorgen Corporation, Baltimore, MD, USA] and oxidized rhizospheres were common; but no pattern of iron depletions or masses (mottling) was observed (Atkinson et al. 1998a, b). Most adjacent uplands were characterized by herbaceous species including

Table 1. Site reference number, hydrology (sources of water, duration dry, and hydrology modifier), and age (year formed) for 11 wetlands in this study.

Site	Inlets/outlets	Water depth[a]	Draw-down[b]	Hydrology[c]	Year formed	# Transects	# Plots
One	No/no	40.1	0.0	PEMe	1975	3	9
Two	Temp/perm	89.4	0.0	PEMe	1974	2	15
Three	No/temp	17.7	24.5	PEMc	1974	2	12
Four	No/no	17.8	13.1	PEMc	1973	3	12
Five	No/temp	61.7	12.5	PEMc	1973	2	46
Six	Temp/temp	20.1	4.2	PEMd	1973	3	14
Seven	Perm/perm	44.1	0.0	PEMe	1974	2	18
Eight	Temp/temp	8.4	20.9	PEMc	1970	3	12
Nine	Temp/temp	60.0	4.2	PEMd	1970–1974	2	12
Ten	No/no	32.0	8.3	PEMd	1970–1974	2	14
Eleven	Temp/temp	72.5	0.0	PEMe	1970–1974	2	9

[a] Maximum measured depth in cm.
[b] Percent of 3 growing seasons that the entire site was exposed.
[c] Modifiers based on Cowardin et al. (1979), include: PEM – palustrine system and emergent class; water regime modifiers include: c – seasonally flooded, d – semi-permanently flooded, and e – intermittently exposed.

Festuca rubra L., *Lespedeza cuneata* (Dumont) G. Don, *Trifolium repens* L., and *Solidago* spp. Terrestrial vegetation dynamics on surface mined lands in the region have been discussed elsewhere (Holl and Cairns 1994).

Hydrology

Climate data were obtained from the National Climatic Data Center (1999), which used the period from 1895 to 1990 to determine precipitation data. The Palmer Drought Severity Index (Heddinghause and Sabol 1991) was used to characterize precipitation conditions during the study period (1992–1994).

Depth of standing water was measured at a permanent stake in the deepest point of each wetland every 2 weeks during the 1992, 1993, and 1994 growing season (May 1–September 30). Since augers, and perhaps plant roots, could not penetrate the underlying compacted substrate, subsurface depth to water table was not measured. Depth of standing water relative to the permanent stake was calculated for each plot within each site using mean plot elevation, determined with a transit and stadia rod, based on three estimates per 1.0-m^2 plot. To quantify the exposure (lack of inundation) in each plot, a soil exposure index (SEI) was calculated. SEI was based on relative elevation of each plot and permanent stake, and the calculation also included the mean and standard deviation of water level that was measured at the permanent stake for each of the 1992, 1993, and 1994 growing seasons:

$$\mathrm{SEI} = 100 - x,$$

where x is the percent of 7 water level calculations that were >0 (inundated, i.e., standing water present), including mean relative plot depth and mean relative plot depth ± 1, 2, and 3 standard deviations of water level estimates, which were recorded at the permanent stake for that site, as described above. Thus, a plot that was inundated for 2 of the 7 (29%) calculations, SEI would be 71%, which means that the plot was exposed for approximately 71% of the growing season. Soil saturation was not measured directly and may have persisted during times of soil exposure.

Vegetation

Vegetative cover of all species was measured in the same 1.0-m^2 plots in which relative depth and SEI were measured. Plots were positioned in 5-m increments along two or three parallel transects per site based on wetland size and in order to represent communities, following Mueller-Dombois and Ellenberg (1974) (Table 1). In 1992 and 1994, vegetation cover was estimated in a total of 173 plots during mid-August (peak growing season).

A weighted average (WA), also known as a Prevalence Index, of the vegetation community was calculated for each of the plots in August 1992 and again in August 1994. The WA was calculated following the conceptual approach of Goff and Cottam (1967). WA has been used to assess the wetland status of vegetation plots along moisture gradients to delineate wetlands under natural conditions (Whittaker 1951; Carter et al. 1988; Wentworth et al. 1988; Eicher 1988; Scott et al. 1989) and as a component of wetland compensation monitoring (Atkinson et al. 1993). The calculation was based on species regional indicator status (Reed 1988) and relative cover of each species in 1.0-m^2 plots:

$$\mathrm{WA} = (y^1u^1 + y^2u^2 + \cdots + y^mu^m)/100$$

where $y^1, y^2, \ldots, y^m$ are the relative cover estimates for each species in the plot, and $u^1, u^2, \ldots, u^m$ are the wetland indicator values of each species (1 – obligate wetland; 2 – facultative wetland; 3 – facultative; 4 – facultative upland; 5 – upland) (Reed 1988). Thus a high WA indicates a plot with less hydrophytic vegetation.

Species richness (SR) was calculated as number of species per 1×1 m plot per year for 1992 and again in 1994. Modified vegetation importance values (IV) (Daubenmire 1959) were calculated for all species in the plots, based on the sum of relative cover and relative frequency. Plant identification followed Radford et al. (1968) and nomenclature was based on The PLANTS Database, version 3.1 (USDA 2001).

Data analyses

All data presented in this manuscript were collected from plots that met the criteria for a wetland (inundation for $> 5\%$ of the 1992 growing season, a 1992 WA < 3.0, and positive indicators of hydric soils) set forth in the US Army Corps of Engineers wetland delineation manual (USACOE 1987). Yearly comparisons of data were based on two-tailed paired t-tests unless stated otherwise. Pearson correlation coefficients were used to test for correlative relationships between the 1992 and 1994 WA. Weighted average and soil exposure were related using simple linear regression (Zar 1984). Variability in the data was described by standard deviation (SD) unless stated otherwise. A p-value of 0.05 was used to determine statistical significance.

Results

Hydrology

According to the National Data Climate Center (1999), 1993 was the second lowest water year on record for Virginia. Total precipitation in 1993 (100.6 cm) was lower than either 1992 (132.6 cm) or 1994 (128.0 cm). In 1993, the Palmer Drought Severity Index (PDSI) for August was classed as 'severe drought' and July, September, October, and November were classed as 'mild drought'.

Soil exposure index during the 1993 growing season (45%, SD 28.1) was higher than either 1992 or 1994 (16%, SD 23.9 and 26%, SD 26.0, respectively) ($p < 0.0001$). All surface water (inundation) disappeared at some time during 1993 from 7 of the 11 wetlands, but inundation was reestablished in every site by January 1994. Mean surface water depth during the growing season was 25.2 cm in communities dominated by obligate wetland species and 0.60 cm in communities dominated by facultative wetland species.

Weighted average

SEI was a good predictor of WA in both 1992 and 1994 (R^2 of 0.57 and 0.53, respectively) (Figures 1a, b). Mean plot WA increased from 1.45 in 1992 to 1.52 in 1994 ($p < 0.0001$). The Pearson correlation coefficient for WA in 1992 and 1994 was 0.91 ($p < 0.0001$). Site WA unchanged at most sites in 1994, but increased slightly at site 5 ($p < 0.001$) and site 3 ($p < 0.03$) (Figure 2).

Species richness

Mean plant SR among plots increased from 1992 to 1994 (mean of 3.91 species per plot to 4.62, respectively) ($p < 0.0001$), which represents an 18% increase in the number of species per plot. SR values in 1992 were good predictors of 1994 SR ($R^2 = 0.63, p < 0.0001$). Total number of species present in the 173 plots remained nearly constant both years: 56 and 57 species in 1992 and 1994, respectively. SR increased in four sites based on means for all plots in each site (Figure 3). There was no difference in mean number of species per

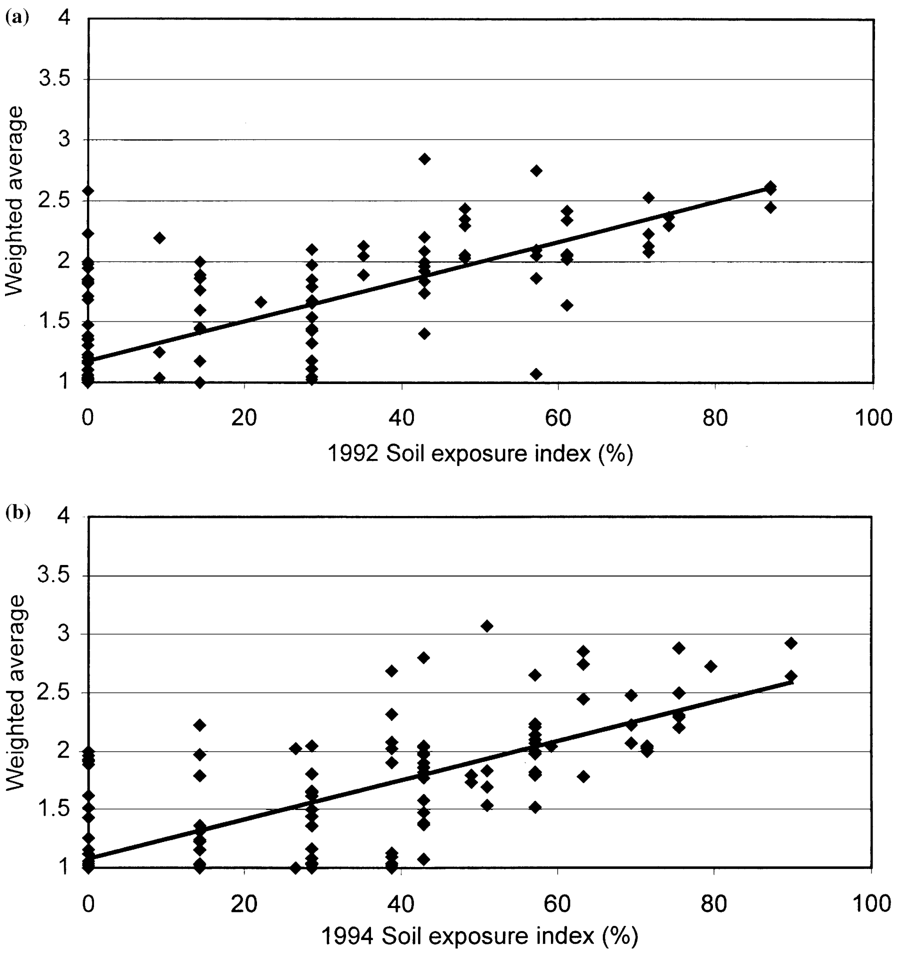

Figure 1. (a) Soil exposure index and weighted average for all 173 wetland plots in 1992 (before drought). Regression of weighted average and soil exposure index yields $R^2 = 0.57$ ($p < 0.0001$). (b) Soil exposure index and weighted average for all 173 wetland plots in 1994 (after drought). Regression of weighted average and soil exposure index yields $R^2 = 0.53$ ($p < 0.0001$).

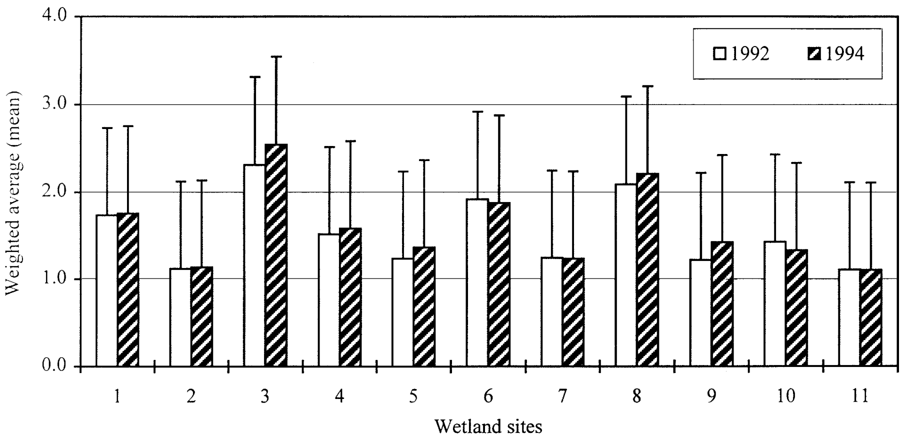

Figure 2. Mean plot weighted average in 1992 and 1994 for 11 wetlands. Means in 1992 and 1994 did not differ, except for sites 3 and 5.

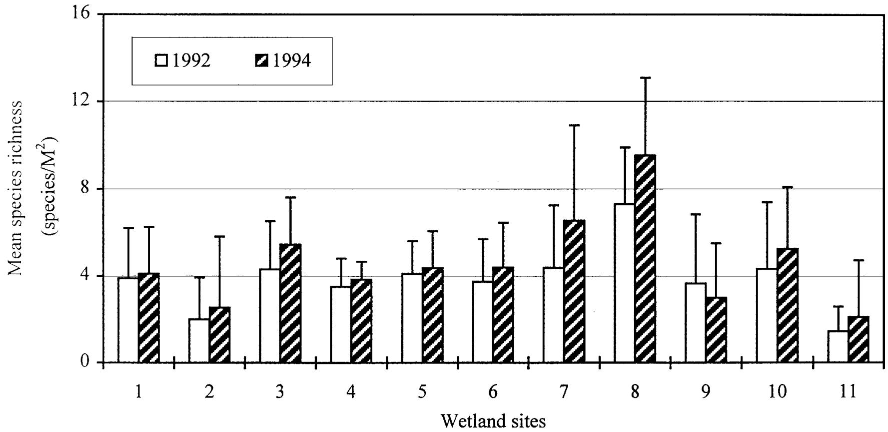

Figure 3. Mean plot species richness in 1992 and 1994 for 11 wetlands. Means in 1992 and 1994 are different for sites 3, 7, 8, and 10.

site in 1992 (18.8 species) compared to 1994 (21.3 species) (p = 0.30).

SR and SEI were positively related in 1992 and 1994 (R^2 = 0.28, p < 0.0001; and R^2 = 0.26, p < 0.0001, respectively). The lowest SR was found in the 32 continuously inundated plots (SR = 1.56 in 1992 and 2.06 in 1994, respectively).

Vegetation cover and importance values

Vegetation cover among obligate wetland and facultative upland indicator status categories showed the greatest changes in relative cover over the study period. In the 75 plots that exhibited increased WA, obligate wetland species relative cover decreased an average of 18%, while relative cover of facultative upland species increased 6% in these plots (Figure 4).

The dominance ranking by IV is similar between the 2 years. *Typha latifolia*, an obligate wetland species, had the highest IV rank and *Scirpus cyperinus*, a facultative wetland species, had the second highest for both 1992 and 1994. IVs for these two species sum to 86.3 in 1992 and 81.2 in 1994, which is 5-fold higher in IV than the third most important species each year (Table 2). Of the 10 species with highest IVs, 9 were perennials and 1, *Eleocharis obtusa*, was an annual. There were only 5 annuals among the 57 species present in the 11 sites.

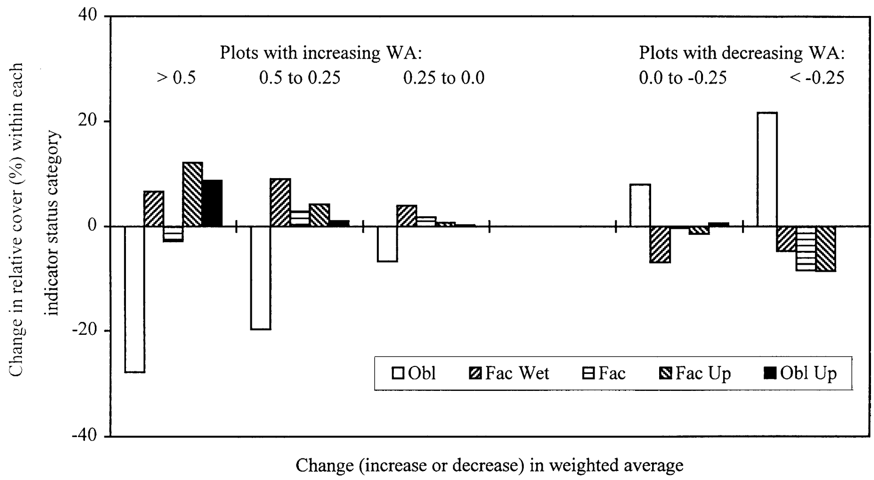

Figure 4. Change in percent cover among indicator status categories for each 0.25 increment of change in plot weighted average. Weighted average increased in 75 plots, was unchanged in 61 plots (not shown), and decreased in 37 plots.

Table 2. Species with 10 highest IVs in 1992 and 1994, based on 173 permanently marked 1.0 m^2 plots.

1992 Species	IS	H	RC	RF	IV	1994 Species	IS	H	RC	RF	IV
Typha latifolia L.	1	E	34.9	15.6	50.5	*Typha latifolia* L.	1	E	32.0	14.4	46.4
Scirpus cyperinus (L.) Kunthe	2	E	23.8	12.0	35.8	*Scirpus cyperinus* (L.) Kunthe	2	E	23.0	11.8	34.8
Potomogeton pulcher Tuckerman	1	S	8.5	8.0	16.6	*Juncus acuminatus* Michaux	1	E	6.0	7.9	13.9
Juncus acuminatus Michaux	1	E	4.0	7.3	11.3	*Eleocharis obtusa* (Willd.) Schultes	1	E	3.8	5.9	9.7
Eleocharis obtusa (Willd.) Schultes	1	E	3.1	6.4	9.5	*Juncus effusus* L.	2	E	5.5	4.1	9.6
Sparganum americanum Nuttall	1	E	6.9	2.4	9.2	*Sparganum americanum* Nuttall	1	E	6.5	2.1	8.7
Juncus effusus L.	2	E	3.7	4.2	7.8	*Potomogeton pulcher* Tuckerman	1	S	3.8	4.5	8.3
Salix nigra Marshall	2	W	2.5	3.1	5.6	*Solidago gigantea* Aiton	3	E	1.7	4.4	6.1
Ludwigia palustris (L.) Ell	1	E	1.1	4.3	5.4	*Festuca rubra* L.	3	E	1.4	3.6	5.0
Potamogeton foliosus Raf.	1	S	1.4	3.9	5.3	*Ludwigia palustris* (L.) Ell.	1	E	1.4	3.3	4.7

IS = Indicator status of 1 = obligate wetland, 2 = facultative wetland, 3 = facultative, 4 = facultative upland, 5 = obligate upland based on Reed (1988). H = habit or growth form, including S = submersed, E = emergent and herbaceous, and W = woody, based on Radford et al. (1968). RC = relative cover; RF = relative frequency; and IV = importance value (sum of RC and RF, totals 200). For complete species list, see Atkinson and Cairns (1994).

Mean relative cover for *T. latifolia* plots in 1992 was 57.1% per plot, which decreased to 49.5% in 1994 ($p = 0.005$). Most species exhibited increased frequency such that relative frequency for *T. latifolia* decreased from 15.6 to 14.4%, even though it became established in 16 plots and disappeared from 5 plots (Table 2). SEI of these 16 plots increased by 22% in 1993 compared with 1992 ($p < 0.0001$). Also in these 16 plots, SR increased from 3.5 species per plot in 1992 to 4.00 in 1994. For the 105 plots in which *T. latifolia* remained, SR also increased from 5.69 to 7.81 species per plot in 1992 and 1994, respectively. In spite of the changes in SEI and SR in these 16 plots, mean WA did not increase from 1992 (1.73) to 1994 (1.87) ($p = 0.14$).

There were 14 plots comprised of *T. latifolia* monocultures in 1992 (i.e., 100% relative cover by *T. latifolia*). A total of 16 plots with *T. latifolia* monocultures were recorded in 1994, including 13 of the original and 3 additional plots. The persistence of *T. latifolia* in these plots may be a result of persistent inundation, since only one of the original 14 plots exhibited a SEI > 0% at any time during the 3 years of the study. SEI among the remaining 91 plots increased from 5% in 1992 to 17% in 1994 ($p < 0.0001$), which indicated that increased soil exposure may have led to the decline in *T. latifolia* relative cover from 1992 (51%) to 1994 (42%) ($p < 0.001$).

Scirpus cyperinus became established in more plots than it disappeared from but the expansion was not associated with an increase in WA. *Scirpus cyperinus* occurred in 80 plots in 1992 compared with 93 plots in 1994, and only disappeared from 1 plot during this study. Mean relative cover for the original 80 plots of this species was unchanged, averaging 49% in 1992 and 46% in 1994 ($p = 0.13$).

In the 14 plots in which *S. cyperinus* became established in 1994, the 1992 species composition had been dominated by obligate wetland species and mean SEI increased from 9% in 1992 to 55% in 1993 ($p < 0.0001$). Mean SR increased from 5.1 to 7.6 species per plot, but obligate wetland species were among the colonizers such that mean WA did not increase in these 14 plots (WA = 1.31 in 1992 and 1.39 in 1994, $p = 0.09$).

Discussion

Vegetation composition

Annuals often dominate young created wetlands, but were not important in these 20-yr-old sites. Seven of the 10 species with highest importance values in a study of a 4-yr-old created wetland in a Petersburg, VA, USA, were annual species, including the two most important species (Atkinson et al. 1993). In Wise County, VA, USA, six small depression wetlands were created for wildlife post-mining land uses. Those sites were dominated by two annuals, *Echinochloa walteri* and *Eleocharis obtusa* (Atkinson et al. 1998a, b). However, in the 20-yr-old sites, *Eleocharis obtusa* was the only annual of the 10 most important species, and 52 of the 57 species were perennials.

Typha latifolia was the most dominant perennial species in the current study and may have been important throughout the entire vegetation development sequence. Fowler et al. (1985) found 11 vascular plant species in 9 newly constructed coal sediment ponds in a surface mined landscape in Tennessee, but only *T. latifolia* was present in all sites. Reinartz and Warne (1993) working in 11 created small depressions in southeastern Wisconsin found that *T. latifolia* cover increased over each of the 3 years in their study. *Typha latifolia* was among the first species that colonized newly created small depressions in southwestern Virginia (Atkinson et al. 1998a, b).

Typha latifolia monocultures are widely considered ecologically undesirable because of their exclusion of other macrophyte species (Odum 1987). However, it is unclear whether it is as aggressive in the 20-yr-old small depressions in the current study. It only occurred as a true monoculture in 14 of the 114 plots where it was found during 1992. The remaining 100 plots that contained *T. latifolia* exhibited a mean SR of 4.2 species m^{-2}, which is similar to SR for the 59 plots in which *T. latifolia* did not occur (4.7 species m^{-2}). Furthermore, it decreased in IV over the short duration (3 years) of the current study.

Many wetland compensation sites are intended to undergo succession from emergent to forested physiognomy. In this study, only one woody species was among the 10 most dominant species. However, caution is advised in interpreting the results of this study since modern construction techniques may differ, including the addition of soil amendments, seed banks, and/or the planting of woody species.

Evidence for equilibrium

Mitsch and Wilson (1996) postulated that, in created emergent wetlands, little change would occur after the wetland reached 20 years of age. Their postulate is supported by our structural data including SR, IV, life history strategy, and WA. There was an 18% increase in the mean number of species present in each plot in 1992–1994, however, there was no change in total species present on each site and only one new species appeared between 1992 and 1994. Other studies have reported increased germination rates with increased soil exposure, as was associated with the drought in 1993 (van der Valk 1981); however, shifts in vegetative dominance were minor. IV rank did not change for the two most dominant species (*T. latifolia* and *Scirpus cyperinus*) and 9 of the top 10 most dominant species were little changed over the 3 years. Perennial, non-woody species remained the most important life history strategy and WA did not change.

The wetlands in our study exhibited hydrogeomorphology similar to prairie glacial marshes. Weller and Spatcher (1965) described a cyclical successional sequence in prairie glacial marsh vegetation, which was expanded by van der Valk and Davis (1978) based on detailed seed bank and floristic cover analyses. Prairie glacial marshes exhibit a four-stage cycle, including (1) dry marsh, (2) regenerating marsh, (3) degenerating marsh, and (4) lake marsh. The 1993 drought could have initiated the dry stage and more normal precipitation rates in 1994 correspond to the regenerating stage, which resulted in increased SR as predicted by van der Valk and Davis (1978). However, the wetlands in the current study do not appear to exhibit two of the hydrologically linked stages in prairie pothole succession, including the deeper 'lake marsh' and 'degenerating marsh'. This may be related to the fact that droughts in prairie potholes are often of far longer duration than that of the current study.

Evidence for stability

Stability cannot be directly evaluated in this study because the drought was classified as 'severe' for only one month and because only structural parameters were presented. However, the 20-yr-old wetlands may possess mechanisms which foster tolerance of at least minor droughts. Biological mechanisms may include the dominance of established perennial species, water holding potential of the accumulated plant litter as reported by Atkinson and Cairns (2001), and the seed bank and/or seed dispersal from adjacent wetland vegetated zones. The persistence of perennial species in the present study suggests a similarity to the inhibition model of succession described by Connell and Slayter (1977). Because the perennial species appear to be tolerant of significant allogenic disturbance, these wetlands may have reached a form of 'relative stability' (Niering 1990), which may continue until the occurrence of a more effective disturbance such as fire, disease, persistent drought, or altered land use.

The sites in this study have been pre-exposed to dry/wet fluctuations, which may have reduced the effect of drought on the vegetation. Mean annual precipitation in 1989 was lower than that for the 1993 drought (National Climate Data Center 1999) and similar events may have selected for more tolerant plant communities and reduced short-term response, termed 'reactivity' by Neubert and Caswell (1997).

Conclusion

The communities of these 20-yr-old wetlands appeared to have reached a stage of equilibrium that is dominated by herbaceous perennial vegetation. However, no reference sites were monitored and this study does not claim successful compensation. Most of the wetlands in southwestern Virginia are associated with riparian systems and our sites lacked this source of water. Natural wetlands might not contain the same vegetative structure nor demonstrate a drought response that is similar to what we measured in these sites, i.e., natural wetlands might exhibit a greater response to the drought. It is possible that a longer drought, such as the multi-year droughts common among prairie pothole wetlands, would have had a greater effect on vegetative structure.

Success in wetland compensation is often measured by assessing vegetative structure, such as plant cover and density. However, some authors have reported that the connection of structural parameters to ecological function is often overstated (Perry et al. 2001). In this work we found some components of wetland vegetative structure; however, published studies from the same sites have indicated that some functions have not been replaced, i.e., decomposition (Atkinson and Cairns 2001). Future studies need to better identify the connectivity between measures of structure and specific functions so that the success of wetland compensation can be improved. As suggested by other studies (e.g., Mitsch and Wilson 1996; Simenstad and Thom 1996), monitoring terms should be extended beyond the typical 5 years that is currently required in order to support claims of success in created wetlands.

Acknowledgments

John Crisafulli, John Heckman, Karen Holl, David Jones, Greg Noe, Colin Rosenquist, and Mara Sabre assisted with fieldwork, and Darla Donald provided editorial assistance. David Knepper provided useful comments on an earlier draft manuscript. Assistance from the Powell River Project and the Biology Department of Virginia Tech is gratefully acknowledged. The Office of Surface Mining Reclamation and Enforcement, US Department of the Interior provided financial support for this project.

References

Atkinson R.B. and Cairns J. Jr. 1994. Possible use of wetlands in ecological restoration of surface mined lands. J. Aquat. Ecosyst. Health 3: 139–144.

Atkinson R.B., Perry J.E., Smith E.P. and Cairns J. Jr. 1993. Use of constructed wetland delineation and weighted averages as a component of assessment. Wetlands 13(2): 185–193.

Atkinson R.B., Daniels W.L. and John Cairns Jr. 1998a. Hydric soil development in depressional wetlands: a case study from surface mined landscapes. In: Majumdar S.K., Miller E.W. and Brenner F.J. (eds), Ecology of Wetlands and Associated Systems. Pennsylvania Academy of Science, pp. 182–197.

Atkinson R.B., Daniels W.L. and Cairns J. Jr. 1998b. Hydric soil development in depressional wetlands: a case study from surface mined landscapes. In: Majumdar S.K., Miller E.W. and Brenner F.J. (eds), Ecology of Wetlands and Associated Habitats. Pennsylvania Academy of Science, Philadelphia, Pennsylvania, pp. 182–197.

Bedford B.L. 1996. The need to define hydrologic equivalence at the landscape scale for freshwater wetland mitigation. Ecol. Appl. 6(1): 57–68.

Brinson M.M. 1993. A hydrogeomorphic classification for wetlands. Technical Report WRP-DE-4. US Army Engineer Waterways Experiment Station, Vicksburg, Mississippi.

Carter V., Garrett M.K. and Gammon P.T. 1988. Wetland boundary determination in the Great Dismal Swamp using weighted averages. Water Res. Bull. 24(2): 297–306.

Connell J.H. and Slayter R.O. 1977. Mechanisms of succession in natural communities and their role in community stability and organization. Am. Nat. 111: 1119–1144.

Cowardin L.M., Carter V., Golet F.C. and LaRoe E.T. 1979. Classification of Wetlands and Deepwater Habitats of the United States. US Fish and Wildlife Service, Washington, DC.

Daniels W.L. and Amos D.F. 1982. Generating productive topsoil substitutes from hard rock overburden in the southern Appalachians. Environ. Geochem. Health 7: 8–15.

Daubenmire R.F. 1959. Canopy coverage methods of vegetation analysis. Northwest Sci. 33: 43–64.

Eicher A.L. 1988. Soil–vegetation correlations in wetlands and adjacent uplands of the San Francisco Bay Estuary, California. US Fish and Wildlife Service, Washington, DC.

Fowler D.K., Hill D.M. and Fowler L.J. 1985. Colonization of coal surface mine sediment ponds in southern Appalachia by aquatic organisms and breeding amphibians. In: Brooks R.P., Samuel D.E. and Hill J.B. (eds), Wetlands and Water Management on Mined Lands. The Pennsylvania State University, State College, Pennsylvania, pp. 261–285.

Goff F.G. and Cottam G. 1967. Gradient analysis: the use of species and synthetic indices. Ecology 48(5): 793–806.

Gosselink J.G. and Turner R.E. 1978. The role of hydrology in freshwater wetland ecosystems. In: Good R.E., Whigham D.F. and Simpson R.L. (eds), Freshwater Wetlands: Ecological Processes and Management Potential. Academic Press, New York, pp. 63–78.

Heddinghause T.R. and Sabol P. 1991. A review of the Palmer drought severity index and where do we go from here? Proceedings of the Seventh Conference on Applied Climatology. American Meteorological Society, Boston, Massachusetts.

Herricks E.E., Cairns J. Jr. and Shanholtz V.O. 1974. Preplanning mining operations to reduce environmental impact of mine drainage on streams. Conference Proceedings. American Water Resources Association, Washington, DC.

Holl K.D. and Cairns J. Jr. 1994. Vegetational community development on reclaimed coal surface mines in Virginia. Bull. Torrey Bot. Club 121(4): 327–337.

Hollands G.G. 1990. Regional analysis of the creation and restoration of kettle and pothole wetlands. In: Kusler J.A. and Kentula M.E. (eds), Wetland Creation and Restoration, the Status of the Science. Island Press, Washington, DC, pp. 281–296.

Kusler J.A. and Kentula M.E. 1990. Executive summary. In: Kusler J.A. and Kentula M.E. (eds), Wetland Creation and Restoration, the Status of the Science. Island Press, Washington, DC, pp. 17–25.

Mitsch W.J. and Wilson R.F. 1996. Improving the success of wetland creation and restoration with know-how, time, and self-design. Ecol. Appl. 6(1): 77–83.

Mueller-Dombois D. and Ellenberg H. 1974. Aims and Methods of Vegetation Ecology. Wiley, New York, NY.

National Climatic Data Center. 1999. Historical climatology series, online. National Oceanic and Atmospheric Administration US Department of Commerce, Washington, DC.

Neubert, M.G. and Caswell H. 1997. Alternatives to resilience for measuring the responses of ecological systems to perturbations. Ecology 78(3): 653–665.

Niering W.A. 1990. Vegetation dynamics in relation to wetland creation. In: Kusler J.A. and Kentula M.E. (eds), Wetland Creation and Restoration, the Status of the Science. Island Press, Washington, DC, pp. 479–486.

Odum W.E. 1987. Predicting ecosystem development following creation and restoration of wetlands. In: Zelazny J. and Feierabend J.S. (eds), Wetlands: Increasing Our Wetland Resources. National Wildlife Federation, Washington, DC, pp. 67–70.

Perry J.E., Barnard T., Bradshaw J., Friedrichs C., Havens K., Mason P., Priest W. and Silberhorn G. 2001. Created tidal wetlands in the Chesapeake Bay: a case study. J. Coast. Res. Special Issue 27: 170–192.

Race M.S. and Fonseca M.S. 1996. Fixing compensatory mitigation: what will it take? Ecol. Appl. 6(1): 94–101.

Radford A.E., Ahles H.E. and Bell C.R. 1968. Manual of the Vascular Flora of the Carolinas. University of North Carolina Press, Chapel Hill, North Carolina, USA, 1183 pp.

Reed P.B. Jr. 1988. National List of Plant Species that Occur in Wetlands: Virginia. NERC-88/18.46. US Fish and Wildlife Service, St. Petersburg, Florida.

Reinartz J.A. and Warne E.L. 1993. Development of vegetation in small created wetlands in southeastern Wisconsin. Wetlands 13(3): 153–164.

Scott M.L., Slauson W.L., Segalquist C.A. and Auble G.T. 1989. Correspondence between vegetation and soils in wetlands and nearby uplands. Wetlands 9(1): 41–60.

Simenstad C.A. and Thom R.M. 1996. Functional equivalency trajectories of the restored Gog-Le-Hi-Te estuarine wetland. Ecol. Appl. 6(1): 38–56.

United States Army Corps of Engineers (USACOE) 1987. Corps of Engineers Wetlands Delineation Manual. Technical Report Y-87–1, Washington, DC, p. 99, Appendices.

United States Department of Agriculture (USDA) 2001. The PLANTS Database, Version 3.1 (http://plants.usda.gov). NRCS National Plant Data Center, Baton Rouge, LA 70874–4490 USA.

van der Valk A.G. 1981. Succession in wetlands: a Gleasonian approach. Ecology 62(3): 688–696.

van der Valk A.G. and Davis C.B. 1978. The role of seed banks in vegetation dynamics of prairie glacial marshes. Ecology 59(2): 322–335.

Vivian-Smith G. and Handel S.N. 1996. Freshwater wetland restoration of an abandoned sand mine: seed bank recruitment dynamics and plant colonization. Wetlands 16(2): 185–196.

Weller M.W. and Spatcher C.S. 1965. Role of habitat in the distribution and abundance of marsh birds. Special Report No. 43. Iowa Agriculture and Home Economics Experiment Station, Ames, Iowa, USA.

Wentworth T.R., Johnson G.P. and Kologiski R.L. 1988. Designation of wetlands by weighted averages of vegetation data: a preliminary evaluation. Water Res. Bull. 24(2): 389–396.

Whittaker R.H. 1951. A criticism of the plant association and climatic climax concepts. Northwest Sci. 25: 17–31.

Zar J.H. 1984. Biostatistical Analysis. Prentice-Hall, Englewood Cliffs, New Jersey.

Zedler J.B. and Weller M.W. 1990. Overview and future directions. In: Kusler J.A. and Kentula M.E. (eds), Wetland Creation and Restoration, the Status of the Science. Island Press, Washington, DC, pp. 405–413.