RESEARCH ARTICLE

Using a Floristic Quality Assessment Technique to Evaluate Plant Community Integrity of Forested Wetlands in Southeastern Virginia

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ABSTRACT: Given the continuing degradation of freshwater wetland ecosystems throughout the Southeast, there has been significant interest in developing methods and indices to evaluate and monitor wetland biological integrity. The purpose of this study was to adapt and test the ability of a vegetation-based assessment technique known as Floristic Quality Assessment to detect the level of human impact in hardwood flat wetlands of Southeastern Virginia. We measured plant species diversity and composition within each vertical stratum [herbaceous, woody understory (shrub and sapling), and canopy] of 11 wetlands. We calculated a Floristic Quality Index (FQI) for each layer, and tested for relationship to land use disturbance patterns within defined site buffer and watershed areas. We found floristic quality of the herbaceous layer and the sapling portion of the woody understory layer to be negatively correlated with level of land use disturbance at both buffer and watershed scales, suggesting that FQI scores within these strata reflect current anthropogenic stress. While FQI of the canopy layer and the shrub portion of the woody understory layer were not reliable indicators of current land use disturbance, we found that a comparison of sapling and canopy layer FQIs gave insights into historic vs. recent floristic integrity of sites. Overall, our findings support the use of floristic quality assessments in evaluating wetland biological integrity when sampling and index calculation methodology are carefully adapted to local flora and community types.

Index terms: evaluation, forested wetlands, FQI, natural habitat

INTRODUCTION

Within the United States, total present-day wetland acreage is less than one-half of its former amount (Dahl 1990). Remaining wetlands are often faced with a number of anthropogenic stresses and disturbances, such as hydrologic modifications, non-point source pollution, and introduction of alien species (Barbour et al. 2000). There is a growing awareness of these trends by the general public and an increased recognition of economically valuable ecosystem functions (*sensu* Brinson 1993) that wetlands can provide. However, protective policies vary widely by state and wetland type. In Virginia, isolated wetlands (i.e., wetlands with no surface hydrologic connection to other waters of the United States) no longer receive federal protection from development as a result of recent case law decisions, and new state policies have been developed to fill the gap. In order to provide effective future conservation efforts, scientists must supply wetland managers with techniques for accurately assessing the overall health of wetland sites over time.

One such technique is Floristic Quality Assessment (FQA). Originally devised by Swink and Wilhelm (1979, 1994), this assessment technique uses information from a site floristic survey to generate a Floristic Quality Index (FQI). The index incorporates two measures of a site's integrity: (1) the biodiversity of a site and (2) its "species conservatism." To derive the latter, a species is assigned a "coefficient of conservatism" that reflects its tolerance to disturbance and fidelity to specific habitat integrity. With this approach, every species thus becomes an "indicator" of some degree of biological integrity. Implicit in FQA application is the assumption that areas with a high degree of biological integrity are those that have species assemblages truly reflective of native, non-disturbed habitat (Swink and Wilhelm 1994). This assumption is therefore predicated on the concept that anthropogenic disturbance represents a mode of introduction for "non-conservative" (e.g., invasive or ruderal) species (Hobbs and Huenneke 1992).

Studies in midwestern states that have tested this method have shown promising results. Mushet et al. (2002) found that the assessment consistently reflected the level of plant community quality in natural and restored wetlands, while Lopez and Fennessey (2002) showed that the index was significantly correlated with the "disturbance rank" of a wetland (calculated from adjacent land use, type of vegetated buffer, and degree of hydrologic modification). However, the reliability of FQA has yet to be tested for wetlands in other areas of the country. A possible factor behind its lack of use in most regions is that many wetlands include significant woody shrub, sapling, and canopy layers – whereas these layers are almost completely absent in the herbaceous wetlands of the midwestern states for which the index was originally designed (Mitsch and Gosselink 1993). With respect to FQA in forested wetlands, the presence of woody plants represents a potential problem related to differential growth rates and longevity. In stratified communities dominated by trees, the response time following disturbance will be different for different strata. This is attributed to the ecological inertia (*sensu* Lopez et al. 2002) exhibited by woody plants, which are slow-growing relative to herbaceous plants and, therefore, exhibit longer response times. Thus, application of FQA in forested wetlands should consider the differential growth and response of the various strata present.

In the southeastern region of Virginia, the predominant palustrine wetland community (*sensu* Cowardin et al. 1979) is the hardwood flat (also known as hardwood mineral flat), which is a wetland type comprised of a stratified community with generally well-developed layers (canopy, sapling, shrub, herbaceous). These areas occupy expansive, nearly level regions formed on low marine terraces in the Coastal Plain physiographic province (Rheinhardt and Rheinhardt 2000), and are subject to a range of land use practices including preservation, silviculture, agriculture, and development.

Given that palustrine forested wetlands account for 64% of the remaining wetland area within Virginia (Tiner et al. 1994) and that hardwood flats represent a considerable portion of that area, we developed a modified version of the index to reflect floristic quality of each layer (herbaceous, woody understory, canopy) in these wetlands and to give an overall assessment of their biological integrity, the principal component of ecosystem health (Yoder 1995, Barbour et al. 2000). We then tested the reliability of our modified FQA by examining its relationship with the level of disturbance resulting from nearby land use patterns at 11 sites in the coastal plain of Virginia. We hypothesized that a negative correlation with land use disturbance would provide evidence for the index's effectiveness in assessing the biological integrity of these forested wetlands and, therefore, its possible use in similar systems.

METHODS

Study Site Selection

We used 11 sites located in the Southeastern Coastal Plain physiographic province of Virginia (Figure 1). The sites represent the available pool of reference sites identified by the U. S. Environmental Protection Agency (EPA) to represent the hardwood flats wetland type in EPA's hydrogeomorphic assessment program (Havens et al. 2004). We chose to use only one community type, hardwood flats, since different community types could potentially show differences in FQI based solely on natural properties independent of anthropogenic influence. We selected sites to represent a gradient of surrounding land use patterns from natural forest to agricultural fields and heavily urbanized areas.

Vegetation Sampling

We sampled all sites at randomly established sampling points from 12-22 July 2002. After collecting data from the first sampling point, we moved to a second randomly placed point and recorded data in the same manner. Our intent was to move to new points as long as we continued to find new species or obvious shifts in community structure. We found that using two points was sufficient for each site; whenever we moved to the third point within a site, we observed the same set of species already seen and with approximately the same relative abundances at the earlier sampling points.

For the herbaceous layer (defined as all vascular plants less than 1 m tall), we recorded the presence of every species observed within sight of the sampling point to compile a list of the species found. Relative abundance data were not deemed necessary for this layer because of the difficulty in interpreting how abundant a particular species "should be" in a natural community of high biological integrity (Swink and Wilhelm 1979, 1994).

However, we did collect abundance data for analyzing the floristic quality of the woody understory (i.e., shrubs and saplings combined) and canopy layers, since dominant species in these layers are likely to exert a stronger influence on ecosystem dynamics than others (see "Calculation of the FQI" below). Therefore, for the understory layer [defined as any woody species, including shrubs and saplings, greater than 1 m tall but with a diameter at breast height (dbh) of less than 10 cm], we conducted stem density counts for each species within a 5 m radius of the sampling point. For the canopy layer (defined as all trees with a dbh greater than or equal to 10 cm), we took a plotless sample from the sampling point using a 10 basal area factor (BAF) angle-gauge prism (following Mitchell et al. 1995) and recorded the dbh of each individual "included" in the sample.

Assignment of Species "Coefficients of Conservatism" (C-values)

At the core of FQA is the coefficient of conservatism (C-value) assigned to each species. These values are based on species tolerance to disturbance and fidelity to specific habitat integrity, with more "weedy," cosmopolitan species receiving lower values, and species restricted to very natural habitats receiving the highest values [see Swink and Wilhelm (1979) for more detailed assignment criteria]. Before sampling, we assigned C-values on a scale of 1-5 to species we expected to find using a number of literature sources that cover the regional flora of Virginia (e.g., Fernald 1950, Radford et al. 1968, Gleason and Cronquist 1991, Harvill et al. 1992) and several local botanists to help with the assignments.

Calculation of the FQI

We calculated the average C-value of each layer within a site from the individual C-values of species present in that layer. For the herbaceous layer, we used a simple, non-weighted average. We weighted the understory layer average by relative abundance of each species and the canopy layer average by the relative basal area of each canopy species as calculated from the dbh of each individual. We aggregated data collected from sampling points for a particular layer in a site when calculating

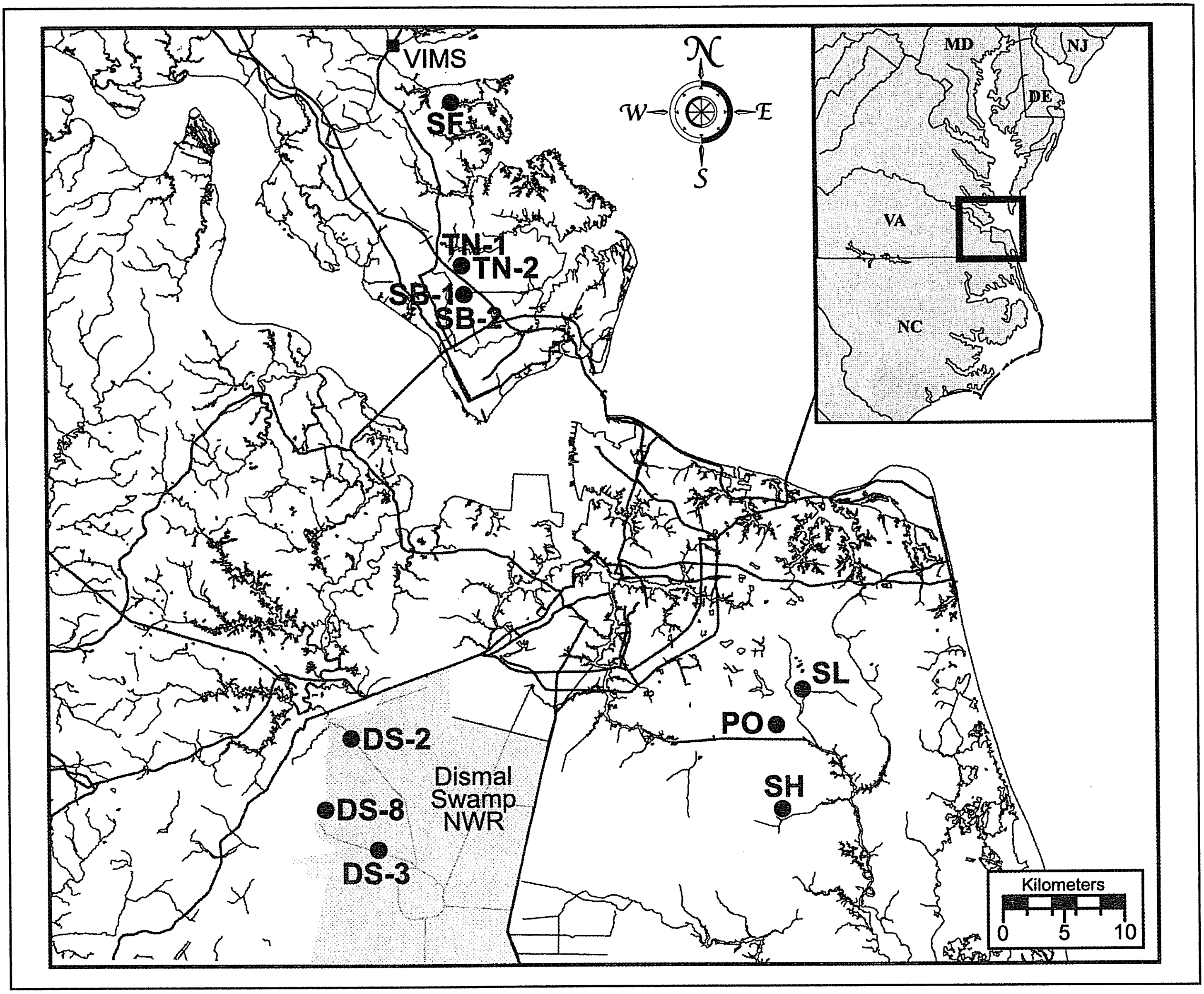

Figure 1. Location of Study Sites. Sites included Dismal Swamp National Wildlife Refuge (DS-2, 3, 8), Pocaty Creek (PO), Sandy Bottom Nature Preserve (SB-1, 2), Seaford Elementary School (SH), Sherman (SF), Stumpy Lake Golf Course (SL), and Thomas Nelson Community College (TN-1, 2).

its average C-value.

We calculated FQI as the product of this average C-value and the square root of the total number of native species present (i.e., native species richness, N) (Swink and Wilhelm 1979): $FQI = \overline{C} * \sqrt{N}$.

Including the square root of native species richness in FQI calculations has been suggested as a way to differentiate between sites with greatly different diversities without "overweighing" sites with a high number of species but low average C-value (Wilhlem and Ladd 1988).

We included only those species native to Virginia in our calculations of $\overline{C}$ and FQI. Because the ecological role of introduced species is very difficult to determine, their presence is not factored into the numerical calculation of the index (Wilhelm and Ladd 1988, Swink and Wilhelm 1994). However, the net effect that introduced species have on the community can be indirectly seen in the index if they reduce the native diversity of a site or entirely replace some of its more conservative species (Mushet et al. 2002).

Land Use/Landcover Assessment

In order to assess the degree of land use disturbance present at each site, we evaluated landcover patterns present in the surroundings of each site at two different scales: (1) watershed and (2) buffer. We defined the "watershed" of each site as the circle inscribed by a 500 m radius emanating from the wetland center. Since all sites were hardwood flats, we felt this distance would adequately represent the area drained by the site without extending into neighboring watersheds. Using a GIS software package (ERDAS 1994),

Table 1. Ranking system for land use categories at watershed-scale and buffer-scale (from Lopez and Fennessey 2002).

Land cover / Land use Category	Disturbance Ranking
Watershed Scale:	
Wetland	1
Forest	2
Artificial Ponds and Clearings	3
Agricultural Fields, Manicured Lawns	4
Developed	5
Buffer Scale:	
Forest	1
Dirt or Gravel Road	2
Old Field	3
Manicured Lawns	4
Developed	5

we determined the relative amount of each landcover category present in this circle based on data from the 2002 National Land Cover Dataset (NLCD) (Vogelmann et al. 2001). We assigned a "disturbance rank" to each landcover category found, using a scale of 1 to 5 (5 being the highest disturbance) to represent the relative degree of ecosystem stress caused by that type of landcover (Table 1; following Lopez and Fennessey 2002).

We followed a similar approach for land use within the buffer surrounding a site (defined here as the non-wetland area immediately surrounding the contiguous forested cover of a site). We determined the landcover present on each side of the wetland during our vegetation sampling. We then used a ranking system to categorize the level of disturbance for each site's buffer with the same approach taken for the watershed-scale data (Table 1). We then calculated overall "land use disturbance scores" for a site at both watershed and buffer levels by taking the average of the results of the disturbance ranking for each category multiplied by the relative amount of land cover for that category .

Statistical Analyses

We used a Spearman's rank correlation test ($\alpha = 0.05$) to evaluate the relationship between FQI value and land use disturbance score. All statistics were performed on StatMost for Windows software (DataMost 1994). We analyzed this relationship both with all layers aggregated and for each layer of vegetation individually (herbaceous, understory, and canopy), checking the relationship at each land use scale separately (watershed and buffer). An understory (i.e., shrubs and saplings combined) FQI: canopy FQI ratio was correlated with land use disturbance. To investigate the chronological relationship of the woody vegetation, we also performed the Spearman's rank correlation test where saplings were considered separately rather than together with shrub species. The decision to evaluate saplings separately was based on the assumption that saplings are more likely to reflect regeneration in the community (Spencer et al. 2001). Finally, we evaluated the usefulness of a sapling-canopy comparison by calculating the sapling:canopy FQI ratio for each site and checking for correlation of these ratios with the land use disturbance scores.

RESULTS

Overview

A total of 108 species representing 55 families was collected in the 11 study sites. Twelve of these species are not native to the state of Virginia. Species having a C-value of 1 were the largest group, whereas only 4 species had a C-value of 5. Landcover disturbance values ranged from 1 to 3.9 at the watershed level and from 1.25 to 5.0 at the buffer level (Table 2, Figure 2).

Herbaceous Layer

We found a significant negative correlation between herbaceous layer FQI and buffer-scale land use disturbance score ($r = -0.67$, $p = 0.03$) (Table 3, Figure 3A). Herbaceous layer FQI scores were also negatively correlated with disturbance scores at the watershed-scale, although this trend was not statistically significant ($r = -0.46$, $p = 0.16$).

Woody Understory Layer

We failed to detect a significant relationship between land use disturbance and woody understory (shrub/sapling) FQI (Table 2) at either scale ($r = -0.10$, $p = 0.78$ for the watershed-scale and $r = -0.25$, $p = 0.43$ for the buffer-scale) (Table 3). Rank order of sites by FQI value for this layer did not appear to reflect any consistent decrease in the land use disturbance score. When saplings were considered separately from the shrub species, however, there was a distinct correlation between sapling FQI and land use disturbance at both scales ($r = -0.72$, $p = 0.01$ for the watershed-scale, and $r = -0.86$, $p < 0.001$ for the buffer-scale) (Table 3, Figure 3B).

Canopy Layer

The correlation between canopy layer FQI and land use disturbance scores was significant at the watershed-scale ($r = 0.77$, $p < 0.01$) but not significant at the buffer-scale ($r = 0.57$, $p = 0.07$) (Table 3, Figure 3C). It should be noted, however, that unlike the negative correlations observed

Table 2. Land use disturbance rankings for watershed and buffer scale disturbance of the 11 hardwood flat sites used in this study. The watershed was defined as a 500 m radius area around each site and the buffer as the non-wetland area immediately surrounding the contiguous forested cover of each site. Approximate size is given in hectares.

Site Designation	Site Name	Size	Longitude / Latitude	Watershed	Buffer
DS-2	Dismal Swamp NWR	3	40° 65' 18" / 36° 31' 72"	1.00	1.25
DS-3	Dismal Swamp NWR	3	40° 56' 83" / 36° 13' 76"	1.14	1.25
DS-8	Dismal Swamp NWR	3	40° 53' 62" / 36° 07' 31"	2.40	1.25
PO	Pocaty Creek	2	40° 59' 01" / 39° 95' 47"	3.24	3.5
SB-1	Sandy Bottom Nature Preserve	3	41° 02' 98" / 37° 30' 32"	3.32	5
SB-2	Sandy Bottom Nature Preserve	1	41° 02' 82" / 37° 29' 47 "	3.16	4
SF	Seaford Elementary School	2	41° 16' 66" / 37° 14' 99"	2.31	3.75
SH	Sherman	3	40° 67' 48" / 39° 71' 96"	2.63	3.33
SL	Stumpy Lake Golf Course	2	40° 69' 96" / 39° 57' 67"	2.67	4
TN-1	Thomas Nelson Community College	1	41° 02' 92" / 37° 34' 52"	3.94	5
TN-2	Thomas Nelson Community College	3	41° 02' 99" / 37° 40' 34"	3.01	3.75

for herbaceous and shrub/sapling layers, canopy correlations were positive.

Aggregate

The Spearman's correlation between an aggregation of FQI for all layers and land use disturbance scores (Table 2) was not significant at the watershed-scale ($r = -0.10$, $p = 0.68$) or buffer-scale ($r = -0.09$, $p = 0.59$) (Table 3, Figure 3D).

Sapling-Canopy Comparison

The sapling-canopy FQI comparison revealed a significant negative correlation between the ratio and land use disturbance at both the watershed- and buffer-scales ($r = -0.82$, $p < 0.01$ for the watershed-scale

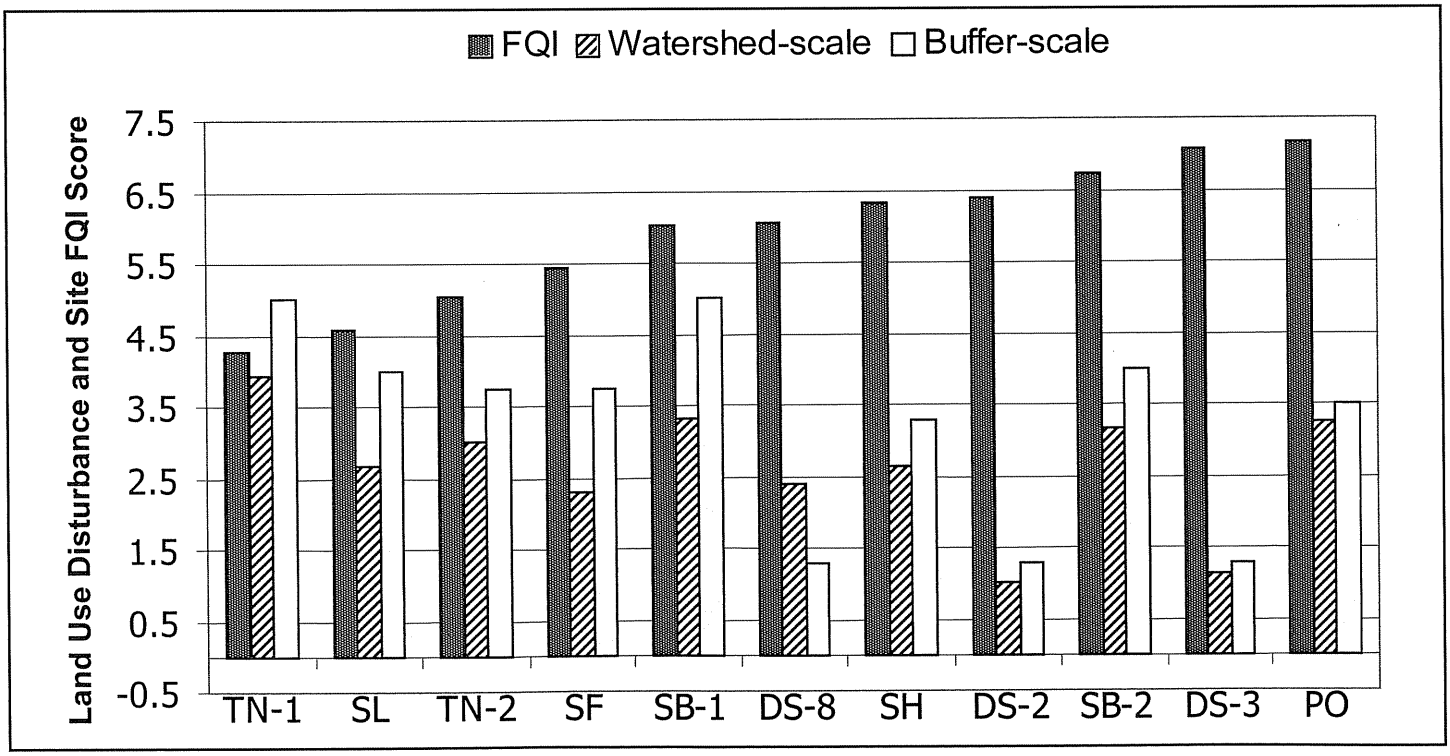

Figure 2. Sites ranked in ascending order by FQI. Watershed- and buffer-scale scores in this figure were calculated with all vegetation layers included. There was no correlation between site FQI and either watershed- and buffer-scale scores (p>0.05).

Table 3. Summary of results for Spearman's Rank Correlation tests between FQI of different vegetation layers and land use disturbance score at both watershed- and buffer-scale. Statistically significant correlations indicated with an *.

Vegetation Layer	Land use Data	Spearman's r	p-value
Herbaceous	Watershed-scale	-0.46	0.16
	Buffer-scale	-0.67	0.03*
Shrub/Sapling	Watershed-scale	-0.10	0.78
	Buffer-scale	-0.25	0.43
Sapling	Watershed-scale	-0.72	0.01*
	Buffer-scale	-0.86	< 0.001*
Canopy	Watershed-scale	+0.77	< 0.01*
	Buffer-scale	+0.57	0.07
Sapling: Canopy	Watershed-scale	-0.82	< 0.01*
	Buffer-scale	-0.83	< 0.01*
Aggregate	Watershed-scale	-0.10	0.68
	Buffer-scale	-0.09	0.49

and $r = -0.83$, $p < 0.01$ for the buffer-scale) (Table 3, Figure 4).

DISCUSSION

We observed differences in FQI values among different vegetative layers at a given site. For example, Dismal Swamp sites had some of the highest herbaceous layer FQI values, yet some of the lowest canopy layer FQI values. Based on these observations, it appears as though correlations between FQI and land use disturbance were dependent on the vegetation layer being analyzed.

Lack of a clear relationship between FQI and land use disturbance scores when layers were aggregated may be due to key differences in life history of plants within each layer. Herbaceous layer species are typically sensitive to current environmental and biological conditions (van der Valk 1981). These traits make the herbaceous layer a good candidate for evaluating the current biological integrity of a site – which seems to be supported by the negative correlation we observed between herbaceous layer FQI and buffer-scale land use disturbance.

The relationship between FQI and land use disturbance was less clear for the understory layer (i.e., shrubs and saplings combined). Woody understory species are likely to be less responsive to actual land use in the surroundings of a wetland and more affected by gap dynamics (King and Allen 1996). Furthermore, understory shrubs and saplings are perhaps too different to be considered in aggregate. Understory shrubs are typically shade-tolerant; and, in the absence of disturbances that would otherwise alter the light environment at a given location, can presumably maintain their populations in situ for a long period of time (Bazzaz 1996). Many understory saplings, by contrast, are highly favored in a regeneration niche such as a canopy gap (Shugart and West 1980).

Analyzing understory saplings separately revealed a negative correlation between FQI and land use disturbance at both the watershed- and buffer-scale, while the FQI of shrub species still showed no overall relationship with land use disturbance. Thus, because saplings represent an understory community guild (*sensu* Barbour et al. 1999) that is more reflective of disturbance related to gap dynamics, sapling FQI values appear more responsive to community disturbance than shrubs and, by analogy, may prove useful indicators of a site's biological integrity following a disturbance regime. Sapling assessment, therefore, could complement assessments of the herbaceous layer, since herbaceous species may be more reflective of "current" disturbance conditions.

Given that a wetland canopy layer typically contains the oldest individuals, canopy adults are more likely to reflect historic disturbance conditions rather than current and recent conditions (Lopez et al. 2002). Further, as Rheinhardt and Rheinhardt (2000) indicate, mineral flat wetlands in Southeastern Virginia are typically much younger than the projected climax state (>300 years) for the region. Thus, canopy composition in mineral flats tends to be dominated by a few mid-successional species such as *Acer rubrum* L., *Liquidambar styraciflua* L., and *Pinus taeda* L., with other non-dominants such as *Fagus grandifolia* Ehrh. and various species of oak (*Quercus* spp.).

The relative dominance of canopy inhabitants depends in part on historic disturbance regime. For example, a highly disturbed condition in the past (e.g., agriculture) would tend to lead to a higher proportion of mid-successional species (e.g., *A. rubrum*, *L. styraciflua*) in the canopy, whereas a less intensive historic disturbance regime (e.g., selective timbering) may contribute to a higher proportion of late successional species in the canopy (e.g., oaks) (Rheinhardt and Rheinhardt 2000). In either case, the difference between sites may be subtle and undetectable when FQI values are calculated. These subtle, compositional distinctions across sites were observed in our study and ostensibly reflected a condition where canopy assemblages approached a uniformly consistent FQI expression beyond a certain age following some disturbance in the past (Hobbs and Huenneke 1992). Based on this, the FQI of canopy species cannot be expected to give a good indication of current disturbance regimes (Lopez et al. 2002). Therefore, it

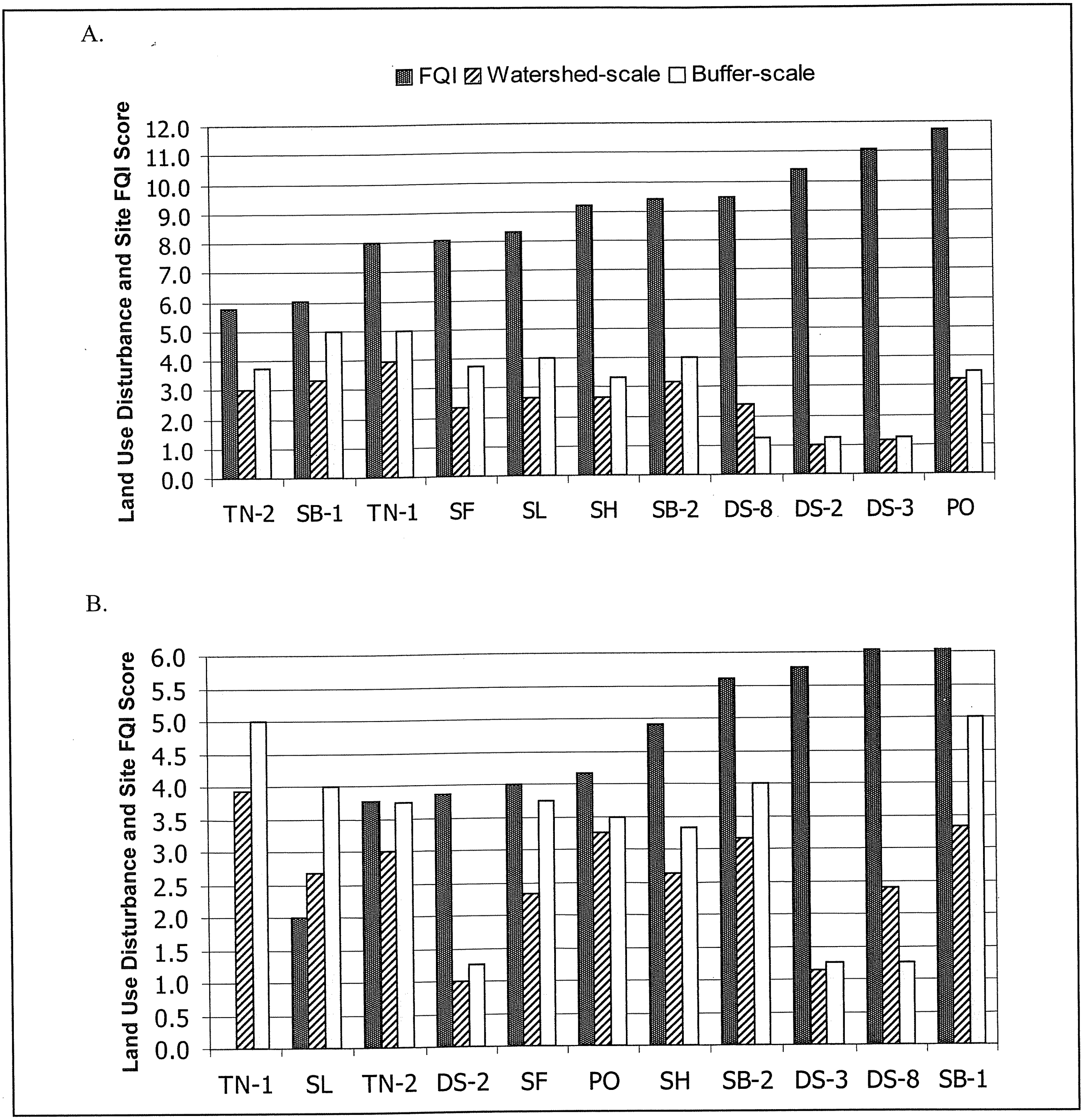

Figure 3. Site watershed- and buffer-scale scores arranged by FQI. A negative correlation ($p<0.05$) was found between the buffer scale of the herbaceous layer and a positive correlation ($p<0.05$) between the FQI and watershed-scale canopy layer. A = herbaceous layer, B = Shrub/understory layer, C = canopy layer, D = aggregate of all layers. (Continued on next page.)

is plausible that the positive relationship we observed between canopy FQI and land use disturbance is due to the inclusion of sites with recent high land use disturbance regimes that did not remove canopy dominants (e.g, Thomas Nelson Community College, Sandy Bottom Nature Park).

However, an assessment of the canopy layer may still have merit. As we have noted, although the distinctions are subtle, canopy FQI can still provide some indication of historical conditions at a site. It may, therefore, be useful in evaluating how the current biological integrity of a site (as

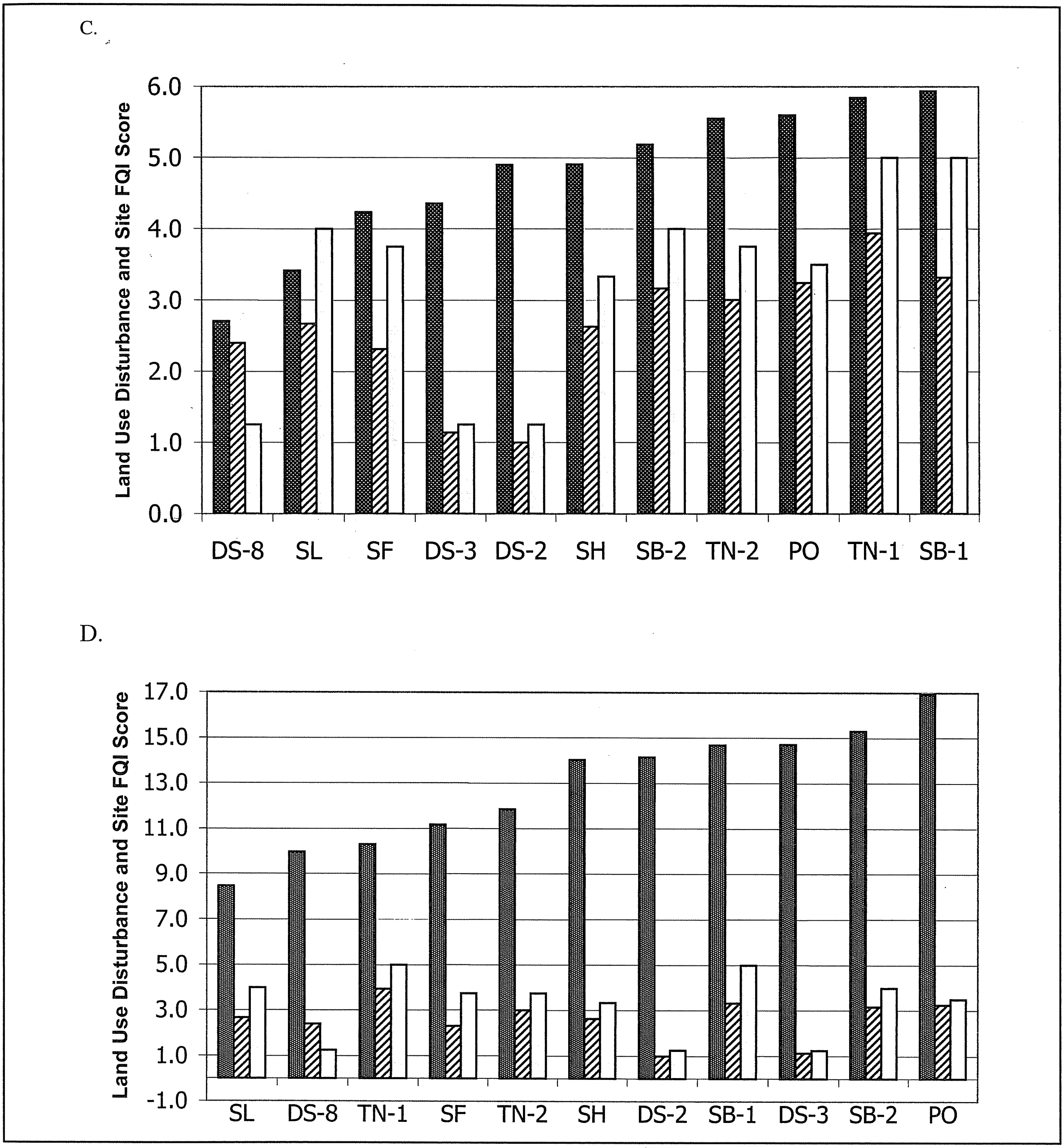

Figure 3. (Continued from preceding page.) Site watershed- and buffer-scale scores arranged by FQI. A negative correlation ($p<0.05$) was found between the buffer scale of the herbaceous layer and a positive correlation ($p<0.05$) between the FQI and watershed-scale canopy layer. A = herbaceous layer, B = Shrub/understory layer, C = canopy layer, D = aggregate of all layers.

measured by the sapling FQI) compares to what it was when the canopy species were established. If a site is being subjected to higher levels of land use disturbance today than in past years, we should expect to see a relatively low sapling FQI compared to a relatively higher canopy FQI. The strongly negative correlation of the sapling: canopy FQI ratio with land use disturbance at

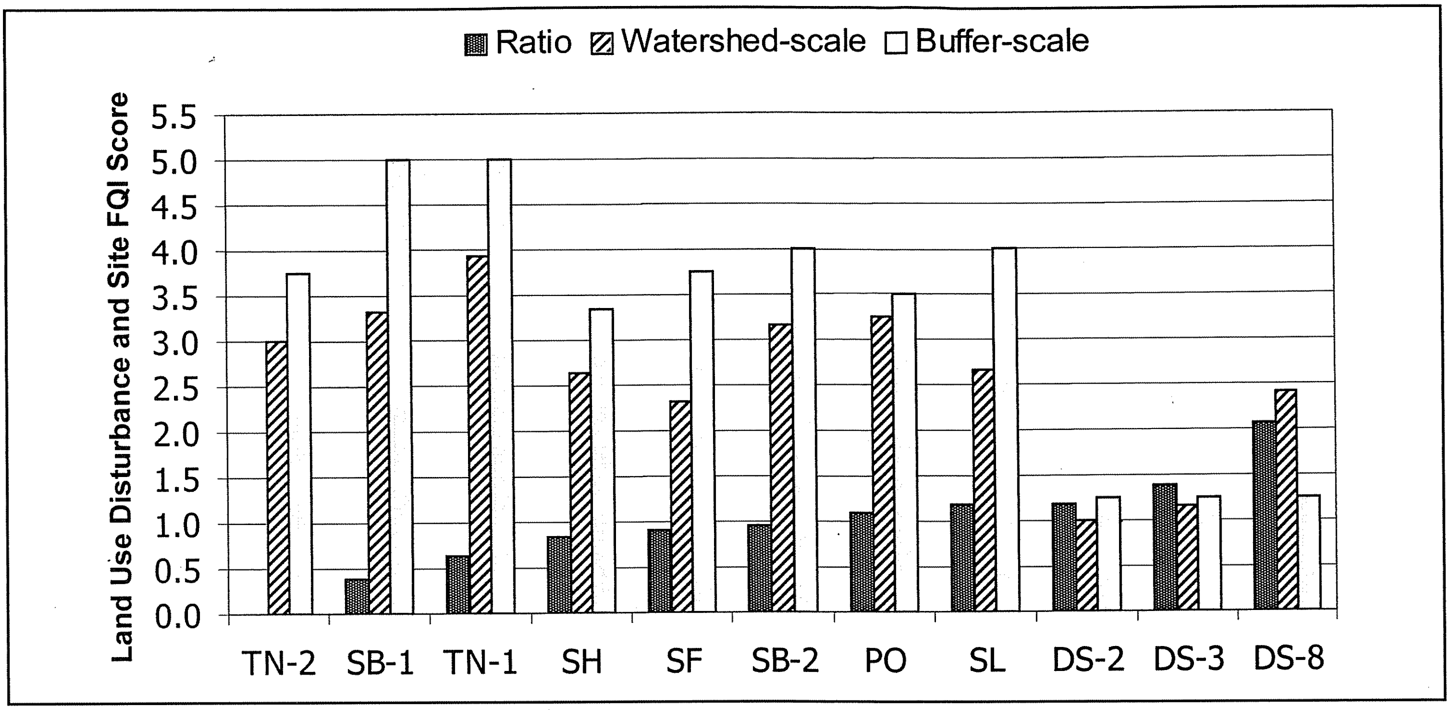

Figure 4. Sites ranked in ascending order by ratio of saplings to canopy. The ratio was negatively correlated ($p<0.05$) with both watershed- and buffer-scale disturbance scores.

both scales suggests that this may be a useful comparison in assessing changes to biological integrity over time.

Overall, we have found considerable evidence from previous studies, as well as our own, suggesting that FQA can be a useful tool in assessing the biological integrity of wetland ecosystems. Although the basic design of the assessment remains true to its original form, using FQA for forested wetlands requires careful adaptations to sampling and calculation protocols and an understanding of what relationships should be expected for each layer. We propose further testing of the index, through re-sampling in subsequent years and different seasons, to confirm the resilience of index values to variation from natural processes. Our hope is that this version of floristic quality assessment will be able to find use in this region. If further research is invested in refining the index and giving close attention to its accuracy, there is also potential for using it in management regimes to monitor changes in biological integrity over time, as well as the possibility of use in other wetland community types.

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