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Synoptic assessment of wetland function: a planning tool for protection of wetland species biodiversity *

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Abstract. We present a synoptic assessment intended to maximize the benefits to wetland species biodiversity gained through Clean Water Act regulatory efforts within 225 sub-basins in Missouri, Iowa, Nebraska and Kansas (US Environmental Protection Agency, Region 7), USA. Our assessment provides a method for prioritizing sub-basins potentially critical for supporting wetland species biodiversity and may assist environmental managers and conservationists constrained by limited resources. We prioritize sub-basins based on the projected increase in the risk of wetland species extirpation across Region 7 that would be avoided by applying a unit of regulatory protection effort within a sub-basin. Because the projected increase in risk avoided per unit effort has not been directly measured, we represent this quantity with an index of indicators drawn from readily available data. A conceptual model incorporating landscape and anthropogenic factors guides index development *via* a series of simple benefit-cost equations. We rank and map the final index scores to show the relative priority among sub-basins for protection effort. High priority sub-basins appear to be concentrated along the major river systems within the region, where sensitive wetland species and intensive agriculture tend to coincide. Protection of wetland species biodiversity is an important, but not exclusive, attribute around which priorities should be set. Nevertheless, incorporation of our results into management strategies should allow managers to cast their local decisions in the context of regional scale maintenance of wetland species biodiversity, increasing ecological benefits for a given protection effort.

Key words: ecological indicators, extirpation risk, geographic prioritization, Section 404, synoptic assessment, wetland biodiversity

Introduction

Geographic prioritization – or the ranking of environmental decisions based on spatial patterns in environmental stressors and resources (Franklin 1994; Csuti et al. 1997; Landis and Wiegers 1997) – can serve as a comprehensive and proactive approach to resource management. However, due to severe information constraints, overriding socioeconomic concerns and a general lack of applicable ecological theory, most

conservation decisions remain reactive, do not include consideration of cumulative impacts and do not adequately account for potential ecological benefits (Scodari 1997; McAllister et al. 2000).

In partial response to these constraints, the Landscape Function Project of the US Environmental Protection Agency (EPA) developed a synoptic framework for geographic prioritization (Abbruzzese and Leibowitz 1997; Hyman and Leibowitz 2000; McAllister et al. 2000). The framework applies ecological theory and benefit-cost considerations (Turner 1991; Macmillan et al. 1998) to assess and prioritize sites when information is limited. Sites are ranked based on the relative benefit gained per unit of management effort, where benefit is expressed in terms of a change in an ecological endpoint. The approach was specifically developed for situations where an endpoint cannot be directly estimated because available information and ability to gather new data are limited. Judgement-based indicators of the projected change in the endpoint per unit of effort are combined into a synoptic index used to compare and rank sites (Leibowitz and Hyman 1999; Hyman and Leibowitz 2000). The approach is applicable to a variety of conservation issues, and it employs an explicit, repeatable and common framework, with necessary assumptions highlighted.

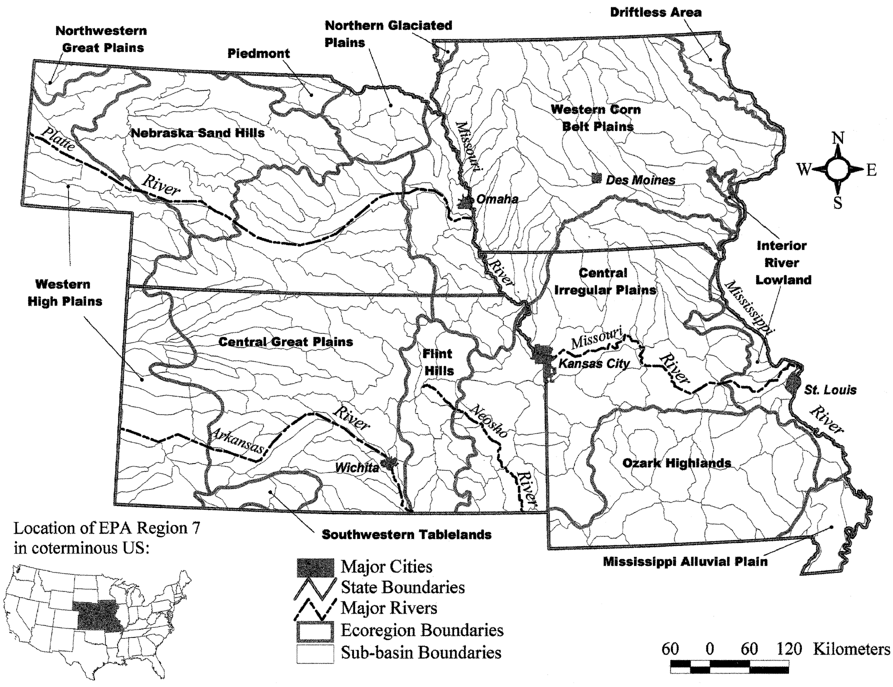

Figure 1. General overview map of EPA Region 7. Features shown include State and ecoregion boundaries (Omernik level 3; Omernik 1995), major rivers, USGS eight-digit sub-basin boundaries and major population centers. All features are clipped by regional boundaries. The inset shows the EPA Region 7 States within the coterminous United States.

We apply the synoptic framework to the management of wetland resources within EPA Region 7 (Missouri, Iowa, Nebraska and Kansas; Figure 1), USA. The goal of our prioritization is to select sites for protection efforts such that we maximize the region-wide ecological benefit of these efforts. We define ecological benefit as the projected increase in the risk of losing wetland species diversity in Region 7 that can be avoided by applying a unit of wetland protection effort. We assume that the risk of species extirpation increases in the absence of such protection effort; that is, we assume that protection avoids an inevitable increase in risk, but does not reduce risk, relative to current levels. We specify this effort as the review of Clean Water Act Section 404 permits (33 USC 1344) for the discharge of dredged or fill material into the wetlands of Region 7, hereinafter referred to as 404 permits. Therefore, we assume that this discharge of material into wetlands, if allowed to proceed, will increase the risk of species extirpation over the current risk. In this paper, we describe the development and estimation of the synoptic index and its potential use for prioritizing wetland management decisions within Region 7. Hyman and Leibowitz (2000, 2001) provide a thorough explanation of the synoptic framework (see also Abbruzzese and Leibowitz 1997; McAllister et al. 2000).

Environmental and regulatory background

Natural setting

The 74 million ha expanse of Region 7 is characterized by several distinct ecoregions (Omernik 1995; Figure 1). Based on the 1992 National Resources Inventory, the primary land use in the region is agriculture, with roughly 34 million ha of land under row-cropping or another agrarian use (covering 47.6% of the landscape) (USDA 1994; Nusser et al. 1998). Widespread conversion to agricultural land uses has been directly responsible for the loss of a large proportion of the historic areal extent of wetland within Region 7 (Tiner 1984). Conservative estimates suggest 95% (IA), 87% (MO), 48% (KS) and 35% (NE) of pre-settlement wetland area was lost by 1990 (Dahl and Johnson 1991). As of 1992, there were roughly 2.5 million ha of wetland in the region, covering 3.4% of the landscape (USDA 1994; Nusser et al. 1998). Moreover, the rate of wetland conversion has exceeded changes in other habitat types (Dahl and Allord 1996).

Wetland biodiversity support

Wetlands typically support a high diversity of species (generalized across all taxa) compared to surrounding uplands due to their relatively high habitat diversity, extent of resources, ecotones and refugia (Reed and Porter 1988; Mitsch and Gosselink 1993). For example, over 55% of the species cataloged by Natural Heritage Programs within

Region 7 are obligate or facultative wetland species (see below for a description of the Heritage data; Niering 1988; Doylan and MacLean 1997). Moreover, wetlands often support relatively dense populations of species. For example, wetland complexes on the Platte River in central Nebraska support over half a million sandhill cranes and several million ducks during their annual migration (Sidle et al. 1993).

Administrative context

Section 404 of the Clean Water Act establishes a program to regulate the discharge of dredged or fill material into waters of the US, including wetlands. Regulated activities are evaluated through a permit review process administered by the US Army Corps of Engineers (COE), which is responsible for issuing or denying permits. EPA has the authority to review 404 permits and designate unacceptable areas for discharges, to veto a COE decision and to take enforcement actions on violations of Section 404. The outcome of EPA review can directly influence the relative density and quality of wetland habitat, including its capacity to support species biodiversity (Votteler and Muir 1996). Efficiently allocating such regulatory effort is crucial for a variety of reasons: First, resources for regulation and conservation are limited. Second, the maintenance of biodiversity has paramount ecological and evolutionary importance (Ehrlich and Wilson 1991). Finally, the protection and restoration of biological diversity is valued by society (UNEP 1992).

Both EPA and the COE receive a heavy load of 404 permits to review (USACOE 1995). For example, the COE received an average of 1878 permits in KS, MO and NE (Iowa data unavailable) per year from 1988–1996 (Environmental Working Group 1999). Similarly, from 1988 to early 1999, EPA Region 7 received an average of 447 permits per year (including Iowa) (note that the lower total number of applications received by EPA, relative to the COE, is in accordance with nationwide permit regulations). Given often severe resource constraints, 404 permits must be prioritized for review by Region 7, with only a portion receiving protracted attention (based on Region 7 internal records, roughly 15% of the permits received from 1988 to 1999 were intensively reviewed; Schweiger, unpublished analysis).

Region 7 wetland staff currently prioritize 404 permits by qualitatively evaluating – through their best professional judgement – localized ecological and biophysical conditions (when available), administrative constraints and political criteria (see also Margules and Usher 1981). This approach has been successful, but it cannot easily incorporate cumulative effects or landscape-scale processes and generally lacks rigor. Given the degree of support of regional biodiversity by wetland habitat, high wetland losses and a need to prioritize regulatory efforts, we conducted a synoptic assessment to complement current ranking procedures. Our objective was to maximize the benefit of a given regulatory effort, where benefit is in terms of avoided increase in the risk of wetland species extirpation from Region 7. This objective was developed in conjunction with Region 7 Section 404 wetland staff with the intention of improving the

process by which 404 permits are prioritized for review. By working closely with the actual user, we hope to minimize our assumptions about the way in which scientific information is or should be used by policy makers.

Assessment units

A critical step in a synoptic assessment is the selection of the spatial units that define areas to be compared and prioritized. We used 225 eight-digit US Geological Survey (USGS) sub-basins (Seaber et al. 1984) as the spatial unit for all synoptic index calculation and establishment of ranks (Figure 1). Our choice was a compromise between a set of practical limits and the spatial bounds on ecological functions important in wetland species biogeography.

Level IV ecoregions (Omernik 1995) would be the preferred unit for our assessment as they correspond by definition to ecological response and are delineated at a high enough spatial resolution to be useful for the management of wetland resources (see also Kiester et al. 1996). However, ecoregions delineated at this scale were not available across Region 7. As an alternative, sub-basin boundaries can constrain wetland occurrence and therefore wetland species distribution, dispersal and community composition (Omernik and Bailey 1997). Therefore, sub-basins were preferable to an entirely arbitrary unit such as a county. While sub-basins often do not correspond to actual watersheds (Omernik and Bailey 1997), based on surficial geology, glaciation patterns and surface water hydrology roughly 70% of the eight-digit sub-basins in Region 7 appear to approximate true watersheds (Schweiger, unpublished analysis). Accordingly, eight-digit sub-basins were our best available choice (higher resolution, smaller scale sub-basins were also unavailable region-wide). The spatial unit used in a synoptic assessment can have a strong impact on results (McAllister et al. 2000), and therefore represents a key assumption of any such analysis. Future work should include other spatial units (e.g., Bailey 1995; Kiester et al. 1996) and comparative analyses of results from these approaches.

Synoptic prioritization: index development and calculation

In the following section we describe the application of the synoptic framework to prioritize Section 404 wetland permit review effort. First, we formally define the criterion for ranking sub-basins. Because data for directly evaluating our criterion are not available, we expand the expression into a series of related terms that are more easily estimated. Next, we present judgement indicators for each of these terms. These indicators are then combined into an index to represent the prioritization criterion. Finally, we estimate three alternative index formulations, based on different options for indicator selection and combination. Figure 2 presents our conceptual model of the relationships among our endpoints and their indicators.

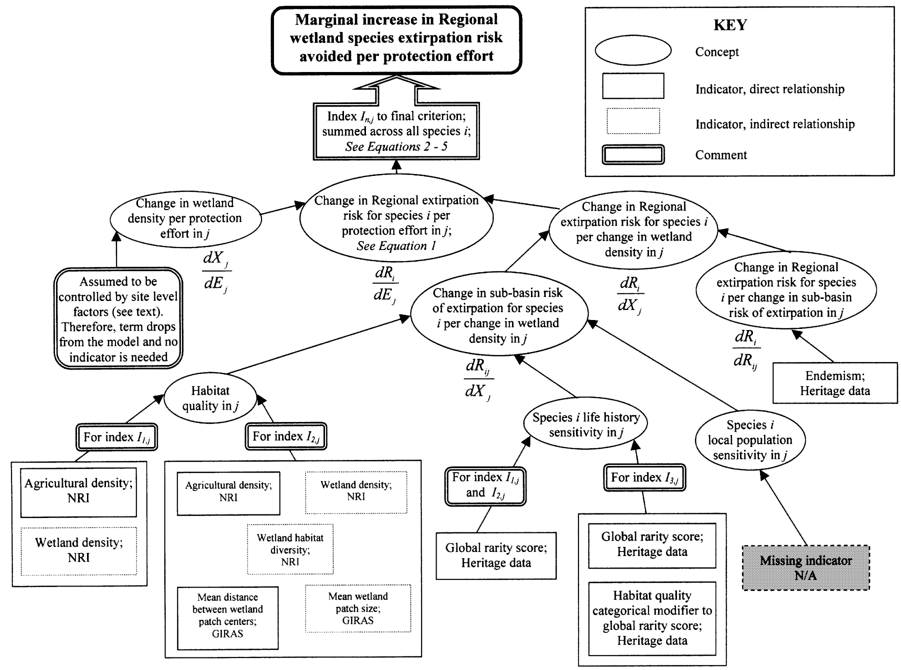

Figure 2. Conceptual model showing how ecological concepts and indicators are linked to the final prioritization criterion. The figure may also be used as a 'road map' to help locate relevant equations throughout the paper. All concepts (unmeasurable endpoints) are in ovals while indicator data for these concepts are in rectangles. Data sources are included with each indicator. The terms used in Equations (1) and (2) are given immediately below their relevant concepts. Comments, including the three alternate sets of indicator combinations used to generate $I_{n,j}$, are given in double lined boxes (note that for $I_{3,j}$ habitat quality is included through the modifier to the species sensitivity score and no separate habitat indicators are used). All indicators assumed to vary inversely with species extirpation risk are given in rectangles with dotted lines. Shaded rectangles show indicator data unavailable at the time of our analysis.

Prioritization criterion

Our assessment addresses the following general question: given a constrained level of regulatory effort, in which sub-basins should protection efforts be targeted to maximize the ecological benefit for a given total amount of effort? Answering this question requires that we define a prioritization criterion that will allow sub-basins to be ranked in such a way that we meet this objective. In such cases, the proper generalized prioritization criterion is the marginal change in ecological benefit per unit effort (Hyman and Leibowitz 2000).

In this application of the synoptic framework, protection efforts provide ecological benefits by avoiding wetland habitat losses, as well as subsequent increases in risk of wetland species extirpation, which we assume would otherwise result from anthropogenic development in sub-basin j. We assume that each wetland species i

has some initial risk of region-wide extirpation, denoted by R_i, which exists even in pristine settings due to factors such as competition, predation and natural catastrophe. The expected increase in this risk with development can be reduced by expending 404 regulatory effort, E_j, thereby reducing wetland impacts in sub-basin j. Because 404 permit applications can be submitted for the dredge or fill of non-wetland aquatic habitat (e.g. lotic systems), it is important to clarify that E_j includes only the effort for 404 permits dealing with wetland habitat as defined by Cowardin et al. (1979). Since total effort is limited, benefit will be maximized for a given total effort if we target sub-basins where we can avoid the greatest increase in extirpation risk per unit of effort. Thus, for our specific case the prioritization criterion is the marginal increase in regional risk to wetland species i that is avoided per unit 404 permit review effort in sub-basin j, which we denote as $\mathrm{d}R_i/\mathrm{d}E_j$. Although these terms consider risk for each species individually, and thus treat the species-specific risks as independent (ignoring species interactions), we will later combine these terms into an index of combined risk for all species. Note that the scale of our effort is at the individual sub-basin, while our risk endpoint is at the regional scale. In addition, our endpoint does not incorporate any ecological or regulatory components expressed at a scale larger than Region 7 (e.g., the proportion of the range of species i outside regional boundaries).

A number of ecological and anthropogenic conditions affect the relationship between R_i and E_j, some of which vary geographically by sub-basin. This variance across sub-basins forms the basis for our ranking of avoided increase in risk (Hyman and Leibowitz 2000). Given that actual 404 permit decisions occur at the local wetland site, and not at the sub-basin level, geographic prioritization is only appropriate if we assume variance in $\mathrm{d}R_i/\mathrm{d}E_j$ within sub-basins is small compared to the variance between sub-basins. We must also assume that $\mathrm{d}R_i/\mathrm{d}E_j$ is independent across sub-basins, and that a unit of R_i is comparable over all sub-basins. These three assumptions allow us to geographically prioritize protection efforts based on variance in $\mathrm{d}R_i/\mathrm{d}E_j$ among sub-basins (Hyman and Leibowitz 2000).

Expansion of prioritization criterion

Because directly estimating $\mathrm{d}R_i/\mathrm{d}E_j$ is difficult, we expand the prioritization criterion into a series of intermediate terms which, when multiplied together, equals $\mathrm{d}R_i/\mathrm{d}E_j$. This expansion is produced by proposing two additional variables in the link between R_i and E_j – wetland habitat density in sub-basin j (X_j) and local risk of species loss in sub-basin j (R_{ij}). Equation (1) shows the expansion of $\mathrm{d}R_i/\mathrm{d}E_j$ to include these terms:

$$\frac{\mathrm{d}R_i}{\mathrm{d}E_j} = \frac{\mathrm{d}X_j}{\mathrm{d}E_j}\,\frac{\mathrm{d}R_{ij}}{\mathrm{d}X_j}\,\frac{\mathrm{d}R_i}{\mathrm{d}R_{ij}} \tag{1}$$

where $\mathrm{d}R_i/\mathrm{d}E_j$ is the change in regional extirpation risk for species i per protection effort in sub-basin j; $\mathrm{d}X_j/\mathrm{d}E_j$, the change in wetland density per protection effort in

sub-basin j; dR_{ij}/dX_j, the change in local extirpation risk for species i in sub-basin j per change in wetland density; and dR_i/dR_{ij}, the change in regional extirpation risk for species i per change in local extirpation risk.

A key component of the synoptic framework is the development of indicators to represent terms in the expanded prioritization criterion. Although the use of indicators is often questioned (e.g., Landres et al. 1998), they are useful when the endpoint of interest is difficult or costly to measure directly, the risk from a wrong decision is low and the management concern calls for a relative rather than absolute assessment of alternatives (Abbruzzese and Leibowitz 1997). Indicator selection is often dominated by data availability or other logistical constraints. However, this focus on practicality can ignore important issues, such as whether ecologically irrelevant variables are included or whether important variables are omitted (Hyman and Leibowitz 2000; McAllister et al. 2000).

General justification for intermediate terms and indicators

We identified general and integrative attributes of sub-basins and wetland species that prevailing research, the literature and consultation with a variety of resource experts suggest may be at least grossly correlated with the terms of Equation (1) (e.g., Gilpin 1986; Harris 1988; Angelstam 1992; Boyce 1992; Doylan and MacLean 1997; Ando et al. 1998). The specific factors that influence species–habitat relationships are complex and well beyond the scope of the synoptic approach (Dobson et al. 1997; Landres et al. 1998). However, because our assessment is both general and intended as a first cut, the relatively simple and intuitive intermediate terms (Equation (1)) and indicators (below) are appropriate for our goals. The next three sub-sections present the indicators developed for each term in Equation (1), briefly justify our choices and describe data sources as needed.

Change in wetland density per protection effort in sub-basin j

The first term in Equation (1) (dX_j/dE_j) represents the avoided loss of wetland per unit of regulatory effort. By denying a 404 permit, conversion of wetland to an alternate land use is avoided. For simplicity, we assume that the denial of a 404 permit prohibits the loss of a unit of wetland habitat (there are no partial denials, mitigation or permit trading). If the avoided loss per effort varies across sub-basins, then this term will have a direct effect on the final rank of each sub-basin. However, we assume that site level factors (e.g., the type of proposed permit action) dominate the variation in both avoided loss per permit and number of permits processed per effort (mostly desk time). Therefore, variance in dX_j/dE_j does not affect the rankings, and the term drops from the final index. However, explicitly including this term in the model allows an indicator to be included if it is determined that E does vary across sub-basin (e.g., due to distance dependent travel costs).

Change in local risk for species i in sub-basin j per change in wetland density

The second term in Equation (1) (dR_{ij}/dX_j) represents the increased risk of species loss from sub-basin j that is avoided by preventing loss of wetland in j. Selected indicators for this term fall into two general categories: (1) measures of habitat quality (six indicators) and (2) measures of species sensitivity to habitat loss (two indicators). Our measures of habitat quality are limited to the sub-basin (landscape, *sensu* Wiens 1995) scale for three reasons: First, we prioritize relatively large areas of land (sub-basins), with a mean area of over 320 000 ha. Second, most available indicator data are coarse or aggregated to the sub-basin scale. Finally, (and most importantly) a body of evidence (e.g., Pearson 1993; Wiens 1995; Schweiger et al. 1999) suggests landscape structure and composition can be more important than local habitat quality in predicting species distribution and persistence, especially at regional scales. We use the term 'habitat quality' interchangeably with 'landscape habitat quality'.

Agricultural density
Relatively high levels of agriculture generally increase the risk of local wetland species extirpation (Harris 1988). Flather et al. (1994) cite agricultural development as the primary cause of species endangerment. Tiner (1984) concluded that 87% of wetland degradation can be directly traced to agricultural land use. Moreover, the detrimental effects of habitat fragmentation, which generally accompany agricultural development, are particularly important for specialized taxa such as facultative and obligate wetland species (Reed and Porter 1988; Mitsch and Gosselink 1993). We used data from 145 914 (real and imputed; H. Bogash, pers. comm.) 1992 National Resources Inventory (NRI) sample locations to estimate the proportion of each sub-basin with row-cropping (Nusser et al. 1998). NRI data were the most current and accurate information available on agricultural extent (nearly 75% of our estimates have confidence bounds within 20% of the parameter value).

Wetland density
Available habitat area is often cited as another important explanatory variable in population and community level studies of species persistence and distribution (e.g., Fahrig and Merriam 1985; Kotliar and Wiens 1990). High wetland density within a sub-basin may reduce dispersal costs (Morris 1992), produce hydrological regimes closer to reference conditions (Mitsch and Gosselink 1993) and reduce predation by upland generalists (Angelstam 1992) – all of which decrease the risk of local species extirpation. Thus, species in sub-basins with a relatively high density of wetland should have a reduced dR_{ij}/dX_j relative to sub-basins with low density. We estimated wetland density by calculating the proportion of sub-basin area that is wetland using 1992 NRI data. These data were the best available information for estimating wetland density across all sub-basins in Region 7. However, because of the relative rareness of

wetland in Region 7, randomized NRI point samples have a low probability of falling on wetland, reducing estimate quality (68% of our estimates have a margin of error that exceeds 50% of the parameter value). A potential alternative data layer, the US Fish and Wildlife Service's National Wetland Inventory (NWI), generated from 1984 aerial photography, may provide more accurate estimates of wetland density than the NRI (Tiner 1984; Swanson and Duebbert 1989). However, NWI-based estimates of wetland density were only available for 116 sub-basins within Region 7 (primarily in Iowa, Nebraska and Missouri) at the time of our analysis. A comparison of the NRI and NWI in sub-basins in which their wetland density coverage overlapped suggests the two data layers are similar (log transformed data: $r = 0.57$), with sub-basin estimates from the NRI only marginally higher than those from the NWI ($t = 1.9$, d$f = 229$, $P = 0.06$; mean difference = 3181.5 ha). Therefore, NRI-based wetland density estimates appear to be sufficient for our purposes.

Human population density

Relatively high human density tends to reduce biotic integrity within a sub-basin, increasing the risk of species extirpation (Soule 1986). This is due to direct (e.g., habitat loss) and indirect effects (such as increased competition from human commensals) often associated with high human population density. Tiner (1984) attributes nearly 10% of wetland degradation to urbanization and increased human density. We estimated human population density per sub-basin using data from the 1990 US Census. Census Bureau data were the only region-wide direct estimates of population size available and have an acceptable error rate for our efforts (in rural areas, Census counts underestimate true values by approximately 5.9%; US Census Bureau 1997).

Average wetland patch size

Many studies have suggested that patch size has a fundamental effect on species performance (e.g., Fahrig and Merriam 1985; Robinson et al. 1992). For many wetland species, larger less fragmented wetland complexes reduce the risk of local extirpation (Bellrose 1977). We estimated wetland patch size per sub-basin using an arithmetic average of the area of each wetland polygon in the USGS Geographical Information Retrieval and Analysis System (GIRAS) database. GIRAS land cover data were developed from late 1970s remotely sensed imagery (Fegeas et al. 1983). GIRAS data were used instead of NRI- or NWI-based estimates because they were the only contiguous region-wide data available. Patch-scale attributes such as size are more readily derived from data like GIRAS than from point-based statistical samples like the NRI.

Average distance between wetland patches

Numerous studies suggest that distance is negatively associated with dispersal success (e.g., Morris 1992; Diffendorfer et al. 1995). Dispersal over longer distances often increases exposure to hostile environments and can increase extirpation risk

(Gustafson and Gardner 1996). We estimated the mean distance between wetland patches by averaging all possible linear distances between the centers of wetland polygons within the GIRAS database (used instead of the NRI or NWI for the reasons given above). Note that this is the average distance between wetlands in a sub-basin, and not a nearest neighbor distance (which could include patches in different sub-basins). This value is weighted by the inverse of wetland patch frequency to correct for differences in the density of wetland patches among sub-basins.

Wetland type diversity

Species often depend on different habitat types for components of their life cycle (Turner 1996). Therefore, sub-basins with lower wetland type diversity may have a higher risk of local extirpation (Krapu and Duebbert 1989; Pearson 1993). We estimated wetland habitat diversity (all Cowardin types; Cowardin et al. 1979) using 1992 NRI data and the Shannon–Wiener equation (Pielou 1966).

Patterns among habitat indicators

We conducted a principal components analysis (PCA) of the six habitat indicators to describe multivariate structure, identify correlated measures and assist in categorizing potential landscape-scale habitat quality impacts on wetlands species. All habitat indicators were first examined for univariate normality and transformed as needed. Correlation matrices were used to equalize the influence of variables with highly different absolute ranges (James and McCulloch 1990). Resulting axes were retained based on standard protocols (Kachigan 1982), and rotated using an equamax normalized procedure.

The PCA results (Table 1a) suggest that sub-basins with high human density have diverse wetland types, high wetland density and short distances between wetland centers. This pattern indicates our original expectation – that increased human density should decrease the quality of wetland habitat within a sub-basin – may have been too general. We therefore recalculated the PCA without human population density and excluded it as an indicator thereafter. Because Region 7 is highly agrarian, with only two population centers over 1 million people (the Kansas City and St Louis metropolitan areas with 1.42 and 1.90 million people, respectively; 1990 US Census Bureau, Figure 1), most wetland impacts are probably agricultural in nature (see also Tiner 1984). Removing this indicator should therefore be relatively inconsequential, which was confirmed in the second PCA (Table 1b); two factors were generated with nearly identical loadings (excluding human density) as the first PCA.

Life history sensitivity

Differences in population size, reproductive potential for recovery, range size and ecological specialization can all contribute to species responses to shifts in habitat quality and quantity (Gilpin and Soule 1986; Boyce 1992; Robinson et al. 1992). To develop an indicator for this collection of traits (collectively denoted as species sensitivity),

Table 1. Results from PCA of six (a) and five (b) habitat quality indicators.

Principle explanatory variables (loading)	Eigenvalue	% Explained variance
(a)		
Log human population density (0.744), wetland type diversity (0.747), −log mean distance between wetland centers (−0.649), log wetland density (0.846)	2.47	41
Agricultural density (0.812), −log mean wetland patch size (−0.531)	1.11	19
(b)		
Wetland type diversity (0.761), −log mean distance between wetland centers (−0.715), log wetland density (0.858);	2.04	41
Agricultural density (0.846), −log mean wetland patch size (−0.500)	1.03	22

All indicator variables were transformed as needed prior to analysis. Axes were rotated using an equamax procedure; however, rotations did not produce any significant correlations among factor scores (all $r < 0.0001$). Terms given include: principle explanatory variables for each axis with respective loadings, the eigenvalue for each axis, and the proportion of the total variance explained by each axis. A '−' indicates a negative correlation between an indicator variable(s) and an axis. The analysis presented in 1(a) explained 60% of the total variation in six habitat quality indicators. The analysis presented in 1(b) (human population density removed) explained 63% of the total variation in five habitat quality indicators.

we used rarity classifications within the 1995 Natural Heritage Program data base (Ostlie et al. 1997; hereinafter referred to as 'Heritage data'). The Heritage rarity scores describe the conservation status of species at the State, National and Global scale (Jenkins 1988). Five classes of rarity exist at each scale, with a score of 1 equivalent to the rarest class. For example, at the Global scale, a G1 designation is given to a critically imperiled species with less than six viable occurrences, less than 1000 individuals, or less than 810 ha occupied worldwide. In contrast, a G5 designation is given to a demonstrably secure species commonly found throughout the world.

Several studies (e.g., Niemi 1982; Burke and Humphrey 1987) have suggested that Heritage rarity scores reflect both intrinsic and synecological characteristics related to species viability. Specifically, Millsap et al. (1990) found a strong inverse relationship between median biological vulnerability scores developed for 668 Florida taxa through expert ranking and Heritage rarity classes. These studies conclude that G1 species typically have characteristics such as reduced ranges, multiple ecological specializations, small disjunct populations, or reduced gene flow that lead to increased risk of local extirpation (see also Gilpin 1986). G5 species have on average the opposite set of attributes.

We identified 612 species in the Region 7 Heritage data that based on literature and expert opinion were classified as wetland facultative or obligate (Johnsgard 1979; Bee et al. 1981; Reed and Porter 1988; Conant and Collins 1991; consultation with Dr C. Freeman of the Kansas Biological Survey). These species included members of Amphibia (24), Aves (80), Bryophyta (2), Crustacea (11), Dicotyledonae (155),

Diplopoda (1), Insecta (18), Lichens (1), Mammalia (21), Mollusca (49), Monocotyledonae (135), Osteichthyes (80), Pterophyta (14), Reptilia (33) and Turbellaria (1). We then used the median number of viable occurrences as defined for each Global rarity class to create and scale our sensitivity indicator for each species. G2 and G3 species have (by definition) four and 20 times the median number of occurrences of G1 species (Jenkins 1988). Using the ratios of median number of occurrences among these three classes of species and an arbitrary sensitivity score of 1000 for G1 species, G2 and G3 species were assigned scores of 250 and 50, respectively (higher scores correspond to higher sensitivity). Basing sensitivity on the median number of occurrences within species rarity classes lends some realism to the indicator scale relative to the attributes that lead to rarity (Leibowitz and Hyman 1999). Because the number of viable occurrences for G4 and G5 species are not available in the Heritage data, we estimated these species sensitivity scores by extrapolation from the sensitivity scores of G1–G3 species, giving values of 25 for G4 and 10 for G5 species.

The Heritage database is derived from a compendium of museum collections, published reports, geographically referenced sightings and extensive targeted field inventories. Species found in the database include both rare and common members of all taxa. Heritage data are not a spatially unbiased sample nor does the design account for detection probability (Nichols et al. 1998); thus they probably underestimate species richness in each sub-basin. However, Heritage data were the only comprehensive and accurate (Noss 1987; Jenkins 1988) biogeographic data available for a wide set of Region 7 species. Moreover, Pearson and Cassola (1992) and Dobson et al. (1997) suggest that a subset of species (here the Heritage species subset of all wetland species that exist in Region 7) can be used in analyzing risks to biodiversity, especially for assessments at large spatial scales.

Local population sensitivity

The previous factor, life history sensitivity, is meant to represent how sensitive a species is to a given change in habitat area. For a particular species, life history sensitivity is constant over all sub-basins; i.e., there is no local variability in our indicator. In our model life history sensitivity only varies by species. It therefore represents a way of weighting the response of an individual species to habitat loss, relative to other species. Yet local (i.e., sub-basin) populations will vary in their response to a given level of habitat loss. While a number of factors contribute to sub-basin variability (e.g., interspecific interactions) probably the most important is the abundance or density of the local population (Boyce 1992). Risk of sub-basin extirpation should generally be higher in sub-basins with lower population densities, in much the same way that stochastic extinctions can occur at low population numbers. A higher density results in a more robust population that is less at risk for extirpation (Gilpin and Soule 1986; Burke and Humphrey 1987). Unfortunately, data on local species abundance or density were not available for our study area, nor were other indicators of local population sensitivity. Data on occurrences (and often,

abundances) of threatened and endangered species are available, but we feel the use of such data are inappropriate since the focus of our assessment is overall biodiversity of wetland species. Also, threatened and endangered species are covered by other EPA programs.

Given the lack of an appropriate indicator, local population sensitivity is omitted from our final indices. Therefore, sub-basin variability in $\mathrm{d}R_{ij}/\mathrm{d}X_j$ is largely a function of habitat quality (when summed over all species for a given sub-basin, $\mathrm{d}R_{ij}/\mathrm{d}X_j$ also varies according to the species present in that sub-basin, as weighted by their life history sensitivity – see section on index calculation below). We include local population sensitivity in our conceptual model (Figure 2) in order to make its omission explicit (see also McAllister et al. 2000). This allows this factor to be added at a later time should sub-basin population abundances or other appropriate data become available.

Change in regional risk for species i per change in local risk

The third term in Equation (1) ($\mathrm{d}R_i/\mathrm{d}R_{ij}$) represents the change in regional risk of species extirpation per unit change in sub-basin *j*. The regional impact of the local loss of species *i* is related to its endemism (Anderson 1994; Koopowitz et al. 1994; Dobson et al. 1997). If species *i* is endemic, occurring in only one sub-basin, then its loss from that sub-basin represents a regional extirpation. At the other extreme, if species *i* is common and occurs in all sub-basins, then its local loss in a sub-basin will have minimal impact on regional extirpation risk. We calculate endemism scores for each wetland species within the Heritage data as $1/N_i$ (following Kerr 1996), where N_i is the number of sub-basins in which species *i* occurs.

Index calculation

To generate numerical estimates of our prioritization criterion, we first replace the terms in Equation (1) with the indicator data described in the preceding sub-sections (recall that the first term of Equation (1) drops out of the index since it is assumed that local factors dominate its variance). We then sum the product of the indicators over all species by sub-basin, since the index represents risk to all wetland species:

$$I_j = \sum_i \left(\frac{\mathrm{d}R_i}{\mathrm{d}E_j}\right)' = \sum_i \left[\left(\frac{\mathrm{d}R_{ij}}{\mathrm{d}X_j}\right)' \left(\frac{\mathrm{d}R_i}{\mathrm{d}R_{ij}}\right)'\right] \tag{2}$$

where I_j is the index of change in regional extirpation risk for all wetland species per protection effort in sub-basin *j*; and $(\mathrm{d}R_i/\mathrm{d}E_j)'$, $(\mathrm{d}R_{ij}/\mathrm{d}X_j)'$, and $(\mathrm{d}R_i/\mathrm{d}R_{ij})'$ are indicators of $\mathrm{d}R_i/\mathrm{d}E_j$, $\mathrm{d}R_{ij}/\mathrm{d}X_j$, and $\mathrm{d}R_i/\mathrm{d}R_{ij}$, respectively.

We chose simple combination rules (e.g., arithmetic averages, no interaction terms, linear functions) because our indicators were not scaled to one another and we had no reason to expect there were any limiting factors in our included set of

indicators. Moreover, the synoptic framework was developed to facilitate such a parsimonious approach. See Leibowitz and Hyman (1999) for a detailed description of the necessary mathematical assumptions within our indicator combination strategy.

Because our index is based largely on judgement, a contrast of results using different formulations for combining indicators allows us to evaluate the overall robustness of our index (McAllister et al. 2000). Thus we formulated the index using three approaches, denoted as $I_{1,j}$, $I_{2,j}$ and $I_{3,j}$. For all three indices, $(\mathrm{d}R_{ij}/\mathrm{d}X_j)'$ is the term modified and $(\mathrm{d}R_i/\mathrm{d}R_{ij})'$ is set equal to the endemism score $(1/N_i)$. We calculated Pearson product–moment and non-parametric γ correlations (Siegel and Castellan 1988) to compare results from the three alternate approaches. The γ statistic accounts for skewed distributions and a high number of tied ranks. We also qualitatively compared the spatial pattern of the results from each method. The following sub-sections describe each approach to index calculation.

Index 1: two habitat indicators

The term $\mathrm{d}R_{ij}/\mathrm{d}X_j$ is a function of habitat quality and sensitivity to habitat loss. $I_{1,j}$ and $I_{2,j}$ use different indicators to represent habitat quality. PCA analysis of our landscape-scale habitat quality indicators suggests that wetland and agriculture density have the highest loadings on their respective axes (Table 1b). Because the remaining habitat quality indicators may be somewhat redundant, index $I_{1,j}$ uses only agricultural density and wetland density to depict landscape habitat quality. We first standardize agricultural and wetland density by dividing by their respective maximum values and then calculate the inverse of wetland density (change in extirpation risk per wetland loss is assumed to vary inversely with wetland density). These two values are averaged and then multiplied by the sensitivity and endemism score for species i. Finally, to estimate $I_{1,j}$ we sum across all species i in basin j. Substituting these indicators into Equation (2) gives the following:

$$I_{1,j} = \sum_i \left[\left(\frac{\mathrm{d}R_{ij}}{\mathrm{d}X_j} \right)' \left(\frac{\mathrm{d}R_i}{\mathrm{d}R_{ij}} \right)' \right] = \sum_i \left[\left(\frac{A_j^* + 1/W_j^*}{2} S\{G_i\} \right) \left(\frac{1}{N_i} \right) \right] \tag{3}$$

where $I_{1,j}$ is the first index of change in regional risk for all wetland species per protection effort in sub-basin j; A_j^* and W_j^*, the standardized agricultural and wetland densities in sub-basin j, respectively; $S\{G_i\}$, the sensitivity value for the Global rarity class of species i; and N_i, the number of sub-basins in which species i occurs.

Index 2: five habitat indicators

Our second index differs from the first by including five landscape indicators of habitat quality. Change in extirpation risk per wetland loss is assumed to vary inversely

with patch size and diversity. All measures are standardized and then averaged. The calculation of $I_{2,j}$ follows similar steps as outlined for $I_{1,j}$ above:

$$\begin{aligned} I_{2,j} &= \sum_i \left[\left(\frac{\mathrm{d}R_{ij}}{\mathrm{d}X_j} \right)' \left(\frac{\mathrm{d}R_i}{\mathrm{d}R_{ij}} \right)' \right] \\ &= \sum_i \left[\left(\frac{A_j^* + 1/W_j^* + 1/P_j^* + D_j^* + 1/H_j^*}{5} S\{G_i\} \right) \left(\frac{1}{N_i} \right) \right] \end{aligned} \tag{4}$$

where $I_{2,j}$ is the second index of change in regional extirpation risk for all wetland species per protection effort in sub-basin j; P_j^*, the standardized average wetland patch size in sub-basin j; D_j^*, the standardized average distance between wetland patches weighted by the inverse of patch frequency in sub-basin j; H_j^*, the standardized Shannon–Wiener diversity of wetland types in sub-basin j; and other variables as previously defined.

Index 3: habitat categories

For the first two indices, habitat quality and species sensitivity both have equally important and proportionate effects on the change in local risk per wetland loss ($\mathrm{d}R_{ij}/\mathrm{d}X_j$). For the third index, we assume that life history sensitivity is the dominant factor, but that its effect on risk is modified by landscape habitat quality. We assume that life history sensitivity is mostly affected by global factors. Thus, habitat quality in a sub-basin has an effect only if it is of extreme (low or high) quality. We use the location of sub-basin j in the multivariate space defined by the PCA (Figure 3) to categorize each sub-basin into low, high, or neutral quality and to examine its departure from the mean (the origin). Sub-basins falling within quadrant one tend to have relatively reduced wetland density and diversity, increased distance between patches, increased agriculture density and decreased patch size – all of which are assumed to increase sensitivity to habitat loss (i.e., species in sub-basins with these characteristics are considered marginal and susceptible to further wetland loss). For a sub-basin in quadrant one, Q_j is assigned a value of 0, -1, or -2 if it is less than one, between one and two, or more than two standard deviations from the origin, respectively. Sub-basins falling within quadrant three tend to have the opposite set of characteristics – assumed to decrease sensitivity (i.e., species in sub-basins with these characteristics are robust and able to cope with wetland loss). Sub-basins in this quadrant are assigned a Q_j value of 0, 1, or 2 based on their deviation from the origin. Quadrants two and four have a mixture of positive and negative factors, and we assume that habitat of such quality is neutral with respect to sensitivity; sub-basins in these two quadrants are assigned $Q_j = 0$. Adding Q_j to G_i results in a modified Global rarity score with an expanded range of -1 to 7. Sensitivity values for scores outside the original 1–5 range are defined by extrapolation. As with $I_{1,j}$ and $I_{2,j}$, we

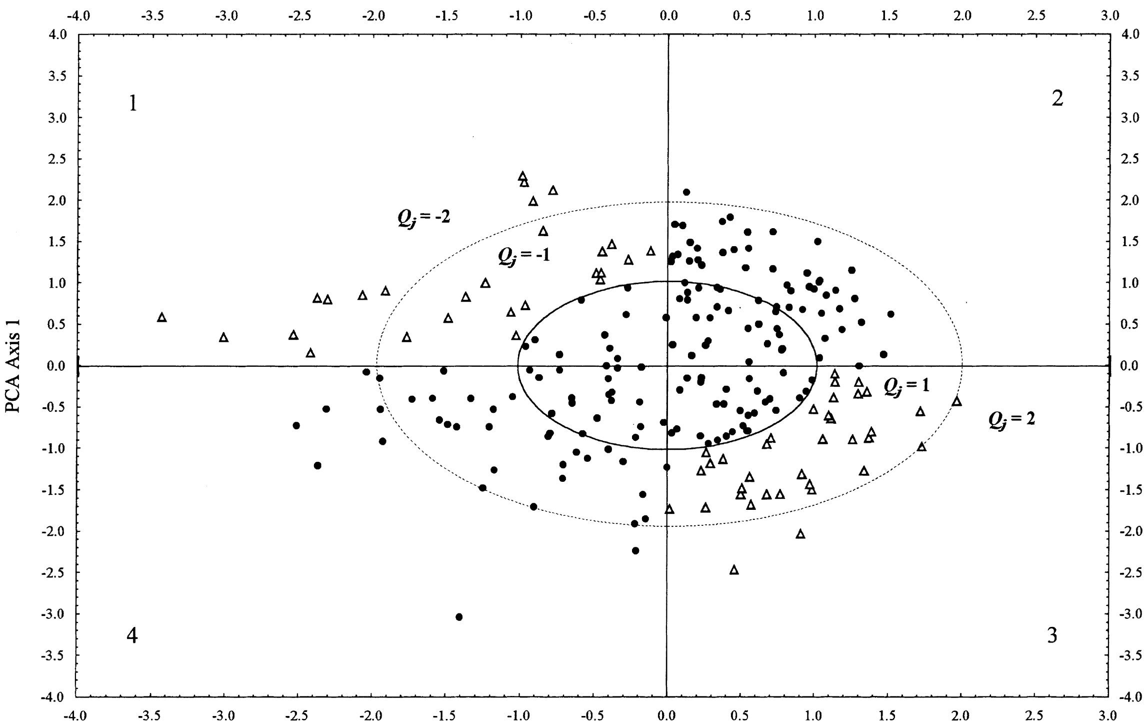

Figure 3. Plot of sub-basin factor scores on first two axes from PCA of five habitat quality indicators. Plot shows four quadrants about mean score on each axis (origin) and ellipses indicating deviation from the origin (solid line = 1 SD; dotted line = 2 SD). Species sensitivity scores in sub-basins that deviate by more than 1 SD from the origin in quadrants one and three (indicated with a '$\triangle$') decrease by Q as shown (Equation (5)). Sensitivity scores for species in sub-basins within quadrants two and four, and within 1 SD in quadrants one and three, are unaffected (indicated with a '•').

multiply the (now modified) sensitivity score for species i by its endemism score and calculate $I_{3,j}$ by summing across all species i in j:

$$I_{3,j} = \sum_i \left[\left(\frac{\mathrm{d}R_{ij}}{\mathrm{d}X_j} \right)' \left(\frac{\mathrm{d}R_i}{\mathrm{d}R_{ij}} \right)' \right] = \sum_i \left[S\{G_i + Q_j\} \left(\frac{1}{N_i} \right) \right] \tag{5}$$

where $I_{3,j}$ is the third index of change in regional extirpation risk for all wetland species per protection effort in sub-basin j; $S\{G_i + Q_j\}$, the sensitivity value for the modified Global rarity class of species i; Q_j, the habitat quality modifier for sub-basin j, as described above; and other variables as previously defined.

Sub-basin ranking and mapping

We classify the distribution of scores from $I_{n,j}$ using the Fisher–Jenks procedure for determining natural break classes (Slocum 1999). The Fisher–Jenks algorithm determines inherent inflection points within a distribution. It is preferable to a quantile or equal area approach as it defines classes based on patterns in a distribution. We present our results as maps of the scores generated for each sub-basin by $I_{n,j}$ grouped and shaded by the Fisher–Jenks classification. We also present the raw numerical ranks of each sub-basin. A rank of one describes a sub-basin where 404 permit review should receive the highest priority in order to maximize the avoided increase in wetland species extirpation risk. However, numerical ranks should be interpreted with caution as they tell us little about the magnitude of differences for individual sub-basins, either between or within a class (Leibowitz and Hyman 1999; Hyman and Leibowitz 2000). Nevertheless, numerical ranks may be useful for management purposes as lower priority classes tend to contain many sub-basins.

Results and discussion

We first briefly discuss differences in results produced by the three indices, $I_{n,j}$. We then summarize the geographic distribution of sub-basin ranks and consider how spatial patterns in individual indicators may have influenced ranks. Finally, we discuss the application and limitations of our results in a management context.

Comparison of indices

Index scores from all three approaches were strongly correlated (Pearson's r ranged from 0.60 to 0.90, all $P < 0.0001$; γ correlations ranged from 0.73 to 0.83, with $P < 0.0001$). Moreover, the general spatial patterning in the ranks was qualitatively similar (Figure 4). Therefore, because selection of either set of indicators for estimating habitat quality produces similar results, errors in the final prioritization are

(a)

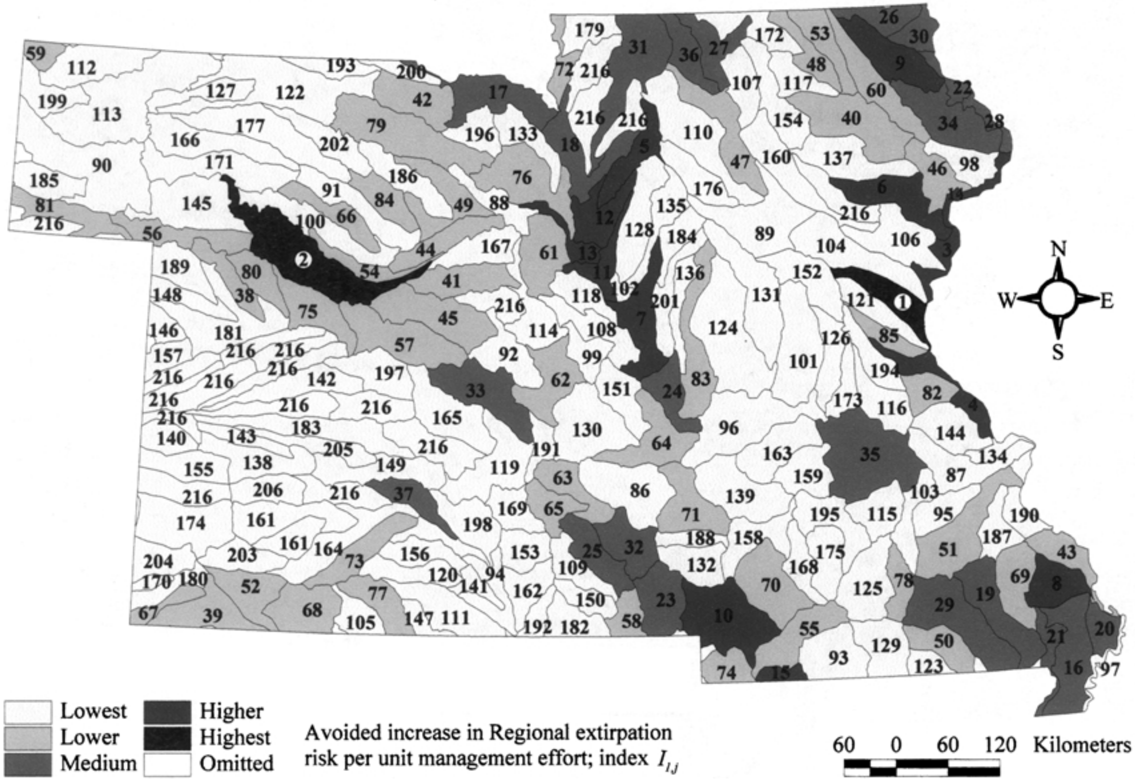

Figure 4. Mapped ranks of index $I_{1,j}$ (a), $I_{2,j}$ (b), and $I_{3,j}$ (c) describing avoided increase in Regional extirpation risk per unit management effort. Classification is by natural breaks. Raw numerical ranks are also given within each sub-basin. The darker shading (i.e., ranking scores near 1) indicate high criterion values or sub-basins within which 404 permits should receive high priority for review to derive the greatest ecological benefit from a given effort. Note that 14 sub-basins less than 43 300 ha in area on the periphery of Region 7 were omitted from analysis because we felt data from such small areas could be unreliable (these sub-basin slivers are left unshaded). This area (43 300 ha) corresponds to the smallest sub-basin entirely contained within Region 7; however, beyond this criteria our choice was somewhat arbitrary.

(b)

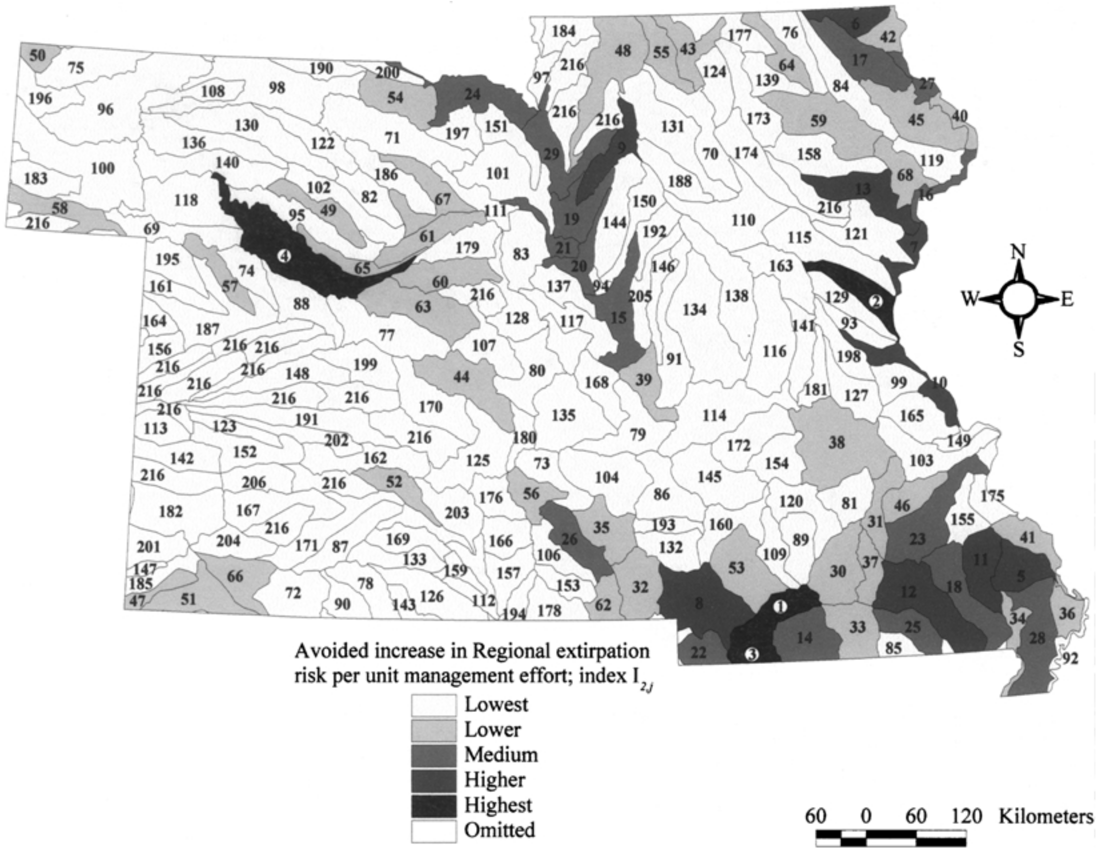

Figure 4. Continued.

(c)

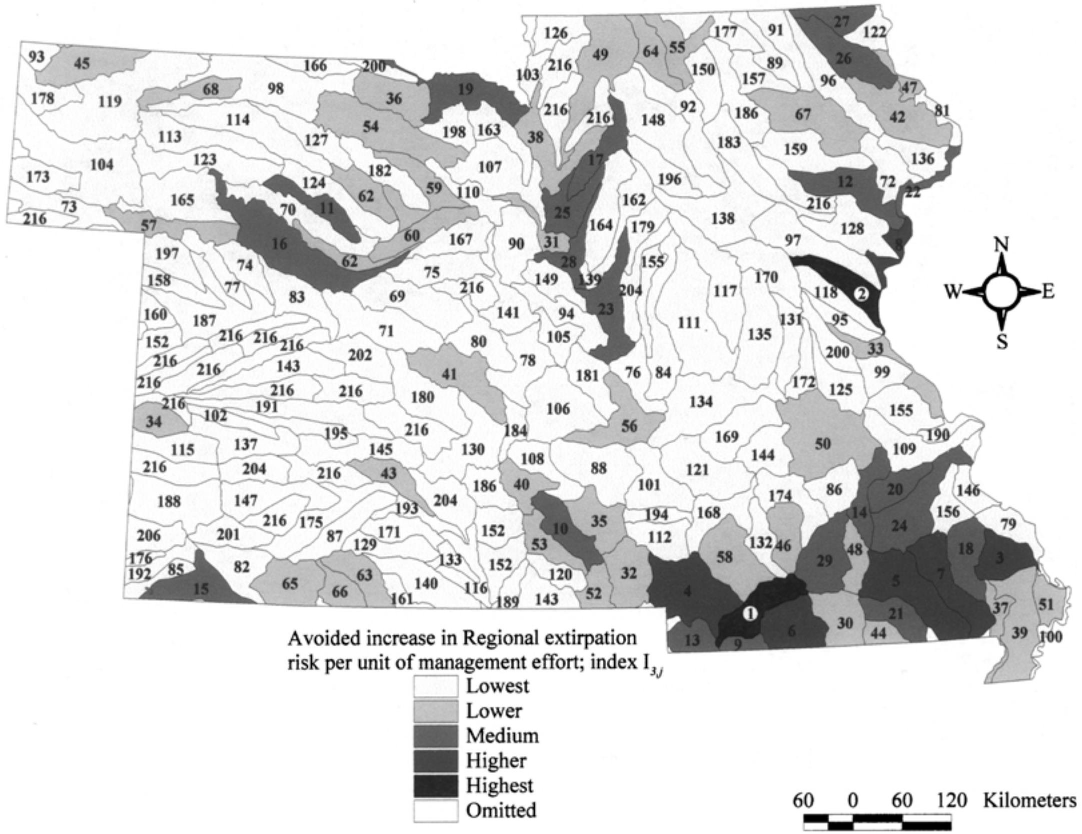

Figure 4. Continued.

more likely to be acceptable for regulatory purposes (McAllister et al. 2000). We choose to focus our interpretation on the results from the most parsimonious index, $I_{1,j}$ (Figure 4a). Focusing on $I_{1,j}$ minimizes the number of constituent layers and facilitates interpretation of spatial patterns in the final ranks. As this choice was somewhat arbitrary, we provide maps of the ranks for all $I_{n,j}$ (Figures 4a–c). Using index $I_{2,j}$ (Equation (4) and Figure 4b) adds complexity to the interpretation of final scores and several additional assumptions about relationships among the five habitat quality indicators in the index. However, the additional detail found in the $(\mathrm{d}R_{ij}/\mathrm{d}X_j)'$ term of $I_{2,j}$, may provide a more realistic characterization of landscape habitat quality for some resource management questions. Using $I_{3,j}$ (Equation (5) and Figure 4c) also adds complexity to the estimation process and assumptions about the structure of the multivariate space we defined (through PCA) to characterize and group sub-basin quality. However, $I_{3,j}$ emphasizes the occurrence of wetland species and the variance among sub-basins in their response to potential habitat loss which may be more appropriate for some management concerns.

Analyses of geographic patterns

Qualitative inspection of the spatial patterning of ranks generated from $I_{1,j}$ suggests that the distribution of priority sub-basins is non-random (Figure 4a). Sub-basins in the top two priority classes appear to fall in five general sub-regions within Region 7. Our results do not allow us to quantitatively explain why these five general areas appear to contain the majority of the high priority sub-basins in the region. However, we can qualitatively describe the more obvious patterns (based on visual comparisons of the appropriate maps) in the hopes of both assisting resource managers in using our results and motivating research into the large-scale and localized dynamics that may lay behind patterns in $I_{1,j}$.

Two of these high priority areas include: (1) the Neosho River in southwest Missouri and southeast Kansas and (2) the Mississippi alluvial plain in southeast Missouri (Figures 1 and 4a). Sub-basins in these areas tend to have high concentrations of sensitive and endemic species (most of the G1 species found in the region occur in these areas), relatively high agricultural land use and relatively low wetland density (except in the extreme southeast corner of the Mississippi alluvial plain). The high ranks of these sub-basins may be due to threats to endemic sensitive species and the lack of alternate wetland to which a displaced species might disperse.

Three additional sub-regions ranked as high priority for 404 permit review by $I_{1,j}$ are (1) along the Mississippi River on the eastern borders of Iowa and Missouri, (2) the Middle Platte River in central Nebraska and (3) along the Missouri River on the Nebraska–Iowa border (Figures 1 and 4a). Sub-basins in these areas tend to have many species with high sensitivity and endemism scores coupled with intensive agricultural land use, yet some of the *highest* density of wetland in the region. This pattern may suggest that the expectation that wetland density is negatively associated

with extirpation risk may be too general – wetland density alone may not describe an attribute of habitat quality that impacts wetland species. We included additional measures of landscape habitat quality in index $I_{2,j}$ (diversity of wetland type, average patch size and mean distance between patches). A visual inspection of the distribution of the constituent indicators in $I_{2,j}$ (not shown) suggests that these three sub-regions contain sub-basins characterized by wetland habitat that is dispersed across the landscape (corrected for sub-basin size). Therefore, while these areas may have relatively high wetland density, patches tend to be well spaced and separated by a matrix of (presumably) inhospitable habitat. The relatively simple set of measures in $I_{1,j}$ does not reveal this and thus the higher ranks of these three sub-regions may be a false positive error. Alternatively, a more species-specific measure such as the density of a specific type of wetland or a more dynamic measure of species responses over time may be more relevant.

A final pattern worth noting in the spatial distribution of $I_{1,j}$ and its constituent indicators is shown by sub-regions with sub-basins ranked as low priority for 404 review effort. These include the Sand Hills of north central Nebraska, the Western High and Central Great Plains of Kansas and the Western Corn Belt Plains of north central Iowa (Figures 1 and 4a). These areas tend to have relatively few sensitive or endemic species, a low density of intensive row-crop agriculture (with the exception of Iowa) and relatively low wetland density. With few wetland species exposed to anthropogenic pressures there may be little increase in risk of losing regional biodiversity by shifting 404 permit review effort to higher ranked sub-basins.

Assessment quality

Our methods by necessity incorporate a potentially sizable amount of error and several major assumptions. Every choice of an indicator for use in an assessment involves an implicit assumption about the validity of the data and the construct for its regional extrapolation. Moreover, assessments involve assumptions about the ways in which data are used and structured to provide the conclusions of the study. The synoptic framework explicitly formalizes these concerns. Leibowitz and Hyman (1999), using scale invariance and measurement theory (Stevens 1946), discuss the quality of inferences made within a synoptic assessment. Hyman and Leibowitz (2001) develop an approach for evaluating indicators when the specific mathematical relationship between indicators and concepts is unknown. Nevertheless, given the potential error in both the indicators we use and how we combine them, our final ranks should not be viewed as measuring actual change in risk, rather they are only an approximation to this reality.

Future work could include a variety of additions to our existing assessment. First, additional indices could be built to incorporate wetland functions other than biodiversity support – such as hydrologic and chemical buffering. This could contribute to a more complete picture of management priorities and enable different functions

to be optimized. Ando et al. (1998) show that the results of setting priorities change as a prioritization scheme accounts for more factors. Second, use of our sub-basin rankings could be extended beyond the goal of assisting EPA staff with prioritizing wetland 404 permitting. Rankings could serve as a criterion for prioritizing COE 404 review, approving grant proposals for a variety of EPA assistance programs that incorporate biodiversity support functions, or the listing of Superfund sites on the National Priority List based on potential impacts to wetland species. Third, our index does not consider the legal status of any species, e.g., whether or not a species is state or federally threatened or endangered. The occurrence of a status species within a management area is a paramount concern of decision-makers, as they are required by law to protect such a species. This could be addressed by performing a supplementary screening using threatened or endangered species status as a weighting factor. However, as we noted in the section on local population sensitivity, this information was considered inappropriate for our particular application. Fourth, species alliances or some other aggregation of taxa could be considered in place of individual species. This adds the challenging step of identifying these associations in the available data. However, the Heritage data currently contain information on the occurrence of unique community types within sub-basins. A synoptic assessment of the avoided risk to these communities with a unit of management effort may lead to useful classifications of the region. Fifth, indices could be constructed that incorporate alternate spatial units (ecoregions) or measures of habitat quality relevant to non-wetland habitat, allowing additional comparisons and uses of our results. An analysis of interactions among upland and wetland habitat, as many wetland species use both of these habitats to complete their life cycles, may be the most useful next step. Finally, obtaining data on the distribution of abundance or density across sub-basins would allow the assessment to incorporate local population sensitivity, which is an important source of between sub-basin variability.

Conclusions

A synoptic assessment is a management tool intended to assist in making decisions about resource allocation for maintenance of ecological function on a regional level. Our assessment does not prioritize actions on individual wetlands. Rather, our results should enable resource managers to place wetland site-specific decisions within a regional context and focus their efforts on sub-basins where functional potential is highest. Because our assessment is based on average conditions, the protection of wetlands in higher ranked units should, on average, avoid a larger increase in the risk of wetland species extirpation than protection in lower ranked units. However, there may be *individual* wetlands in lower ranked units that would provide greater biodiversity benefit than specific wetlands in higher ranked units. Thus, our assessment should serve as a screening tool to target overall effort and to identify areas

where more costly site-specific information should be obtained. Our rankings should be used in conjunction with political, social and economic information to support final decisions. Such an approach should result in enhanced protection of Region 7 wetland species biodiversity for a given cost, or lower cost for a given level of protection.

The synoptic approach was specifically designed to make use of judgement and available data in circumstances where scientific knowledge and opportunities for data collection are limited. Its use is appropriate when such information is required (for example, by regulation), the cost of improving existing information is prohibitive and the consequence of a wrong answer is low (Abbruzzese and Leibowitz 1997). The need for prioritizing 404 permitting efforts fits these requirements. However, it is important to emphasize that our results should not be treated as empirical or field-tested findings. The conclusions of the assessment are based on judgement guided by scientific principles and a general understanding of the relevant ecological processes. We have tried to use these judgement-based indicators in such a way so as to minimize certain errors (Leibowitz and Hyman 1999). Thus the results are somewhat akin to the conclusions of a scientist providing expert testimony at a trial. The assumptions incorporated into the assessment can be tested over time to improve the reliability of the results (Hyman and Leibowitz 2001). Improved data can be readily substituted for existing indicators as they become available. Verification of results through field studies and testing of assumptions is also critical to improve the assessment and our understanding of relevant regional processes.

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