11-19-2021

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Ecological Indicators 133 (2021) 108391

Contents lists available at ScienceDirect

Ecological Indicators

journal homepage: www.elsevier.com/locate/ecolind

Original Articles

Using multiple taxa and wetland classification schemes for enhanced detection of biological response signatures to human impairment

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ARTICLE INFO

Keywords:
Anuran communities
Avian communities
Indices of biological integrity (IBIs)
Macroinvertebrate communities
Vegetation communities
Wetlands

ABSTRACT

Wetland indices of biological integrity (IBIs) are a common component in monitoring the wetland water resources as required by the United States' Clean Water Act (CWA). The effectiveness of an IBI to monitor disturbance is dependent on the metrics being consistently responsive to measures of human disturbance within a described classification category. We present IBIs designed for two types of commonly used wetland classification systems – the hydrogeomorphic (HGM) and the National Wetlands Inventory (NWI). The metrics making up the IBIs were derived from anuran, avian, macroinvertebrate, and vegetation communities; each representing increasing levels of resources associated with gathering the necessary data. Knowing which communities' data best corresponds to impairment can maximize limited wetland monitoring resources, especially if the response differs based on the wetland vegetation type NWI or HGM position on the landscape. By combining these two classification schemes together, to better define a wetland's form and context on the landscape, more of the variability in community metrics are explained by human impairment. Moreover, when multiple taxa are used within a single wetland classification scheme, the response of the multi-taxa community IBI to the human disturbance gradient is often more sensitive than one-taxa group alone. This approach, a combination of taxa, in hybrid 2-system classification schemes, creates additional utility in measuring the effectiveness of wetland assessments and, or restoration success.

1. Introduction

Under the United States' Clean Water Act (CWA), individual states have the right to determine water quality standards with respect to differing types of waterbodies (*e.g., streams, lakes, wetlands*). Within wetlands, the CWA directive to restore and maintain the chemical, physical, and biological integrity of the United States' waters (33 U.S.C. §1251- Section 101(a)) does not define how each of these quantitative criteria should be developed. Despite supportive scientific literature describing a consistent link between aquatic communities and human impairment (Wardrop et al., 2007a; Raab and Bayley, 2012, Kutcher and Bried, 2014, Magee et al., 2019), little regulatory headway has been made using wetland communities as an indicator of wetland value, especially in terms of mitigation or the regulatory requirement to replace wetland area lost to development on the landscape.

Vegetation community characteristics are only one component of performance success criteria for restored and created wetlands, mitigation or otherwise; and often take longer to develop meaningful metrics than the allotted required monitoring period for success (Van den Bosch and Matthews, 2017). As such, the vegetation criteria tend to be tied to subjective numbers like stems per acre rather than ecological thresholds that must be calibrated for each region (DeBerry et al., 2015). This lack of using species assemblages in monitoring comes despite the U.S. Environmental Protection Agency (EPA) previously announcing standardized guidelines based on three tiers to link a wetland's condition to its ability to perform wetland functions with the hopes to lead to an improved regulatory framework (U.S. Environmental Protection Agency, 2003). This framework is based on spatial scales. For example,

; CWA, Clean Water Act; EPA, Environmental Protection Agency; HGM, Hydrogeomorphic; IBI, Index of Biological Integrity; NWI, National Wetlands Inventory; ORAM, Ohio Rapid Assessment Method.

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https://doi.org/10.1016/j.ecolind.2021.108391
Received 7 September 2021; Received in revised form 15 November 2021; Accepted 16 November 2021
Available online 19 November 2021

Level 1 is a landscape-level evaluation of a wetland's potential capacity to perform specific ecosystem services within a watershed and although the level of wetland functions is static with Geographic Information System (GIS) based variables, it is useful in describing the drivers in vegetation community assemblages and capacity to buffer nutrients (Alamanos and Papaioannou, 2020, Allen et al., 2021). Level 2 is a rapid, onsite, assessment of wetland condition that is meant to provide insight to the level of wetland functions (*e.g.,* sediment retention, nutrient processing, ecological condition, floodwater attenuation, etc.) that cannot be determined from the Landscape level data. However, these assessments must first be verified, validated, and then calibrated within more nuanced site-level research, or Level 3 surveys, for the scoring to be meaningful (DeBerry and Perry, 2015, Stein et al., 2009, Sifneos et al., 2010, Gallaway et al., 2020). These Level 3 efforts are intensive biological and physiographic studies within a wetland over time to quantify levels of function (Fennessy et al., 2007, Wardrop et al., 2007b, Deller-Jacobs et al., 2010). This format is intended to guide the establishment of wetland monitoring programs that mandate regulators and practitioners to report on the level of function within a wetland and evaluate restoration success (U.S. Environmental Protection Agency, 2003). However, wetlands are typically regulated in the United States based on vegetative characteristics (U.S. Army Corps of Engineers, 1987), but a wetland's ability or capacity to perform functions varies by hydrogeomorphic position (Brinson, 1993, Cole et al., 1997).

Classification schemes form the categorical bins for calibrating and developing Indices of Biological Integrity (IBIs) responses to disturbance. While most IBIs are categorized based on landscape position (Mack, 2007), we attempt to reconcile this disconnect by deliberately using both classification techniques, and even merging them to describe community response to disturbance (Mack, 2009, Veselka and Anderson, 2013).

Our objective is three-part in the evaluation of the responses to disturbance by avian, anuran, vegetation, and macroinvertebrate communities. Firstly, we examine the relation between each individual species assemblage and the disturbance gradient by wetland classification schemes, one based on vegetation structure and the other on wetland landscape position, to understand consistency of the response metrics within each taxa group to disturbance based on classification system.

Secondly, we examined different combinations of species assemblages to create a multi-species IBI that can be as effective as one assemblage but varies on level of monitoring commitment and proficiency. Combining taxa data can increase the accuracy of the Index of Biological Integrity (Medeiros et al., 2015) and can enable monitoring choices based on season or professional resources (Veselka and Anderson, 2013). For example, vegetation metrics may be the best single taxa indicator of human disturbance; but combining the avian and anuran metrics (via addition) may result in a multi-taxa IBI that is more sensitive and cost-effective than the single taxa vegetation-IBI.

Thirdly, based on the unique response to disturbance by species assemblages within different wetland classification schemes, we use additive properties of indices to improve resource management decisions (Gerritsen, 1995). This has led us to develop hybrid indices based on the geomorphic landscape position of the wetland and the vegetation structure to measure the biological integrity of multiple species assemblages. This grows the number of options for monitoring based on species assemblages and wetland type to allow flexibility in monitoring by resource managers. For example, with a limited budget and a large number of wetlands requiring monitoring, a subset of sites can be prioritized by sensitivity requiring professional skills (*e.g.,* botany identification), while volunteers are assigned to collect meaningful anuran or avian data over the majority of wetland monitoring locations.

We developed a series of indices to measure the biological integrity (Level 3) of wetlands in West Virginia by collecting biological data that varies seasonally (collection time) but responds consistently to disturbance (Veselka et al., 2010a; b, Veselka and Anderson, 2013). Common wetland taxa (avian, anuran, macroinvertebrate, and vegetation communities) were selected to measure their varied responses to stressors over distance and time to localized human disturbance (Miller et al., 2006). Although in the United States, wetland permitting starts with defining a wetland's edge jurisdictionally (U.S. Army Corps of Engineers, 1987); altering areas around a wetland can diminish the functionality and ecological importance, even if the physical size of a wetland remains intact (Gascoigne et al., 2011, O'Connell et al., 2012). As adjacent uplands are modified by landuse changes, the wetland vegetative community may be subject to invasion by exotic and noxious species (Galatowitsch et al., 2000, Allen et al., 2021), as well as changes in community composition of amphibian and avian communities (Bryce et al., 2002; Houlahan and Findlay, 2003, Stapanian et al., 2004). Sedimentation from upland disturbances can reach waterways, covering benthic macroinvertebrate habitat and stifling macrophytic transpiration – reducing the wetland capacity to filter and trap pollutants (Martin and Neely, 2001; Mahaney et al., 2004). Our hope in emphasizing ecological communities as the target indicating successful restoration will promote considerations for habitat features that are a component of natural wetlands, rather than performance criteria strictly on design attributes for restored or created wetlands, like stems per acre, for example.

2. Methods

The selection of methods was intentional to promote the likelihood of management agencies incorporating taxa assemblages into wetland restoration and mitigation success criteria. We developed a series of wetland indices of biological integrity using anuran, avian, macroinvertebrate, and vegetative communities (Veselka et al., 2010a; b, Veselka and Anderson, 2013) in 151 wetlands located throughout West Virginia during 2005–2006. These sampling techniques mimicked methods that are commonly used by both professionals and volunteers to collect data, represent approved surveying protocols by U.S. government agencies and volunteer groups, and have been used in numerous other peer-reviewed methodologies (Shirose et al., 1997; Pellet and Schmidt, 2005; Weir and Mossman, 2005). For example, our anuran acoustically-based IBI (AA-IBI) were derived from methods used by the North American Amphibian Monitoring Protocols (NAAMP, 2005) and are the easiest data to collect in terms of expertise and effort (Veselka and Anderson, 2013). The avian wetland IBI (AW-IBI) are arguably next in terms of expertise and resources needed to evaluate wetlands (Balcombe et al., 2005a). Birds can be identified by sight and sound but is a more challenging process for people to learn. These data can be collected by volunteers, but these volunteers must be trained and checked to ensure the quality of their data (Hicks and Nedeau, 2000). The identification of macroinvertebrates is professional-level work, as communities were identified and quantified to Family, and collected with a water-column sampler and benthic sampling corer (Balcombe et al., 2005b). It should be noted that not all wetlands could be sampled for water column macroinvertebrates because not all had standing water during the sampling period (July – August). The vegetation-based IBIs (Veg-IBI) were based on data that are the most laborious and require professional skill. Identifying plants to the species level, especially grasses and other monocots, is necessary to develop meaningful metrics from the raw data. The relative cover of the dominant wetland communities was measured using the wetland delineation circular plots (U. S. Army Corps of Engineers, 1987) and a complete walk-through census of other species was tallied for use in Floristic Quality Index (FQI) metrics (Lopez and Fennessy, 2002; Miller and Wardrop, 2006; Rentch and Anderson, 2006).

We chose *a-priori* to develop wetland IBIs based on vegetation-structure driven National Wetland Inventory (NWI) (i.e., Cowardin et al. (1979) class; hereafter referred to as "Cowardin"), and the landscape-setting hydrogeomorphic (HGM) class (Brinson, 1993) to better partition the response to disturbance (Gerritsen et al., 2000).

Table 1
Total number of sites by hydrogeomorphic (HGM) class and Cowardin et al. (1979) class by ecoregion for use in developing indices of biological integrity (IBIs) in West Virginia, USA from 2005 to 2006.

	Level 3 U.S. Environmental Protection Agency aquatic ecoregion[a]			
Wetland Classification SystemWetland ClassRidge and Valley		Central Appalachian	Western Allegheny Plateau	Total
HGM Class				
Depression	10	28	34	72
Floodplain	12	17	6	35
Impoundment	1	14	8	23
Fringing[b]	0	2	11	13
Slope[b]	4	4	0	8
Cowardin Class				
Emergent	15	34	26	75
Scrub-shrub	6	17	21	44
Forested	6	14	11	31
Aquatic bed[b]	0	0	1	1
Total	27	65	59	151

[a]Omernik, (1987), modified by Woods et al. (1999).
[b]Removed from analysis due to small sample size.

Wetlands were probabilistically stratified by ecoregions (Omernik, 1995; Woods et al., 1999), as well as by NWI wetland class occurrence in these ecoregions (Table 1).

The HGM classifications were kept as class designations, rather than the more detailed regional HGM subclass (Cole et al., 1997), to simplify for water resource practitioners. We selected an independent disturbance gradient made up of components in the Ohio Rapid Assessment Method (Mack, 2001) for wetlands that were specific to signs of human impairment (buffer and land use, hydrology, and habitat alternation and development). The metrics included in each of these indices were extensively evaluated to ensure the capability to discriminate between reference and stressed sites throughout the state of West Virginia; and at the same time able to provide consistent scores for each class-specific (Cowardin or HGM-based) IBI regardless of the wetland's alternative classification (Veselka et al., 2010a; b). For example, metrics capable of differentiating between reference and stressed in an emergent wetland, were able to do so consistently regardless if the wetland was a depression or a floodplain. These methods are discussed in-depth in Veselka et al. (2010a; b), but resulted in each selected metric being scaled from 0 to 10, and the cumulative total of all the metrics for each taxa group was used to create a final IBI score, which was then evaluated in relation to the disturbance score using simple linear regression (Veselka and Anderson, 2013).

We then combined metric scores of the individual taxa specific IBIs, via addition, to create multi-species IBIs to reflect different levels of committed resources to wetland monitoring; comparing them to each classification scheme, Cowardin or HGM, individually. The resulting wetland class-specific, multi-taxa IBIs were each evaluated against the disturbance scores using simple linear regression to provide a relative measurement of sensitivity.

Thirdly, we combined metric scores from single and multiple taxa groups that were responsive to the Cowardin or HGM classification schemes and combined them to develop IBIs for a hybrid classification scheme that reflects both the Cowardin and the HGM setting (*e.g.*, a floodplain-emergent wetland vs. a floodplain-forested wetland). Again, using simple linear regression to be consistent with our previous IBI development (Veselka et al., 2010a; b), we compared the effectiveness of using metrics from both classification schemes to our class-specific and multi-taxa IBIs to determine a relative measure of disturbance

Table 2
Significant R^2 for single and multi-taxa indices of biological integrity (IBIs) for individual and integrated hydrogeomorphic (HGM) and Cowardin classes for wetlands of West Virginia, USA, 2005–2006.

		Cowardin			Combined Cowardin – HGM						HGM	
Taxa groups		Emergent	Scrub-shrub	Forested	Emergent-Depression	Emergent-Floodplain	Scrub shrub-Depression	Scrub shrub-Floodplain	Forested-Depression	Forested-Floodplain	Depression	Floodplain
	Sample size*	75	44	31	38	16	19	8	14	11	72	35
Anurans												0.18
Birds		0.11	0.25	0.24		0.52		0.85		0.42	0.12	0.46
Macroinvertebrates				0.14							0.11	0.47
Vegetation		0.14	0.20	0.35		0.55	0.46	0.59	0.42	0.68	0.31	0.56
Anurans - Birds												0.43
Anurans - Birds -Macroinvertebrates						0.59						0.59
Anurans - Birds - Vegetation						0.58		0.68				0.63
Anurans - Macroinvertebrates - Vegetation						0.44						0.54
Anurans - Vegetation						0.35						0.5
Birds - Macroinvertebrates				0.35	0.30	0.75				0.76	0.16	0.72
Birds - Vegetation		0.21	0.34	0.48	0.11	0.75	0.26	0.86	0.32	0.70	0.32	0.75
Birds - Macroinvertebrates - Vegetation				0.52	0.33	0.78		0.86		0.79	0.35	0.77
Macroinvertebrates - Vegetation				0.36	0.25	0.65	0.46			0.58	0.37	0.61

sensitivity. The resulting IBIs, in combination of one another or based on multiple wetland classifications, can then be used to track restoration progress on specific mitigation projects and serve to calibrate and give credibility to the Level 2 rapid assessment ecological condition inferences.

3. Theory

When multiple wetland taxa groups are combined as part of a response index, the cumulative response to predictor variables is often stronger than a single taxa approach (Medeiros et al., 2015). Our research focused on quantifying elements of human disturbance and the taxa response, rather than habitat characteristics. As such, the disturbance gradient was made from three internal metrics (*e.g.,* land use, hydrology, and habitat alternation) that evaluated wetland characteristics based on human impairment based on Ohio Rapid Assessment Method (ORAM) (Mack, 2001). We expected, by evaluating community response to this impairment on multiple scales, we would have a more consistent response to disturbance and single environmental variables (Rooney and Bayley, 2012). For example, benthic macroinvertebrate communities can be limited by the hydrological disturbance to the wetland ecosystem (Vineetha and Nandan, 2021) but demonstrate resiliency towards overall landscape-level factors (Growns et al., 2020). Whereas bird communities demonstrate responses to site-level habitat changes (Gillings, 2019), as well as greater landscape characteristics (Hanioka et al., 2018). As such, using several assemblages to evaluate the response of multiple sources of human impairment is intuitive for an overall wetland assessment, or tracking the overall ecological community response of a wetland over time.

However, some assemblage-specific metrics used in this analysis were not responsive to the overall disturbance index, but still have value in promoting future wetland management research. The consistent ecological response to overall disturbance in wetlands is important, but it is not the same as indicating other functions of the wetland (*e.g.,* floodwater attenuation, sediment filtering, pollutant sequestration, etc.) beyond ecological and biological integrity. Further research using similar methods should re-examine these community metrics by species assemblages for links to these other functions, not intrinsically linked to biological integrity. This may include evaluating vegetation communities for metrics indicating a tolerance for sedimentation (Saaltink et al., 2018), or macroinvertebrate communities responding to pollutant loadings (Mehring et al., 2017). These types of metric comparisons to wetland function are valuable to future research, as a wetland's functional value can be economically important even if the biological integrity of the wetland is degraded.

3.1. Results

A complete listing with details of the relations between taxa group and disturbance is found in Appendix A; however, significant R^2 results for both the individual and combined taxa IBIs and singular and hybrid classification schemes are summarized in Table 2. There were no significant relations within the impoundment class, so we have omitted the class in presenting the results.

For each classification, the metrics from the vegetation IBI or combinations of taxa groups that included vegetation metrics were the most sensitive to the disturbance score; however, the variation in community attributed to disturbance was only marginally better than some of the other taxa. For example, within floodplain wetlands; the vegetation and invertebrate IBI scores, collected based on skills requiring professional (and more expensive) resources, disturbance explained 56% and 47% of the variation in scores, respectively. Alternatively, the disturbance level accounts for 46% and 18% of the variation in avian and anuran IBI scores, respectively in the same floodplain wetlands. The data collected to develop these avian and anuran metrics were derived from volunteer-friendly methods. However, the utility and consistency of the vegetative responses is evident as other wetland classes are considered.

Moreover, regardless of initial wetland classification used to develop an IBI (i.e., HGM or Cowardin) the response to disturbance could be improved with the hybrid HGM $\times$ Cowardin approach to IBI development. For example, if the prior mentioned floodplain wetland was also an emergent wetland; by adding the avian and vegetation metrics of both floodplain and emergent IBIs, we form a hybrid class, multi-taxa IBI where the disturbance scores explained 75% of the variation in IBI scores.

4. Discussion

The development of HGM $\times$ Cowardin class IBIs provide us with better understanding of biological responses to impairment than traditional one classification techniques. The simplicity of our approach, integrating classification systems, provides a framework that can and should be expanded to other regions. For example, our approach may elucidate the ecological response characteristics of an emergent-depression prairie pothole region of the United States, from an emergent wetland associated with a floodplain in the same region.

This research has two applied lessons that can be applied to wetland monitoring programs. When entities are developing IBI monitoring criteria, additional analyses to develop IBIs for both the HGM and Cowardin class, as well as multiple species assemblages, can provide tremendous resolution in determining the level of response between communities and human impairment (Mack et al., 2008, Veselka and Anderson, 2013). This can even be completed with *ex post facto* data used to develop existing IBIs, as the human impairment independent variable remains consistent, and metrics evaluated again against the alternative classification. The multi-species approach creates options in monitoring and can increase the predictability of the response in measuring the trends of biological integrity in wetlands (Veselka et al., 2010a; b, Medeiros et al., 2015). For example, forested wetlands may be problematic applying Veg-IBIs regardless of HGM class because the older, more established canopy species do not exhibit responses from current impairments under the same timeline as herbaceous and understory species (Mack, 2009). We see marked improvement in our forested response to disturbance when we incorporate other taxa groups in addition to vegetation, specifically birds (Veselka et al., 2010a); or alternatively combined birds and macroinvertebrate metrics can explain the same amount of variation within a forested wetland under a single classification scheme (Veselka et al., 2010b). With this, the utility works both ways; as alternatively within other vegetation classes (emergent and scrub-shrub), the predictability of species assemblages' responses to disturbance is greatly improved when HGM is taken into context. This is consistent with a plant-based IBI in neighboring Pennsylvania indicated around 80% of variability in wetland plant communities is related to disturbance (Miller et al., 2006) when isolated to a specific regional HGM classification (Cole et al., 1997). Within our study, we can only account for disturbance accounting for 21% of the variability in emergent wetlands, even if evaluated using a combination of avian and vegetation metrics. However, if we know that these emergent wetlands are also floodplains, the additional metrics included for floodplains for birds and vegetation attribute 75% in the variation in IBI scores to disturbance. The finer the resolution of the lens to evaluate metric sensitivity to disturbance, the more variability can be explained.

This first applied lesson is wetland monitoring efforts can be tiered, based on the multiple classification schemes and species assemblages to maximize survey effort based on allotted resources (Veselka and Anderson, 2013), as different bioassessment methods can yield similar results (Herbst and Silldorff, 2006). We have demonstrated a mechanism using multiple species assemblages to predictably measure the impairment the same. This can embolden wetland resource managers to make an educated, proactive choice in regard to committing resources to wetland monitoring based on effort and resources available. The results show not all wetlands need to be monitored using the same criteria when

Table 3
Recommendations for allocating monitoring resources-based wetland classifications and results from the development of wetland indices of biological integrity for wetlands in West Virginia.

Wetland Type	Survey Resources Recommendations	Taxa Groups
Emergent	Professionals and volunteers	Vegetation and birds
Scrub-shrub	Professionals and volunteers	Vegetation and birds
Forested	Professionals and volunteers	Vegetation and birds
Emergent-Depression	Volunteers and professionals*	Birds and macroinvertebrates
Emergent-Floodplain	Volunteers and professionals*	Birds and macroinvertebrates
Scrubshrub-Depression	Professionals	Vegetation
Scrubshrub-Floodplain	Volunteers	Birds
Forested-Depression	Professionals	Vegetation
Forested-Floodplain	Volunteers and professionals*	Birds and macroinvertebrates
Depression	Professionals and volunteers	Vegetation and birds
Floodplain	Professionals and volunteers	Vegetation and birds

*Professional-level identification of collected samples required.

looking for a response to human impairment, and our research attempts to provide a measure of clarity for options to determine survey effort (Table 3).

The second applied lesson pertains to future research efforts evaluating ecological communities and species assemblages as predictors of wetland function. We understand the importance of the landscape context demonstrated by the significant gains in sensitivity to disturbance with the additional floodplain metrics. This is especially relevant to areas typified by steep terrain where flat land is a valuable resource and commonly found in the valley bottoms, associated with meandering streams and rivers. Many of these seasonal floodplain wetlands are missed on maps (Serran and Creed, 2016), confounding efforts to put a functional value on wetlands that are not identified or quantified. Worldwide, these floodplain wetlands are susceptible to land-use changes brought on by human development, and many historically have suffered from being drained for agriculture or disconnected from flooding regions as rivers were channeled and dredged to facilitate commerce. Our research provides a framework on how to improve the monitoring of the biological integrity of regionally-specific wetlands over time in response to impairment. However, it also can provide the design to model and monitor the relation between species assemblages and function, which is not always tied to biological integrity. Biologically degraded floodplain wetlands still have value in trapping sediment and filtering out pollutants to improve water quality (Whigham and Jordan, 2003; Fleming-Singer and Horne, 2006; Kovacic et al., 2006). Therefore, subsequent expansion of this framework should include the evaluation of species assemblages for predictable relations to water quality or another wetland function, unrelated to biological integrity.

5. Conclusions

Our research provides wetland resource managers with guidance on increasing wetland monitoring efficiency by using two classification schemes and multiple species assemblages to allocate professional and volunteer services to monitor wetlands. Building a wetland monitoring program with these guidelines in mind not only maximizes the data utility in terms of resources, but also enables the ability to leverage data to evaluate wetland function in the future. Additionally, the ability to provide meaningful data collection opportunities for volunteers, can have a two-fold compounding effect. Public involvement can create a greater sense of public empathy for the plight of wetlands; while simultaneously increasing sample sizes to continually strengthen the relation between monitoring and ecological research.

CRediT authorship contribution statement

Walter Veselka IV: Conceptualization, Formal analysis, Investigation, Methodology, Writing – original draft. **Walter S. Kordek:** Data curation, Supervision. **James T. Anderson:** Funding acquisition, Project administration, Resources, Writing – review & editing.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgments

We thank Adrianne Brand, Mark Hepner, Joe Osbourne, and the late Bill Grafton for assistance in data collection. Geographic information system and database management assistance was provided by Ben Gilmer. All research data is available for review in its entirety at https://researchrepository.wvu.edu/etd/2619/. Funding was provided by the West Virginia Division of Natural Resources with assistance from U.S. EPA State Wetland Program Development Grant CD 973080-01-0.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecolind.2021.108391.

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