FUNCTIONAL ASSESSMENT OF URBAN FORESTED WETLANDS

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Abstract: Wetlands perform various functions of vital socio-ecological significance. To avoid further loss of functions, functional assessment techniques for management purposes are important to develop for different wetland classes. Our aim was to assess the biotic functions of urban-forested wetlands, and to evaluate specific functional assessment models in an urban setting. The models were adopted from the low gradient riverine wetlands hydrogeomorphic (HGM) functional assessment guidebook of Western Kentucky of the US Army Corps of Engineers. Three bottomland hardwood wetlands were chosen for assessment and models evaluation in East Baton Rouge Parish (EBRP), Louisiana. Fourteen out of 17 variables for nutrient cycling, maintenance of native plant community and provision of habitat for wildlife functions were applicable to the selected wetlands. Three surrogate variables were developed to fill identified gaps in the existing models and provide more accurate assessment of urban forested wetlands. Litter layer depth was found to be a more reliable assessment variable for quantifying O-horizon biomass production than the presence/absence of an O-horizon. Dominant wetlands plant species list was adjusted to accurately reflect the flora of the urban forested wetlands of EBRP. An additional variable for characterization of forest strata as a factor of wildlife habitat provision was developed, and added to the model. Overbank flood frequency variable was not applicable to the fragmented urban wetlands and was removed from the models. The amended assessment models accurately captured existing wetland conditions and the effects of site alterations due to urbanization. These alterations caused significant differences ($p < 0.05$) in wildlife habitat provision, maintenance of characteristic plants community and nutrient cycling functions among the three sites. Further work on the application of these models in similar urban forested settings in the southeastern US is recommended.

Keywords: Hydrogeomorphic functional assessment; bottomland hardwood forest; wetland assessment; Urban-forested wetlands; ecological functions

Introduction

Wetlands perform key ecosystem functions that maintain the ecological integrity of the wetland ecosystems [1]. They provide wildlife habitat, recharge ground water and deep aquifers, recycle nutrients, ameliorate downstream flooding and protect water quality and produce biomass [2,3]. The level of a function performed by a wetland is the result of its biotic and abiotic structural characteristics as well as their interactions [4].

Since European colonization, nearly 80% of the forested wetlands in the Lower Mississippi River alluvial valley have been lost due mainly to drainage and conversion to croplands [5,6]. Loss of wetland acreage means loss of specific functions that the wetland could have sustained [7,3].

The ecological importance of wetlands was recognized and protected through Section 404 of the Clean Water Act (CWA) and subsequent judicial decisions [8]. In 1980, the Federal

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Government adopted a ‘no net wetlands loss’ policy to ensure mitigation of wetland functions loss due to development. Under Section 404 of CWA, the US army corps of engineers (COE) is responsible for assessing impacts to wetland functions from development activities and requiring mitigation of unavoidable impacts. However, assessing wetland functions is more complicated than identifying wetland boundaries and lost acreage. Various assessment techniques have been developed since 1970’s for the determination of wetland functions such as the habitat evaluation procedure (HEP), wetland evaluation techniques (WET), and hydro-geomorphic functional assessment (HGM) [9].

The COE has adopted the HGM approach for wetlands regulatory purposes [10]. The HGM approach classifies and then functionally characterizes wetlands on the basis of the hydrogeomorphic setting, water source and hydrodynamics [10]. Wetlands are classified based on these attributes and the functions supported by different wetland classes are quantified. Local and regional guidebooks have been developed that provide protocols and models for quantifying specific functions for the various wetland classes and subclasses [11]. The ability of a specific wetland to perform functions naturally varies within a wetland class. Data are collected on specific functions from a number of sites (pristine to disturbed) encompassing the natural variability within the HGM class and are called the reference wetlands. Sites among the reference wetlands which sustain a function optimally are called the reference standard wetlands [12]. The models developed for specific functions designed to result in a functional capacity index (FCI), which rank the functions on a scale from 0.00 to 1.0. FCIs indicate the capacity of wetlands to perform a function.

Each model is comprised of variables, which are analytic structural criteria of the wetlands and surrounding landscapes that influence a function. Variables are measured and ranked on a sub-index scale of 0.0 to 1.0 relative to reference standard wetlands. The sub-index values of the variables are put into the assessment model and FCIs for different functions are determined [1]. An FCI of 1.0 indicates optimal performance of a specific function comparable to that of the reference standard wetland [10].

Urbanization activities in East Baton Rouge Parish (EBRP) of Louisiana has led to the loss, fragmentation, and hydrologic isolation of forested wetlands, besides forest conversion to croplands as elsewhere in the southeast [13]. Although forested wetlands represent 33% of the total land area of the EBRP [14], they are important for flood control, nutrient cycling, sediment and metal retention [15], habitat for native and migratory wildlife species and pollution control. Applicable and feasible ecological functional assessment models are needed for these urban wetlands for making informed management decisions. Our objective was to evaluate the applicability of three biotic functional assessment models for assessing the biotic functions of urban forested wetlands of EBRP.

Materials and Methods

Research Sites

Three forested wetland sites were selected for functional assessment in EBRP. The parish is part of the Lower Mississippi River alluvial valley, which was formed by fluvial sediments deposited by Mississippi River, about 6000 years ago [16]. The alluvium is mainly comprised of montmorillonite, mica, illite, vermiculite, feldspar, quarts and iron oxides [17]. The hardwood forests are dominated by typical bottomland hardwood tree species (*Ulmus americana*, *Quercus* species, *Acer* species, *Celtis* species). These poorly drained soils have permeability less than 5.08 cm/hour [17,18]. Two of the three sites selected for this study were located in the southern more developed section of EBRP (Burbank and Siegen sites), while the third site was located in the northern more rural part of EBRP (Flonacher site).

The Burbank site was located on 528 ha tract surrounded by roads on three sides. The major source of water at this site is precipitation and run-off from adjacent croplands. The soil of the site belongs to the Sharky series (very fine, montmorillonitic, nonacid, thermic, Vertic Haplaquepts).

The Siegen site (70 ha) is in an early secondary successional stage, after being abandoned as agricultural field. Shallow surface ditches that were dug in the past for agricultural purposes cover the site and drain into a main channel. Precipitation and run-off from adjacent uplands are the major sources of water. The soil type is Sharky series and is poorly drained [17].

The Flonacher site (24 ha) is located in the more rural northwest section of the parish. The forested wetlands of the site are flat and depressional regions surrounded by upland forests and rangelands. Precipitation, run-off from forest and rangelands and overbank flooding from a small seasonal creek are the major sources of water of the site. The soil type of the site is Sharky and Zachary series (Typic Albaqualfs). The soils are formed in silty alluvium deposited by the Mississippi River.

Assessment Models

We used the Western Kentucky functional assessment guidebook [19] to assess wetland functions as it was the only regional guidebook available at the time of our study. The forested wetlands of EBRP originally developed under conditions similar to the low-gradient riverine forested wetlands of Western Kentucky, therefore, the models should be applicable.

Three biotic functions were selected for assessment in the three selected sites. The functions were 1) cycling of nutrients, 2) maintenance of characteristic plant community, and 3) provision of habitat to wildlife. Assessment models for each function and their underlying variables were adopted from Ainslie *et al.* [19]. Where necessary, surrogate variables were developed to fill any gaps in the existing models.

Variable values were measured onsite and by referring to published literature. After we measured the variables, their values were converted to sub-index values on a scale of 0.0 to 1.0 based on the reference data in Ainslie *et al.* [19]. The range of sub-index values represents the contribution of the value of a variable to the functional capacity for a function by a wetland [19]. The FCIs for the three functions were determined by using the following models.

a) *Cycling of nutrients*

$$FCI = 1/2\left[\left(V_{TBA} + V_{SSD} + V_{GVC}\right)/3 + \left(V_{OHOR} + V_{AHOR} + V_{WD}\right)/3\right] \quad (1)$$

where
V_{TBA} = Tree basal area
V_{SSD} = Understory vegetation density
V_{GVC} = Ground vegetation cover
V_{OHR} = O-horizon biomass
V_{AHOR} = A-horizon biomass
V_{WD} = Woody debris biomass

b) *Maintenance of characteristic plant community*

$$FCI = \left[\frac{(V_{TBA} + V_{DEN})/2) + V_{COMP}}{2} \times \frac{(F_{SOILINT} + V_{WTD})}{2} \right]^{1/2} \qquad (2)$$

where

V_{TBA} = Variable tree basal area
V_{DEN} = Trees density
Vcomp = Plant species composition
Vsoilint = Soil integrity of the assessment area
V_{WTD} = Depth to water table from soil surface

c) *Provision of habitat for wildlife*

$$FCI = \left[\frac{V_{MACRO} + ((V_{TRACT} + V_{CONNECT} + V_{CORE})/3)}{2} \times \frac{(V_{COMP} + V_{TBA} + V_{TDEN} + V_{SNAG} + V_{STRATA}) + (V_{LOG} + V_{OHOR}/2)}{6} \right]^{1/2} \qquad (3)$$

where

V_{MACRO} = Macro-topographic features
V_{TRACT} = Wetland tract area
$V_{CONNECT}$ = Percent of wetland tract connected to other suitable wildlife habitats
V_{CORE} = Interior core area
V_{COMP} = Vegetation composition
V_{TBA} = Tree basal area
V_{TDEN} = Tree density
V_{SNAG} = Snag density
V_{STRATA} = Vegetation strata
V_{LOG} = Log biomass
V_{OHOR} = O-horizon biomass

Twelve circular plots of 11.3 meters radius were randomly established in the three sites for measuring the values of the variables. Variable tree basal area ($V_{TBA,}$ $m^2 ha^{-1}$) was determined by measuring the diameter at breast height (DBH) of trees greater than 10 cm. The diameter values were converted to area by using the following formula.

Area = 5/6 Diameter^2

Tree density ($V_{DEN,}$ stems/ha) was determined by counting the number of trees having DBH greater than 10 cm and multiplied by 25 to convert it to stems/ha. Similarly snag density (V_{SNAG}, snags/ha) with DBH greater than 10 cm were counted in the plots. Plant species composition (Vcomp) for the three strata was determined for each site and ranked in descending order of dominance. The relative plant species dominance was calculated by using the 50/20 rule of the Federal Wetland Delineation Manual [20]. However, the updated dominant plant species list for EBRP was used for the purpose (Table 2). Shrub density (V_{SSD}, shrubs/ha) was determined in 3.6 meter radius subplots established in the main plots. Shrubs having a minimum height of 1 meter with DBH less than 10 cm were counted in the subplots. Percent ground vegetation cover was determined visually in four subplots (1 m^2) in the main plot. O-horizon biomass (V_{OHR}) was determined by taking leaf litter depth in 4 1-m^2 subplots in the main plot. A sub-index scale developed for EBRP was used to assign sub-index values to the variable (Fig. 3). The presence, absence and thickness of soil A-horizon (V_{AOHR}) was determined by taking 15 cm deep soil cores through a hand auger in the 1-m^2 sub-plots. Woody debris biomass (V_{WD}, tons/ha) in each plot was determined by counting the number of stems intersecting 2 transects of 125 cm.

The depth to seasonal high water table (V_{WTD}) was determined by digging ground water table monitoring wells in spring 2001. The depth to the ground water table was determined and averaged for the spring season in the main plots. Soil integrity ($V_{SOILINT}$) variable was determined visually on-site by looking for any soil disturbance indicators (excavation material, fill, plowing, and compaction). Absence of these indicators in the plots indicated that the soils were intact. Forest strata variable (V_{STRATA}) was determined based on the absence/presence of the forest stand stratification in the assessment plots (Table 2).

The landscape scale variables (wetland tract area (V_{TRACT}), habitat connections (V_{CON}), interior core area (V_{CORE}), and macrotopographic features (V_{MACRO})) for the three assessment sites were determined by using Louisiana GIS database maps and USGS quad maps [21]. Sub-index values determined for the 17 variables of the 12 plots for the three sites were averaged and FCIs were calculated with the assessment models (Eqns. 1, 2, and 3).

Statistical Analysis

Linear regression of litter layer depth, leaf litter weight and tree basal area was analyzed with the SAS REG procedure. One-way ANOVA was used to test for significant differences among the three sites for the three biotic functions [22]. Post-hot Fisher's protected pairewise comparisons were done for finding differences among the sites for the selected functions. The data was checked for normality of the residual assumptions of the general linear model procedures of ANOVA and regression.

Results and Discussions

Cycling of Nutrients

Out of the 6 variables of this model (Eqn.1), only the characterization technique of the O-Horizon biomass variable given in the Western Kentucky guidebook was not applicable to the wetlands of EBRP. The O-horizon is the partially decomposed organic matter and recognizable twigs and leaves overlying the A-horizon [23]. The O-horizon biomass is an active pool of nutrients in the litter layer, which decomposes and releases the bound nutrients and make them bioavailable [24]. The measurement protocol for this variable is to visually determine the percent cover of ground surface by leaf litter [19]. The generally high litterfall and moist to wet soil conditions in bottomland hardwood forests in EBRP yield an O-horizon. Percent cover by an O-horizon was 100% for all the sites.

We developed an alternative approach for the O-horizon variable by quantifying the mass of the litter layer/O-horizon per unit area and determining whether this variable could be accurately represented by depth of the litter layer/O-horizon. The dry weight of the litter layer/O-horizon was determined in four randomly selected 1-m^2 subplots within each main plot. Average depth of the litter layer/O-horizon was recorded for each subplot before collection. Average weight, depth, and tree basal areas for the three research sites are shown in Table 1. Linear regression of the average litter layer/O-horizon depth on litter/O-horizon weight identified a significant linear relationship (Fig. 1) between the litter layer/O-horizon depth and weight ($r^2 = 0.50$, $p < 0.01$). The linear relationship of litter layer/O-horizon mass and depth captured the assessment sites condition. Assessment plots with high mass of litter layer/O-horizon had a high tree basal area (TBA) than plots of low litter layer/O-horizon mass. A linear regression (Fig. 2) identified a significant relationship of TBA and litter layer/O-horizon weight ($p < 0.01$ and r^2 of 0.51). Research sites in EBRP with TBA of 15 m^2/ha or more had a litter layer biomass of more than 500g m^{-2}, which indicated optimally functional sites. This result suggested that litter layer/O-horizon depth accurately represented O-horizon biomass variable.

Table 1.
Average litter layer weight and depth of urban-forested wetlands in EBRP, Louisiana (± values are standard errors of the mean).

Research plots	**Mean litter layer weight (g/m^2)**	**Mean litter layer depth (cm)**	**Mean Tree Basal Area (m^2/ha)**
Burbank Site	513 ± 44.2	2.20 ± 0.13	17.1 ± 0.50
Flonacher site	626 ± 110	2.51 ± 0.33	24.1 ± 10.1
Siegen Site	422 ± 30.1	2.11 ± 0.42	7.0 ± 1.45

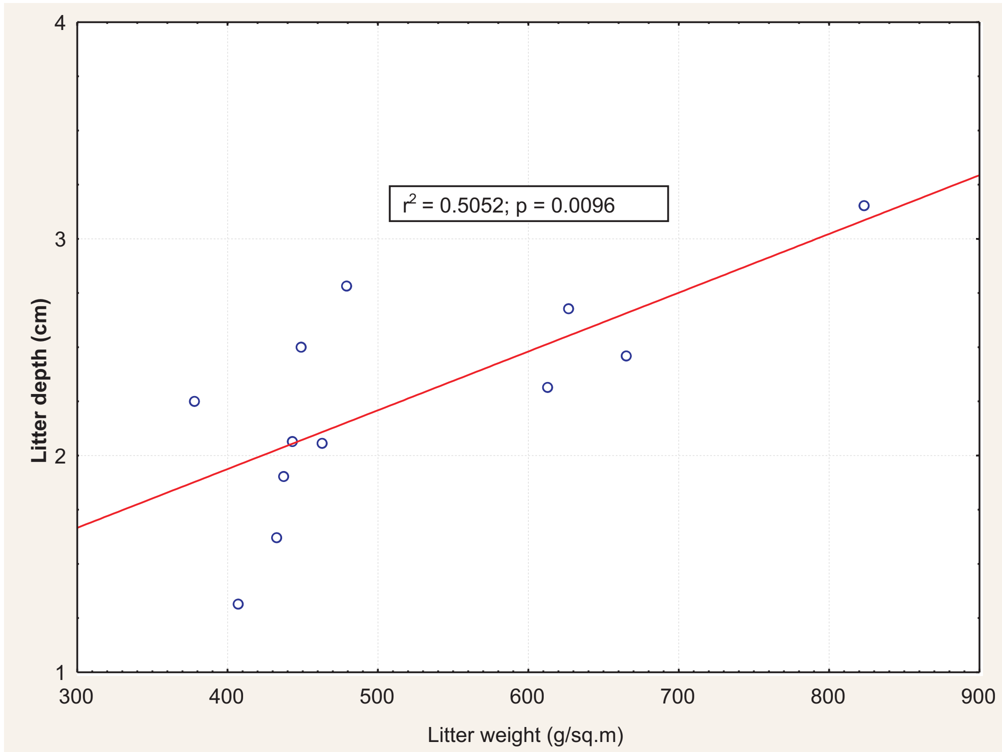

Figure 1. Linear regression graph of litter layer depth on litter layer weight of urban forested wetlands of EBRP, Louisiana.

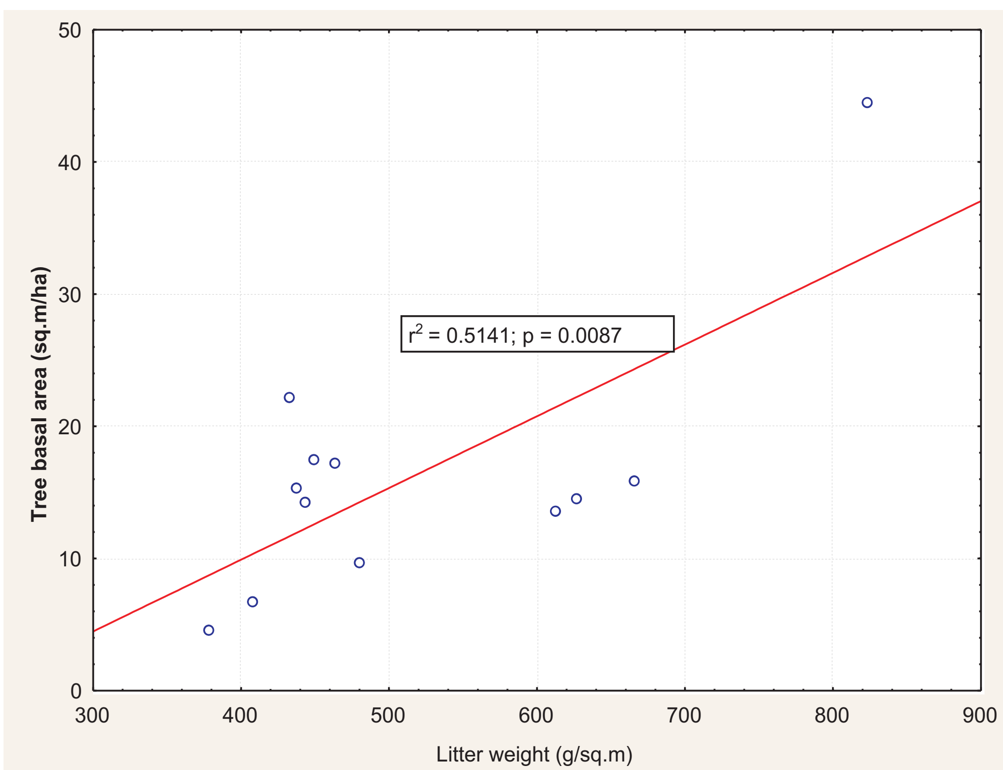

Figure 2. Regression graph of tree basal area on leaf litter weight of urban forested wetlands of EBRP, Louisiana.

Average litter layer/O-horizon mass for all the three sites before the autumn litter fall was 519 g m^{-2}. Litter layer/O-horizon production per year at the end of the autumn leaf fall would be higher than the recorded value. Litter fall production of bottomland hardwood forests of the southeast is 574 g m^{-2} y^{-1}, while for naturally flooded swamps the value is 418 gm m^{-2} y^{-1} [25, 26]. For periodically flooded riverine forest systems in Louisiana, a leaf litter production of 725 gm m^{-2} y^{-1} has been reported [27]. Similarly, naturally flooded swamps were found with 630 g m^{-2} y^{-1} of leaf litter production in Southwestern Louisiana [28]. Leaf litter accounts for 41% of the net primary productivity in bottomland hardwood forests [25]. The litter layer/O-horizon biomass determined in this study is comparable to that reported by Conner [25], Megonigal [27] and Hoeppner [28] for bottomland hardwood forests in the Southeast.

Thus litter layer/O-horizon mass of 500 g m^{-2} or more indicates an optimally functioning site regarding the role of O-Horizon biomass in EBRP. Based on the significant linear relationship of litter layer/O-horizon weight and depth, depth could be used as a surrogate indictor of the O-horizon biomass production and it contribution to the nutrient cycling function. Litter layer depth as a measurement technique for the O-horizon characterization was also developed for the HGM assessment guidebook of the forested wetlands of Yazoo delta, Mississippi [29], which also falls in the Lower Mississippi alluvial valley like that of EBRP. Fig. 3 was used to scale this variable for the research sites. Litter layer/O-horizon depth may decrease in spring, yet the scaling range is large enough to (2 to 5cm litter depth for optimally functioning sites) to account for minor seasonal changes in litter layer/O-horizon depth.

Maintenance of Characteristic Plant Community

In this model, the overbank flood frequency variable was removed from the model because of the absence of a hydrological connection between the wetlands in the EBRP and active streams or rivers.

The major sources of hydrological input into the wetlands are precipitation, and run-off from

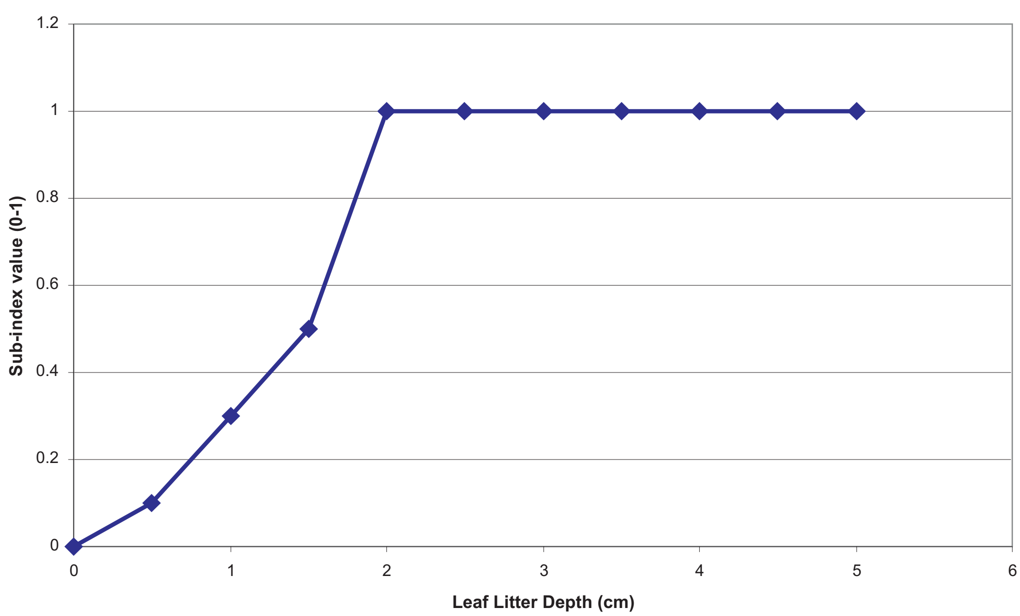

Figure 3. Litter layer/O-horizon biomass sub-index score determination graph.

roadside ditches and small creeks that carry rainfall and urban run-off. The small seasonal channels and creeks that empty into the forested wetlands are not monitored for their flow regime by any state or federal agency. Therefore, the overbank flood frequency variable was dropped from the model to make it representative of urban forested wetlands.

Plant species composition variable (V_{COMP}) of this model was adjusted for EBRP by adding and removing species from the plant list. Some of the dominant species given in the Western Kentucky guidebook do not occur as dominant species in the wetlands of the EBRP. Moreover, some species in EBRP that occur as dominants are not listed as such in the Western Kentucky guidebook. An updated dominant plant species list developed for EBRP is shown in Table 2.

Table 2.
Dominant plant species by strata in urban-forested wetlands of the EBRP, Louisiana.

Tree	Shrubs/understory	Ground vegetation
Acer rubrum	*Acer rubrum*	*Acer rubrum*
Carya x lecontei	*Carya x lecontei*	*Arundinaria gigantean*
Celtis leavigata	*Carpinus caroliniana*	*Aster sp.*
Fraxinus pensylvanica	*Celtis leavigata*	*Boehmaria cylindrical*
Liquidambar styraciflua	*Fraxinus pensylvanica*	*Campsis radicans*
Quercus nigra	*Ilex deciduas*	*Carex squarosa*
Quercus michauxii	*Liquidamabar styraciflua*	*Eragrostis alba*
Q.pagodaeifolia	*Lagustrum sinense*	*Clyxeria striata*
Quercus phellos	*Quercus nigra*	*Hypericum sp.*
Quercus lyrata	*Quercus phellos*	*Impatiens capensis*
Quercus laurifolia	*Quercus lyrata*	*Panicum sp.*
Salix nigra	*Quercus laurifolia*	*Pathenocissus quiquefolia*
Ulmus americana)	*Quercus pagodifolia*	*Pilea pumila*
	Nyssa sylvatica	*Quercus phellos*
	Salix nigra	*Sabal minor*
	Ulmus Americana	*Salix nigra*
	Ilex deciduas	*Sauraurus cernuus*
		Smilacina recemosa
		Smilax rotundifolia
		Sparganium sp.
		Toxicodendron radicans

(Plant names are according to Harrar and Harrar [33]).

Species like water oak (*Quercus nigra*), American Elm (*Ulmus Americana L)*, Laural oak (*Quercus laurifolia Michx*), Bitter Pecan (*Carya x lecontei Little)*, Dwarf Palmetto (*Sabal minor*), Chinese privet (*Ligustrum sinense*) and Red maple (*Acer rubrum L*) were found dominant in one or more than one strata of trees, shrubs or ground vegetation in the research plots. These species are reported as representative of the forested wetlands of EBRP [30,31,32]. Based on the dominance of these species in a stratum according to the 50/20 rule [19], all these species were added to the dominant species list of trees, shrubs or ground vegetation (Table 2).

River birch (*Betula nigra*), Shellbark hickory (*Carya laciniosa Schneid*), Shingle Oak (*Quercus imbricaria Michx),* Post Oak (*Q.stellata* Wang), Pin Oak (*Q.palustris Muench*) and Hackberry (*Celtis occidentalis L*) are dominant species in the forested wetlands of Western Kentucky. These species were not found during the surveys in the research sites in EBRP. The distribution of these species does not extend to EBRP [31,32,33].Therefore these species were removed from the dominant plant species list to make the list representative of the vegetation type of the forested wetlands of EBRP.

Provision of Habitat for Wildlife

In this model (Eq. 3), the overbank flood frequency variable was not applicable to the wetlands of the EBRP and was removed from the model. Standard techniques given in the Western Kentucky guidebook [19] for the VMACRO, VTRACT, VCONNECT, VCORE, VTBA, VTDEN, VSNAG and VLOG variables were found applicable to the wetlands of EBRP. The changed measurement techniques of O-horizon and plant species composition variables discussed in the previous models (Eqs. 1 and 2) also apply to this model.

Depending on the age, hydrology and disturbance level, some forests in EBRP have three vegetation strata (Burbank site), while others lack one or more strata (Siegen and Flonacher sites). The presence or absence of vegetation strata (spatial stratification) affects the microhabitats available for wildlife. According to Lynch and Whigham's [34] study on habitat requirements of 15 birds, abundance was high in mature forests with tall canopies and well-developed herb and shrub layers. Bushman and Therres [35] studied 19 bird species in Maryland and concluded that the preferred habitat was a closed canopy, mature forest, with a mix of dense and open understory conditions. Some forest organisms are limited to a particular stratum, and differences in plant community structure and stratification between sites can lead to differences in animal diversity and composition [2]. The richness and diversity of consumer species is dependent on plant diversity that produces structural niche differentiation [36]. Understory and ground vegetation [37] including ferns and lichens are important components of a stand for a variety of taxa [38], and the quantity of ground vegetation cover is highly correlated with richness of small mammal species [39,40]. Bird species composition is more diverse when horizontal forest structure (or even the landscape) structure is more heterogeneous [41]. Contribution of herb, shrub and tree strata in providing habitat to vertebrates is used in the HGM assessment guidebook for riverine wetlands of Northern Rocky mountains [42]. Difference in stratification is likely to influence the habitat quality; therefore, a new variable, the vegetation strata (VSTRATA) variable was added to the model (eq. 3) and a sub-index value table was scaled for the variable accordingly (Table 3).

Table 3.
Calibrated scale for the characterization of vegetation strata variable.

Vegetation status	Sub-index values
All layers present	1.0
Herb layer absent (<5% vegetation cover)	0.9
Shrub layer absent (<50 stems/ha)	0.7
Tree canopy absent (<50 stems/ha)	0.3
Newly regenerating site (restoration possible)	0.1
Converted sites (restoration not possible)	0.0

The absence of herb layer indicates that the canopy is closed and dominated by mature trees. As a result there is not enough light reaching the forest floor to promote ground vegetation growth. Consequently, this absence does decrease the diversity of habitats (heterogeneity) in the forest, yet it is not an indicator of disturbance to vegetation. Therefore, a decrease of 0.1 in sub-index value is assumed to occur in terms of provision of habitat to wildlife. Similarly, with the absence of shrub layer, it is assumed that a sub-index value decrease of 0.3 occurs. Less than 50 stems per hectare of trees or shrubs is a very sparse density [19] to be qualified as optimally functional strata regarding the provision of habitat to wildlife [43] with reference to a reference standard wetland. If tree layer is absent, this indicates either a newly regenerating site or a damaged site. In such circumstances, the sub-index value is dropped by a factor of 0.7. If all the layers are absent, and the site has a potential for restoration, then a value of 0.1 could be assigned to the site.

Functional Capacity Indices of the Selected Sites

The cycling of nutrients function varied among the sites (Fig. 4). The variability in function among the sites was mainly affected by

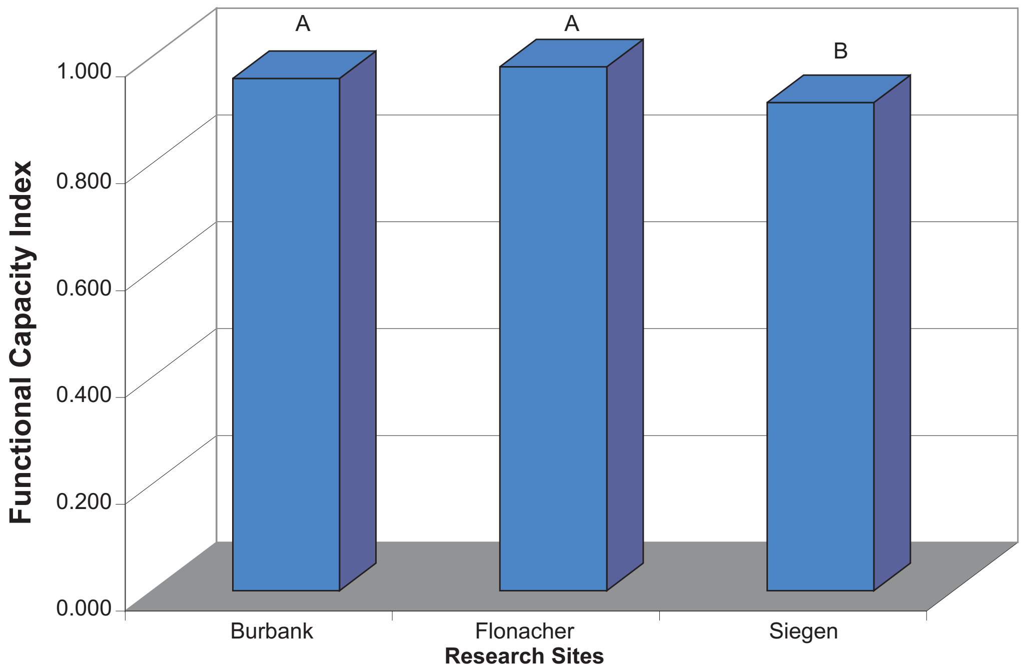

Figure 4. Mean nutrient cycling FCIs of urban forested Wetlands in EBRP, Louisiana (means with the same letters are not significantly different from each other at p= 0.05).

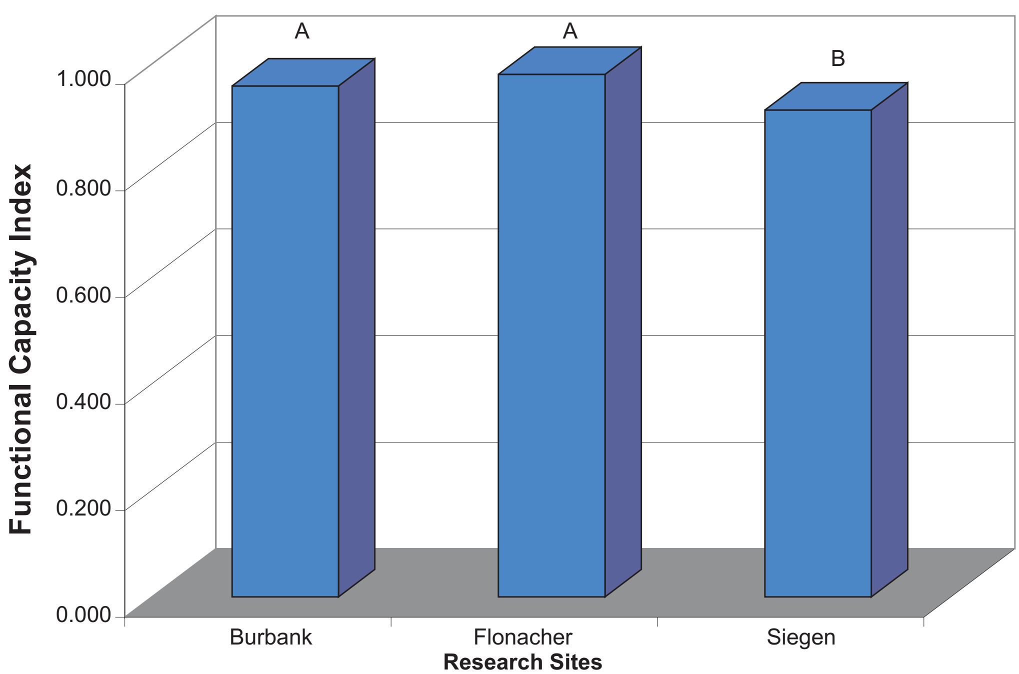

Figure 5. Mean plants habitat maintenance FCIs of urban forested wetlands in East Baton ouge Parish, Louisiana (means with the same letters are not significantly different from each other at p = 0.05).

the variability in the values of the biotic component of the assessment model. The FCI was 0.73 for the Burbank site, 0.68 for the Flonacher site, and 0.59 for the Siegen site. The Burbank site was significantly higher than the Siegen site (p value <0.01), but not the Flonacher site (p =0.49).

The maintenance of the characteristic plant community function for the three sties (Fig. 5) was found to be above the 90th percentile on the functional scale. The Burbank and Flonacher sites were significantly higher than the Siegen site (p = 0.02 and 0.04 respectively). All the plant species reported from the three sites were facultative, facultative wet and obligate wetlands species of the region [32].

The wildlife habitat function varied significantly among the sites (Fig. 6). The function was influenced by hydrologic, landscape and biotic variables and their values were different for the three sites. The difference was clearly reflected in the average FCIs for the three sites. The hydrologic and biotic components of the assessment model did not vary among the three assessment sites, while the landscape scale variables component varied for all the sites. The differences in the landscape scale variables led to significant functional differences in the FCIs among the three sites. The Burbank site was significantly higher than the Flonacher and Siegen sites (p <0.01). The FCI of the Flonacher site was significantly higher than the Siegen site (p <0.01).

Urbanization has isolated and fragmented the wetlands in EBRP, which has affected the spatial scale variables such as fragmentation, habitat connection, and interior core area. Consequently, the wildlife habitat quality FCI dropped below the 50th percentile for the three sites compared to maintenance of native plant community function (FCIs for the three sites were above the 90th percentile), which has a major role in providing wildlife habitat. Urban and

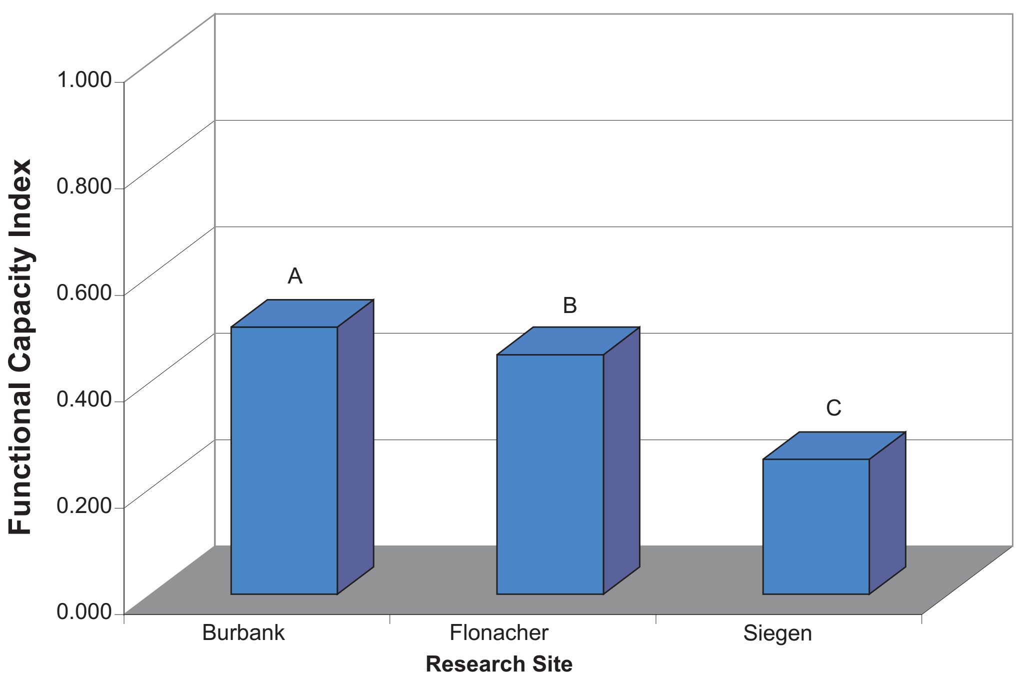

Figure 6. Mean Wildlife Habitat Provision FCIs of Urban Forested Wetlands of East Baton Rouge Parish, Louisiana (means with the same letters are not significantly different from each other at p = 0.05).

suburban development affected the spatial scale variables of the wildlife habitat in EBRP. Similar habitat degradation trends by urban development as a threat to wildlife habitat, and native flora and fauna is reported by Marzluff [44].

From the management perspective, the provision of habitat to wildlife function in an urban environment is very important. Urban and suburban fragmented forest patches provide important staging, feeding and resting areas for migratory birds, besides serving as a habitat hotspot for native flora and fauna in an altered landscape [45]. Though the urban forested wetlands function at a lower level in providing habitat for wildlife compared to reference wetlands, these habitat patches needs to be protected/mitigated from further functional losses.

Applicability of the Adjusted Models

The Western Kentucky guidebook served as the basic assessment template for assessing the biotic functions of the urban- forested wetlands of EBRP. Out of the 17 variables given in the Western Kentucky guidebook for the selected functions, standard measurement techniques of 15 variables were found to be applicable to the urban forested wetlands of EBRP. COE used 10 variables associated with the development of wetland vegetation out of the 17 variables of the Western Kentucky guidebook for monitoring the restoration of bottomland hardwood forests in western Tennessee, western Kentucky and eastern Arkansas. The variables were found to be applicable to the low gradient riverine wetlands of the selected sites, although majority of the sites were outside of the reference domain of the Western Kentucky guidebook [46]. The study indicated that the Western Kentucky guidebook's recommended variables associated with the development of biotic characteristics were consistent and applicable to low gradient riverine wetlands in the southeast region [46]. It is not surprising then that the current study found these same variables applicable to the research sites.

Functional capacity indices for the three functions (Figs. 4 to 6) were determined through the adoption of the adjusted models. The FCIs were significantly different for the three sites based on their existing topo-hydrologic, edaphic and biotic characteristics. The differences between the level of functioning for the three functions were indicative of the difference between the biotic and abiotic characteristics, disturbance level and size of the three sites. On the whole, the functional level of Siegen site was lower than the Burbank and Flonacher sites. This is consistent with the young homogeneous plant community dominated by Salix nigra, an indicator of disturbed and newly restoring sites. Siegen site was more disturbed hydrologically, had a non-stratified monotype young forest stand, fragmented and had litter layer/O-horizon mass which significantly reduced its nutrients cycling, maintenance of characteristic plant community and provision of habitat to wildlife functions ($p < 0.01$) compared to Burbank and Flonacher sites.

In conclusion, the average FCIs determined for the selected functions were representative of the sites conditions and accurately captured site specific differences in hydrologic regime, vegetation type, composition, density and stand stratification, litter layer/O-horizon biomass and landscape scale variables. This indicates the suitability and feasibility of the adjusted assessment models for functional assessment of urban forested wetlands in EBRP. Further research on the refinement and validity of these models for urban forested wetlands in the Southeastern US and similar riverine forests elsewhere in the world is recommended.

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