Ecological Indicators 94 (2018) 454–467

Contents lists available at ScienceDirect

Ecological Indicators

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

Review

Indicators of vegetation development in restored wetlands

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ARTICLE INFO

Keywords:
Restoration indicators
Monitoring
Trajectories
Remote sensing
Landscape metrics
Structure

ABSTRACT

Significant wetland losses across the globe have motivated large-scale restoration efforts to improve the quality of wetland habitats. However, previous studies have shown a high variability in the outcomes of restoration treatments. Post-restoration monitoring is critical to identifying factors constraining wetland recovery and diverting sites away from restoration goals but is often limited by a lack of funding. To circumvent limitations to the large-scale monitoring of wetlands, it is pivotal to identify metrics that can be implemented at low cost yet provide a reliable signal of restoration progress. We review scientific literature on methods to appraise post-restoration progress in wetland ecosystems, focusing on vegetation-based indicators. We present a synthesis of demonstrated relationships between these indicators, site conditions, landscape context, and key ecosystem functions to highlight benefits and potential limitations to the widespread applications of these indicators to post-restoration monitoring. Based on this literature synthesis, we suggest adopting a multi-metric approach to fully measure ecosystem recovery. Potential solutions identified in this review to reduce costs associated with large-scale monitoring include: identifying correlation among indicators, focusing on the most widespread species, and using remote sensing to expand the spatiotemporal scope of monitoring and inform monitoring efforts.

1. Introduction

Wetlands play a key role in supporting biological diversity and providing ecosystem services, but have been critically impacted by global land conversions and increasing ecosystem stress (Allan et al., 2013; Costanza et al., 1997; Zedler, 2003). In response, various organizations, from local to nationwide, have initiated large-scale restoration efforts to rehabilitate depleted ecosystem functions and increase the quality and extent of wetland habitats (Bedford, 1999; Moreno-Mateos et al., 2015; Suding, 2011). The Society for Ecological Restoration defines ecological restoration as the assisted recovery of an ecosystem towards a desired state or ecological condition (SER, 2004). In wetland ecosystems, restoration interventions can include removing non-native species, planting to accelerate recovery, grading to create topographic heterogeneity, and site breeching to increase tidal prism (Simenstad et al., 2006; Zhao et al., 2016). Common goals of wetland restoration projects include extending the quality and extent of wetland habitats, restoring ecosystem services, or promoting the resilience of local communities to climatic changes and sea level rise (Kentula, 2000; Simenstad et al., 2006). Yet, recent studies have revealed a substantial variability in restoration outcomes, even among similar habitat types and restoration treatments (e.g., Berkowitz, 2013; Matthews et al., 2009), with projects sometimes falling short of restoration targets (Brudvig et al., 2017; Van den Bosch and Matthews, 2017). As a result, it is challenging for project managers to predict the outcomes of restoration treatments or identify factors that could divert their sites away from targets (Suding, 2011).

Without a consistent monitoring, it becomes difficult to identify the site characteristics, landscape factors, or management decisions that impact the post-restoration trajectory of sites (Brudvig et al., 2017). For example, hydrological connectivity between wetlands and adjacent land covers through surface flows, runoff, and groundwater can facilitate the transport of pollutants, nutrients, or non-native species (Baldwin, 2004; Moreno-Mateos et al., 2008), increasing the potential for unexpected changes and chaotic fluctuations in post-restoration trajectories. Furthermore, the need to account for variability in climatic, hydrologic, and anthropogenic factors affecting site properties can make post-restoration monitoring and site planning even more challenging (Wilcox et al., 2002; Xiong et al., 2003). Generating long-term and geographically comprehensive ecological datasets from existing sites could improve the planning and design of future projects and help managers identify the most appropriate spatiotemporal scale for post-restoration monitoring (Brudvig, 2011; Kentula, 2000; Suding, 2011). However, such a rigorous documentation of post-restoration

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https://doi.org/10.1016/j.ecolind.2018.07.010
Received 9 December 2017; Received in revised form 4 July 2018; Accepted 5 July 2018

Table 1
Vegetation indicators reviewed in this study, with appreciation of their predictability, ease of monitoring and interpretation.

Type	Indicator	Methods	Ease of Monitoring	Ease of Analysis/Interpretation	Predictability
Structural	**Plant coverage**: proportion of the ground covered by vegetation.	Estimation using cover classes (e.g., Braun-Blanquet); digital photography and analysis; remote sensing.	**Rapid**; can be conducted through rapid visual assessments or photography analysis.	**Easy**; can be compared to reference sites, baseline data, or previous years of data.	Responds rapidly to restoration treatments but can stabilize quickly or decline over time (Matthews, 2015; Raab and Bayley, 2012; Staszak and Armitage, 2013).
	Plant biomass: total mass of vegetation within a given area.	Direct methods (harvesting and weighing of plant biomass).	**Difficult**; direct monitoring is time consuming and potentially disruptive.	**Easy**; can be compared to reference sites, baseline data, or previous years of data.	Rapid response to environmental stressors and restoration treatments (Mollard et al., 2013; Raab and Bayley, 2012); important year-to-year fluctuations in response to abiotic and climatic variability (Anderson et al., 2016; Wilcox et al., 2002; Zedler and Langis, 1991).
		Indirect methods (leaf area index, allometric equations, analysis of remotely sensed data).	**Difficult**; presence of standing litter may affect indirect measurements.	**Variable;** correction factors specific to individual species or functional groups may be needed to account for the effect of litter and water level.	Rapid response to environmental stressors (Mollard et al., 2013; Raab and Bayley, 2012).
Species composition	**Species richness:** number of species in a plot of site.	Species identification.	**Moderate to difficult**; requires experts or well-trained personnel to identify species; risk of overlooking rare species if area monitored is too small.	**Easy to moderate;** can be compared to reference sites, baseline data, or previous years of data.	Some studies have shown increases over time (Gonzalez et al., 2014), but other observed later declines as a result of biological invasions (Yepsen et al., 2014). Sites can recover slowly (Galatowitsch and Van der Valk, 1996; Moreno-Mateos et al., 2012).
	Species evenness: distribution of abundances within an ecological community.	Species identification along with assessment of abundance through stem count or coverage estimation.	**Moderate to difficult;** requires experts or well-trained personnel to identify species; risk of overlooking rare species if area monitored is too small.	**Easy to moderate;** can be compared to reference sites, baseline data, or previous years of data.	Sensitive to biological invasions (Yepsen et al., 2014).
	Species diversity: number of species present and the relative abundance of individuals per species.	Species identification along with assessment of abundance through stem count or coverage estimation.	**Difficult;** sensitive to sampling size and taxonomic knowledge.	**Easy to moderate;** can be compared to reference sites, baseline data, or previous years of data.	Responds to landscape context, management, and local environmental conditions (Bernhardt and Koch, 2003; Galatowitsch and Van der Valk, 1996; Seabloom and van der Valk, 2003), but shows variable response time to restoration treatments (Matthews and Spyreas, 2010; Moreno-Mateos et al., 2012).
	Coefficient of conservatism: expert-based ranking of species from highly tolerant species (score of 0) to specialized species (score of 10).	Species identification.	**Moderate**; little sensitivity to sampling size (complete sampling size not required); requires capacity to accurately identify species.	**Moderate to difficult**; requires pre-established ranking of species; variable results where two jurisdictions are overlapping; biased where new species (not yet ranked) are occurring.	Sensitivity to landscape composition, site conditions, and restoration treatments (Bourdaghs et al., 2006; Matthews, 2015).
	Floristic quality index: calculated by multiplying the mean coefficient of conservatism of a given plot or site by the square root of its species richness.	Species identification.	**Moderate to difficult**; requires complete site inventory or well-established species-area curves; requires capacity to accurately identify species.	**Moderate to difficult**; requires pre-established ranking of species; variable results where two jurisdictions are overlapping; biased where new species (not yet ranked) are occurring.	Significant variation with species composition (e.g., richness, diversity, evenness) and site characteristics (age, soil chemical composition) (DeBerry and Perry, 2015) and landscape context (Lopez and Fennessy, 2002).
	Indicator species analysis: subset of species representative of habitat characteristics, site conditions, or plant community type.	Identification of indicator species, and in most case, measurement of their abundance.	**Easy to moderate**; simplifies species composition assessment by focusing on a small set of species.	**Moderate to difficult**; requires initial sampling across many sites and species to identify the species most responsive to specific conditions or habitat types.	Sensitive to site conditions (Gonzalez et al., 2014).

(*continued on next page*)

Table 1 (*continued*)

Type	Indicator	Methods	Ease of Monitoring	Ease of Analysis/Interpretation	Predictability
Functional	Diversity and abundance of plant functional traits.	Measurement of functional traits (either mean functional trait value in community, trait of dominant species, or functional divergence).	**Difficult**; need in-depth monitoring of plant functional traits but, with a better understanding of correlations in between indicators, a few traits could be monitored to evaluate several ecosystem functions.	**Difficult**; need for statistical analyses, baseline/historical conditions, or existing trait databases.	Can reveal environmental filters and stressors (Galatowitsch and Van der Valk, 1996; Hedberg et al., 2014).
Spatial	**Landscape metrics**: describe the spatial distribution, diversity, and structure of land covers.	Classification and monitoring of patches through field visits or analysis of aerial photographs and remote sensing data.	**Moderate to difficult**; analysis of aerial photo/remote sensing offers a low-cost option but requires ground-truthing and correction for atmospheric effect, water level, and litter; sensitivity to resolution of datasets.	**Moderate**; can be achieved through simple landscape statistics and GIS analyses.	Current limited use in wetland restoration makes it difficult to assess predictability.

outcomes relies on high quality data which can increase the financial and logistical burden on project managers (Bernhardt et al., 2007), particularly under challenging field conditions and limited wetland access.

These issues raise an important question of which ecological approaches and metrics provide the most useful, cost-effective indicators of post-restoration progress in wetlands. A careful selection of indicators and reference points for restoration progress assessment is critical as it can impact project evaluation and likelihood of meeting restoration goals (Ehrenfeld, 2008; Kentula, 2000). Ideally, such indicators should be easy to measure and relate to clear restoration goals (Doren et al., 2009; Matthews et al., 2009). Metrics should have a demonstrated correlation to important ecological properties and show a rapid response to changes in site conditions or mounting ecosystem stressors (Diefenderfer et al., 2011; Noss, 1990). Yet, selecting robust indicators and an adequate monitoring period can be challenging under limited budgets or human resources available for post-restoration monitoring.

To circumvent current limitations to the long-term monitoring of wetland ecosystems, we review scientific literature on approaches to measure restoration progress. Focusing on vegetation-based indicators as the most common attribute in restoration assessments, we synthesize current evidence of their responses to restoration treatments, time, and ecosystem stressors. We also discuss their relationships with ecosystem functions and services typically targeted by restoration efforts (e.g., denitrification, habitat provisioning, soil rebuilding) and list potential limitations to their widespread implementation and interpretation (Table 1). Finally, we present strategies to improve the spatiotemporal scope of monitoring despite limited budgets and research directions to facilitate coherent and informative wetland monitoring efforts.

2. Methods

We researched the Thomson Reuter's Web of Knowledge database and the journal Ecological Restoration to identify previous applications of indicators of post-restoration vegetation dynamics. The following keywords were used to select relevant studies published between 1990 and 2018: restoration or rest*, monitoring, indicator, wetland* and marsh*. We focused on studies conducted in wetland ecosystems and accounted for a variety of wetland types (e.g., freshwater, brackish, tidal, peatland, vernal pools). Within this set of scientific studies, we identified relevant vegetation indicators and noted examples of their applications and responses to site treatment and landscape dynamics. To gain additional insights, we then researched the Web of Knowledge database using each indicator as a keyword (e.g., plant coverage, species richness) along the words restor*and wetland*. With these research criteria, we obtained a total of 99 published studies published between 1990 and 2018 (Fig. 1).

Through our review of literature, we identified four main categories of post-restoration indicators: structural, compositional, functional, and spatial indicators. We reviewed each indicator based on: (1) whether it demonstrated a significant and rapid response to changes in site conditions; (2) whether it correlated with ecosystem properties or functions of interest to project managers; (3) whether it showed a predictable and continuous response to time, and (4) what were potential limits to the widespread implementation of this indicator, if any (Table 1). Among the 99 selected studies, 36 focused on structural indicators of vegetation recovery, 46 studies focused on indicators of species composition, nine on functional indicators, and eight on spatial indicators or satellite-based wetland monitoring. Seventy-six of these studies were based in North America, eight in Europe, five in Asia, one in South America. The remaining nine studies were global in scope, providing either a review of literature or a meta-analysis of published studies. Of the 106 studies analyzed in the meta-analysis conducted by Moreno-Mateos et al. (2012), 16 are also included in this review, while the other studies focused on wetland components not covered by our study (e.g., fauna, soil properties).

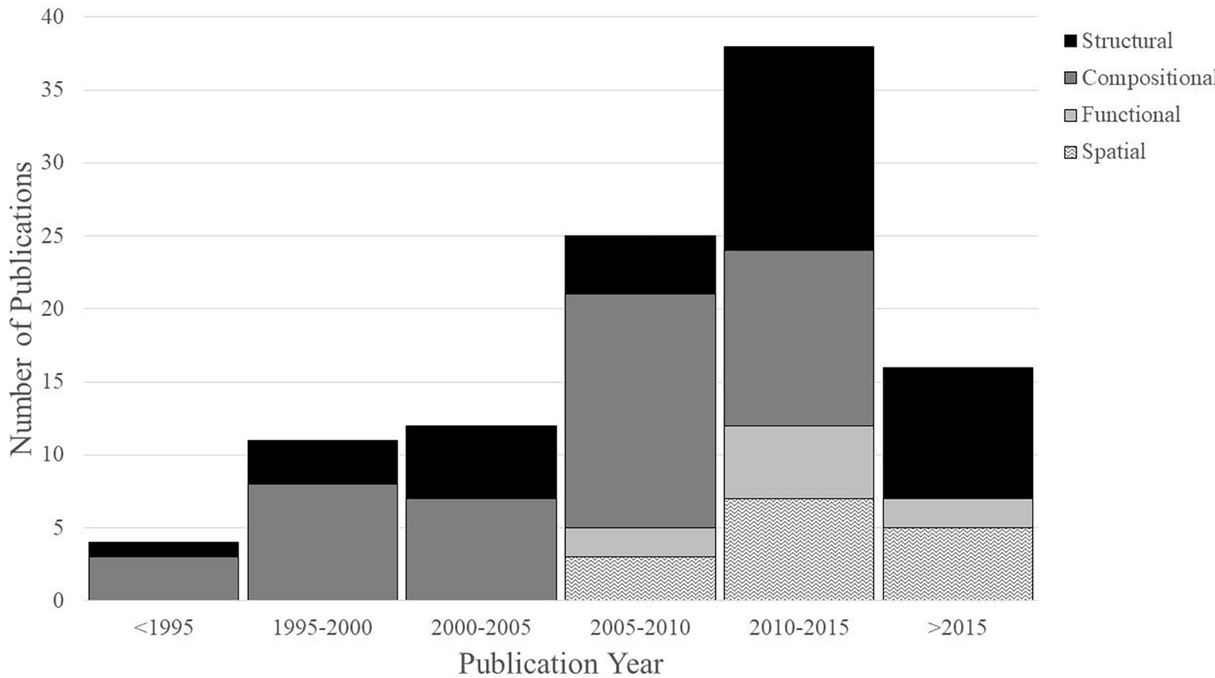

Fig. 1. Count of papers reviewed by year and indicator type.

3. Restoration indicators

3.1. Structural indicators

Structural indicators characterize the amount and distribution of plant biomass throughout the canopy and include above and belowground biomass, leaf area index (LAI), and canopy height among others. A rapid structural response of the plant canopy after restoration helps secure available resources, provide habitat cover, and reduce the potential for colonization by undesirable species (Byun et al., 2013; Funk et al., 2008). Structural indices can also be key indicators of ecosystem services such as productivity, carbon sequestration, and soil rebuilding (Elmore et al., 2015) and are frequently included in the long-term forecasting of these services (e.g., Findlay et al., 2002).

3.1.1. Metrics

Plant coverage is the most commonly used indicator of restoration progress. It refers to the proportion of the ground covered by the foliage of all species or functional types (Wilson, 2011). Plant coverage can be estimated though visual assessments using cover classes (e.g., Braun-Blanquet), the pin-interception method, or the analysis of ground-level digital photography or aerial images (Klemas, 2013a; Wilson, 2011). While rapidly implementable, field-level and visual-based plant coverage estimation can vary significantly from one observer to another (Wilson, 2011). Few strategies can reduce the time needed to measure plant coverage while facilitating monitoring efforts in sites with difficult field access. In small wetlands, observations from the site's edge can provide a reasonable estimation of the plant coverage of dominant functional types (Tavernia et al., 2016). In large wetlands or sites with difficult access, high-resolution aerial or satellite remote sensing images can yield a more precise estimation of total vegetated coverage at broader spatial extents (Mo et al., 2018; Shuman and Ambrose, 2003; Tuxen et al., 2008).

Remote sensing data can also help reduce the subjectivity and variation in cover estimation among different observers (Wilson, 2011) by using spectral indicators of vegetation extent or statistical mapping of plant distributions (Knox et al., 2017; O'Connell and Alber, 2016; Tuxen and Kelly, 2008). Identifying individual species from remote sensing data can, however, be difficult, unless those species are dominant (Shuman and Ambrose, 2003) or presenting very distinct phenological characteristics (Dronova et al., 2015). Some studies have surveyed post-restoration variation in plant height (Craft et al., 2003; Zedler et al., 1999; Zedler and Langis, 1991) as a key predictor of avian occupancy and diversity (Bradbury et al., 2005; Zedler et al., 1999; Zedler and Langis, 1991). Craft et al. (2003) found a strong correlation between the shoot height of *Spartina alterniflora* and its aboveground biomass, while Dronova and Taddeo (2016) found a weak to moderate positive correlation between canopy height and plant area index in wetlands dominated by *Typha* spp., *Schoenoplectus acutus*, or *Sarcocornia pacifica.*

Leaf area index (i.e., total canopy leaf area per unit ground surface, often one-sided), is also used as a proxy of photosynthetically active canopy surface and aboveground biomass, which can account for multiple layers in the canopy (both overstory and understory) (Wilson, 2011). A related metric, plant area index, measures the surface of all above-ground canopy elements (e.g., leaves, stems). LAI is a common vegetation parameter in models of ecosystem functions and biogeochemical cycling, which in terrestrial studies showed positive correlations with plant biomass, aboveground productivity, foliar nutrient content, and other functional properties (Asner et al., 2003; Schulze, 2006). LAI can be measured directly (through harvesting or allometric measurements) or indirectly, via less invasive optical measurements of canopy gap fraction and attenuation of solar radiation or empirical relationships between field LAI and remote sensing data (Dronova and Taddeo, 2016; Elijah et al., 2004; Mo et al., 2018; Nagler et al., 2004). Indirect approaches are generally less time-consuming and more practical for field monitoring (Bréda, 2003; Dronova and Taddeo, 2016; Nagler et al., 2004). However, they typically cannot distinguish between green photosynthetic matter and dead biomass, which may lead to an overestimation of LAI in wetlands with litter accumulation (Dronova and Taddeo, 2016).

Plant biomass can be measured directly by harvesting and subsequent drying and weighing of below and aboveground parts or by using allometric equations based on key plant traits measured in the field (e.g., leaf width, height, culm width; Byrd et al., 2014; Miller et al., 2008). Both direct and indirect methods can be time-demanding, logistically challenging, and disruptive to plant canopies and vulnerable species due to harvesting and trampling. Belowground biomass can be

Table 2
Post-restoration trajectories reported in the scientific literature with their explanatory factor, per indicator and wetland type.

Indicator	Trajectory/Response	Wetland type	Explanatory variable
Native plant coverage	Unimodal (i.e., initial increase followed by decline)	Forested wetland (Berkowitz, 2013); freshwater wetlands (Anderson et al., 2016); compensatory mitigation (Matthews, 2015; Matthews et al., 2009).	Increase in non-native coverage (Matthews, 2015; Matthews et al., 2009); canopy closure (Berkowitz, 2013).
	Asymptotic (i.e., rapid increase in first years, followed by stabilization)	Salt marshes (Bernhardt and Koch, 2003; Erfanzadeh et al., 2009; Staszak and Armitage, 2013).	Reintroduction of natural flooding and periodic grazing (Bernhardt and Koch, 2003); site connectivity promoting seed dispersal (Erfanzadeh et al., 2009).
Plant biomass and canopy architecture	Increase in biomass	Salt marshes (Castillo et al., 2008; Craft et al., 2003, 1999).	Sediment accretion favors plant expansion (Castillo et al., 2008); nutrient availability (Craft et al., 1999).
	Decline in gross ecosystem productivity	Freshwater emergent wetland (Anderson et al., 2016).	Increase in litter accumulation potentially affecting plant growth (Anderson et al., 2016).
	Fluctuations in height	Salt marshes (Zedler et al., 1999; Zedler and Langis, 1991).	Soil properties and nutrient availability (Zedler and Langis, 1991); climatic fluctuations (Zedler et al., 1999).
Species richness	Asymptotic, below restoration targets	Compensatory mitigation (Matthews et al., 2009) (Matthews and Spyreas); global meta-analysis (Meli et al., 2014; Moreno-Mateos et al., 2015); prairie wetlands (Galatowitsch and van der Valk, 1996).	Dominance by non-native species (Matthews and Spyreas, 2010); low diversity and abundance in seedbank (Galatowitsch and Van der Valk, 1996); lack of connectivity and physical constraints to seed dispersal (Galatowitsch and van der Valk, 1996).
	Asymptotic, exceeding reference sites	Compensatory mitigation (Balcombe et al., 2005; Matthews, 2015; Matthews et al., 2009); freshwater wetlands (Yepsen et al., 2014); peatlands (Poulin et al., 2013).	Influence of seed bank (Yepsen et al., 2014); importance of plant type (i.e., certain functional groups recovering faster than others) (Balcombe et al., 2005; Poulin et al., 2013).
	Unimodal (i.e., rapid increase followed by decrease)	Wet meadows (Meyer et al., 2010); prairie wetlands (Aronson and Galatowitsch, 2008); riparian wetland (Lu et al., 2007).	Potential incidence of droughts (Meyer et al., 2010); succession (Lu et al., 2007); increase coverage by non-natives (Aronson and Galatowitsch, 2008).
	Asymptotic (no comparison to reference conditions)	Salt marshes (Bernhardt and Koch, 2003).	Dike opening and introduction of cattle grazing promoting habitat diversity (Bernhardt and Koch, 2003).
Species diversity	Asymptotic (no comparison to reference conditions)	Salt marshes (Bernhardt and Koch, 2003).	Dike opening and introduction of cattle grazing promoting habitat diversity (Bernhardt and Koch, 2003).
	Asymptotic, exceeding reference sites	Compensatory mitigation (Balcombe et al., 2005).	Site connectivity to water bodies favoring seed dispersal (Balcombe et al., 2005).
	Unimodal	Peatlands (d'Astous et al., 2013).	Successional transitions (d'Astous et al., 2013).
Species similarity to reference site	Asymptotic increase	Wet meadows (Meyer et al., 2010); emergent freshwater marshes (Brown, 1999; Galatowitsch, 2006).	Distance to nearest wetland affects species accumulation rate (Galatowitsch, 2006).
	Divergence from reference sites	Prairie wetlands (Galatowitsch and van der Valk, 1996).	Some guilds present in reference site are underrepresented in restored wetlands (Galatowitsch and van der Valk, 1996); less seeds and less seeds diversity in the seed bank (Galatowitsch and van der Valk, 1996); lack of site connectivity (Galatowitsch and van der Valk, 1996).
Coefficient of conservatism	Asymptotic increase	Compensatory mitigation (Matthews et al., 2009).	Local site conditions (Matthews et al., 2009).
	Unimodal	Compensatory mitigation (Matthews et al., 2009).	Increase in non-native species (Matthews et al., 2009).
Floristic quality index	Asymptotic (below restoration target)	Compensatory mitigation (Matthews et al., 2009); freshwater depressional wetlands (Yepsen et al., 2014).	Absence of certain functional groups in seed bank (Yepsen et al., 2014); surrounding land uses acting as barriers to dispersal of certain functional groups (Yepsen et al., 2014).
Functional diversity	Unimodal	Peatlands (d'Astous et al., 2013).	Succession (d'Astous et al., 2013).

particularly time-consuming to measure (O'Connell et al., 2015), which has motivated efforts to model it based on aboveground biomass or leaf nutrient content. Lauck and Benscoter (2014) have developed statistical relationships between belowground biomass and the number of leaves intercepting a randomly placed pin for a more cost-effective quantification. However, the strength of this correlation varies among different plant functional types (Lauck and Benscoter, 2014). Field observations by Tobias and Nyman (2017) suggest that leaf concentrations in manganese and calcium could be used as a less disruptive indicator of belowground biomass in *Spartina patens*. Additional work is needed to assess the strength of this relationship in other wetland species.

3.1.2. Applications of structural indicators

Current literature reports a significant response of structural indices to restoration treatments and changes in local or regional environmental conditions. Previous studies have observed a rapid positive response of plant coverage in the years immediately following restoration (e.g., Bernhardt and Koch, 2003; Craft et al., 2003; Staszak and Armitage, 2013; Table 2), but these trajectories can stabilize quickly (Meyer et al., 2010; Staszak and Armitage, 2013) or decline in subsequent years following canopy closure (e.g., Berkowitz, 2013). Environmental stressors within the site or the adjacent landscape can also trigger a rapid structural response of the plant canopy although the magnitude of this response may vary among functional types. For example, Raab and Bayley (2012) found correlations between non-native species coverage and physico-chemical stress in reclaimed wetlands of Alberta, Canada, suggesting that tracking non-native coverage could help detect ecosystem stressors. Mollard et al. (2013) detected significant relationships between the morphology of six common wetland species and landscape context: individuals in wetlands surrounded by over 50% of cultivated or urban land had smaller, narrower, and less abundant leaves than individuals in wetlands surrounded by over 50% of forest cover. Zedler and Langis (1991) found that poor substrate quality and nitrogen level prevented a restored site in San Diego, Califronia, USA from meeting the structural characteristics (height, biomass) of a reference site. These results demonstrate the sensitivity of structural indicators to abiotic factors and their potential to identify constraints in ecosystem recovery.

Structural indices can reveal a site's capacity to fulfill key environmental goals such as providing faunal habitats, sequestering carbon, improving water quality, and promoting soil accretion (e.g., Bradbury et al., 2005; Craft et al., 2003; Zedler and Langis, 1991). Plant coverage and height heterogeneity have been used as key predictors of bird diversity and abundance as well as invertebrate richness (Bradbury et al., 2005; St. Pierre and Kovalenko, 2014; Zedler et al., 1999). Above and belowground biomass production stimulates soil accretion in wetlands and enhances their capacity to resist sea level rise in coastal regions due to organic substrate accumulation and root-mediated retention of soil particles (Miller et al., 2008; Nyman et al., 2006). Together, plant productivity and rates of sediment accretion control the capacity of wetlands to sequester carbon as live aboveground (shoots, leaves) and belowground (roots) biomass, non-living biomass (litter), and soil carbon pools (Chmura et al., 2003; Kayranli et al., 2010; McLeod et al., 2011; Miller et al., 2008). As such, structural indicators are frequently used in studies modeling carbon sequestration at larger scales (Kollmann et al., 2016).

Structural indicators have also shown a relatively low sensitivity to changes in plant community composition in both experimental studies and field observations. A mesocosm experiment by Hong et al. (2017) showed that increased soil fertility could favor the proliferation of exotic species and negatively impact species richness, while increasing plant biomass. In other studies, plant coverage remained stable over time despite a shift in plant dominance from natives to non-natives following disturbances (Gaertner et al., 2014; Raab and Bayley, 2012). This lack of sensitivity to variation in other vegetation-based indices suggests a limited ability to characterize long-term changes in site functions affected by species richness and composition (Meyer et al., 2010; Staszak and Armitage, 2013).

Regional and site-specific environmental variations in abiotic conditions may affect structural variables and, consequently, their interpretation as indicators of restoration progress. For instance, Wilcox et al. (2002) found in a 3-year monitoring study significant year-to-year biomass fluctuations due to hydrologic conditions and management and could not detect a significant increase in biomass through time. Anderson et al. (2016) observed a decline in the aboveground productivity of a freshwater wetland in California, USA potentially attributed to litter accumulation increasing plant competition for light. Berkowitz (2013) observed a unimodal response of plant cover in forested wetlands of the Mississippi Valley, USA: the total coverage peaked in the 10th post-restoration year, followed by a slow decline until wetlands reached a canopy closure in the 15th year. Light limitations then constrained the abundance and diversity of understory species, triggering a decrease in plant cover. These observations highlight the need for long-term monitoring of both restored and reference sites to distinguish between annual fluctuations and significant trends in canopy structure and biomass accumulation.

3.2. Composition

Species composition refers to the taxonomic identity and abundance of species in a plant assemblage and characterizes the richness, diversity, evenness, and nature of species (Noss, 1990). Indicators of composition are commonly compared to local reference sites to measure restoration progress and evaluate the compliance of mitigation projects (Hill et al., 2013; Van den Bosch and Matthews, 2017). Other projects use documented historical composition as a reference point, which may be difficult in heavily altered sites or landscapes where novel species and abiotic conditions can preclude the recovery of historical assemblages (Seastedt et al., 2008; Suding et al., 2004).

3.2.1. Metrics of composition

Species richness refers to the number of species within a sampling unit while species diversity includes metrics (e.g., Shannon's diversity index) that account for species richness and evenness (i.e., the relative abundance of individuals per species; Hamilton, 2005; Noss, 1990). Calculating species diversity can be more demanding as it requires both identifying species and measuring their abundance as a stem count or visual estimation of their coverage. The accuracy and robustness of diversity, richness, and evenness metrics rely on sample size and area as these affect the probability of observing rare species (Noss, 1990).

Recent studies have shown that restored and reference sites can maintain similar levels of species richness or diversity while remaining compositionally and functionally dissimilar (Jaunatre et al., 2013; Yepsen et al., 2014). For example, sites can sustain high richness and diversity despite increases in the proportion of non-native or terrestrial species (Jaunatre et al., 2013; Roy et al., 2016) with potential consequences for ecosystem functions (Roy et al., 2016). Alternative indicators aim to provide a more qualitative portrayal of species composition by accounting for the specific taxonomic identity of plants. The coefficient of conservatism ranks species from 0 to 10 based on their tolerance to habitat disturbances and fidelity to remnant habitats (Catling, 2013). A score of zero would be allocated to species with a broad tolerance to anthropogenic disturbances while a score of 10 would be allocated to species sensitive to these disturbances and restricted to intact remnant habitats (Catling, 2013; Matthews et al., 2015). To compare sites or plots across space and time, scientists and project managers often report the mean coefficient of conservatism. The ranking is typically established at the state level and has been implemented in most, but not all, American states (Freyman et al., 2016). Other countries including Canada (Wilson et al., 2013), China (Tu et al., 2009), and Italy (Landi and Chiarucci, 2010) have tested or adopted the coefficient. Where the ranking exists, the presence of novel species (i.e., not yet scored) may impact the reliability of the mean coefficient value of a plot or site (Raab and Bayley, 2012) and certain plant groups (e.g., woody species and perennial herbs) tend to be under and overestimated (Bourdaghs et al., 2006). Adjacent jurisdictions can score the same species differently (Bourdaghs et al., 2006), which can affect the consistency of regional assessments. Despite its subjectivity, the coefficient showed a better capacity to distinguish habitats along a gradient of anthropogenic development compared to species richness (Bourdaghs et al., 2006; Matthews et al., 2015). In contrast to species richness, the mean coefficient of conservatism shows little sensitivity to sampling area, implying that it can serve as a sufficient plot-scale indicator for monitoring restoration success (Bourdaghs et al., 2006; Spyreas, 2016). In addition, Chamberlain and Brooks (2016) observed that omitting rarer species had little effect on the mean coefficient value of a sample of 87 wetlands in Pennsylvania, USA.

The floristic quality index (FQI) is derived from the coefficient of conservatism by multiplying the square root of a site's species richness by its mean coefficient of conservatism (Catling, 2013). High FQI scores are allocated to sites with a high abundance of species historically present in a region and are characteristic of undisturbed areas. While the performance of the FQI is restricted by the limitations of the coefficient of conservatism, it shows a significant response to ecosystem stressors (Bourdaghs et al., 2006; Chu and Molano-Flores, 2012; Lopez and Fennessy, 2002; Yepsen et al., 2014) enabling project managers to rapidly identify constraints to restoration progress. Bourdaghs et al. (2006) showed that FQI is more responsive to anthropogenic stress (e.g., modification of hydrological regime, nutrient, chemical loading) than species richness or the mean coefficient of conservatism. Kutcher and Forrester (2018) noted that a modified version of the FQI—the adjusted FQI, which accounts for the proportion of both native and non-native species (Miller and Wardrop, 2006)—improved its capacity to distinguish anthropogenically-disturbed sites from undisturbed ones. However, a small sample size can decrease the index performance as it affects the likelihood of observing rare species associated with a higher coefficient of conservatism (Bourdaghs et al., 2006; Spyreas, 2016). As such, Bourdaghs et al. (2006) suggested assessing FQI over entire sites or relying on well-established species-area curves.

Dissimilarity measures are also employed to compare the composition of restored and reference wetlands without relying on expert ranking. Such statistics include the Sørensen dissimilarity index based on presence/absence data (e.g., Balcombe et al., 2005; Galatowitsch and van der Valk, 1996) and the Bray-Curtis dissimilarity index based on both presence and abundance (e.g., Chapple et al., 2017; Larkin et al., 2014), among several other indices (see Legendre and Legendre, 2012). Alternatively, the community structure integrity index can be used to compare the species composition and abundance of restored and reference wetlands (Jaunatre et al., 2013). This index varies between 0 and 1; a score of 1 is attributed to a restored site in which species show abundances similar or equal to the reference site's. This index can help identify species missing from the restored community and barriers to the proliferation and persistence of specific functional types (Jaunatre et al., 2013). Lastly, indicator species analyses focus on smaller sets of species representative of habitat characteristics (Johnston et al., 2007), community types (Dufrêne and Legendre, 1997), or site conditions (Stapanian et al., 2013). Gonzalez et al. (2014) used a linear discriminant analysis within completed projects to identify sets of species associated with desirable and undesirable conditions, where the latter could indicate divergence from restoration goals. Stapanian et al. (2013) took the inverse approach and showed that the occurrence of a set of three species could be correlated to a high biological integrity across 353 wetlands of Ohio, USA.

3.2.2. Applications of composition metrics

Improving species diversity and recreating historical species assemblages are common goals of restoration projects (Brudvig, 2011). Species richness and diversity have been linked to increases in resource use efficiency and productivity as the enhanced niche partitioning maximizes resource harvesting (Cardinale et al., 2012). However, the strength of the relationships between diversity, resource use efficiency, and productivity can vary from one wetland type to another (Waide et al., 1999). Evidence from mesocosm experiments suggests a positive impact of species richness on ecological resilience, with more diverse plant communities showing a faster biomass recovery following disturbances (Means et al., 2017). However, other authors have argued that functional diversity has a greater impact on resilience than taxonomic diversity alone (Elmqvist et al., 2003; Mori et al., 2013).

While compositional indicators are often considered in wetland restoration monitoring, their relationships with desired ecosystem functions or services are complex and often still unclear in the context of specific applications and projects. An increase in species diversity generally enhances ecosystem processes including productivity (Callaway et al., 2003), although these relationships are nonlinear and tend to saturate at an intermediate number of species (Cardinale et al., 2012). Some studies have observed little to no effect of diversity on functional indicators such as aboveground productivity and soil organic matter content (e.g., Petersen et al., 2015). Others showed a positive impact of species diversity on nitrogen retention (Callaway et al., 2003). The correlation between species diversity and ecosystem services also varies with ecosystem type and management context, and some ecosystem services show a negative response to diversity (Cardinale et al., 2012; Kayranli et al., 2010). Doherty et al. (2011) noted that the relationship between biodiversity and ecosystem function can vary throughout the post-restoration phase. In their study, dominant species had a greater influence on ecosystem functions (e.g., biomass production, canopy layering) in later phases of wetland development than did indicators of species diversity. Zhang et al. (2015) also showed that dominant species had a greater effect on ecosystem functions than species diversity. Few of the reviewed studies assessed the relationship between qualitative indicators of species composition (e.g., indicator species, floristic quality index, mean coefficient of conservatism) and ecosystem functions, but Jessop et al. (2015) observed a positive relationship between mean coefficient of conservatism and avian conservation score, and a negative correlation to nutrient cycling in a sample of wetlands in Illinois. Petersen et al. (2015) found no significant relation between FQI, net primary productivity, and nutrient removal in a controlled experiment. Further testing the relationship between qualitative indices of species composition and ecosystem functions could expand potential applications for the evaluation of restoration progress.

The reviewed literature reveals a substantial variability in the post-restoration trajectories of species composition indicators, demonstrating their sensitivity to site conditions and differences in restoration contexts (Table 2). These trajectories range from asymptotic increases to declines as well as more complex non-linear and unimodal dynamics (Table 2) which are attributed to differences in restoration treatments (e.g., intensity of initial planting), landscape legacies, disturbances, or connectivity to propagule sources. In their meta-analyses of 621 restored wetlands, Moreno-Mateos et al. (2012) noted that species richness generally recovers slowly and can take up to 100 years to meet the richness of reference sites. In contrast, some of the long-term restoration studies reviewed here observe a rapid initial increase in plant richness due to successful species dispersal and seedbank emergence (e.g., Balcombe et al., 2005; Matthews et al., 2009; Meyer et al., 2010). Chaotic or non-linear variations in richness have also been documented at later stages of wetland development due to the proliferation of non-native species facilitated by habitat openings and disturbances caused by restoration actions (Doren et al., 2009; Gaertner et al., 2014; Zedler et al., 1999). Lastly, Matthews and Endress (2010) identified a significant incidence of landscape context on species turnover in a sample of restored wetlands in Illinois, with sites surrounded by a greater proportion of agricultural crops showing a more rapid successional change.

The composition of a plant community can vary significantly throughout post-restoration phases depending on the interactions among species, individual dispersal capacity, environmental fluctuations, and local management activities (Brown, 1999; Laughlin et al., 2017). As a result, it can be difficult for managers to distinguish a specific response to restoration from natural fluctuations in species composition if the post-restoration monitoring period is too short. Moorhead (2013) posited that differences in successional stages between restored and reference sites may limit the capacity for a restored site to meet species composition targets within a short monitoring period. Studies by Chapple et al. (2017) and Wilcox et al. (2002) also revealed important year-to-year fluctuations in the species composition of both restored and reference sites in response to climatic and hydrologic variability. These results demonstrate the importance of long-term monitoring for a robust assessment of plant recovery and understanding of post-restoration dynamics.

3.3. Functional indicators

A growing body of restoration and conservation literature is using functional assessments—the inventory of traits within a plant community—to measure ecosystem functions and evaluate ecosystem resilience (Kollmann et al., 2016; Perring et al., 2015). Recent theoretical models and field evidence suggest that functional traits may offer stronger predictors of ecosystem stability, productivity, and functioning than taxonomic diversity alone (Cadotte et al., 2011; Roscher et al., 2012; Zhang et al., 2015). As such, researchers have argued for a more rigorous incorporation of functional indicators within restoration planning and monitoring (e.g., Kollmann et al., 2016; Perring et al., 2015).

3.3.1. Functional metrics

Metrics of functional diversity (e.g., functional richness, evenness, diversity) describe a plant's allocation of resources throughout the niche space (Mason et al., 2005). Functional diversity can be measured by grouping species into "functional types" sharing similar trait values and then counting the resulting number of groups (Petchey and Gaston, 2006). Functional diversity can also be quantified by combining functional evenness (i.e., degree to which biomass is equally distributed across a trait range) and richness (i.e., the number of values or expressions of a trait) (Mason et al., 2005; Petchey and Gaston, 2006). Other studies have focused on functional divergence, which characterizes the degree of difference among the range of functional traits in a community (Mason et al., 2005; Mouchet et al., 2010).

Post-restoration assessments can focus on one specific functional characteristic (e.g., height, dispersal strategies, life cycle) or characterize plant communities in a multidimensional space. Multi-trait metrics include functional dispersion, which measures the mean distance between traits of an individual species and the mean trait value of the community (Laliberté and Legendre, 2010). Assessments of functional diversity and divergence in a multidimensional space may be sensitive to the number of individual traits considered, which can affect functional indices (Petchey and Gaston, 2006). Mean trait value is commonly used to quantify ecosystem functions and can be determined by identifying the mean trait value across the trait range in a sample (i.e., community-weighted mean) or by focusing on the trait value of dominant species (Mason et al., 2005; Mouchet et al., 2010). Measuring functional traits can be time-consuming and logistically challenging, often requiring both field measurements and lab manipulations. Cornelissen et al. (2003) recommended collecting samples from 10 individuals per plot and species, with the specific number of plots varying with site size and abiotic heterogeneity. To limit the logistical burden associated with functional assessments, previous studies have identified correlations among plant functional traits (McCoy-Sulentic et al., 2017), making it possible to measure several traits from one single leaf sample. For example, specific leaf area (SLA) can correlate with leaf nitrogen, leaf C/N, and leaf height (McCoy-Sulentic et al., 2017). Alternatively, focusing on dominant species can often provide a reliable estimation of the main site-level plant functions (Zhang et al., 2015).

3.3.2. Applications of functional indicators

Both dominant traits and their divergence metrics can signal how environmental conditions are influencing mechanisms of species assembly (response traits) and ecosystem functions (effect traits) (Lavorel, 2013; Lavorel et al., 2011; Suding et al., 2008). Functional traits have shown stronger predictive relationships to ecosystem functions than biomass or species composition alone (Cadotte et al., 2011; Tilman, 1997). As such, monitoring functional diversity, functional divergence, or the mean value of functional traits may help managers track and forecast ecosystem functions within their sites (Zirbel et al., 2017). This strategy, however, relies on well-established correlations between traits and functions, which are still lacking in wetland ecosystems (Perring et al., 2015).

Species assembly and environmental constraints drive the functional composition of plant communities either by promoting certain traits or filtering against them. Monitoring the functional composition of a site or comparing it to a reference point may help identify shortcomings in site design, management (e.g., Zirbel et al., 2017), or environmental constraints to recovery (D'Astous et al., 2013; Hedberg et al., 2014). For example, Hedberg et al. (2014) observed that a lack of topographic heterogeneity impedes functional diversity. Differences in dispersal capacity—particularly in fragmented or isolated sites—may further impact the abundance of certain functional groups and overall functional diversity. As an example, Galatowitsch and van der Valk (1996) observed a slow recovery of *Carex* in a prairie wetland of Iowa, USA, likely due to a low annual seed production and reliance on vegetative reproduction limiting their capacity to colonize newly opened and poorly connected sites.

Detecting changes in the functional traits of dominant species can highlight ecosystem stressors with a potential to jeopardize ecosystem recovery. In particular, McCoy-Sulentic et al. (2017) observed a significant impact of soil dryness on the height of wetland-indicator species. Responses to environmental stressors, however, may differ from one species or functional trait to another (Pivovaroff et al., 2015), which emphasizes the need for more in-depth and long-term analyses of the relationships between functional traits and environmental changes to accurately interpret post-restoration trends.

Lastly, functional indicators may help project managers evaluate the resilience potential of their site to long-term changes, because functional traits influence the range and magnitude of ecosystem functioning. For instance, high functional richness implies that most of the niche space in a community is utilized, which improves a community's resistance to biological invasions as less niche space is available to invaders (Byun et al., 2013; Funk et al., 2008). High functional evenness also ensures a more complete utilization of resources throughout the niche space (Mason et al., 2005). While several papers have presented theoretical relationships between functional diversity, functional divergence, ecosystem stability, and resilience, few studies have tested how specific plant traits can increase the resilience of restored wetlands to ecosystem stressors (e.g., climate change) (Perring et al., 2015). A better understanding of which specific functional indices foster resistance and resilience against environmental perturbations could help managers plan for changing conditions but would first benefit from advanced research in a variety of wetland types and restoration contexts.

Recent population growth models (Loreau and de Mazancourt, 2013, 2008) and empirical evidence from grasslands (Isbell et al., 2013) and forests (Aussenac et al., 2017) suggest that among all components of species diversity (e.g., species richness, evenness, functional diversity), response diversity may be the primary driver of ecosystem stability. Response diversity pertains to the variability of responses to environmental fluctuations shown by a plant community. A more asynchronous response to disturbances ensures the stability (Loreau and de Mazancourt, 2008, 2013) and resilience (Elmqvist et al., 2003; Mori et al., 2013) of ecosystem functions over time despite fluctuations in the abundance of individual species. Species asynchrony can be measured in terms of population growth, survival rates, or phenophases (e.g., Mori et al., 2013; Tredennick et al., 2017). However, specific effects of response diversity on the resilience of restored ecosystems have not yet been tested in wetlands, and no study to date has evaluated response diversity as a metric of wetland restoration success or used it as a restoration target.

3.4. Spatial metrics

Some researchers have advocated for broadening the spatial scope of restoration planning, recognizing that multiple restoration projects are needed to fulfill regional restoration objectives (Kimmerer et al., 2005; Simenstad et al., 2006). Expanding the scale of post-restoration assessments can be financially and logistically challenging, and strategies must be adopted to ensure a uniform and consistent monitoring of sites at a landscape scale. Recent studies have tested spatial metrics (derived from remote sensing or geospatial data) as a tool to monitor site-specific and landscape-scale changes in vegetation structure and composition (Chapple and Dronova, 2017; Kelly et al., 2011; Tuxen et al., 2008), assess ecosystem service provisioning (Almeida et al., 2016), and quantify faunal habitats (Dronova et al., 2016; Moffett et al., 2014). Yet, landscape-scale metrics of spatial distribution, diversity, and structure of land cover types (Turner et al., 1989) remain marginally applied in restoration ecology despite their common use in the planning and monitoring of other ecological habitats (e.g., Botequilha Leitao and Ahern, 2002; Colwell and Lees, 2000; Fahrig et al., 2011).

3.4.1. Metrics

Metrics of landscape structure and configuration describe the spatial or geometric characteristics of land cover patches for a given cover type class or whole landscape composed of multiple land cover types (Botequilha Leitao and Ahern, 2002). Metrics such as perimeter to area ratio, fractal dimension, and shape index characterize the spatial complexity of patches and are used to delineate patch edges and model the exchange of nutrients and species at the wetland edge. Metrics of habitat connectivity are based on the shortest Euclidean or functional distance between patches of the same land cover type and impact species dispersal (Botequilha Leitao and Ahern, 2002). Metrics of patch composition describe the nature and diversity of cover types within a given area. They include dominant patch, patch richness and evenness, as well as the Shannon's and Simpson's diversity indices (Botequilha Leitao and Ahern, 2002). Analyses of patch-level diversity require a detailed prior land cover classification which often relies on high-resolution remote sensing data. While the analysis of landscape metrics can provide a low-cost opportunity to monitor the full extent of sites, such metrics can be sensitive to the minimum mapping unit, i.e., the size of the smallest detectable landscape patch or grain size in the input raster data (Kelly et al., 2011).

3.4.2. Applications of landscape metrics

Landscape metrics are especially useful for the repeated monitoring of sites using remote sensing data to detect changes in the quality, abundance, and connectivity of habitat patches or compare the spatial structure of restored and reference sites (Dale and Beyeler, 2001; Nagendra et al., 2013). For example, Van Meter and Basu (2015) used landscape metrics to compare the structure of restored sites with the historical distribution of wetlands in the Prairie Pothole region, USA. Restored wetlands were bigger, more isolated, and less complex than historical ones, which has important implications for ecosystem processes including species recruitment (dependent on connectivity), denitrification potential (increasing with greater edge irregularity), and habitat heterogeneity (promoting faunal diversity). Temporal dynamics in patch metrics may also be used to quantify and forecast ecosystem service provisioning through time. For example, Almeida et al. (2016) tracked long-term changes in the patch connectivity, shape, and spatial complexity of Portuguese wetlands to quantify their erosion prevention capacity.

Landscape metrics have also been reported as important predictors of animal habitat availability and species dispersal. For example, Moffett et al. (2014) used three landscape metrics—mean core area index, mean shape index, and patch core area—to predict the distribution of the Alameda song sparrow (*Melospiza melodia pusillula*) in wetlands of Northern California, USA. Tuxen and Kelly (2008) used a multi-scale object-based analysis to map suitable habitats for the Salt marsh harvest mouse (*Reithrodontomys raviventris*) and account for its habitat requirements and preferred landscape context (i.e., proximity to elevated patches providing refuge during high tides).

Lastly, inter-annual changes in landscape metrics can be used to characterize post-restoration trajectories and elucidate the incidence of environmental stressors on patterns of vegetation development. For example, Chapple and Dronova (2017) observed a slower rate of lateral vegetation growth during drier years. Allard et al. (2012) used landscape metrics to track the effect of goose populations on the vegetation composition of Canadian wetlands and their capacity to limit shoreline erosion. Tuxen et al. (2011) compared spatial characteristics of different aged wetland sites to characterize their post-restoration trajectories and found higher variability in patch diversity, evenness, and density in younger sites compared to older restored and reference sites. These results illustrate the potential role of early succession and colonization in modulating the spatial characteristics of restored wetlands.

4. Discussion

Our review of literature highlights the advantages and limitations of restoration indicators commonly applied by practitioners and scientists to measure wetland restoration progress (Table 1). We reviewed four groups of indicators describing structural, compositional, functional, and landscape-scale spatial characteristics of wetland vegetation. Structural indices generally offer a rapid response to restoration treatments and ecosystem stressors but can stabilize quickly or show little sensitivity to changes in species composition. Compositional indices are typically compared to reference sites or historical data to measure the success of restoration treatments and can show positive relationships to ecosystem functions, although the strength of these relationships can vary with time, wetland type, and ecosystem function (Cardinale et al., 2012; Doherty et al., 2011; Kayranli et al., 2010). Few studies have addressed the associations between qualitative indicators of species composition and ecosystem functions (but see Jessop et al., 2015 and Petersen et al., 2015), leaving some relationships unaddressed in the context of wetland restoration. Substantial variability due to climatic fluctuations (Chapple et al., 2017), hydrology (Mulhouse and Galatowitsch, 2003; Wilcox et al., 2002), and successional changes (Moorhead, 2013) suggest that a long-term monitoring approach is critical when using species composition as a metric of restoration success. Functional indices offer the strongest correlations to ecosystem functions (Cadotte et al., 2011; Roscher et al., 2012; Zhang et al., 2015) and a promising opportunity to measure ecosystem resilience (Elmqvist et al., 2003; Mori et al., 2013) but can be more logistically demanding to measure at a large spatiotemporal scale. In addition, specific relationships between key ecosystem functions and functional traits have yet to be established for an array of wetland types and landscape contexts. Lastly, spatial patch metrics offer a potential method to quantify ecosystem services and faunal habitats (e.g., Almeida et al., 2016; Stralberg et al., 2010) or infer ecosystem processes including species dispersion and colonization patterns (e.g., Tuxen et al., 2008), but their informative value relies on the quality and resolution of data (Kelly et al., 2011).

4.1. Strategies to alleviate monitoring costs

Differences in response rate and sensitivity to ecosystem processes among the reviewed indicators suggest that a multi-metric approach may be most useful in assessing key objectives of habitat availability or detecting environmental filters to species diversity. A multi-metric approach is also critical in situations where several restoration goals (e.g., species diversity and productivity, faunal diversity, soil rebuilding) must be met by the same project, which may make post-restoration assessments particularly cumbersome. Our review highlights the important effects of sampling size, frequency, and temporal scope on indicators of vegetation recovery, which must clearly be balanced against available resources, monitoring budgets, and site characteristics affecting surveying strategies.

Thus, to reduce monitoring costs, it may be useful to adjust monitoring schedules to a specific spatiotemporal scale at which the response of an indicator is likely to be most representative of the processes of interest. For example, structural indicators often show a rapid response to environmental stress and restoration treatments. As such, it may be beneficial to sample these indicators frequently in the initial post-restoration phase and later use remote sensing images for a cost-effective long-term monitoring. Meanwhile, indicators of species composition tend to recover more slowly (Berkowitz, 2013; Moreno-Mateos et al., 2012) and can show important variability in the later post-restoration phases (Chapple et al., 2017; Matthews et al., 2009; Moorhead, 2013; Mulhouse and Galatowitsch, 2003). As compositional indices are more resource demanding, project managers may choose specific indicators of species composition and functions that can be monitored less frequently but over a longer period. The monitoring schedule could also be adapted to landscape context, considering that it may increase variability in ecological properties and species turnover rates (Matthews and Endress, 2010). Long-term and repeated monitoring is crucial to account for environmental variability and successional differences in site composition (Berkowitz, 2013; Moreno-Mateos et al., 2015; Zedler et al., 1999), particularly for indicators with a slow recovery (Moreno-Mateos et al., 2012).

More research and documentation of existing projects are clearly needed to determine the optimal spatiotemporal scales for measuring indicators of recovery. These scales must be specific to wetland type and restoration context as they shape post-restoration trajectories. One promising approach is to apply statistical modeling to identify the characteristic length scales of wetland properties (Johnson, 2010; Johnson et al., 2017), which examines variations in the abundance or distribution of a single species as proxies of typical variations in the entire community. This method compares the real distribution of a species with the model's capacity to predict the distribution of that species; the scale (spatial or temporal) at which the prediction error stabilizes is then declared as the characteristic length scale or ideal temporal/spatial window at which to monitor a species.

Some particularly promising indicators, including metrics of functional diversity and landscape patch structure, remain under-studied in terms of their responses to restoration treatments and time. In part, this can be explained by the challenge of choosing among multiple possibilities of defining functionally relevant groups, or among a large number of landscape metrics, some of which may be correlated and redundant (Cushman et al., 2008). This issue calls for more in-depth research on such indicators specifically in the context of wetland restoration to enable project managers to (1) identify the most informative and relevant functional traits or landscape metrics to measure, and (2) more accurately interpret temporal fluctuations in functional traits and landscape metrics. As such, it would be useful for the scientific community to study correlations and redundancies among functional traits or landscape metrics to identify the most practical metrics in different wetland types and contexts. For example, Findlay et al. (2002) reported correlations among several wetland ecosystem functions, which suggests that measuring a few informative functional traits may suffice in assessing multiple functions. Field evidence from Zhang et al. (2015) in wetlands and Garnier and Navas (2011) in grasslands suggest that the characteristics of dominant plants have a greater effect on ecosystem properties (e.g., primary productivity, accumulation of organic matter, soil organic content) than the overall richness, evenness, or diversity.

4.2. Remote sensing to improve spatiotemporal scope

Given the high spatial complexity of wetland landscape composition, comprehensive assessments of floristic functions and composition may require repeated monitoring over broad spatial scales (Hickson and Keeler-Wolf, 2007); yet, this task is inherently costly in wetlands due to limited field access and reduced mobility within sites. Remote sensing technology offers a powerful opportunity to embrace wetland complexity at different scales and complement plot-level monitoring efforts at low cost. However, the use of remote sensing by practitioners remains limited. This tendency is paralleled in the literature: only a few published studies to date have used remote sensing data to monitor wetland restoration progress (e.g., Klemas, 2013a,b; Schile et al., 2013; Shuman and Ambrose, 2003), despite the wide range of remote sensing applications in wetland analyses outside the restoration context (e.g., Dronova et al., 2015; Klemas, 2013b; Ozesmi and Bauer, 2002).

Although remote sensing applications bring their own challenges due to potential atmospheric interference (Song et al., 2001), limited spatial resolution (Pettorelli et al., 2005), or difficulties in detecting specific environmental properties or plant functional traits (Andrew et al., 2014), they offer an invaluable potential to link ecological properties assessed in the field with their broader landscape context. Instantaneous capture of whole sites at the same phenological state by remote sensing images provides a critical basis to quantify various metrics of patch composition and structure as indicators of post-restoration habitat and vegetation development (Adam et al., 2010; Dronova et al., 2015; Mo et al., 2015). Spectral vegetation indices can be used to separate bare locations from vegetated areas and track annual vegetation development at the entire site-scale (Tuxen et al., 2008). Quantifying fluctuations in the spatial extent of vegetation can then help land managers upscale plot-level measurements of faunal diversity and abundance (Dronova et al., 2016; Moffett et al., 2014; Stralberg et al., 2010), soil accretion (Kulawardhana et al., 2015), greenhouse gas fluxes (Knox et al., 2017; Kulawardhana et al., 2014), or floristic diversity (Brandt et al., 2015) into site-level estimation of ecosystem functions.

However, the upscaling of plot-level measurements to site-level estimates can sometimes require more costly high resolution datasets (Brandt et al., 2015; Shuman and Ambrose, 2003) while vegetation properties detectable from remote sensing data do not always correlate with the abundance and diversity of other taxa (Dronova et al., 2016; Leyequien et al., 2007). Correlations among spectral characteristics or patch properties (e.g., shape for clonal species, phenology, texture) can thus be leveraged to identify individual species or plant functional groups. This then provides useful base data to evaluate the impact of specific restoration treatments on species distribution and growth, including invasive species (Lishawa et al., 2017; Maheu-Giroux and De Blois, 2005; Rosso et al., 2006) or specific functional groups providing habitats for species of concern (Kelly et al., 2011). Fluctuations in vegetation indices and metrics of mapped vegetated patches can also help detect ecosystem stressors and quantify their impacts on vegetation abundance and composition (Allard et al., 2012). Lastly, recent attempts to estimate species richness from high or medium-high resolution and hyperspectral data using spectral indices, image texture, or a combination of both have shown promising results (Cabezas et al., 2016; Rocchini et al., 2016). Advances in very high resolution sensing, particularly from the novel unmanned aerial vehicle platforms (Anderson and Gaston, 2013), will facilitate monitoring efforts by providing unprecedented spatial detail on wetland surface cover and structure. The important gaps and research needs identified by our review strongly suggest the promise of such technology not only to assist with specific monitoring tasks, but also to make these approaches more coherent and comparable among different efforts.

4.3. Trajectories as a tool for monitoring

Sensitivity of different metrics to the overall time frame and frequency of pre- and post-restoration monitoring strongly advocates for a more rigorous adoption of a "trajectory"-based approach in assessing restoration success and conformance to targets. The wide variety of trajectories of post-restoration indicators identified by this review (Table 2) illustrates the effect of both natural dynamics in restored and reference sites and site-specific conditions on wetland recovery. Site history and landscape dynamics mediate post-restoration dynamics and affect an individual project's likelihood of meeting fixed targets (Hobbs et al., 2014; Jackson and Hobbs, 2009; Simenstad et al., 2006). This suggests that focusing on a specific post-restoration trajectory, rather than a fixed ecosystem state, could provide a more realistic target for wetland restoration. Characterizing sites in term of their trajectories may also provide a framework to identify local and landscape constraints to wetland recovery, or ineffective restoration treatments (Suding, 2011). For example, Matthews (2015) reported a significant effect of landscape composition on post-restoration trajectories in species richness, number of sedge species, mean coefficient of conservatism, and native plant coverage among 54 restored wetlands of Illinois, USA. Matthews et al. (2010) observed a convergence in compensatory mitigation projects five to eleven years after restoration as non-native species progression led to the homogenization of species composition.

Studying the shape of the trajectory itself may provide additional information on the impact of site variations, landscape context, and year effect on the ecological properties of a site. For example, the "year effect", or the environmental conditions during restoration treatments, may influence the trajectory of a restored site and explain why two sites exposed to similar restoration treatments may diverge over time (Suding, 2011). Climatic conditions during the initial years following restoration may influence germination, recruitment, and emergence of seeds and affect the survival of plantings and a site's capacity to meet targets (Stuble et al., 2017). The shape of trajectories may also reflect the influence of local conditions and landscape dynamics on plant community development. For example, indicators related to structure or composition (e.g., plant coverage, species richness, species diversity) can inflate in the post-restoration phase due to the combined effect of initial plantings and seedbank legacy (Yepsen et al., 2014). Other indicators may follow an asymptotic trajectory but take much longer to meet goals or stabilize below targets (Bullock et al., 2011). Unimodal trajectories, which meet or surpass targets rapidly to decline in the following years, have been attributed to a rapid utilization of resources by early successional species (Matthews et al., 2009, Berkowitz, 2013). An increase in the abundance of non-native species can also result in a unimodal response of native species richness, diversity, and coverage (Matthews et al., 2009; Matthews and Spyreas, 2010; Jaunatre et al., 2013).

To establish an "ideal" trajectory as a restoration target, as well as a range of acceptable trajectories, project managers can study the characteristics of several reference sites or completed restoration projects. This would allow accounting for the full array of environmental, geomorphological, and hydrological conditions that can impact site trajectory (Matthews et al., 2009). Matthews and Spyreas (2010) observed a wide range of ecological conditions among reference wetlands used to measure restoration success in Illinois and noted that restored wetlands tend to resemble the mean of these reference sites. Some authors suggested studying old projects or "less desirable" reference sites to establish a lower limit of expectation (Kentula, 2000; Matthews and Spyreas, 2010). This approach would provide more realistic restoration goals in heavily modified landscapes where current conditions may preclude site recovery toward a fixed ideal state.

5. Conclusion

Our review of 99 papers on wetland restoration projects and their monitoring reveals a broad variety of post-restoration indicators available to characterize the structure, composition, function, and landscape-scale configuration of vegetation as measures of project recovery and success. However, not all of them have been utilized to the same degree in previous efforts, and some potentially promising yet logistically demanding metrics, such as trait-based functional diversity indices, remain less well understood as indicators of recovery in the context of wetland restoration. Our review also highlights the need to further document the response of various indicators to specific restoration treatments, ecosystem stressors, and landscape context, as these important determinants of post-restoration performance vary greatly among projects and wetland types. A more profound understanding of such responses would help project managers more accurately interpret variation in site properties as a response to restoration treatments or change in ecosystem conditions. It is also important to continue developing reproducible approaches to identify the optimal spatial and temporal scales for monitoring specific indicators to maximize the representation of key wetland processes and functions in a cost-effective way. Finally, setting restoration and monitoring targets as post-restoration trajectories of recovery rather than fixed-end targets is a promising strategy to incorporate characteristic wetland dynamics into interpretations of recovery and detection of unexpected trajectories.

While we did not limit our selection of peer-review publications to any specific region or country, most of the studies we reviewed have been conducted in North America. As a result, it is possible that the trends identified here predominantly reflect the methods and regulatory framework used in Canada and US. As ecological restoration research continues to expand, it will be useful to develop more geographically comprehensive syntheses in the future.

Acknowledgements

This work was supported by California Sea Grant Delta Science Awards R/SF-71 and R/SF-52. The authors thank Isabel Sablan Johnson and Alexis Andrea Bernal for their help with preliminary literature assessments. We thank two anonymous reviewers for their useful comments that greatly improved the manuscript.

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