Comparison of soil organic matter in created, restored and paired natural wetlands in North Carolina

Gregory L. Bruland^{1,2,*} and Curtis J. Richardson¹

¹Duke University Wetland Center, Nicholas School of the Environment and Earth Sciences, Box 90333, Durham, North Carolina 27708-0333, USA; ²Present address: University of Hawaii at Manoa, Department of Natural Resources and Environmental Management, 1910 East-West Rd., Honolulu, HI 96822, USA; *Author for correspondence (e-mail: bruland@hawaii.edu; phone: +1-808-956-8901; fax: +1-808-956-6539)

Received 1 June 2004; accepted in revised form 15 July 2005

Key words: Hydrogeomorphic subclass, Non-tidal forested wetland, Soil organic matter, Wetland creation, Wetland restoration

Abstract

Soil organic matter (SOM) content is a key indicator of soil quality and is correlated to a number of important soil processes that occur in wetlands such as respiration, denitrification, and phosphorus sorption. To better understand the differences in the SOM content of created (CW), restored (RW), and paired natural wetlands (NWs), 11 CW/RW-NW pairs were sampled in North Carolina. The site pairs spanned a range of hydrogeomorphic (HGM) subclasses common in the Coastal Plain. The following null hypotheses were tested: (1) SOM content of paired CW/RWs and NWs are similar; (2) SOM content of wetlands across different HGM subclasses is similar; and (3) interactions between wetland status (CW/RW vs. NW) and hydrogeomorphic subclass are similar. The first null hypothesis was rejected as CW/RWs had significantly lower mean SOM ($11.8 \pm 3.9\%$) than their paired NWs ($28.98 \pm 8.0\%$) on average and at 10 out of the 11 individual sites. The second and third null hypotheses were also rejected as CW/RWs and NWs in the non-riverine organic soil flat subclass had significantly higher mean SOM content ($31.08 \pm 14.2\%$) than the other three subclasses (8.18 ± 2.5 , 11.18 ± 8.2 , and $10.38 \pm 4.2\%$). Individual sites within this fourth subclass also had significantly different SOM content. This indicated that it would be inappropriate to include the organic soil flat subclass with either the riverine or non-riverine mineral soil flat subclasses when considering restoration guidelines. These results also suggested that if there is a choice in mitigation options between restoration or creation, wetlands should be restored rather than created, especially those in the non-riverine organic soil flat subclass.

Introduction

Galatowitsch and Van der Valk (1996) stated that the definitive test of success for created (CWs) and restored wetlands (RWs) is how closely they function like natural wetlands (NWs). Currently,

the ability of CWs and RWs to replace NW functions is a topic of considerable debate (Zedler and Callaway 1999; Stolt et al. 2000; Hunter and Faulkner 2001; Spencer et al. 2001). Soil properties of CW/RWs have almost always been shown to differ from NWs (Verhoeven et al. 2001). For

example, CWs usually have lower soil organic matter (SOM) content than NWs (Confer and Niering 1992; Bishel-Machung et al. 1996; Galatowisch and Van der Valk 1996; Shaffer and Ernst 1999; Whittecar and Daniels 1999; Campbell et al. 2002). Such soil conditions can lead to low growth and survival of planted and colonizing species. Litter layers in CWs and RWs are often poorly developed or absent in comparison to those in NWs (Hunter and Faulkner 2001; Verhoeven et al. 2001). As a result of the low SOM content and paucity of litter, the microbial communities in the soil of RWs and CWs are considered less viable than those of NWs (Duncan and Groffman 1994). As very few created or restored wetland projects take any consideration of soil characteristics (Shaffer and Ernst 1999), additional data on the development of soil properties in CW/RWs are needed to indicate whether they are actually proceeding toward natural wetland ecosystems and if they will be able to replace the functions performed by the impacted wetlands (Spencer et al. 2001). Furthermore, soil characteristics are useful parameters for comparison across sites, as some can be easily measured in the field or laboratory (Bishel-Machung et al. 1996).

For this investigation, we chose to measure SOM content, shown to be: (1) an excellent indicator of soil quality; (2) well correlated with a number of important wetland processes such as respiration (Howard and Howard 1993), denitrification (Brettar and Höfle 2002), and P sorption (Axt and Walbridge 1999); (3) significantly different in CW/RWs and NWs (Confer and Niering 1992; Bishel-Machung et al. 1996; Shaffer and Ernst 1999; Stolt et al. 2000; Campbell et al. 2002); and (4) a limiting factor in the development of CW/RWs (Shaffer and Ernst 1999). This investigation was limited to non-tidal forested CW/RWs for two reasons: (1) non-tidal CW/RWs have been studied much less than their tidal counterparts; and (2) natural non-tidal forested wetlands have been shown to perform important nutrient transformation and retention functions (Lowrance 1992; Walbridge and Struthers 1993; Axt and Walbridge 1999). To control for differences in geology, soils, vegetation, and climate, site pairs were located in areas of similar topographic settings, identical mapped USDA soil series when possible, or in a closely related soil series.

Thus, the objective of this study was to compare the differences in SOM content of CW/RWs to those of paired NWs across a range of four hydrogeomorphic (HGM) settings common in the Southeastern Coastal Plain of the United States (Brinson 1993). Previous research has established that wetlands should be grouped by HGM setting when comparing ecosystem properties and functions (Brinson and Rheinhardt 1996; Rheinhardt et al. 1998; Verhoeven et al. 2001). The following null hypotheses were tested: (1) SOM content of paired CW/RWs and NWs are similar; (2) SOM content of wetlands across different HGM subclasses is similar; (3) interactions between wetland status (CW/RW vs. NW) and hydrogeomorphic subclass are similar, i.e. the differences between CW/RWs and NWs will be consistent across all subclasses.

Methods

Study sites

All sites selected for this study were compensatory wetland mitigation sites and were located in the Coastal Plain physiographic region of North Carolina (Table 1, Figure 1). Nine of the sites belonged to the North Carolina Department of Transportation (NCDOT), while the other two sites, Bailey Creek and Parker Farm, belonged to the PCS Phosphate Company. The sites were grouped into four different hydrogeomorphic subclasses common in the southeastern Coastal Plain including: (1) headwater riverine (stream order ≤ 2); (2) mainstem riverine (stream order >2); (3) non-riverine mineral soil flat; and (4) non-riverine organic soil flat (Brinson 1993; Cole et al. 1997). Paired plots were used to control for differences in geology, soils, and vegetation that occur across the Coastal Plain. To narrow the causes of edaphic differences between the CW/RW and NW plots to prior land-use and mitigation activities, sites were selected that contained both CW/RW and NW areas located in similar HGM settings, in a closely-related or identical mapped USDA soil series. All the CW/RWs sampled in this study were relatively young, between three and nine years since creation or restoration. A more detailed description of each of the site pairs can be found in Bruland (2004).

Figure 1. Locations of the 11 paired wetland study sites in the Coastal Plain of North Carolina.

Soil sampling

At each site, two transects were established, one in the CW/RW and the other in the NW. At the riverine sites (Rowel Branch, Wiggins Mill, Bailey Creek, Grimesland, and Haws Run), transects ran parallel to the direction of water flow, while at the non-riverine sites (ABC, Balance Farm, Mildred Woods, Barra Farms, Parker Farms, and Dismal Swamp), transects were oriented in a random direction. Along each transect, three to six soil cores were collected based on the size of the site, with a greater number of cores collected at the larger sites. During the spring and summer of 2002 cores were collected from the upper 20 cm of the soil profile with a 5 cm diameter piston corer. The upper 20 cm was chosen for sampling as this zone is the most biologically active and is subject to pedogenic processes of erosion and sedimentation. Upon arrival at the Duke Wetland Center Laboratory, the cores were extruded from plastic sleeves and oven dried at 105 °C for 24 h. Once dried, the cores were ground and passed through a 2 mm sieve to remove rock fragments and other large organic debris. The sieved soil was then used for the determination of soil organic matter (SOM) content by loss on ignition at 450 °C for 4 h (Campbell et al. 2002). The SOM data was reported on a mass basis to facilitate comparisons with other studies of created and restored wetlands.

Statistical analyses

The SOM data was analyzed with a general linearized model (PROC GLM, SAS for Windows

Version 8.2, SAS Institute, Cary, North Carolina). The GLM approach allowed for an unbalanced sampling design (five riverine and six non-riverine sites) as well as the different number of cores (from 3 to 6) collected at each site. The GLM tested the effects of wetland status (created/restored vs. natural), HGM setting (headwater riverine, mainstem riverine, non-riverine mineral soil flat, non-riverine organic soil flat), individual sites within HGM settings (Site[HGM]) effects for Rowel Branch, Wiggins Mill, Bailey Creek, Grimesland, Haws Run, ABC, Balance Farm, Mildred Woods, Barra Farm, Parker Farm, and Dismal Swamp), and the status by site within HGM setting interaction (Status*Site[HGM]). Differences between mean values of created/restored and natural wetlands, the four HGM subclasses, individual sites, and their interactions were analyzed with a least squares test in SAS. Log-transformation of the data was necessary to partially satisfy the GLM assumptions of normality and homogeneous variances. Despite the somewhat skewed data following transformation, multiple factor ANOVAs are robust in handling such slightly non-normal distributions (Darke and Walbridge 2000). Data in the figures was presented in the untransformed rather than the log scale for ease of interpretation.

Results

The mean SOM content for all 11 CW/RWs sampled in this study was $11.8 \pm 3.9\%$ (1 standard error). This was significantly lower ($p < 0.001$) than the mean value for SOM content in the paired NWs at $28.9 \pm 8.0\%$ (Figure 2a, Table 2). When

Table 1. Site characteristics of the 11 paired sites sampled in this study including name, HGM subclass, size, construction dates, and mapped soil series present.

Site name	HGM subclass	Size (ha)	Constructed	Soil series present	Prior Land Use (PLU) and Mitigation Activities (MA)
Rowel Branch	Headwater Riverine	6.5	2000–2001	Foreston, Muckalee, Rains	PLU: FP † ditched and filled MA: Fill removed and stream rerouted into relic channel
Wiggins Mill	Headwater riverine	36	2000–2001	Bibb, Rains, Tomotely	PLU: Headwater wetlands cleared, ditched, used for AG‡, SILV§, pasture MA: Ditches filled, sinuous stream channels restored
Bailey Creek	Headwater riverine	1.2	1995	Dorovan, Muckalee	PLU: Upland and wetland subject to phosphate mining MA: Excavation, creation of new channel and FP
Grimes-land	Mainstem riverine	3.1	1999–2000	Lakeland, Portsmouth, Swamp	PLU: Upland subject to sand mining MA: Borrow pit lake filled, graded to FP elevation
Haws Run	Mainstem riverine	11	1999	Muckalee, Foreston	PLU: Upland and wetland cleared for bison pasture, MA: Upland excavated to create FP
ABC	Non-riverine mineral soil flat	39	2000–2001	Leaf	PLU: Wetland cleared, ditched, converted to AG MA: Ditches plugged and filled, MT¶ reestablished
Balance Farm	Non-riverine mineral soil flat	163	1998	Roanoke	PLU: Wetland cleared, ditched, converted to AG MA: Ditches plugged and filled, risers added to perimeter ditches
Mildred Woods	Non-riverine mineral soil flat	240	1995	Altavista, Roanoke	PLU: Wetland cleared, ditched, converted to AG MA: Ditches plugged and filled
Barra Farm	Non-riverine organic soil	240	1997–1998	Croatian	PLU: Wetland cleared, ditched, converted to AG MA: Ditches filled
Parker Farm	Non-riverine organic soil	160	1995	Ponzer	PLU: Wetland cleared, ditched, converted to AG MA: Ditches filled
Dismal Swamp	Non-riverine organic soil	250	1996–1997	Scupper-nong	PLU: Wetland, cleared, ditched, converted to AG and SILV MA: Ditches filled, soil contoured to reestablish MT

The PLU and MA are described briefly in the final column.

†FP = floodplain; ‡AG = agriculture; § SILV = silviculture; and ¶MT = microtopography.

Figure 2. Comparison of SOM in CW/RWs and NWs (a), of SOM in CW/RWs and NWs across the four hydrogeomorphic subclasses (HW RIV, MS RIV, NR MSF, NR OSF) (b), and of SOM across the 11 CW/RWs and paired NWs (Willi, RoBr, BaCr, GrLa, HaRu, BIFm, ABC, MiWo, Bann, PaFm, DiSw) (c). Bars represent mean values ($\pm$ standard error). Bars with different letters are significantly different ($p < 0.05$).

the data from both CW/RW and NW plots were pooled together by HGM subclass, the headwater riverine ($8.1 \pm 2.5\%$), mainstem riverine ($11.1 \pm 8.2\%$), and non-riverine mineral soil ($10.3 \pm 4.2\%$) subclasses had significantly lower ($p < 0.001$) mean SOM content than the non-riverine organic soil subclass ($31.0 \pm 14.2\%$) (Table 2). After stratifying the data from the HGM subclasses by wetland status, the SOM content was significantly lower in the CW/RWs across all four HGM subclasses (Figure 2b, Table 2). Also, in the non-riverine organic subclass, mean SOM at Dismal Swamp was significantly higher than that of Parker Farm and

Table 2. Results of the analysis of variance testing for significance of wetland status (created/restored vs. natural), hydrogeomorphic subclass (headwater riverine, mainstem riverine, non-riverine mineral soil flat, non-riverine organic soil flat), the status by HGM subclass interaction, the site within HGM subclass effect, and the status by site within subclass effect for soil organic matter (SOM).

Soil property	Source of variation	df†	F	p-value
SOM† (%)	Status	1	218.5	<0.001
	HGM subclass	3	152.4	<0.001
	Status * HGM	3	12.1	<0.001
	Site (HGM)	7	4.7	<0.001
	Status*Site (HGM)	7	7.3	<0.001

Significant effects ($p < 0.05$) are indicated by bold type.

† SOM was log-transformed to better meet the GLM assumptions.

‡ Error degrees of freedom = 66.

Barra Farm, and mean SOM at Parker Farm was significantly higher than that of Barra Farm. Finally, the effects of the status by site within subclass interaction were also significant (Table 2), indicating that differences in SOM of CW/RWs and NWs were not always consistent across the subclasses (Figure 2c). For instance, at 10 sites, mean SOM content was significantly lower in CW/RWs than in their paired NWs, but at Bailey Creek, there was no significant difference between SOM content of the CW and NW.

The created wetland with the lowest mean SOM content was Haws Run at 1.5%. The restored wetlands at Mildred Woods (2.2%), Rowel Branch (2.9%), and the created wetland at Grimesland (3.7%) also had quite low SOM contents. In contrast, the restored wetlands with the highest SOM content were Parker Farm (24.2%), Barra Farm (25.7%), and Dismal Swamp (44.9%). The NWs with the lowest mean SOM content were Wiggins Mill (8.6%), Bailey Creek (8.9%), and Balance Farm (10.1%). In contrast, the NWs with the highest mean SOM content were Barra Farm (43.7%), Parker Farm (77.4%), and Dismal Swamp (80.2%). When compared to their paired CW/RWs, mean SOM values in the NWs were three times higher at Grimesland, ABC, and Parker Farm, five times higher for Rowel Branch, six times higher for Mildred Woods, and 18 times higher for Haws Run. Finally, after excluding the non-riverine organic soil flats, the SOM content of the CWs and the RWs was not significantly different ($p = 0.39$).

Discussion

Despite some problems (Axt and Walbridge 1999), the loss of ignition technique for measuring SOM content has been widely used in studies of wetland soils. The technique is simple, quick, and does not require the use of expensive laboratory equipment. Our results of lower SOM in CW/RWs than in NWs are similar to previous studies (Bishel-Machung et al. 1996; Gallatowitsch and Van der Valk 1996; Shaffer and Ernst 1999; Campbell et al. 2002). The lower SOM in the CW/RWs may be due to increased decomposition during prior land-uses, or to the removal of organic-rich topsoil during excavation into subsurface horizons that often occurs in wetland creation. Low levels of SOM in CW/RWs may hinder the development of microbial communities that are critical to wetland function (Bruland 2004). Interestingly, the mean SOM content of 11.8% reported for the 11 CW/RWs sampled in this study was nearly double the mean SOM contents of 6.2% reported for a study of 44 created wetlands in Pennsylvania (Bishel-Machung et al. 1996) and 6.7% reported in a study of nine created wetlands in Oregon (Shaffer and Ernst 1999). The main reason for the higher mean SOM content in the North Carolina CW/RWs was that, unlike the studies from Pennsylvania and Oregon, this study sampled restored wetland sites located on former agricultural land with histosol substrates. These types of sites have very high SOM content, and more recalcitrant fractions persist even after decades of agricultural activity (Bridgman et al. 1991).

When CW/RW and NWs were pooled by HGM subclass, the non-riverine organic soil subclass had significantly higher mean SOM content than the other three subclasses. Studies of forested wetlands in the southeastern Coastal Plain should differentiate between sites with organic and mineral soils. While this study did not show significant differences in SOM content between headwater and mainstem riverine sites, this was most likely due to the limited sample size and high within subclass variability of the sites that were sampled.

When mean SOM content of CW/RWs and NWs were compared within subclasses, SOM was lower in the CW/RWs than the NWs across every subclass, suggesting that the effects of prior-land use and mitigation were at least consistent across these Coastal Plain forested wetland types. The

study of SOM content of wetlands in Oregon also reported significant differences in SOM content of CW/RWs and NW across HGM subclasses, with CW/RWs having lower within subclass mean SOM content in four out of five subclasses for which this comparison could be made (Shaffer and Ernst 1999). Despite the efforts to group the sites into similar HGM subclasses, SOM contents within subclasses varied considerably. In the riverine subclasses, certain CW/RWs had significantly lower mean SOM content than their NW pairs, while other CW/RWs had SOM content closer to, if not comparable, to the SOM content of their NW pairs. For the non-riverine sites, the NW areas in the organic soil subclass showed significantly higher mean SOM content at Parker Farm and Dismal Swamp than at Barra Farm. After excluding the non-riverine organic flats, SOM content of the CWs and RWs were similar.

Given a choice in mitigation options between restoration sites of different substrates, sites with histosol soils in the non-riverine organic soil flat subclass should be targeted for restoration for a number of reasons. First, these sites contain pools of SOM which, despite their recalcitrance, provide an energy source to microbes that may serve to “jump-start” wetland function once hydrology is restored. Second, under agriculture these pools of SOM would be subject to oxidation and loss to the atmosphere as CO₂, contributing to the greenhouse effect and global climate change (Richardson 1983; Richardson and Gibbons 1993). Third, by taking these areas out of agricultural production, they appear to provide not only flood control but also water quality benefits (Bruland et al. 2003). Without ditches, these sites can retain enormous quantities of water in microtopographic depressions and surface soil horizons.

Conclusions

Created/restored wetlands had significantly lower SOM than their paired NWs on average, across all HGM subclasses, and at 10 out of the 11 individual sites sampled. Restored wetlands and NWs in the non-riverine organic soil flat subclass also had significantly higher SOM content than the other three subclasses. For restoration guidelines, the organic soil flat subclass should not be included with either the riverine or non-riverine

mineral soil flat subclasses. As the levels of SOM found in NWs can take decades to develop in CWs, wetland restoration is the preferred mitigation technique for forested wetlands in the southeastern coastal plain. A better understanding of the differences in SOM content among CWs, RWs, and NWs will help us to target optimal sites for restoration and to improve our ability to create better mitigation wetlands.

Acknowledgements

Funding for this project was provided by the Duke Wetland Center Case Studies Program and by a Graduate Research Fellowship to the first author from the Center for Transportation and the Environment (Raleigh, NC). We thank L. Paugh, D. Schiller, G. Lewis, and M. West of NCDOT and Jeff Furness of PCS Phosphate for helping us with site identification. Dr. D. Burdick of the Institute of Statistics and Decision Sciences at Duke University provided guidance in developing the model for the statistical analysis. H. Bruland provided useful comments on earlier versions of the manuscript.

References

- Axt J.R. and Walbridge M.R. 1999. Phosphate removal capacity of palustrine forested wetlands and adjacent uplands in Virginia. *Soil Sci. Soc. Am. J.* 63: 1019–1031.
- Bishel-Machung L., Brooks R.P., Yates S.S. and Hoover K.L. 1996. Soil properties of reference wetlands and wetland creation projects in Pennsylvania. *Wetlands* 16: 532–541.
- Brettar I. and Höfle M.G. 2002. Close correlation between the nitrate elimination rate by denitrification and the organic matter content in hardwood forest soils of the upper Rhine floodplain (France). *Wetlands* 22: 214–224.
- Bridgman S.D., Richardson C.J., Maltby E. and Faulkner S.P. 1991. Cellulose decay in natural and disturbed peatlands in North Carolina. *J. Environ. Qual.* 20: 695–701.
- Brinson M.M. 1993. A Hydrogeomorphic Classification for Wetlands. U.S. Army Corps of Engineers, Vicksburg, MS, USA. Wetlands Research Technical Report WRP-DE-4.
- Brinson M.M. and Rheinhardt R. 1996. The role of reference wetlands in functional assessment and mitigation. *Ecol. Appl.* 6: 69–76.
- Bruland G.L., Hanchey M.F. and Richardson C.J. 2003. The effects of agriculture and wetland restoration on hydrology, soils, and water quality of a Carolina bay complex. *Wetlands Ecol. Manage.* 11: 141–156.
- Bruland G.L. 2004. An Observational, Geostatistical, and Experimental Assessment of Edaphic Properties and Processes in Created, Restored, and Natural Wetlands of the Southeastern Coastal Plain. Ph.D. dissertation. Duke University, Durham, North Carolina, USA.
- Campbell D.A., Cole C.A. and Brooks R.P. 2002. A comparison of created and natural wetlands in Pennsylvania, USA. *Wetlands Ecol. Manage.* 10: 41–49.
- Cole C.A., Brooks R.P. and Wardrop D.H. 1997. Wetland hydrology as a function of hydrogeomorphic subclass. *Wetlands* 17: 456–467.
- Confer S.R. and Niering W.A. 1992. Comparison of created and natural freshwater emergent wetland in Connecticut. *Wetlands Ecol. Manage.* 2: 143–156.
- Darke A.K. and Walbridge M.R. 2000. Al and Fe biogeochemistry in a floodplain forest: implications for P retention. *Biogeochemistry* 51: 1–32.
- Duncan C.P. and Groffman P.M. 1994. Comparing microbial parameters in natural and constructed wetlands. *J. Environ. Qual.* 23: 298–305.
- Galatowitsch S.M. and Vander Valk A.G. 1996. Vegetation and environmental conditions in recently restored wetlands in the prairie pothole region of the USA. *Vegetatio* 126: 89–99.
- Howard D.M. and Howard P.J.A. 1993. Relationships between CO₂ evolution, moisture-content and temperature for a range of soil types. *Soil Biol. Biochem.* 25: 1537–1546.
- Hunter R.G. and Faulkner S.P. 2001. Denitrification potential in restored and natural bottomland hardwood wetlands. *Soil Sci. Soc. Am. J.* 65: 1865–1872.
- Lowrance R. 1992. Groundwater nitrate and denitrification in a Coastal Plain riparian forest. *J. Environ. Qual.* 41: 401–405.
- Rheinhardt R.D., Rheinhardt M.C., Brinson M.M. and Faser K. 1998. Forested wetlands of low order streams in the inner coastal plain of North Carolina, USA. *Wetlands* 18: 365–378.
- Richardson C.J. 1983. Pocosins: vanishing wastelands or valuable wetlands? *Bioscience* 33: 626–633.
- Richardson C.J. and Gibbons J.W. 1993. Pocosins, Carolina bays, and mountain bogs. In: Martin W.H., Boyce S.G. and Echternacht A.C. (eds), *Biodiversity of the Southeastern United States*. John Wiley & Sons, Inc, New York NY, USA, pp. 257–309.
- Shaffer P.W. and Ernst T.L. 1999. Distribution of SOM in freshwater wetlands in the Portland, Oregon area. *Wetlands* 19: 505–516.
- Spencer D.R., Perry J.E. and Silberhorn G.M. 2001. Early secondary succession in bottomland hardwood forests of southeastern Virginia. *Environ. Manage.* 27: 559–570.
- Stolt M.H., Genthner M.H., Daniels W.L., Groover V.A., Nagle S. and Haering K.C. 2000. Comparison of soil and other environmental conditions in constructed and adjacent palustrine reference wetlands. *Wetlands* 20: 671–683.
- Verhoeven J.T.A., Whigham D.F., van Logtestijn R. and O'Neill J. 2001. A comparative study of nitrogen and phosphorus cycling in tidal and non-tidal riverine wetlands. *Wetlands* 21: 210–222.
- Walbridge M.R. and Struthers J.P. 1993. Phosphorus retention in non-tidal palustrine forested wetlands of the Mid-Atlantic region. *Wetlands* 13: 84–94.
- Whitcar G.R. and Daniels W.L. 1999. Use of hydrogeomorphic concepts to design created wetlands in southeastern Virginia. *Geomorphology* 31: 355–371.
- Zedler J.B. and Callaway J.C. 1999. Tracking wetland restoration: do mitigation sites follow desired trajectories. *Restoration Ecol.* 7: 69–73.