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Expert-based measures of human impact to vegetation

Jack Zinnen[1] | **Greg Spyreas**[2] | **László Erdős**[3,4] | **Christian Berg**[5] | **Jeffrey W. Matthews**[1]

[1]Department of Natural Resources and Environmental Sciences, University of Illinois at Urbana-Champaign, Urbana, IL, USA

[2]Illinois Natural History Survey, University of Illinois at Urbana-Champaign, Champaign, IL, USA

[3]MTA Centre for Ecological Research, Institute of Ecology and Botany, Vácrátót, Hungary

[4]MTA-DE Lendület Functional and Restoration Ecology Research Group, Debrecen, Hungary

[5]Institute of Biology, University of Graz, Graz, Austria

Correspondence
Jack Zinnen, Department of Natural Resources and Environmental Sciences, University of Illinois at Urbana-Champaign, 1102 S. Goodwin Ave., Urbana, IL 61801, USA.
Email: jzinnen2@illinois.edu

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Abstract

Background: Human impact has had a profound influence on modern vegetation. Expert-based vegetation indicator systems have been developed to measure and characterize human impacts on vegetation. Floristic Quality Assessment (FQA) is a widely used, primarily North American system for assessing degradation and natural quality through species-based values called coefficients of conservatism. The hemeroby and naturalness indicator value (NIV) systems were independently developed for users to assess human impacts in Europe. Despite the similarities among these indicator systems, there is no mutual recognition among them, and they have developed and operate in relative isolation from one another.

Approach: We review the FQA, hemeroby, and NIV systems to provide a basic summary of the three systems and to evaluate them, highlighting their nearly identical core mechanics and conceptual commonalities. We also compare and contrast the three systems to less integrative ecological indicator systems that can be used to indirectly measure human impact, notably the Ellenberg system.

Findings: We describe how FQA and naturalness indicator values, in particular, could be considered twin systems. Despite these core similarities, users of these systems do not cite each other, potentially overlooking benefits from applying methods and concepts from separate systems. The FQA, hemeroby, and NIV systems have unique weaknesses, strengths, and primary applications compared to other ecological indicator systems.

Conclusions: We conclude by discussing the future role and utility of the three specialized human impact indicator systems for scientists and practitioners. Human impact indicator values may be valuable for use in basic research, but arguably their most important applications are in the practice of conservation, such as monitoring restored ecosystems.

KEYWORDS

coefficient of conservatism, ellenberg indicator values, Europe, floristic quality assessment, grassland utilization values, hemeroby, indicator values, naturalness, North America, social behavior types

1 | INTRODUCTION

Each plant species can be a rich source of information about its habitat (Didukh, 2013). Using plants for indication predates modern ecology, because plants have long been known to reflect latent environmental conditions (Diekmann, 2003). Early ecologists capitalized on this by developing quantitative indicator systems based on plant occurrences (e.g., moisture tolerance, Iversen, 1936). The effects of human influence on historical and modern vegetation attracted the attention of plant ecologists (Sukopp, 1969; Olaczek, 1982), and plant-based bioindicators can also be used to assess human environmental impacts. While anthropogenic impacts are variable in identity, intensity, duration, and scale, in most modern ecosystems their influence is profound. And, although some individual anthropogenic disturbances may be like natural disturbances in their ecological effects, in general the effects of modern human disturbances on the landscape and on plant communities are quite distinct from those of natural disturbance regimes (Kowarik, 1990). Multiple authors in different fields have noted the similarity of their resultant environmental legacies (e.g. Ramensky et al., 1956; Curtis, 1959; Faliński, 1966). One specific result is that plant communities lose species that are sensitive to human disturbance and are replaced by species more tolerant or uniquely suited to anthropogenic change. In response to these observations, early ecologists developed new indicator systems, such as Ramensky's pasture degradation scale (Ramensky et al., 1956) and the cultural impact values of Eklund (1958), to characterize changes wrought by humans based on the plants in an area.

Expert-based species indicator systems are still important tools for ecologists and land managers, fulfilling the same purpose of describing human impacts, as well as being used in novel contexts. For example, Floristic Quality Assessment (FQA), which is primarily a North American vegetation-based indicator system, has many users in ecology and natural areas management, and a scientific literature supporting its use (DeBerry et al., 2015; Matthews et al., 2015). The FQA system is frequently lauded by its users for integrating a broad spectrum of human disturbance impacts via individual species. However, the integrative nature of FQA is also shared with two European indicator systems: the hemeroby and naturalness indicator value systems (Jalas, 1955; Borhidi, 1995). Here, we review these three systems and emphasize their commonalities, revealing what we believe to be fundamental similarities among them. In doing so, we provide a review of the major measures of human impacts to plant communities and vegetation via their plant species.

Our specific objectives were to: (a) provide overviews of FQA, hemeroby, and the naturalness indicator systems; (b) discuss the shared underlying principles, applications, and philosophies of these systems; (c) contextualize and contrast these three systems with respect to other ecological indicator systems alongside which they may be used; and (d) discuss the future of integrative human impact indicators.

2 | INTRODUCTIONS TO THE HUMAN IMPACT INDICATOR SYSTEMS

2.1 | Floristic quality assessment

Floristic Quality Assessment (FQA) is a prominent, primarily North American indicator system created to determine the degree of anthropogenic disturbance to, and conservation value of, natural areas (Table 1). Botanists Gerould Wilhelm and Floyd Swink developed FQA for the Chicago region to identify high-quality natural areas for preservation (Wilhelm, 1977; Swink and Wilhelm, 1979). Coefficients of conservatism (*C*-values) provide the framework for FQA: *C*-values are expert-assigned scores, ranging from 0 to 10, that indicate a plant species' relative intolerance to anthropogenic disturbance based on its fidelity to high-quality natural areas (Swink and Wilhelm, 1979; Taft et al., 1997; Table 1 and Appendix S1). Expert botanists assign low scores to non-conservative species that tolerate or benefit from anthropogenic disturbance and have low fidelity to remnant natural areas. Conversely, a higher score indicates a more conservative species with less tolerance to anthropogenic disturbance. Non-native species are either ignored or are all assigned *C*-values of 0 because such species have no historical relationship with native plant communities and are assumed to lower an area's conservation value (Taft et al., 1997).

Individual *C*-values are inputs for the two basic FQA metrics, mean *C*-value (mean *C*) and the Floristic Quality Index, which are calculated as:

$$\overline{C} = (\sum C_t)/N, \text{ and}$$

$$\text{Floristic Quality Index} = \overline{C} \times \sqrt{}N,$$

where C_i represents the assigned *C*-value of each *i*th species and *N* is the observed species richness of the plants in the surveyed area. Multiplying the square root of the species richness to the mean *C* was an attempt by Swink and Wilhelm (1979) to yield higher valuation for more diverse natural areas.

C-values are typically assigned to all individual species within a defined area, typically a political state or ecoregion. Assignment occurs by averaging the score assignments of a group of expert field botanists working independently (e.g. Cohen et al., 2004) or through unanimous panel decisions (e.g. Taft et al., 1997). Empirical testing of individual *C*-values is generally rare after their assignment, but even if some individual species are less accurate (Matthews et al., 2015), they have repeatedly been shown to work well overall as part of community-level FQA metrics (e.g. Taft et al., 2006; Mack, 2007).

FQA metrics capture integrative information about anthropogenic disturbance and habitat quality. Some authors suggest that the metrics reflect the intensity of accrued disturbances within a given plant community in the modern era (e.g. Wilhelm and Rericha, 2017). These impacts include various forms of habitat destruction, such as

TABLE 1 A generalized description of the three integrative indicator systems FQA, hemeroby, and NIVs

System	Primary concern	Original developer(s)	Origin and year of development	Basic scale	Values for individual species?	Method of scale assignment	Direct linkage to other concepts?
Floristic Quality Assessment	Natural quality based on conservatism, i.e. species' and communities' responses to human disturbance	Gerould Wilhelm and Floyd Swink	Midwestern United States (1977 & 1979)	C-values: 0–10	Yes	Expert opinion	No
Hemeroby	Degree of human influence on ecosystems	Jaakko Jalas	Finland (1955)	Hemeroby categories or values: H1–H5 or H1–H9 commonly	Sometimes	Expert opinion, averaging species distributions	Has been linked to the naturalness concept (Winter, 2012) and sometimes potential natural vegetation
Naturalness Indicator Values	Natural quality based on "social behaviors," i.e. the role of a species in a community and its relative response to human disturbance	Attila Borhidi	Hungary (1995)	NIVs: −3–6	Yes	Expert opinion	Linked closely to Simon (1988) and Grime's (1979) C–S–R, and numerical assignment like other EIVs

through agricultural and urban development, but also fundamental shifts in ecosystem processes, such as altered fire, nutrient, or hydrologic regimes (Taft et al., 1997; Herman et al., 2001; Freyman et al., 2016). Thus, users should be aware that conjectures about the state of North American ecosystems prior to European colonization underlie the FQA system.

There are well-established applications of the FQA system. FQA is typically used in management and conservation decision-making contexts. FQA users have extensively studied its utility to reflect conservation value, characterize degree of habitat degradation (Lopez and Fennessy, 2002; LaPaix et al., 2009), and assess management effectiveness (Maginel et al., 2016). Often, managers calculate FQA metrics in conjunction with other measures (Taft et al., 2006), such as invasive species abundance or species richness (Taddeo and Dronova, 2018), or apply them in multimetric indices (Mack, 2007; Taddeo and Dronova, 2018). FQA has also been used to compare ecological restorations with target or reference plant communities (Mushet et al., 2002; Matthews et al., 2009; Taddeo and Dronova, 2018). Some institutions even require FQA as a performance standard in legally mandated restoration, namely wetland offsetting (Matthews and Endress, 2008). As a consequence, FQA is disproportionately used and studied in wetlands compared to other ecosystems (see DeBerry et al., 2015).

Ecologists have employed the FQA system, and particularly *C*-values, in basic research. For example, some authors have used *C*-values to understand community assembly (Matthews, 2004; Spyreas and Matthews, 2006). Others have applied FQA to examine succession (Spyreas et al., 2012; Koziol and Bever, 2017). Despite these examples, the value of FQA for research may be limited compared to other ecological indicator values systems used in Europe, such as Ellenberg values (see Section 3.4 on *Considering human disturbance indicators alongside ecological indicator values* below).

FQA has a geographically broad user base. Originally developed for the Chicago, Illinois, USA region, FQA has expanded through the assignment of *C*-values to the floras of additional regions (usually at the state or provincial level) and now covers the majority of the continental United States and parts of Canada (Figure 1; Spyreas, 2019). The use of FQA has also expanded to Italy (Landi and Chiarucci, 2010), the Middle East (Mirazadi et al., 2017), China (Hou et al., 2018), and Africa (Alemu et al., 2018), though these FQA systems are generally less complete and less studied compared to those in North America.

2.2 | Hemeroby

The hemeroby system is primarily European; it is used to describe the degree of current and past human *influence* on vegetation or landscapes (Table 1). Hemeroby arguably doubles as both an indicator system and a conceptualization of nature because it can be related to an area's naturalness (Winter, 2012). Finnish botanist Jaakko Jalas (1955) created the hemeroby system and modeled it as a scale of human impact after the early, pre-existing scale of moisture in Iversen's (1936)

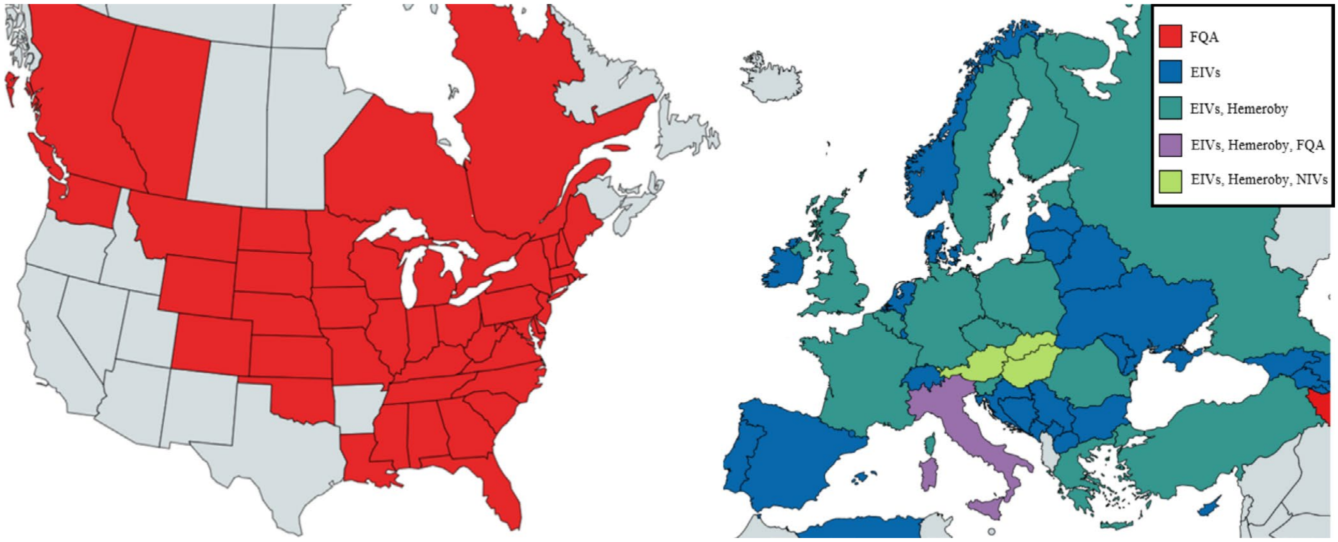

FIGURE 1 The geographic extent of the regions where the four systems have been applied in Europe and North America. During our bibliographic analysis, we recorded the locations of the studies applying the systems at a national level for Europe, or state/provincial level in North America (Spyreas, 2019). We used a liberal approach to counting where a system was applied, counting all countries when a study aggregated relevés from multiple countries and applied indicator values. Thus, the color shading does not mean the systems are frequently applied in the area, but rather it has been applied in that region in some context. Here, FQA, NIVs, and EIVs refer to Floristic Quality Assessment, naturalness indicator values, and Ellenberg indicator values, respectively (Appendices S1 and S2)

system. German authors later promoted hemeroby, increasing awareness and use of the system (Sukopp, 1969; Kowarik, 1990).

Hemeroby metrics are obtained by assigning a sampled plant community to hemeroby degrees (i.e. hemeroby categories). Jalas (1955) originally described communities in four categories, but later authors expanded the number (Kowarik, 1990). The number of applied categories is more variable than in the other systems: a total of 5–10 categories is available to assign to a community; these categories demonstrate the amount of anthropogenic influence on that community (Appendix S1). Categories are assigned using various methods of expert judgment (Kim et al., 2002). Some authors have deconstructed and described the components of human influence in hemeroby categories (Kim et al., 2002; Appendix S1), but more often the categories are less clearly defined. Sites with higher hemeroby values have distinct, degraded communities of non-native and weedy (i.e. ruderal) species (Breg Valjavec et al., 2018b). The vegetation of sites in lower hemeroby categories is characteristic of late-successional communities, some of which can be indicative of past European landscapes (Sukopp, 1969; cf. Kim et al., 2002; Battisti et al., 2016). Note that some hemeroby users suggest the system is not intended to be historically oriented, because hemeroby can assess the human-induced deviation from a site's present “potential natural vegetation,” i.e. the hypothetical end state of the vegetation if all human influence were removed, rather than reflecting deviation from a historical ecological state per se (Kowarik, 2014). Nonetheless, in practice this difference is subtle because hemeroby will still characterize deviation from historical ecological conditions in many communities where the degradation from human impacts is reversible (Walz and Stein, 2014).

There are also methods to assign numeric hemeroby values to individual species, which is more analogous to the other two human impact indicators. Specifically, Kowarik (1988) assigned hemeroby indicator values to individual plant species in Berlin to be used like Ellenberg indicator values. These were generated by subjective assignment of hemeroby classes to vegetation relevés, followed by assigning each species a hemeroby value reflecting its ecological optimum relative to human influence (Kowarik, 1990). Similarly, Klotz and Kühn (2002) have published multiple hemeroby values (Appendix S1) for individual species to account for their variable inhabited conditions, though these scales can be converted to single numeric values for individual species (see Berg et al., 2016). In another case, Fanelli and De Villis (2004) assigned hemeroby values to a species by averaging the community-level hemeroby classes that they inhabited. Thus, subjective expert assignment is used to assign community-level hemeroby scales, and species-based values can be subsequently created from such data.

Hemeroby has been applied in scattered contexts. A common use of hemeroby is to describe species' tendencies to urban habitats (Sukopp, 1969; Hill et al., 2002). For example, urban ecologists have applied hemeroby categories to the vegetation in large European cities such as Rome, Berlin, and Brussels to assess human impact (Fanelli et al., 2006; Godefroid et al., 2007; Sukopp, 2008). Others have used the hemeroby system to compare species richness patterns across cultural ecosystems (e.g. castles; Celka, 2011) and among bryophyte communities (Zechmeister and Moser, 2001). Species-level hemeroby indicator values have also been used to infer environmental change (Berg et al., 2016), and community-level hemeroby scales can be useful for mapping relative human impacts over large areas. In other cases, uses for hemeroby deviate from plant ecology or natural areas management applications by focusing on landscape ecology and cartography, such as mapping land-use or disturbance mosaics on a regional or national scale (e.g. Walz and Stein, 2014).

While the hemeroby system has been applied in many European countries (Figure 1) and occasionally beyond (e.g. Kim et al., 2002), its use is relatively uncommon (Battisti et al., 2016) compared to Ellenberg indicator values and FQA. Perhaps this is because of the lack of clarity or consistency in some of the system's underlying concepts, definitions, and standard practices (Hill et al., 2002).

2.3 | Naturalness indicator values and social behaviors

Attila Borhidi developed the naturalness indicator value (NIV) system for the Hungarian flora to assess the natural quality of communities and the relative conservation value of plant species (Borhidi, 1995; Table 1). Borhidi assigned each species from the Hungarian flora an NIV ranging from −3 to 6. Negative scores were assigned to plant species that indicate ecosystem disturbance and anthropogenic degradation (Török and Szitár, 2010), including native ruderal and non-native species. Non-native species were scored depending on their invasive tendencies (Borhidi, 1995); non-native waifs and garden escapes were scored higher (−1, e.g. *Oenothera* spp., *Ginkgo biloba*) than noxious invasives (−3, e.g. *Robinia pseudoacacia*). Higher values were assigned to competitor and specialized, stress-tolerating species that have a fidelity to ecosystems with desirable, more "natural" conditions. NIVs could thus be considered conceptually complementary to hemeroby, but surprisingly this link has been hitherto unrecognized. Borhidi (1995) did not specify how NIVs were to be used in relevés or communities, but later users averaged species' NIV scores at the community level like Ellenberg (weighted mean) or FQA (arithmetic mean) users.

Like the other two systems, NIVs were assigned by expert judgment. However, the scoring foundation also integrated the frameworks of two other systems, those of Simon (1988) and Grime (1979). Simon (1988) assigned relative conservation values to plant species for the identification of protection-worthy natural areas. Grime (1979) considered how plants allocate resources and acquire energy when growth is limited by disturbance and stress, and posited three primary strategies: competitors (C), stress tolerators (S), and ruderals (R). Borhidi (1995) argued that each plant species has a particular role in a community based on its position in the C–S–R framework. These roles were termed "social behavior types." A species' social behavior category reflects its community's successional state and disturbance level, and thus its conservation value. Each social behavior category was associated with a particular NIV score (Appendix S1).

The application of Borhidi's (1995) NIV system is not particularly common compared to the other systems. It is more geographically limited than FQA or hemeroby, and has been applied only in Austria, Hungary, and Slovakia (Figure 1). Few authors use NIVs; most papers referencing Borhidi (1995) extract the regionalized Ellenberg values for the Hungarian flora from his work, or merely cite the work to mention "social behaviors" to identify weedy species (see Section 3.4 on *ecological indicator values* below). Nonetheless, NIVs have been used in management contexts to assess the relative degradation of habitats (Erdős et al., 2017), habitat quality (Sengl et al., 2017; Erdős et al., 2018), and to indicate prior management history (Sengl et al., 2016).

3 | DISCUSSION

3.1 | Conceptual similarities

FQA, hemeroby, and NIVs could be considered conceptual cousins. At their core is a set of subjectively assigned numeric ranks (Table 1, Appendix S1). The developers of the systems assigned these values based on their intimate and localized knowledge of their respective floras (Swink and Wilhelm, 1979; Kowarik, 1988; Borhidi, 1995; Klotz and Kühn, 2002). The systems act as a nexus between autecology and synecology, with species-specific values used to infer environmental conditions at the community level.

Of these three human impact indicator systems, FQA and NIVs are the most similar. Although the NIV system is relatively inconspicuous as a system compared to FQA, the focus of both systems on human disturbance gradients is virtually identical. The conceptual axis of human disturbance tolerance quantifies a species' conservation value and fidelity to high-quality natural areas (Taft et al., 1997; Erdős et al., 2018). This leads to similar values assigned to ecologically similar species. Low *C*-values and NIVs are usually tied to early successional species (ruderals), non-native species, and generalists (Taft et al., 2006; Török and Szitár, 2010), whereas high values are associated with late-successional and habitat-specialist species (Andreas and Lichvar, 1995; Borhidi, 1995; Koziol and Bever, 2017; but see Spyreas, 2019). In the original forms of both systems, indicator values were even assigned that exceeded the normal scoring range if a species was opined as rare or exceptional (Swink and Wilhelm, 1979; Borhidi, 1995), though later authors abandoned this.

The earliest versions of the FQA and NIV systems also link to traditional ideas of community ecology. The earliest version of FQA posited that the most conservative species reflect mature, stable, climax communities, whereas introduced species "perverted" the "time-honored" development and existence of natural areas (Swink and Wilhelm, 1979). Similarly, Borhidi (1995) commented that the most natural social behaviors are indicative of stability, or "steno-ecological" conditions, whereas the least natural social behaviors act as "obstacles" to natural succession and promote "deficient, alien ecosystems." Moreover, social behaviors and NIVs were meant to capture the wholeness of the community "linkage." Thus, Swink and Wilhelm (1979) and Borhidi (1995) were rooted in Clementsian ecological thinking of community holism and directional climax, although later authors moved away from this thinking (e.g. Taft *et al*, 1997; Spyreas, 2019).

3.2 | Differences among focal systems

These three systems have important differences. First, the basic rule sets of hemeroby are different from those of FQA and NIVs.

This also leads to a difference among the typical applications of the systems. In the FQA and NIV systems, each species is assigned an individual value. In contrast, Klotz and Kühn (2002) suggested that the variable responses of a species to human impacts make it difficult to assign a single numerical indicator value, and therefore published multiple hemeroby values for each species Appendix S1). Furthermore, many hemeroby users do not apply species-based hemeroby values, but rather assign a score to a community by its landscape context and land use (e.g. Celka, 2011). Many studies move beyond the system's original focus on plant ecology and intergrade into large-scale cartography (e.g. Walz and Stein, 2014). Hemeroby could thus be considered a more synecological system with a plastic rule set compared to FQA and NIVs. In contrast to hemeroby, the FQA literature emphasizes management-related questions, as FQA was developed as a practical tool for natural area assessment and management (Andreas and Lichvar, 1995). This is also true for the NIV system, for which recent users have focused on similar management and restoration-based research (e.g. Sengl et al., 2016; Erdős et al., 2017).

Second, FQA and NIVs have an underlying value-laden philosophy, whereas hemeroby users take a neutral view of vegetation change due to modern human impacts. The hemeroby system measures the strength of human impact on vegetation without reference to the perceived natural value of the resultant vegetation, whereas *C*-values and NIVs are fundamentally used to convey subjective judgments about natural quality or conservation value. These philosophical differences could be considered a product of the historical, regional, and ecological contexts of the FQA and NIV creators. FQA developed in the North American Midwest with a primary focus on conservation value, where European colonization had dramatic, devastating, and easily discernible ecological impacts on sensitive species and plant communities over a relatively short period (Curtis, 1959; Andreas and Lichvar, 1995). Developing the FQA system to identify the "small vignettes of [native] diversity" remaining in the rapidly developing Chicago region was imperative for their preservation (Swink and Wilhelm, 1979). Similarly, Borhidi (1995) created his system to identify and protect worthwhile natural areas, specifically by comparing the conservation value of Landscape Protection Districts in Hungary. Borhidi (1995) noted that species with great conservation value were not protected, and some protected species did not necessarily have local ecological importance. This directly parallels FQA's origin and usage, whereby species or communities that are not rare enough to be legally protected could be evaluated for their conservation value (Herman et al., 2001). In contrast, hemeroby was created for description rather than valuation (Jalas, 1955), though users can project evaluative conclusions from hemeroby metrics if they so choose (e.g. Testi et al., 2012). A final example of the philosophical difference between systems can be found in the clear disdain for non-native species in FQA and NIVs (Swink and Wilhelm, 1979; Borhidi, 1995), in contrast to the indifference to species origin in the hemeroby system (Jalas, 1955).

3.3 | Mutual awareness among the users of different systems: A call for intellectual exchange

Human impact indicator systems have independently developed multiple times, clearly demonstrating their utility to scientists and practitioners. Because these systems are similar and still relatively undeveloped, information exchange among them could lead to methodological or conceptual improvements. However, their users seem largely unaware of the other, analogous indicator systems. Spyreas (2014) briefly discussed similarities between hemeroby and FQA, but we are aware of no other acknowledgment in the literature of the similarity of these systems. More authors have discussed or cited FQA and Ellenberg values together (e.g. LaPaix et al., 2009; Landi and Chiarucci, 2010; Spyreas, 2014), though this is still uncommon. A bibliographic coupling analysis of the three focal systems (Figure 2; Van Eck and Waltman, 2019; details in Appendix S1), as well as an analysis that included the related ecological indicator value systems (discussed below; Appendix S1), confirms continental isolation. Some independence among these indicator systems is to be expected due to their geography, background literatures, and the unequal development and size of their user bases. However, complete isolation comes at the expense of sharing lessons among users of these systems.

One area where FQA and NIVs might be criticized and improved is in their reliance on an oversimplified, arguably romantic vision of pristine natural vs. human-degraded habitats. Many of the undisturbed or high-quality plant communities described by Swink and Wilhelm (1979) and Borhidi (1995) relied on traditional anthropogenic disturbances (see Battisti et al., 2016). In other words, these communities were clearly not free of human influence and might be considered "mesohemerobic." Examples include low-intensity agricultural practices in Europe before the Industrial Revolution (Olaczek, 1982) and fire regimes created by indigenous peoples in North America (Cronon, 2003). These nuances and distinctions were not characterized in detail during the initial development of FQA (cf. Swink and Wilhelm, 1979; Swink and Wilhelm, 1994) and NIVs, and remain undeveloped in the subsequent literature (Spyreas, 2019). Landi and Chiarucci (2010) discussed the underrecognition of human impacts on ecosystems as a major shortcoming of FQA when applied to Europe, where such impacts occurred with an intensity that has increased gradually over a much longer period of time compared to North America. The language and concepts describing FQA and NIVs should potentially address this weakness, so that users understand that the systems characterize *modern* (i.e., post-industrial) *anthropogenic disturbance* or the *total degree of human impact on the vegetation*, rather than a natural vs. human-disturbed dichotomy. Users might also interpret the conceptual foundation of the FQA and NIV systems as a reaction to the landscape-scale shift toward euhemeroby.

These systems offer users new literature for borrowing methods and ideas, especially for the underdeveloped hemeroby and NIV literature. A potential lesson from North America is that most regional *C*-value assignments consult multiple experts, whereas NIVs rely

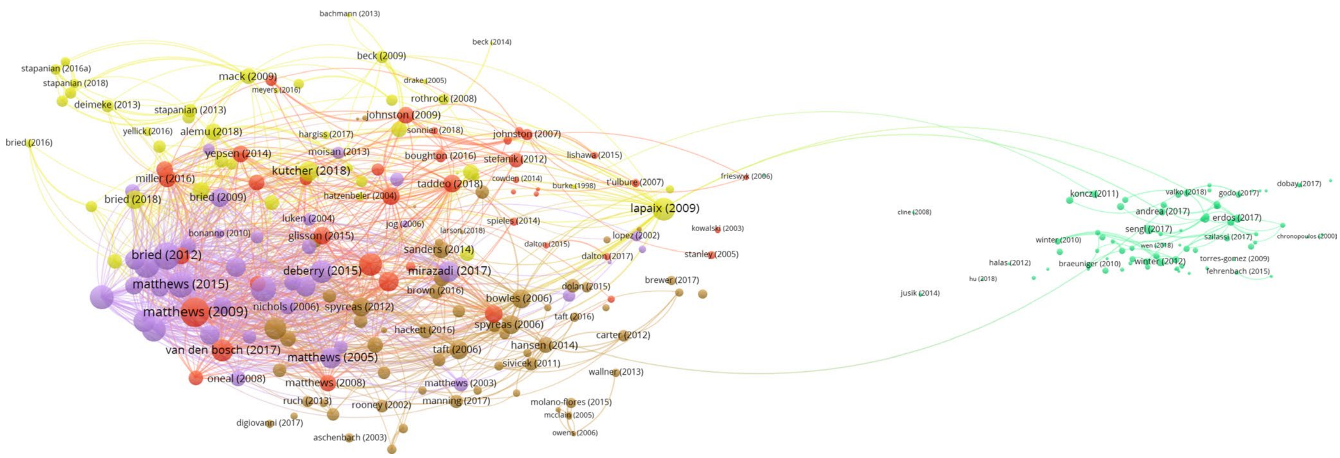

FIGURE 2 A bibliographic coupling analysis of 262 source documents of the Floristic Quality Assessment (FQA) (n = 183), hemeroby (n = 52), and social behavior/NIV (naturalness indicator value) (n = 27) literatures (Appendices S1 and S2). VOSviewer identified fives clusters. Cluster 1 (blue green) was associated with the hemeroby and social behavior/NIV literatures; the four other clusters were exclusively FQA documents. Clusters within FQA were partitioned to some subtopics in the literature. Cluster 2 (brown) was associated with habitat management and restoration documents, particularly in tallgrass prairie ecosystems. Clusters 3 (red) and 4 (yellow) were associated with papers applying FQA in wetland ecology, such as wetland mitigation and riparian assessment. Cluster 5 (purple) was a mix of wetland management and applied ecology, as well as papers establishing regionalized *C*-values. VOSviewer does not label some nodes in order to prevent overlapping labels (Van Eck and Waltman, 2019)

TABLE 2 A generalized description of three EIV systems

System	Primary concern	Select authorities	Environmental axes measured
Ellenberg indicator values	Quantifying species' autecological requirements for particular realized niche dimensions	Ellenberg et al. (1991); Landolt et al. (2010)	Continentality (*Kontinentalitätszahl*):1–9 Light (*Lichtzahl*): 1–9 Moisture (*Feuchtezahl*): 1–9, or 1–12 Nutrient (*Nährstoffzahl*): 1–9 Salinity (*Salzzahl*): 1–9 Soil pH (*Reaktionszahl*): 1–9 Temperature (*Temperaturzahl*): 1–9
Grassland utilization indicator values	Quantifying relative tolerances of delineated disturbance axes common in European grasslands	Briemle and Ellenberg (1994); Briemle et al. (2002)	Grazing (*Weideverträglichkeitszahl*): 1–9 Mowing (*Mahdverträglichkeitszahl*): 1–9 Trampling (*Trittverträglichkeitszahl*): 1–9
Urbanity values	Characterizing species' relative fidelities to urban habitats, and thus presumably their tolerance to urban disturbances	Wittig et al. (1985); Klotz and Kühn (2002)	Urbanity: 1–5 (urbanophobic–urbanophilic)

on Borhidi's (1995) judgment alone. As with the geographic spread of FQA (Figure 1), improved regionalization of NIVs and hemeroby values beyond Hungary and Germany, respectively, could provide greater ecological resolution and expand the user bases. New research efforts to validate, or increase the precision of, hemeroby values or NIVs could use methods applied to FQA metrics (e.g. Lopez and Fennessy, 2002; Mack, 2007; Matthews et al., 2015). Borhidi's (1995) practice of differentiating and standardizing the NIV assignments of non-native species is arguably more informative regarding species' ecology compared to the default assignment of 0 scores to every non-native species in the FQA system (Spyreas, 2014; Matthews et al., 2015). Finally, many hemeroby and NIV users have access to additional ecological indicator values (see Section 3.4 on *ecological indicator values* below), and often simultaneously employ multiple systems in their studies (e.g. Laanisto et al., 2015; Berg et al., 2016; Sengl et al., 2016; Appendix S2). Because North American ecologists are limited to *C*-values and moisture indicator values (an analog of Ellenberg moisture values; Lichvar, 2013), creating more types of ecological indicator values in North America would be a valuable addition.

3.4 | Considering human disturbance indicators alongside ecological indicator values

Environmental indicators that rely on expert-assigned species values is a characteristic of several other systems (Table 2). These differ from the indicators we thus far discussed in that they arrange plant species along single, well-defined environmental gradients. We term these ecological indicator values (EIVs), because they characterize any one of several discrete, ecological parameters compared to the more integrated and complex human disturbance gradient that our focal systems measure. Nonetheless, these systems are sometimes used to characterize human impacts.

The most pre-eminent among the EIV systems is the Ellenberg indicator system. Ellenberg indicator values are specific realized niche optima for a particular and well-defined environmental axis (Ellenberg et al., 1991; Diekmann, 2003). Heinz Ellenberg produced a nine-step scale for EIVs (pointer or indicator values) for the flora of west-central Europe (Ellenberg et al., 1991), consisting of seven indicator categories to reflect different environmental conditions for each species (Table 2; Appendix S1). The ability to easily validate and calibrate Ellenberg values, along with their flexibility and vast diversity of applications (Diekmann, 2003), have made Ellenberg values ubiquitous in European ecology, dwarfing the use of FQA, hemeroby, and NIVs (Figures 1–3, Appendix S1). Because there are up to seven Ellenberg axes for use, one or more values can be applied to reflect anthropogenic influence on the vegetation, such as atmospheric nitrogen deposition (Dupré et al., 2010), fertilizer application (Chytrý et al., 2009), and selective tree cutting (Decocq et al., 2004).

EIVs can also specifically characterize response to management or disturbance. Gottfried Briemle and Heinz Ellenberg (Briemle and Ellenberg, 1994) developed grassland utilization indicator values (hereafter referred to as Briemle EIVs for simplicity). Briemle EIVs employ the scoring mechanics of the Ellenberg system to reflect a species' relative tolerance to univariate axes of anthropogenic disturbance, specifically grazing, mowing, and trampling (Briemle and Ellenberg, 1994). However, Briemle EIVs reflect species-specific responses to *physical* disturbances, or the destruction of vegetation and the soil medium (sensu Grime, 2006), which are not directly characterized by Ellenberg values. Like other ecological indicator values, the authors based their nine-point scales on expert knowledge (Table 2).

The urbanity system is another analog EIV system that is close to FQA, hemeroby, and NIVs. The urbanity system was first developed by Wittig et al. (1985), who characterized the urban tendencies of species in Berlin. In the strictest sense, urbanity indicators reflect species' tolerance to and distribution among urban habitats, though they could be more broadly interpreted as their relative sensitivities to urban disturbance. Although urbanity indicator values were published together with hemeroby values by Klotz and Kühn (2002), urbanity values are rarely used compared to hemeroby values (Appendix S2). Perhaps urbanity indicators are rarely used because they do not directly assess human impacts and only concern urban contexts.

Grime's C–S–R theory and system (1979) are, in some ways, similar to these indicator systems. The C–S–R system provides a functional signature of plant communities regarding resistance, elasticity, or degradation in response to natural or anthropogenic disturbance and recolonization. Grime (2006) described how different types of human impacts, such as eutrophication or dereliction, can lead to predictable functional shifts in the three strategy types. C–S–R strategies do not reflect species' distribution relative to a specific ecological gradient like the ecological indicator value systems described here. Nonetheless, C–S–R strategies can be converted to simple numeric scores and used in similar ways to any of these systems (e.g. Hill et al., 2002; Hunt et al., 2004).

3.5 | Comparing integrative values to ecological indicator values

The three EIV systems share a core foundation of subjective, expert-based numerical assignment with the human impact indicator systems. Thus, FQA, hemeroby, and NIV indicator systems unsurprisingly share many of the characteristics of other EIVs, and the strengths and weaknesses of the FQA and Ellenberg systems are similar and well characterized (Thompson et al., 1993; Diekmann, 2003; Spyreas, 2019). Nonetheless, the human impact indicators are a separate, unique suite of systems which simultaneously consider multiple dimensions of human impacts and project them onto a single numerical axis (Table 1 and Appendix S1).

All of these systems can characterize different types and dimensions of human disturbances. Disturbance, while often considered only in the context of physical destruction of biomass

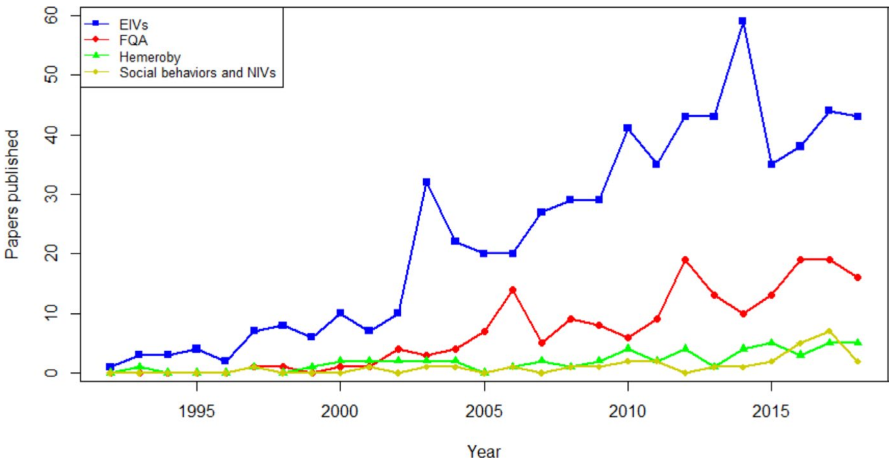

FIGURE 3 A 25-year publication record of the four systems. Even though the Floristic Quality Assessment (FQA) and Ellenberg indicator value (EIV) systems were created in the 1970s, the respective literatures garnered more interest after the 1990s. Numbers of hemeroby and social-behavior papers have increased modestly since the 2000s

(e.g. Grime, 2006), can be more broadly interpreted as any type of pulse event (White and Pickett, 1985). For example, disturbance events can be due to fluctuations in environmental conditions, abiotic changes in resource supplies, alterations of existing ecological regimes, and changes in biotic structure due to demographic events (Jentsch and White, 2019). Some of the indicator systems reviewed here attempt to distinguish naturally occurring disturbances from anthropogenic disturbances. Ellenberg indicator values, however, can be used for either natural or anthropogenic disturbances; they are useful for assessing a subset of disturbance types that involve environmental and abiotic resource changes over time. Briemle EIVs were created for the management of cultural ecosystems (Briemle and Ellenberg, 1994), so this system is focused on the anthropogenic physical disturbances common in grassland communities in Central Europe. Although Briemle EIVs specifically consider human disturbances, this system teases apart specific human impacts (see Table 2) and is therefore more discrete compared to the three human impact indicator systems. Urbanity EIVs could be considered the closest system to FQA, hemeroby, and NIVs because they integrate a wide variety of human disturbances associated with urbanization. Still, the urban emphasis of urbanity values is inherently narrower than the three human impact indicator systems.

There are trade-offs to using integrative human impact systems vs. Ellenberg values. Many widely used (Appendix S1) Ellenberg indicators of basic environmental conditions are easy to validate and calibrate. Ellenberg indicator users can directly compare indicator metrics to the environmental attribute of interest, which has been commonly done with moisture, nutrient, and pH values (e.g. Schaffers and Sýkora, 2000; Wamelink et al., 2002). Empirical validation of integrative human impact indicators is more difficult as anthropogenic effects cannot be precisely measured (Sukopp, 2008). To address this challenge, FQA users have compared proxy anthropogenic impact indices, such as landscape development indices or rapid assessment proxies, as baseline metrics to validate *C*-values (Lopez and Fennessy, 2002; Cohen et al., 2004; Miller and Wardrop, 2006; Gallaway et al., 2019).

Another trade-off of the human impact indicators concerns their simplicity contrasted with ecological vagueness. Compared to multiple EIVs, it may be easier to use single human impact indicators to reflect degradation or perceived natural quality. However, by themselves, human impact indicators cannot highlight the mechanisms of environmental change. For example, Breg Valjavec et al. (2018b) found that hemeroby indicators were suitable to infer the degradation of Slovenian karst depressions resulting from landfilling. The same authors characterized the same areas by applying moisture, nutrient, and pH Ellenberg indicators (Breg Valjavec et al., 2018a), which provided insight into edaphic effects of landfilling. In such cases, Ellenberg indicators may reveal relevant factors underlying a phenomenon of interest beyond that of integrative human impact indicators. Therefore, assigning both discrete and integrative indicator values to regional species lists can improve ecological knowledge (see Landolt et al., 2010).

4 | PROSPECTIVES

4.1 | On the future of human disturbance indicator systems

What are some future prospects for these integrative indicator systems, and what should their roles be compared to other EIV systems? Human impact indicators can be useful inputs for future basic science applications. Examples include assessing the experimental effects of human disturbance (Bernhardt-Römermann et al., 2011), resurvey studies documenting changes in vegetation from urbanization or biological invasion (Dolan et al., 2011; Berg et al., 2016), or conservation risk assessment (Ash et al., 2017; Clark et al., 2019).

The human impact indicator systems have particular value as applied tools for management, and are uniquely suitable for the ecological restoration of plant communities. The three systems produce interpretable metrics for monitoring vegetation trajectories over time (Matthews et al., 2009; Hansen and Gibson, 2014), and they are sensitive enough to detect the effects of recent management on vegetation quality (e.g. Taft et al., 2019). Ambitious restoration practitioners will emphasize the re-establishment of diverse, sensitive plant taxa and achieving a resemblance to historical reference systems (McDonald et al., 2016). Restoration practitioners often seek to establish native plant species characteristic of remnant ecosystems (McDonald et al., 2016), which matches the core philosophies of FQA and NIVs (Allison, 2002; Sengl et al., 2016). Often, however, achieving a resemblance to a historical vegetation community is impossible in practice. Some restoration ecologists have argued that novel ecosystems must be incorporated into restoration paradigms (Hobbs et al., 2009). The hemeroby system, by explicitly considering "potential natural vegetation" without reference to the historical state of the ecosystem, would be consistent with non-traditional or novel restoration targets. Furthermore, human impact indicators can assist managers in identifying and selecting restoration target species (Packard and Mutel, 2005; Brudvig and Mabry, 2008). They could also help managers predict successful species' establishment in ecological restorations, a tactic applied with several Ellenberg and Briemle indicators (e.g. Pywell et al., 2003; Baasch et al., 2016). Using such values could be simpler than strategically choosing multiple niche axes as in EIV bioindication, some of which will likely have low relevance in many scenarios (see Table 2).

The human impact indicator systems could also contribute to regional conservation strategies and environmental policies. For example, Berg et al. (2014) used hemeroby as one criterion in a stepwise index to characterize the conservation requirements of plant community types in Central Europe. FQA metrics have become staple metrics in environmental compensation schemes. The U.S. wetland offsetting scheme often uses FQA metrics as performance standards for environmental compensation to achieve "no net loss" of wetlands (Taddeo and Dronova, 2018). Hemeroby and NIVs have not been used in an analogous regulatory context, but perhaps they, or other EIVs, could be used to develop metrics for conservation or offsetting in Europe.

Future research efforts are needed to improve and characterize what ecological information is associated with *C*-values, hemeroby, and naturalness indicator values. Understanding the functional traits associated with sensitivities to human impacts could lead to improvements in the indicator systems by providing a life history-based foundation for these systems (Fanelli and De Lillis, 2004; Bauer et al., 2017; Ficken and Rooney, 2020). Ultimately, functional traits might be used to generate the indicator values of disturbance tolerance for individual species, in lieu of expert judgment (see Shipley et al., 2017). Ecological strategies, viz. Grime's C–S–R, or mutualism niche characteristics (e.g. Bauer et al., 2017), could be additional traits to link to the values. *C*-values, NIVs, or hemeroby values could be simple starting bioindicators to search for cosmopolitan, or cross-continental, predictors of relative human impact tolerance. Large functional trait and vegetation plot databases (Kattge et al., 2011; Bruelheide et al., 2019), as well as the existing indicator value data sets (e.g. Klotz and Kühn, 2002; Freyman et al., 2016), present an opportunity to study and apply the human impact indicators.

Future challenges related to biodiversity protection, continued anthropogenic land-use pressures, and climate change will necessitate the collection, analysis, and interpretation of vast amounts of vegetation data. Although our synthesis has focused on North America and Europe, we also suggest any of the systems we reviewed could be worthwhile if they were more widely adopted into new areas, such as Latin America, Oceania, or Sub-Saharan Africa. There are promising opportunities for scientists and managers to apply, improve, and investigate integrative human impact indicators. Therefore, human impact indicator systems may very well persist into the future.

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CONFLICTS OF INTEREST

None declared.

AUTHOR CONTRIBUTIONS

J. Zinnen developed the concept, conducted the bibliographic analysis, and was the primary author. All authors assisted with the bibliographic analysis and contributed to the conceptualization, writing, organization, and editing of the manuscript.

DATA AVAILABILITY STATEMENT

The bibliographic data used in the synthesis are openly available in the Illinois Data Bank, https://doi.org/10.13012/B2IDB-9374105_V1.

ORCID

Jack Zinnen https://orcid.org/0000-0002-5551-3781
László Erdős https://orcid.org/0000-0002-6750-0961
Christian Berg https://orcid.org/0000-0002-0587-3316

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SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section.

Appendix S1 Supplemental tables & figures, and bibliographic analyses methods

Appendix S2 Papers used in bibliographic analyses

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