A practical framework for selecting among single-species, community-, and ecosystem-based recovery plans

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Abstract: Science-based approaches for selecting among single-species, community-, and ecosystem-based recovery plans are needed to conserve imperilled species. Selection of recovery plans has often been based on past success rates with other taxa and systems or on economic cost, but less on the ecology of the system in question. We developed a framework for selecting a recovery plan based on the distributions and ecology of imperilled and nonimperilled species across available habitat types and applied it to fishes in the Sydenham River, Ontario, Canada. We first tested whether distributions of fishes were adequately predicted by habitat features hypothesized to limit the distributions of imperilled fishes versus a broader set of habitat features known to predict fish distributions. We then tested whether imperilled species occurred in similar or disparate habitat types. For the Sydenham River, an ecosystem-based recovery plan was deemed most appropriate because imperilled species occur in disparate habitat types. We lastly provide decision criteria to facilitate applications of our framework to the selection of recovery plans for other species and systems.

Résumé : Afin d’assurer la conservation des espèces en péril, il est nécessaire de posséder des méthodes basées sur la science pour choisir entre les plans de récupération centrés sur une seule espèce, sur une communauté ou sur un écosystème. Le choix d’un plan de récupération se base souvent plus sur les taux de succès obtenus avec d’autres taxons et dans d’autres systèmes ou sur le coût économique, mais moins sur l’écologie du système en question. Nous avons mis au point un cadre de sélection d’un plan de récupération basé sur la répartition et l’écologie des espèces menacées et non menacées dans les différents types d’habitats disponibles et nous l’avons utilisé dans le cas des poissons de la rivière Sydenham, Ontario, Canada. En premier lieu, nous avons vérifié si les répartitions des poissons pouvaient être adéquatement prédites d’après les caractéristiques de l’habitat que nous soupçonnions limiter les répartitions des poissons menacés, par comparaison à celles obtenues à partir d’un éventail connu et plus étendu de caractéristiques de l’habitat capable de prédire les répartitions de poissons. Nous avons ensuite vérifié si les espèces menacées se retrouvaient dans des types d’habitat similaires ou disparates. Dans la Sydenham, un plan de récupération à l’échelle de l’écosystème nous semble plus approprié car les espèces menacées vivent dans des types d’habitat disparates. Nous fournissons enfin des critères de choix pour faciliter l’utilisation de notre cadre dans la sélection des plans de récupération pour d’autres espèces et systèmes.

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Introduction

Science-based approaches for selecting between recovery plans are needed to protect the growing number of imperilled species listed by conservation agencies. Federal agencies in Canada, the United States, Europe, Australia, and elsewhere are mandated by legislation to develop recovery plans for species listed as endangered or threatened (Tear et al. 1995). With the rate of species imperilment increasing (Miller et al. 1989; Ricciardi and Rasmussen 1999; Abbitt and Scott 2001), so is the need for management strategies necessary to facilitate recovery (Abell 2002; Lundquist et al. 2002). To date, decisions regarding the selection of recovery plan type have been based on comparisons of past success rates (Boersma et al. 2001; Clark et al. 2002) or economics (Tear et al. 1995), while decisions based on the given biotic community are rarely assessed and are sorely needed (Clark and Harvey 2002).

Recycle of imperilled species is typically implemented by conservation agencies using three recovery planning types: single-species, community-, and ecosystem-based plans. Single-species plans focus on the threats and recovery of an individual imperilled species (Franklin 1993; Caughley 1994). Although single-species plans are relatively successful, they are often expensive to implement (Campbell et al. 2002). Plans that focus on the recovery of more than one imperilled species or on the ecosystem in which those species live have been proposed as alternatives to single-species plans.
recovery plans (Grumbine 1997; Yaffee 1999; Clark and Harvey 2002). Community recovery plans facilitate the recovery of groups of imperilled species that share the same ecosystem or groups of taxonomically related species facing similar influences (Clark and Harvey 2002). Ecosystem-based recovery plans focus more broadly on imperilled and nonimperilled species and their habitats (Franklin 1993). Each type of plan has been used in the management of imperilled species recovery, with varying levels of success (Boersma et al. 2001; Pikitch et al. 2004). For any recovery planning type to be effective, it must consider the biology of the imperilled species, the factors affecting its distribution and abundance within a given community, and the biology of the other nonimperilled species (Boersma et al. 2001; Clark and Harvey 2002).

The development of a science-based framework for the selection of recovery plans intuitively begins with the description of the relationship between imperilled species and their habitat. Loss of critical habitat is the leading cause of species imperilment; consequently, greater than 99% of recovery plans target habitat protection as an objective for species recovery (Tear et al. 1995; Abbott and Scott 2001; Dextrase and Mandrak 2006). However, as is often the case with imperilled species, biological information may be lacking and decisions for species recovery are needed regardless of data paucity (Foin et al. 1998). We use the identification of habitat-related factors (e.g., reduction, fragmentation, or modification) as a key starting point to discriminate between factors influencing imperilled species. We define imperilled species as those species designated as either threatened or endangered or those species given a conservation designation as suggested by the Committee for the Status of Endangered Wildlife in Canada. Although there are notable exceptions in factors that affect imperilled species (e.g., invasive species, disease, and over harvesting), our framework begins with the assumption that habitat-related factors are important first considerations when developing a recovery plan and that other factors can be included as they become known (see Dextrase and Mandrak 2006). Alternatively if other factors are known to influence the species or system in question, they should be included in the scenarios that we describe.

Three scenarios regarding the distribution of imperilled and nonimperilled species across available habitat types can be initially used to select a suitable type of recovery plan (Fig. 1). The first scenario considers an assemblage or community where imperilled species occur in a subset of habitats that are unique relative to those used by nonimperilled species (Fig. 1a). This scenario might be expected in systems where imperilled species are indicative of changes to specific habitats (Landres et al. 1988; Lawler et al. 2002). For example, declines in imperilled or sensitive species are often an indicator of broader habitat changes (e.g., habitat fragmentation in birds; pH in invertebrates and fish), where the larger suite of more common species has yet to be impacted (Jackson and Harvey 1993; Herkert 1994). For this scenario, single-species and community recovery plans could be implemented easily because mitigating the same set of habitat variables could be proposed to facilitate the recovery of all the imperilled species (Fig. 1a). However, if the habitat variables are altered, the risk of nonimperilled species being negatively affected remains.

The second scenario considers an assemblage or community where imperilled species occur together in a subset of the habitats used by nonimperilled species (Fig. 1b). This scenario might be expected where a system-wide habitat change negatively affects the entire suite of species or affects a subset of sensitive species. For example, changes in water temperatures through deforestation or loss of riparian zones are thought to affect both imperilled and nonimperilled species (France 1997; Casselman 2002). In this situation, single-species and community recovery plans would be well suited to facilitate the recovery of imperilled species because habitats could be modified to benefit all imperilled species (or a subset of sensitive species); accordingly, alterations to assist the recovery of imperilled species would have smaller effects on nonimperilled species because of the overlap in habitat preferences (Fig. 1).

The third scenario considers a system where different imperilled species occur in subsets of habitats and their geographic distributions overlap with nonimperilled species (Fig. 1c). This scenario is likely to occur where many environmental factors contribute to species declines. Under this scenario, single-species and community recovery plans may impede recovery of other imperilled species, as well as negatively affect nonimperilled species. For such a scenario, ecosystem-based recovery plans would be better suited than single-species or community recovery plans because improvements at a watershed–ecosystem level would presumably cause improvements to the myriad of species found in that watershed–ecosystem. Single-species or community recovery plans would only be feasible if the distributions of imperilled species were restricted to different geographical units (e.g., subwatersheds), where each could be managed separately for the threats occurring there. These three scenarios represent key examples from a continuum of scenarios, and specific scenarios and variables (e.g., other threats) can always be implemented as they occur.

We used four steps to explicitly apply our framework to imperilled fish species in the Sydenham River, Ontario, Canada (Fig. 1). We applied the framework to the Sydenham River watershed because this 2725 km² watershed is being used as a model system to develop scientific tools needed to support recovery planning for imperilled fish species, as required by the Canadian Species at Risk Act (Dextrase et al. 2003; Staton et al. 2003). The Sydenham River is species-rich for a north temperate system, with over 80 na-
tive and introduced fish species. First, we identified habitat features hypothesized in the literature to influence imperilled fish species in the Sydenham River. These species include the endangered northern madtom (*Noturus stigmosus*), the threatened eastern sand darter (*Ammocrypta pellucida*) and spotted gar (*Lepisosteus oculatus*), and species of special concern: greenside darter (*Etheostoma blennioides*), pugnose minnow (*Opsopoedus emiliae*), bigmouth buffalo (*Ictiobus cyprinellus*), and blackstripe topminnow (*Fundulus notatus*). These species are all currently encompassed within one recovery plan and are therefore considered collectively (Dextrase et al. 2003; Staton et al. 2003). Second, we tested whether those habitat features identified in the literature were good predictors of the types of habitats where the imperilled species were found. We conducted this test because recovery plans typically rely on species accounts from the literature without explicitly examining the adequacy and efficacy of the habitat features hypothesized to be influencing the distributions of imperilled species in the system they are managing. Third, we examined the habitat associations of imperilled and nonimperilled species to test among the three scenarios depicted in Fig. 1. Finally, we examined whether imperilled species were restricted to different subwatersheds that could be managed separately, as posed under scenario 3 (Fig. 1c).

**Materials and methods**

**Site selection**

Seventy-five sites were sampled across the Sydenham River watershed. Fifty sites were sampled in 2002, 25 of these were resampled in 2003 along with 25 new sites (Fig. 2). Sixty-two sites, at least 2 km apart, were selected to achieve uniform coverage across the watershed. Non-wadeable sections of the river near Shetland Conservation Area and in the lower portions of the watershed were not sampled. We defined each site as either a pool–riffle sequence or a stream reach approximately 60 m in length. Because much of the watershed has been impacted by agricultural drainage (Dextrase et al. 2003), many stream reaches have become channelized, lacking pool–riffle sequences. When a sample site was located within a channelized reach, sites were defined as a 60 m stream reach, following Bohlin et al.’s (1989) recommendation to use sites of roughly equal length.

**Fish sampling**

Fishes were collected with an electrofisher, seine nets, gill net and overnight sets of Windermere traps. Each site was sampled using single-pass electrofishing with a backpack Smith-Root model 12 electrofisher and using a seine net.
The type of seine net (bag or straight) used and the deployment of the remaining sampling methods were dependent on the depth of each site. A bag seine (8.2 m × 2 m × 2 m, 7.5 mm mesh), gill nets (5 cm stretched mesh), and Windermere traps (1 m diameter) were used at sites when depths were over 1 m. A straight seine (9.8 m × 2 m, 7.5 mm mesh) was used when depths were 0.5–1 m. A small straight seine (4.3 m × 0.6 m, 7.5 mm mesh) was used when depths were ≤0.5 m. Captured fishes were identified to species, counted, and either returned to the river (as the case for all imperilled species) or preserved as voucher specimens and sent to the Royal Ontario Museum (Toronto, Ontario).

Each site was systematically sampled by the electrofisher (pulsed DC current at 200–225 V, 60 Hz, pulse length = 3 ms) in an upstream direction at a rate of 5 electrofishing seconds per square metre (OMNR 2007). Total sampling effort depended on the area of the site; however, a minimum of 2000 s (mean 4257 ± 130 s) was shocked at each site, exceeding the recommendation of at least 1500 s proposed by Yoder and Smith (1999). The entire area of the site was sampled again by hauling a seine net in the downstream direction. The number of seine hauls varied from five to eight depending on the number of obstructions at the site. In total, the combination of electrofishing and seining captured over 95% of the imperilled species at sites where they were known to occur (Poos et al. 2007). Gill nets (5 cm stretched mesh) were used as block nets during sampling, unless a natural obstruction (e.g., shallow water) was present. Although absences were assumed as true absences because of the relatively large amount of sampling compared with others (e.g., Yoder and Smith 1999, OMNR 2007), we validated this assumption by modeling confidence intervals around each subsection and found all imperilled species had asymptotic relationships with a probability of detection >0.95 at the level we sampled (Poos et al. 2007).

Habitat measurement

Forty-two habitat variables were measured at each site. These represented habitat variables considered to specifically influence the distribution of imperilled fish species in the Sydenham River (Dextrase et al. 2003), as well as habitat variables considered to influence the distribution of stream fishes in general (Richter et al. 1997; Flather et al. 1998; OMNR 2007). Habitat variables were categorized as geomorphological, substrate, and chemical. Geomorphological variables measured attributes of the channel and stream hydrology and riparian buffers (OMNR 2007). Substrate variables measured the physical stream bottom where fishes were sampled (OMNR 2007), while chemical variables measured water quality at the sample sites (Table 1). Geomorphological and substrate variables were measured using the Ontario Stream Assessment Protocol (OMNR 2007). This protocol was selected because it has proven useful for testing hypotheses regarding how fish communities respond to habitat change and for developing conservation strategies (Stanfield and Jones 1998). In-stream measurements were recorded at six equally spaced points along 10 equally spaced transects, totaling 60 in-stream measurements per site. In-stream measurements included hydraulic head (HH) (±1 mm) as an index of velocity, water depth (±1 mm, AvgDepth), average bank undercut (±1 mm), percentage of sample points with nonfilamentous algae (NFL), and percentages of sand (median particle size 0.1 mm; labeled as PrSand), cobble (median particle size 20 mm; labeled as PrCobble), and clay (median particle size 0.01 mm; labeled as PrClay). Riparian measurements included percent riparian buffer (PrVegSq), amount of aquatic grasses (PrGrass), average bank particle size (±1 mm), and average bank angle.

Water quality variables were measured using a HydroLab DataSonde 4a multiprobe sensor. The sensor measured specific conductivity (±0.001 mS·cm⁻¹), turbidity (±50 nephelometric turbidity units, NTU), pH (±0.2), dissolved oxygen (±0.2 mg·L⁻¹), and nitrate concentration (±2 mg·L⁻¹·N⁻¹). These measurements were made approximately 20 m above the up-stream end of the sample site continuously for the duration of sampling. Measurements were averaged (± standard error) over the sampling duration for each site sampled.

Data analysis

Redundancy analysis was used instead of canonical correspondence analysis because a linear response model was considered appropriate given that the extent of species turnover (beta diversity) along the longest gradient of a detrended correspondence analysis (DCA) was less than 3 (Leps and Smilauer 2004). Further, the imperilled species captured in this study were all found in greater than 10% of the sites sampled (using all sampling methods) and as such did not suffer from zero-inflated bias as one would assume with most imperilled species (Martin et al. 2005). Species scores were standardized by dividing by their standard deviation so that species with large variance did not unduly influence the analysis (ter Braak and Smilauer 2002). Scaling was focused on interspecies correlations to interpret spatial relationships among species (ter Braak and Smilauer 2002).

The 42 habitat variables were reduced to a more interpretable number of variables in two steps. First, eight variables that were only suitable for measurement at a small number of samples sites (<0.1% of transect points) were removed. For example, the presence of watercress (Rorippa nasturtium-aquaticum), a groundwater indicator in temperate climates, revealed that the plant was found at only 0.02% of the transect points. Second, four derived variables were created by averaging multiple repeated measurements and using the average, instead of the individual measurements. These derived variables were average bank angle, average percent riparian buffer, average undercut, and average bank particle size. For example, the four riparian measurements of the slope of each bank were summarized as average bank angle. In total, 20 variables were included in the overall redundancy model (Table 1). A reduced redundancy model including only those habitat variables identified in the literature as influencing imperilled species found on the Sydenham River was also developed (Table 1). Statistical significance of individual axes was tested using Monte Carlo permutation tests (Leps and Smilauer 2004). After the first axis was tested, it was used as a co-variable in the test of the second axis, and so on, until all axes were tested for significance in explanatory power (Leps and Smilauer 2004). Only axes significant at p < 0.05 were included in subsequent sections.
The usefulness of habitat variables identified from the literature as influencing the imperilled species found on the Sydenham River was assessed by comparing the reduced redundancy model with the overall redundancy model. The comparison was made by using the variables isolated in the reduced model as co-variables and testing whether addition of the remaining variables in the overall model significantly increased the proportion of variance explained. Statistical significance of the improvement was tested using an unrestricted Monte Carlo permutation test (p < 0.01). The permutation test determined whether the variance in species’ occurrences explained by the additional habitat variables was greater than that expected by chance (Leps and Smilauer 2004).

**Evaluation of the scenarios**

Biplots of the first two habitat axes from the redundancy analysis were used to generate figures comparable to Fig. 1, visualize the habitat associations of imperilled and non-imperilled fishes, and distinguish among the scenarios. Locations where imperilled species were detected were plotted on maps using the geographic information system ArcView 9.1 (ESRI, Redlands, California) to determine whether the imperilled species were aggregations occurring in certain subwatersheds.

**Results**

We collected 67 fish species from the Sydenham River, including five imperilled species. In total 43 928 (2002 = 20 685; 2003 = 23 244) fishes were captured, with the imperilled species eastern sand darter, greenside darter, blackstripe topminnow, bigmouth buffalo and spotted sucker (*Minytrema melanops*) accounting for 0.19%, 9.20%, 1.10%, 0.07%, and 0.04% of the total abundance, respectively. We did not capture three imperilled fishes, the pugnose minnow, spotted gar, and northern madtom. Two of these species (northern madtom and spotted gar) are thought to be extirpated from this watershed, while the pugnose minnow was captured in nonwadeable sections not sampled in this study (N.E. Mandrak, unpublished data).

<table>
<thead>
<tr>
<th>Predicted variable (Measured variable)</th>
<th>Eastern sand darter&lt;sup&gt;a&lt;/sup&gt;</th>
<th>Greenside darter&lt;sup&gt;b&lt;/sup&gt;</th>
<th>Blackstripe topminnow&lt;sup&gt;c&lt;/sup&gt;</th>
<th>Bigmouth buffalo&lt;sup&gt;d&lt;/sup&gt;</th>
<th>Spotted sucker&lt;sup&gt;e&lt;/sup&gt;</th>
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<td>0 (+)</td>
<td>+ (+)</td>
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<td>Flow (HH)</td>
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<td>+ (+)</td>
<td>0 (–)</td>
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<tr>
<td>Nonfilamentous algae(NFL)</td>
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<td>+ (+)</td>
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**Note:** Predicted and observed signs of the associations between presence of an imperilled species and the habitat variable are shown (+, a positive correlation; –, a negative correlation; 0, no correlation). Observed associations appear in parentheses. NA, not applicable.

<sup>a</sup>Williams et al. 1989; Daniels 1993; Holm and Mandrak 1996; Facey 1998.


<sup>c</sup>McKee and Parker 1982; McAllister 1987; Dextrase et al. 2003.

<sup>d</sup>Scott and Crossman 1973; Stewart et al. 1985; Goodchild 1990.

<sup>e</sup>Trautman 1957; Scott and Crossman 1973; Page and Burr 1991.
sandy darter and greenside darter were found in habitats characterized by high hydraulic head (HH), and hence flow, large amounts of nonfilamentous algae (NFL), high proportions of cobble substrate (PrCobble), high amounts of riparian buffers (PrVegSq), and high concentrations of dissolved oxygen (AvgDO). Conversely, the blackstripe topminnow, bigmouth buffalo and spotted sucker were found in habitats characterized by deep water (AvgDepth), high turbidity (AvgTur), and high nitrate concentration (AvgNO3).

The habitat variables hypothesized as important in literature accounts of the imperilled species were found to be good predictors of the occurrence of imperilled species in the Sydenham watershed. The redundancy analyses demonstrated that the model based on a broad suite of habitat variables known to influence stream fishes in general did not provide a significantly better fit than a model based solely on those habitat variables believed to influence imperilled species (14.5% for the reduced redundancy model versus 17.3% for the overall redundancy model, p > 0.05). In addition, there was a significant relationship between the habitat variables believed to influence imperilled species and the occurrence of the fishes in general (axis 1: eigenvalue = 0.1; F = 5.3; p = 0.001; axis 2: eigenvalue = 0.04; F = 2.6, p = 0.002).

Our consideration of variables not identified for imperilled species in the literature identified two additional variables that were helpful in predicting the occurrence of imperilled species in the Sydenham River. The variables were percent clay substrate (PrClay) and pH (Fig. 3d). Adding those variables to the reduced redundancy model explained an additional 2.6% of the variance in the occurrence of fish species (eigenvalue = 0.017, F = 2.07, p = 0.003; eigenvalue = 0.014, F = 1.75, p = 0.01, respectively). Only axis 1 was significant (axis 1: eigenvalue = 0.019; F = 1.9; p = 0.01; axis 2: eigenvalue = 0.012; F = 0.87, p = 0.32). The occurrence of the eastern sand darter, blackstripe topminnow, and bigmouth buffalo tended to be in habitats characterized by low amounts of clay and low pH. Spotted sucker also tended to be in habitats characterized by high amounts of clay and high pH. The occurrence of the greenside darter was unrelated to the amount of clay and water pH (Table 1; Fig. 3).
Discussion

As rates of species imperilment and habitat degradation increase, meeting the challenges of species recovery has become increasingly difficult (Foin et al. 1998; Simberloff 1998). One strategy to circumvent this downward trend has been to reduce the emphasis on species-specific recovery plans (which require more effort per species) and use ecosystem-based recovery plans, where several imperilled species can be managed across entire systems, such as watersheds (Grumbine 1994; Brunner and Clark 1997). Although ecosystem-based recovery plans may be scientifically appropriate, the willingness of managers to adopt these strategies may be difficult given the past success of single-species recovery approaches, which have been four times more likely to exhibit recovering status trends for imperilled species (Boersma et al. 2001; Clark and Harvey 2002). Of the number of limitations addressed for shortcomings of ecosystem-based recovery plans, the lack of integration of ecologically relevant information into recovery plans are almost exclusively linked with lack of improved status (Tear et al. 1995; Clark et al. 2002). This may be more related to discrepancies between the type of recovery plan chosen and the specific conservation tasks identified for recovery in the biological system or species in question rather than a specific type of recovery plan (Lundquist et al. 2002). For example, the lumping of species into community plans is often not based on ecologically defensible criteria (e.g., similarity of threats) or represents inconsistent associations with preferred habitat ranges (Clark and Harvey 2002; Rahn et al. 2006), which is counterintuitive to the goals of that recovery plan. This study introduces a framework to improve the implementation of selecting an appropriate recovery plan type by providing a basis for incorporating species-specific associations directly into decision criteria (Fig. 4). The goal of using decision criteria is to incorporate an understanding of species–habitat relationships for the selection and implementation of an appropriate type of recovery plan.

Classification and validation of habitat variables thought to be critical for the conservation of imperilled species is fundamental to the presented framework because habitat characteristics often become key components of species recovery plans (Tear et al. 1995), under the assumption that they are important, but often without appropriate validation (Gerber and Hatch 2002). Our application quantitatively demonstrated that habitat variables identified in the literature as influencing the distributions of imperilled fish species found in the Sydenham River were good predictors of the habitats where those species were found; however, this scenario may not always occur. Validation of habitat variables can be specifically useful for isolating uncertain predictors or for identifying new habitat predictors not previously recognized in the literature. We consider the classification and validation of habitat characteristics as preliminary steps in the decision framework (Fig. 4). These steps are necessary for the success of the framework because plans that incorporate species-specific biology are typically more successful than those that do not (Boersma et al. 2001; Lawler et al. 2002; Lundquist et al. 2002). Further, identification of new, important habitat predictors of imperilled species (through validation procedures) may lead to the development of better recovery plans by improving our knowledge of habitat associations for imperilled and nonimperilled species. The discovery of previously unimportant habitat predictors will increase knowledge of the biology of species inhabiting the management area, thereby increasing rigor to the assessment of scenarios 1–3.

The framework and decision criteria presented here represent spatially explicit criteria from which to base management decisions for communities with imperilled species. By considering the effect of species threats with geographic location, there is a high likelihood that the success rate of community- and ecosystem-based plans can be improved. For example, we found that imperilled species in the Sydenham River were found in different habitat types and were not restricted to different subwatersheds, which suggests considerable challenges to recovery planning using community recovery plans. Such situations can be expected to pose constraints whereby actions taken to assist the recovery on one imperilled species will potentially hamper the recovery of another. In addition, strict calculation of threat similarity (Clark and Harvey 2002) may not encompass the geographic extent from which those threats should be mitigated. For example, the Sydenham River recovery strategy has focused on reducing turbidity to mitigate threats to critical habitat (Dextrase et al. 2003). Reducing turbidity throughout the watershed is predicted to benefit the eastern sand darter and greenside darter, because both species occur at sites characterized by low turbidity. However, reducing turbidity could negatively affect the blackstripe topminnow, bigmouth buffalo, and spotted sucker because these species occur at sites characterized by high turbidity. Single-species or community recovery plans could be effective if the eastern sand darter and greenside darter were primarily found in the less turbid eastern subwatershed and the blackstripe topminnow, bigmouth buffalo, and spotted sucker were primarily found in the turbid northern subwatershed. Such a scenario would require partitioning of recovery actions for each watershed so that each subwatershed could be managed differently. However, this is not the case in the Sydenham River, whereby, supporting selection of an ecosystem-based recovery plan and not a community plan (Fig. 4). Recovery actions focusing on single factor remediation, such as the reduction of turbidity, need to be tempered with the trade-offs in species-specific responses, specifically that any shift may be at the detriment of at least some imperilled species. Although this type of situation may ultimately doom ecosystem-based approaches to lower success rates (by reducing the viability of some imperilled species), presumably there would be a net benefit to the other imperilled and nonimperilled species by improving habitat quantity and quality through an ecosystem approach.

A stronger, science-based approach using decisions tailored to the organisms and system of concern can help with the selection of a recovery plan type. With the rate of species imperilment increasing (Miller et al. 1989; Ricciardi and Rasmussen 1999; Abbitt and Scott 2001), so is the need for recovery plans defining management strategies necessary to facilitate recovery (Abell 2002; Lundquist et al. 2002). To date, recommendations regarding the choice of recovery plan have been based on either the past success rates with other taxa and systems (Boersma et al. 2001; Clark et al. 2002) or economic cost of the different recovery plans.
I) Classification of factors limiting imperilled species
   1) Have the factor(s) been characterized?
      - YES
      - NO → Characterize factor(s) using best available knowledge.

II) Validation of factors limiting imperilled species
   2) Have the factors thought to influence
      the imperilled species been validated for the study area
      and species in question?
      - YES → Conduct analyses to validate the factor(s).
      - NO

III) Scenario evaluation and selection of recovery plan type
   3) Do the imperilled species share associations with
      common species?
      - YES
      - NO (Scenario 1)

   4) Are the imperilled species
      geographically restricted where
      a subset of factors can be mitigated?
      - YES → 3-5b) Is there more than one imperilled species
              in the system in question?
      - NO (Scenario 2)

   5) Are the imperilled species influenced by contrasting
      factors from one another (e.g. turbidity)?
      - YES (Scenario 3)

- Ecosystem-based recovery plan most appropriate
- Community recovery plan most appropriate
- Single-species recovery plan most appropriate

(Tear et al. 1995) and not necessarily on the ecology of the system in question. While past success and economics are important considerations for choosing a type of recovery plan, they do not explicitly recognize that the success rate and cost of each type of recovery plan may vary according to the ecological situation in which the recovery plan is applied. For example, the higher success rates observed for single-species recovery plans relative to community plans found by Boersma et al. (2001) may be a consequence of the former being implemented in simple systems and of the latter implemented in more complex, challenging systems. Alternatively, the cost effectiveness of community recovery plans should be improved when the recovery plan incorporates knowledge of local conditions, and this should also im-
prove the rate of recovery (Grumbine 1994). In addition, both single-species and community approaches may be more cost effective in situations where factors contribute similarly to communities with imperilled species, such as an increase in water temperature through the loss of riparian buffers, and where recovery can be achieved in a relatively short time frame (Lawler et al. 2002).

Of course, the framework developed has limitations and understanding these can help identify where they can be applied most successfully and how it might be improved. First, implementing the framework requires quantifying the species–habitat associations, but it does not demonstrate cause and effect relationships. Therefore, habitat modification as a recovery action, such as reducing turbidity, may not necessarily lead to corresponding changes in the abundances of imperilled species due to the complex ways in which species interact with the environment and the effects of other factors not measured, such as connectivity among suitable habitats. However, cause and effect could be tested subsequently either in formal field experiments or through adaptive management. Second, the variance explained from multivariate analyses (e.g., redundancy analysis, canonical correspondence analysis) will tend to be lower than their univariate counterparts (e.g., logistic regression, least squares regression, analysis of variance) and should not be taken as a surrogate for poor fit of the model per se (Legendre and Legendre 1998). In this case, the low variance on the multivariate axes represent the overall ability of the environmental correlates in describing the relatively large species assemblages ($n = 67$) across the relatively large number of sites sampled ($n = 75$) and as such is not surprisingly low. Although we should always strive for rigorous species assessments, in many cases extensive sampling of the kind preformed here may not be feasible. As is often the case with endangered species, biological information may be sparse for rare species and decisions for species recovery are needed regardless (Foin et al. 1998). In general, the models represent the integration of the best available knowledge across the study system and species in question. As the decision criteria presented here (Fig. 4) does not incorporate explicitly statistical methods, care should be used to ensure appropriate models are applied. Finally, we did not consider other imperilled taxonomic groups, such as mussels (Metcalfe-Smith et al. 2003) or benthic macroinvertebrates or the imperilled fishes not captured in our sampling. The addition of other imperilled species may provide more insight as to which habitat characteristics would be best mitigated in the situation where several imperilled species need to be alleviated with contrasting habitat types. Indeed, the majority of the declines attributed to imperilled mussels are thought to be due to turbidity (Dextrase et al. 2003). The inclusion of other imperilled species would, therefore, improve the implementation of an ecosystem-based recovery strategy, and such data should be included when implementing the most appropriate type of recovery plan. However, in our situation, adding the imperilled species not sampled would not have altered our results or the choice of which recovery plan type would be most appropriate. The reason for this is that as imperilled species are added to the multivariate data, the more likely that those species will be found with some aspect of habitat that is contrasting with another imperilled species without geographic isolation (scenario 3).

This framework has a number of strengths that may allow it to be transferred to other species and systems. First, the framework for the selection of recovery plan type can be standardized and is straightforward and transparent. Second, the framework is rigorous by using strong science to validate proposed habitat predictors and to test among competing scenarios and is comprehensive by explicitly considering imperilled and nonimperilled species. Third, the framework presents opportunities to learn, for example, through identifying new habitat predictors to include in the community model (e.g., pH or percent clay in our model). Fourth, the framework lends itself to an adaptive approach as biological information becomes more available, limiting factors become validated, and extinction risks become better quantified.

Finally, the framework is flexible in the use of most multivariate statistical methodologies, insofar that they are used appropriately and can distinguish between the scenarios we present. These advantages may lead to improved success with species recoveries by lessening the use of inappropriate recovery plans that are based solely on past success and tight fiscal resources (Grumbine 1994, 1997; Boersma et al. 2001). As the recovery of imperilled species will be hindered if the objectives of recovery plans cannot be met (Boersma et al. 2001; Lundquist et al. 2002) and as loss of imperilled species rises, it is becoming increasingly important that conservation actions, such as the selection of recovery plans, be transparent, sound, and scientifically defensible.

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