Landscape Ecol (2018) 33:2239–2252
https://doi.org/10.1007/s10980-018-0738-5

RESEARCH ARTICLE

Scale-specific land cover thresholds for conservation of stream invertebrate communities in agricultural landscapes

Jeremy P. Grimstead · Edward M. Krynak · Adam G. Yates

Received: 27 June 2018 / Accepted: 29 October 2018 / Published online: 10 November 2018

Abstract
Context In agricultural landscapes, riparian forests are used as a management tool to protect stream ecosystems from agricultural activities. However, the ability of managers to target stream protection actions is limited by incomplete knowledge of scale-specific effects of agriculture in riparian corridor and catchment areas.
Objectives We evaluated scale-specific effects of agricultural cover in riparian corridor and catchment areas on stream benthic macroinvertebrate (BMI) communities to develop cover targets for agricultural landscapes.
Methods Sixty-eight streams assigned to three experimental treatments (Forested Riparian, Agricultural Riparian, Agricultural Catchment) were sampled for BMIs. Ordination and segmented regression were used to assess impacts of agriculture on BMI communities and detect thresholds for BMI community metrics.
Results BMI communities were not associated with catchment agricultural cover where the riparian corridor was forested, but were associated with variation in catchment agriculture where riparian forests had been converted to agriculture. Trait-based metrics showed threshold responses at greater than 70% agricultural cover in the catchment. Increasing agriculture in the riparian corridor was associated with less diverse and more tolerant BMI communities. Eight metrics exhibited threshold responses ranging from 45 to 75% agriculture in the riparian corridor.
Conclusions Riparian forest effectively buffered streams from agricultural activity even where catchment agriculture exceeds 80%. We recommend managers prioritize protection of forested riparian corridors and that restore riparian corridors where agricultural cover is near identified thresholds be a secondary priority. Adoption of catchment management actions should be effective where the riparian corridor has been converted to agriculture.

Keywords Agriculture · Benthic macroinvertebrates · Catchment · Riparian corridor · Scale-specific effects · Threshold analysis

Electronic supplementary material The online version of this article (https://doi.org/10.1007/s10980-018-0738-5) contains supplementary material, which is available to authorized users.

J. P. Grimstead · E. M. Krynak · A. G. Yates
Department of Geography, Western University and Canadian Rivers Institute, London, ON, Canada

A. G. Yates (✉)
Social Science Centre, Western University, 1151 Richmond St., Rm 2332, London, ON N6A 5C2, Canada
e-mail: adam.yates@uwo.ca

Introduction

Riparian corridors are the interfaces between aquatic and terrestrial ecosystems that include the stream channel and adjacent terrestrial area hydrologically influenced by the stream (Naiman et al. 1993). The importance of riparian corridors to the structure and function of stream ecosystems is well established, particularly for small streams with forested riparian corridors (see reviews by Gregory et al. 1991; Naiman and Decamps 1997). Forest canopies control stream microclimates, including moderating water temperature fluctuations and regulating light availability for primary production (Moore et al. 2005). Riparian forests also add woody debris to streams, which plays an important role in determining channel morphology and stream flow by increasing channel heterogeneity, trapping sediment and organic material, slowing and redirecting currents, as well as providing habitat (Gurnell et al. 1995). Moreover, riparian forests perform critical ecological functions including acting as a source of allochthonous organic matter for aquatic organisms through deposition of leaf litter (Benfield 1997). Riparian corridors are also biogeochemical hot spots which, combined with hydrologic processes, regulate nutrient and contaminant loading to streams (Vidon et al. 2010). Cumulatively, the functions of the riparian corridor are predicted to increase stream resilience by buffering effects of catchment disturbances, such as conversion and management of upland areas for agricultural uses (Allan 2004a).

Studies of the buffering effects of riparian forests have focused on the lateral extent of forest required to filter agricultural pollutants and maintain ecological communities (reviewed by Sweeney and Newbold 2014). These studies generally agree that a buffer width of at least 30 m effectively mitigates nutrient and sediment loadings and is capable of protecting ecological communities. In contrast, the longitudinal dimension of the riparian corridor has received less attention (but see Feld 2013), and managers lack knowledge of the proportion of forested riparian corridor required to protect stream communities from agricultural impacts.

Numerous ecological studies have found spatial patterns in stream communities to be associated with agricultural and forest cover in the riparian corridor and catchment areas (e.g., Yates and Bailey 2010; Feld 2013; Marzin et al. 2013). Studies have also used inverse distance models to demonstrate that land cover in the riparian corridor is more strongly associated with community composition than land cover in the catchment area (Van Sickle and Johnson 2008; Peterson et al. 2011). Moreover, studies have assessed ecological benefits of individual riparian management projects, although with generally equivocal results (Greenwood et al. 2012). However, past studies have not simultaneously controlled the amount of agricultural cover at the riparian corridor and catchment scales making it difficult to disentangle cumulative and interactive effects of agriculture on ecological communities. Isolating scale-specific effects of agricultural cover is thus an important step towards development of management schemes that effectively target stream conservation activities in agricultural landscapes.

Environmental thresholds are increasingly used to inform land management decisions as a threshold can serve as an effective target for protection of a known level of ecosystem integrity (Dodds et al. 2010). To date, land use thresholds have typically been identified at the catchment scale (Allan 2004b). Indeed, there has been substantial research linking land use thresholds to biotic integrity of stream ecosystems at the catchment scale, although most of these studies have focused on urban environments (e.g., Hilderbrand et al. 2010; King and Baker 2010), with less work in agricultural landscapes (Utz et al. 2009; Waite 2014). However, there has been limited research to identify scale-specific thresholds at which agricultural land cover at the catchment or riparian corridor scale overwhelms mitigating effects of riparian vegetation and alters aquatic communities (but see Feld 2013; Waite 2014).

In this study, we evaluated scale-specific effects of agricultural land cover in the riparian corridor and catchment areas on stream benthic macroinvertebrates (BMIs) communities to address two research goals. First, we assessed associations between BMI community composition and agricultural cover in the riparian corridor and catchment scales. Second, we identified scale-specific thresholds in associations between twelve common BMI metrics and agricultural cover in the catchment and riparian corridor. We applied our findings to generate a preliminary prioritization scheme to enhance protection and restoration of stream ecological conditions through management of riparian corridors in agricultural landscapes.

Materials and methods

Study area

Our study was conducted on 68 headwater streams in southern Ontario, Canada (Fig. 1). Surrounded by the Laurentian Great Lakes, the southern Ontario region experiences a humid, continental climate with average temperatures ranging from a high of approximately 27 °C in July and a low in January of − 10 °C (Environment Canada and Climate Change 2016). Average annual precipitation is approximately 1025 mm with monthly averages ranging from 35 to 163 mm (Environment Canada and Climate Change 2016). The physiography of southwestern Ontario is comprised of glacial deposits overlying carbonate-rich Paleozoic bedrock. Land cover in the region is characterized by patches of deciduous forests in an otherwise agriculturally dominated landscape. Agricultural activities are a mixture of row crop cultivation (e.g., corn and soybean) and high-density livestock farms, including swine, dairy, and poultry. Regional drainage patterns have been modified by the installation of tile drainage in agricultural lands and channelization of the stream network.

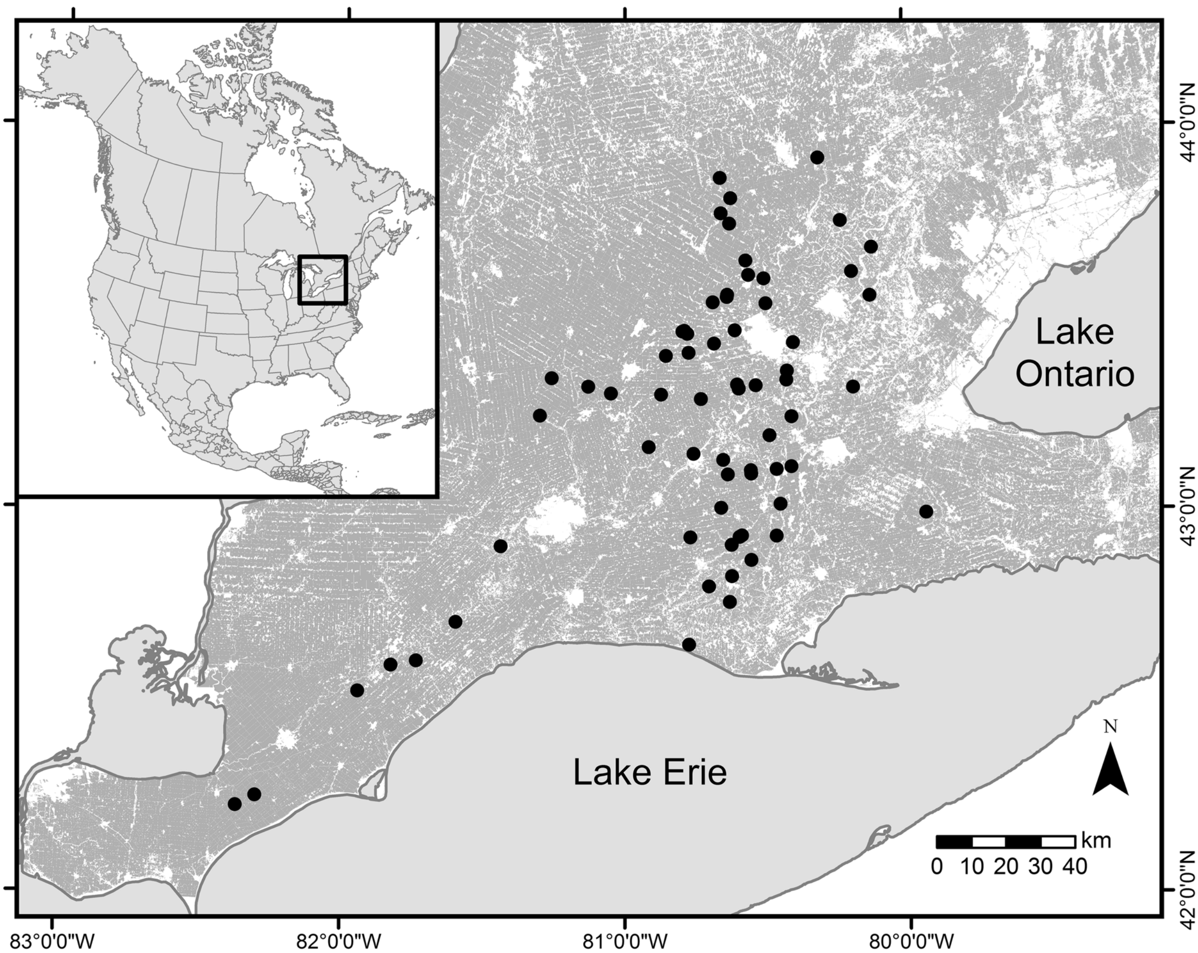


Fig. 1 Location of 68 study sites (black circles) used to assess independent effects of agricultural land cover (gray shading) in the riparian corridor and catchment areas on benthic macroinvertebrate communities collected from headwater streams in the southern Ontario portion of the Laurentian Great Lakes in eastern North America (inset)

Study design and site selection

Our study assessed the responses of BMIs to spatial heterogeneity of agricultural land cover at two spatial scales: the catchment and riparian corridor. We defined the catchment as the land area draining into a specified drainage point that constituted the sampling location. The riparian corridor was defined as a 40 m wide buffer on either side of the stream, extending the length of the stream segment (defined as section of stream between two confluences) upstream of the drainage point. Catchment and riparian corridor areas were defined using a geographic information system (GIS) and available GIS layers describing the stream network (National Hydro Network, Natural Resources Canada) and regional topography (ASTER V2, global digital elevation map; NASA).

Sites were screened using a GIS to minimize among-site variation in environmental attributes unrelated to agricultural land cover by ensuring sites had similar catchment area (5–15 km^2) and physiography (i.e., topography and surface geology). Thus, only sites that drained catchments characterized by coarse, glacial outwash deposits and had sand to fine gravel stream substrate were used. Furthermore, channelized streams and sites with urban land cover in the catchment or riparian corridor were excluded from the study. Lastly, all sites were non-nested to ensure independence among sites.

A space-for-time substitution was used to generate three treatments that evaluated effects of agricultural cover in the catchment and riparian corridor (Fig. 2). For each treatment, variation in land cover at one scale was minimized (hereafter control scale), whereas variation in the other scale was maximized (hereafter gradient scale) (Fig. 2a). The first treatment (hereafter Forested Riparian) included streams (n = 24) with forest cover in the riparian corridor and a gradient of agricultural land cover in the surrounding catchment (Fig. 2b). In contrast, streams (n = 20) in the second treatment (hereafter Agricultural Riparian) were characterized by a riparian corridor dominated (i.e., minimum of 75%) by agricultural cover and a range of agricultural land cover at the catchment scale. Streams (n = 43) in the third treatment (hereafter Agricultural Catchment) were exposed to a gradient of agricultural cover at the riparian corridor scale, whereas agricultural cover dominated (i.e., minimum of 80%) the catchment scale. Characteristics of sites were sometimes suitable for multiple treatments thus seven sites were included in both the Forested Riparian and Agricultural Catchment treatments and a different twelve sites were used for both the Agricultural Riparian and Agricultural Catchment treatments. Extensive agricultural cover within southern Ontario limited the range of agricultural cover in our study at the catchment scale. Indeed, sites exposed to less than 50% agricultural cover at the catchment scale were rare. Furthermore, in some cases, reduced variation in the control scale was traded-off with maintaining a reasonable sample size for analyses.

Land cover at the catchment and riparian corridor scales was described for each site by intersecting a land cover layer generated in 2015 by Agriculture and Agri-food Canada (AAFC) with the defined boundaries. The area of forest and agriculture in the boundaries was then divided by the total area to determine the percentage of the catchment and riparian corridor accounted for by each cover type. Because samples included in this study were collected in different years we assessed temporal changes in land cover between the AAFC layer and the Southern Ontario Land Resource Information System (SOLRIS 2.0) generated by the Ontario Ministry of Natural Resources and Forestry using land cover information from circa 2007. This assessment showed no demonstrable changes in land cover in the studied catchment and riparian areas between 2007 and 2015. As a result, data from the AAFC layer was used for all sites to maintain consistency in cover definitions and layer resolution.

Field sampling protocols

BMI community data was obtained from past monitoring data by querying site information that met the stated study design criteria. Selected samples were collected and processed in 2006, 2007 or 2015 using the CABIN (Canadian Aquatic Biomonitoring Network) protocol (Reynoldson et al. 2012). In brief, a D-frame net equipped with 400 μm mesh was used for a 3-min traveling kick. All habitats present within the defined sampling reach (i.e., six times the bankfull width) were sampled in proportion to their occurrence. Collected invertebrates were fixed using 10% buffered formalin and later preserved in 75% ethanol. Samples were subsampled at random until a minimum of 5% of

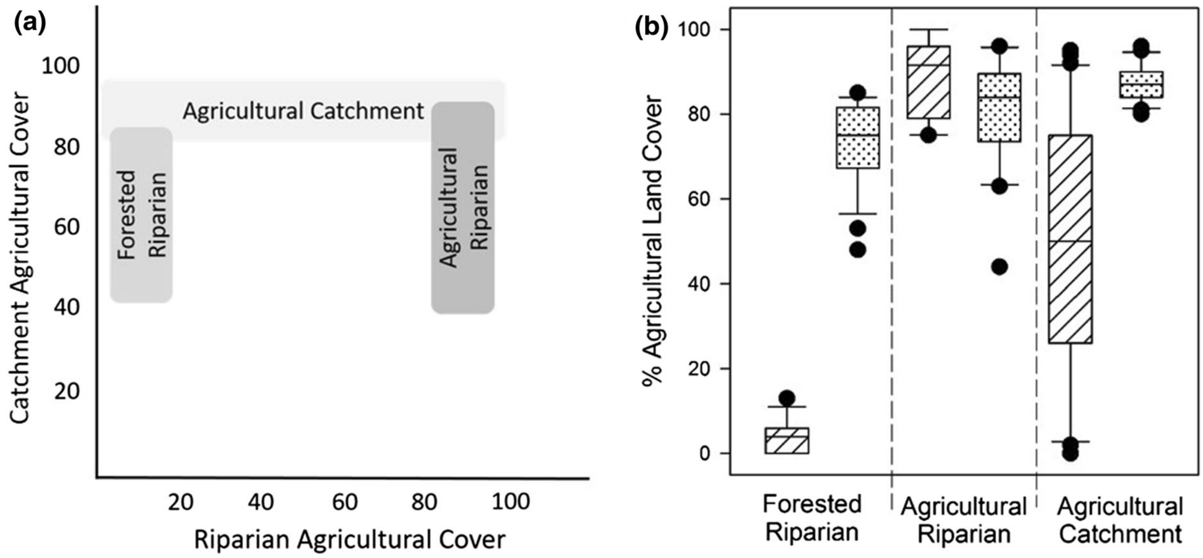


Fig. 2 Schematic plot (**a**) indicating approximate agricultural cover patterns (gray shaded areas) at the catchment and riparian corridor scales for the three experimental treatments. Boxplots (**b**) showing median (middle line), quartiles (box), 95th percentile (whiskers) and outliers (filled circles) for percentage of agricultural land cover in the riparian corridor (hatch pattern) and catchment areas (stippled pattern) for three study treatments: Forested Riparian (n = 24), Agricultural Riparian (n = 20) and Agricultural Catchment (n = 43)

the sample or 300 individuals were subsampled, whichever required more of the sample to be processed. All enumerated invertebrates were identified to the lowest taxonomic unit practical, usually genus or family.

Habitat conditions at each sampling reach were assessed using the United States Environmental Protection Agency's rapid habitat assessment protocol for low gradient streams (Barbour et al. 1999) to ensure among sites differences were primarily associated with spatial heterogeneity of agricultural cover. This habitat assessment qualitatively assigns each of 10 habitat characteristics a score on a scale of 0–20. Because streams were sampled over different years, the water level score was not included in these assessments. Furthermore, the channelization score was not included as the presence of channelization was used to filter sites during the site selection process. To simplify comparison we summed the scores of the remaining eight habitat characteristics into three groups based on whether the characteristics describe aspects of substrate (epifaunal substrate/available cover, pool substrate and sediment deposition), channel form (pool variability, channel sinuosity and bank stability) or riparian vegetation (vegetative protection and riparian vegetation width).

Data analysis

Collinearity between habitat quality and agricultural cover at the treatment scale were assessed using Pearson's correlation analysis in SYSTAT 13 (2015). Individual analyses were carried out for each of the three habitat groups (i.e., Substrate, Channel Form and Riparian Vegetation) within each of the treatments (i.e., Forested Riparian, Agricultural Riparian and Agricultural Catchment). Associations were considered significant for p-values less than 0.05.

Multivariate analyses of BMI community composition for each treatment was conducted using separate non-metric dimensional scaling (NMDS) analyses of taxonomic and trait-based descriptions of the communities. Trait modalities were assigned to each taxon as per Krynak and Yates (2018) using genus level information whenever possible. Family level descriptions were used for taxonomic analyses to reduce the number of rare taxa. Rare taxa were removed if a taxon was found at five percent or less of the sites in each treatment. Taxonomic and trait-based descriptions of the community were represented as abundance (i.e., count) data and as presence/absence data leading to four different matrices describing community composition for each treatment. Abundance data were transformed using a Hellinger transformation prior to calculation of among site dissimilarities using Euclidean distance. Jaccard's distance was used to calculate among-site dissimilarities for presence/absence data.

Relationships between descriptions of BMI assemblage composition and gradients of agricultural cover for each of the three treatments (i.e., Forested Riparian, Agricultural Riparian and Agricultural Catchment) were separately assessed by fitting both linear and non-parametrically smoothed surfaces to NMDS ordinations. Generalized additive models (GAMs) were used to fit smoothed surfaces. GAMs used thinplate splines in two dimensions with the amount of smoothing determined by generalized cross-validation. Coefficients of determination (R^2) were used as a measure of goodness-of-fit and significance was tested using 999 permutations. The resultant R^2 for the linear trend and fitted surface were interpreted following Virtanen et al. (2006). If the response was linear, the fitted surface and vector would have equivalent R^2 values, whereas if the surface R^2 was greater the response was considered non-linear. All ordination and fitting analyses were performed in the R statistical environment, version 3.3.2 (R Core Team 2016) using the R function *ordisurf* in the package 'vegan' (Oksanen et al. 2017).

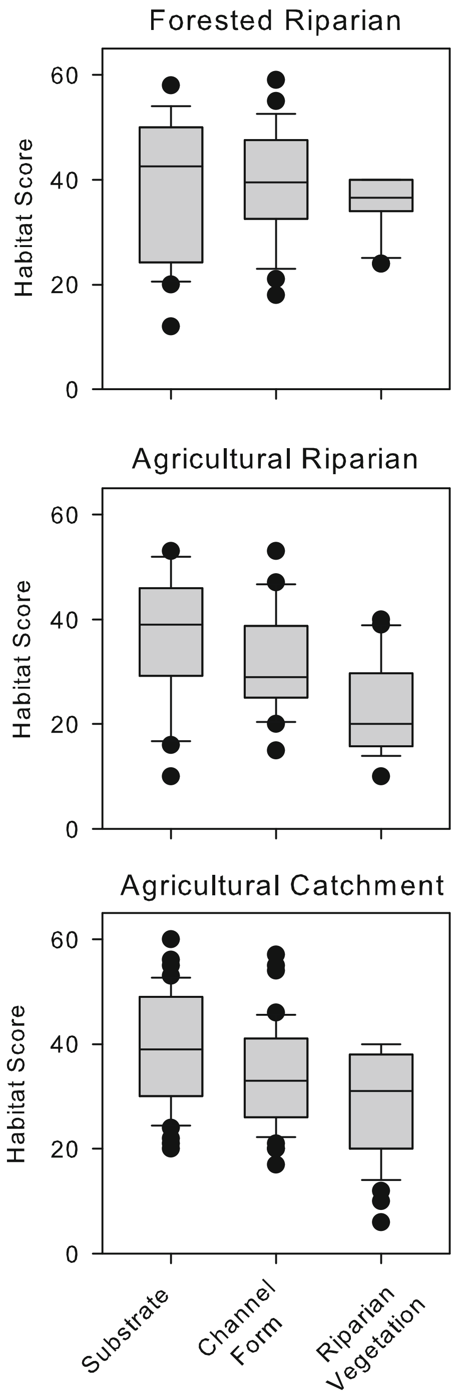
In addition to whole assemblage descriptions of BMI composition, we calculated 12 metrics commonly used in assessment of BMI assemblages (Table S1). Metrics described taxonomic richness and composition, as well as modalities of four functional traits (feeding groups, habitat use, life history strategy, and tolerance). Richness metrics were total community richness [TotalRich], EPT (Ephemeroptera, Plecoptera, and Trichoptera) richness [EPTRich], and Diptera richness [DipteraRich]. Compositional metrics were percentages of the community comprised by EPT [%EPT] and dipteran taxa [%Diptera]. For the functional traits two feeding (i.e., %Herbivores and %Shredders), two life history (i.e., %Multivoltine [%Multivolt] and %Small Body Size [%Small]), two habitat (i.e., %Clingers and %Burrowers), and one tolerance (i.e., Hilsenhoff Family Biotic Index [FBI]) metrics were calculated.

Metrics were calculated using the lowest practical taxonomic resolution, usually genus or family. Consistency in taxonomic resolution among samples was achieved using rules adapted from Vlek et al. (2004). Under these rules, if more than 20% of individuals in a taxon were identified to family level, then those individuals from the lower genus level would be elevated to the family level. In contrast, if less than 20% of individuals in a taxon were identified to the family level, then only individuals of that taxon that were identified to the genus level were retained for analysis. However, in cases where less than 20% of individuals were at the family level, 100% of the sites where that family was collected were required to have at least one individual at the genus level for the taxon to be adjusted to the genus level. This criterion ensured that richness metrics at individual sites were not artificially reduced through taxonomic adjustments. If 100% of the sites did not have a member of the taxa at the genus level then all sites were adjusted to the family level.

Prior to conducting threshold analyses BMI metrics were transformed so that each metric more closely approximated a normal distribution. Log base-10 transformations were applied to diversity and tolerance metrics. Square-root transformations were applied to percent-based composition and trait metrics.

Confounding effects of variation in the amount of agricultural cover at the control scale (i.e., the scale with minimal variation in the amount of agricultural cover) was assessed for each treatment. Measurement of confounding effects was done using ordinary least squares regression analysis in SYSTAT 13 (2015) to test for associations between control scale agricultural cover and each BMI metric. Associations between agricultural land cover in the control scale and BMI metrics were considered significant where the p value was less than 0.1. Significant associations ($p < 0.1$) were detected for %Clingers and FBI for the Forested Riparian treatment and %Multivolt and FBI for the Agricultural Catchment treatment. Residual values (%ClingersRes, FBIRes and %MultivoltRes) were thus used in threshold analyses for the respective treatments.

Potential thresholds in the response of the 12 BMI community metrics to agricultural cover patterns were analyzed with segmented regression using the SegReg 1.7.0 program (Oosterbaan 2017). Segmented regression analysis partitions the independent variable into intervals around breakpoints in the data and each interval is then fitted with a separate line segment. The SegReg program assigns the associations between the independent and dependent variables to one of seven



◀**Fig. 3** Boxplots showing median (middle line), quartiles (shaded box), 95th percentile (whiskers) and outliers (filled circles) for United States Environmental Protection Agency Rapid Assessment Protocol (Barbour et al. 1999) stream habitat scores collected from 68 headwater streams used in three study treatments (Forested Riparian, Agricultural Riparian, Agricultural Catchment). Scores were summed thematically to generate summary scores for each treatment describing, stream substrate quality (Substrate), channel form quality (Channel Form) and condition of the riparian zone (Riparian Vegetation). Maximum attainable scores were 60 for Substrate and Channel Form and 40 for Riparian Vegetation

function types (0–6, Table S2) based on which function maximizes the coefficient of explanation (E). Types 0 and 1 indicate no association and a linear association, respectively, whereas Types 2–4 represent threshold responses where the rate of change in the dependent variable changes about the threshold. In contrast, Types 5 and 6 represent thresholds describing a state change in the data where the mean of the response variable differs about the threshold. Significance of associations was assessed at an alpha of 0.05. SegReg also calculates 90% confidence intervals for threshold values associated with Types 2–4 functions, but not Types 5 and 6.

Results

Habitat scores from stream reaches sampled as part of the Forest Riparian treatment ranged from under 20 to near the maximum possible score of 60 for both substrate and channel form groups (Fig. 3). In contrast, more than 75% of the sites had a riparian vegetation score of greater than 35 out of a possible 40 and all sites had scores of at least 24. The Agricultural Riparian treatment had ranges that were at least two-thirds of the total possible range for all habitat groups. Likewise, the range of habitat scores exceeded two-thirds of the possible scores for the Riparian Gradient treatment. Pearson's correlation analyses found no significant collinearity ($p > 0.05$) between habitat group scores and the amount of agricultural cover at the treatment scale for all three treatments (i.e., Forested Riparian, Agricultural Riparian and Agricultural Catchment).

Benthic macroinvertebrate composition

Sixty-one different macroinvertebrate families and five Chironomidae subfamilies were identified from the samples collected at all sites used in this study. The most common taxa were Elmidae and the Chironomidae subfamilies Chironominae, Orthocladinae and

Tanypodinae, all of which were collected at greater than 85% of the sites. Sites included in the Forested Riparian treatment contained 51 taxa of which 12 were found at less than 5% of the sites. Twelve rare taxa were also identified as rare out of the 47 taxa collected at the Agricultural Riparian sites. The Agricultural Catchment sites had the largest number of taxa (58) of which 17 taxa were considered rare. Taxa identified as rare within each treatment were removed from further analysis.

NMDS analyses and associated surface fitting showed no associations between community dissimilarities and agricultural land cover in the catchment of the Forested Riparian sites ($p > 0.05$). Likewise, agricultural cover in the catchment areas of Agricultural Riparian sites was not associated ($p > 0.05$) with dissimilarity matrices describing taxonomic or trait composition using presence/absence data. However, abundance-based descriptions of taxa and trait composition of Agricultural Riparian sites were associated with agricultural cover in the catchment area (Fig. 4).

Fitted surfaces describing agricultural cover in catchment areas of Agricultural Riparian sites had greater R^2 values than did fitted vectors. Surfaces explained 54% of the deviance in among site dissimilarity for taxonomic abundance ($p = 0.019$; $R^2_{adj} = 0.42$) and 63% of the deviance for trait abundance ($p = 0.006$; $R^2_{adj} = 0.52$). Taxa associated with increased agricultural cover were primarily dipterans (i.e., Ceratopogonidae, Chironomine, Pyschododidae and Tanypodinae), as well as the mayfly Caenidae and hemipteran Corixidae. In contrast, the caddisflies Hydropsychidae and Philapotimidae, as well as the beetle Psephenidae and dipterans Tipulidae and Empididae were associated with less agricultural cover in the catchment area. Trait modalities associated with increased agricultural cover in the catchment area were associated with greater tolerance to poor water quality, preference for slower flowing water and faster generation times. In contrast, trait modalities associated with reduced amounts of agricultural cover in the catchment were increased body size, clinging strategies and greater dispersal capacity.

Dissimilarity among Agricultural Catchment communities was best explained by a fitted surface describing agricultural cover in the riparian corridor for taxonomic and trait composition for both presence/absence and abundance (Fig. 5). For taxonomic composition agricultural cover was more strongly associated with dissimilarities in presence/absence data (% explained = 35; $p = 0.002$; $R^2_{adj} = 0.28$) than

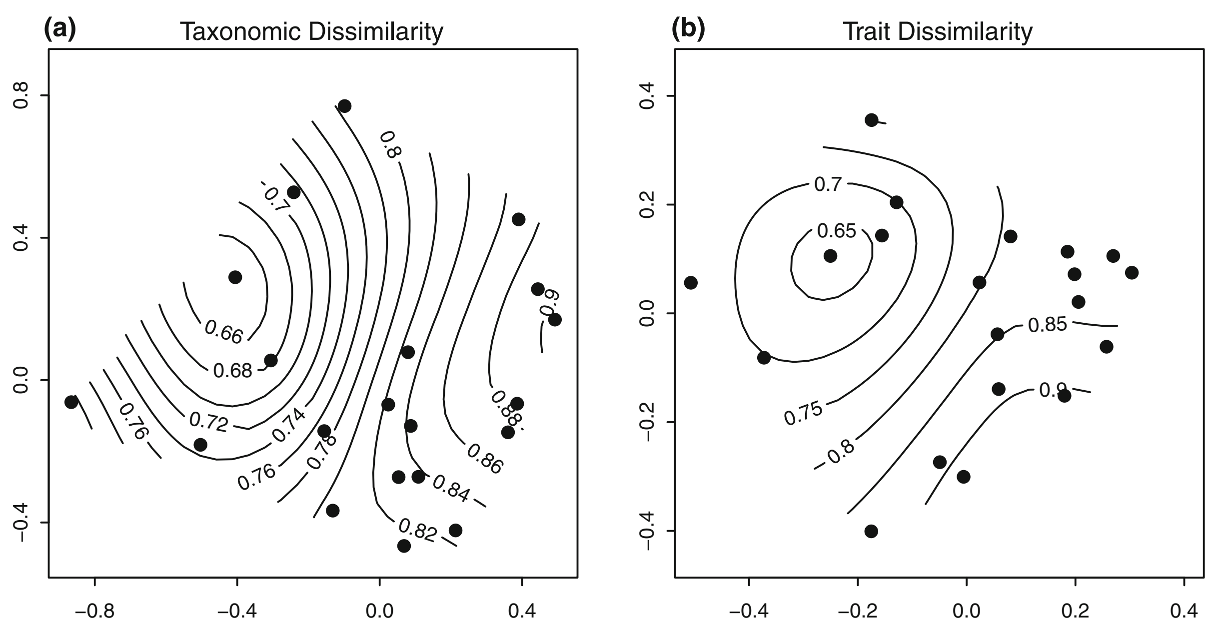


Fig. 4 NMDS ordination plots of taxonomic (**a**) and trait (**b**) dissimilarities for Hellinger transformed abundance data from 20 sites sampled as part of the Agricultural Riparian treatment. Contour lines represent the proportion of agricultural land cover at the catchment scale fitted using generalized additive models

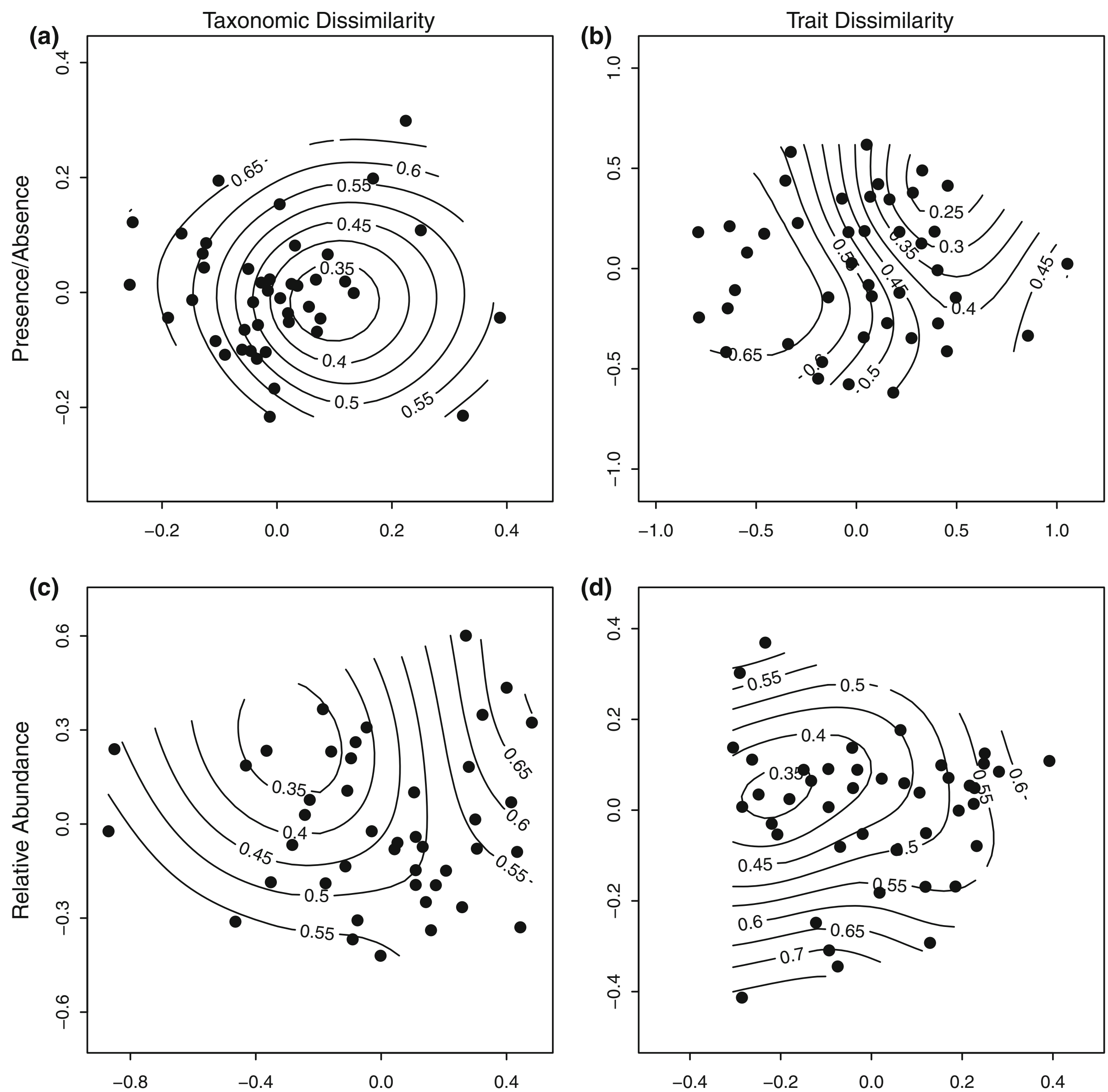


Fig. 5 NMDS ordination plots of taxonomic and trait dissimilarities for presence/absence (**a**, **b**) and Hellinger transformed abundance (**c**, **d**) data from 43 sites sampled as part of the Agricultural Catchment treatment. Contour lines represent the proportion of agricultural land cover at the riparian corridor scale fitted using generalized additive models

abundance (% explained = 22; p = 0.039; $R^2_{adj} = 0.15$). Taxa associated with increased agricultural cover in the riparian corridor were similar for presence/absence and abundance data and included: Caenidae, Coenagriondiae, Corixidae, Hydrophilidae, Haliplidae, Psychododiae, Chironominae and Tanypodinae. Taxa associated with low agricultural cover in the riparian corridor included the odonates Aeshinidae and Calopterygidae, the caddisflies Helicopsychidae and Hydrosychidae and mayflies Heptageniidae and Leptophlebiidae. In contrast, agricultural cover in the riparian zone explained a similar amount of deviance in dissimilarity in presence/absence (% explained = 28; p = 0.01; $R^2_{adj} = 0.21$) and abundance (% explained = 27; p = 0.015; $R^2_{adj} = 0.20$) of trait modalities. Trait modalities associated with increased agriculture in the riparian corridor included increased tolerance to poor water quality, slower moving water, sand habitats, and preference for warmer water temperatures. Trait modalities related to faster moving water, rocky habitats, and cooler water temperatures were associated with limited agricultural land cover in the riparian corridor.

Agricultural cover thresholds

Descriptive analyses of BMI metrics showed that mean TotalRich was about 20 for all treatments and that the majority of taxa were Diptera or EPT taxa (Tables S3, S4, S5). Coefficients of variation were generally smaller for the Forested Riparian treatment than the other two treatments. The FBI indicated that communities were, on average, fairly tolerant of organic pollution for all treatments.

Segmented regression did not identify thresholds or associations ($p > 0.05$) between the twelve BMI metrics and agricultural cover in catchments for the Forested Riparian treatment. In contrast, four trait metrics exhibited associations with agricultural cover at the catchment scale for the Agricultural Riparian treatment. A Type 1 function (linear) with a positive association with agricultural cover best fit %Small ($R^2 = 0.64$, $p < 0.001$) and FBI ($R^2 = 0.478$, $p < 0.001$) (Fig. S1). %Clingers fit a Type 3 function ($E = 0.40$, $p = 0.035$) exhibiting decreasing abundance of clingers after agricultural cover exceeded 72% (90% CI = 65–80%) in the catchment. Last, a Type 5 function best fit the response of %Multivolt ($E = 0.54$, $p < 0.001$), which increased from a mean abundance of 34% to a mean of approximately 70% when agricultural land cover exceeded 84% of the catchment area.

Segmented regression identified thresholds for eight metrics in the Agricultural Catchment treatment. Richness metrics best fit a Type 3 function with declines in richness occurring at greater than 62% (90% CI = 54–69%), 53% (90% CI = 49–59%) and 75% (90% CI = 70–80%) agriculture in the riparian corridor for TotalRich ($E = 0.20$, $p = 0.037$), EPTRich ($E = 0.50$, $p < 0.001$) and DipteraRich ($E = 0.20$, $p = 0.039$), respectively (Fig. S2). %Shredders ($E = 0.20$, $p = 0.039$) and %Small ($E = 0.18$, $p = 0.057$) also best fit a Type 3 function. %Shredders declined beyond 59% (90% CI = 50–70%) agriculture and %Small increased above 44% (90% CI = 35–51%) (Fig. S3). %MultivoltRes ($E = 0.19$, $p = 0.041$), FBIRes ($E = 0.21$, $p = 0.009$) and %Clingers ($E = 0.19$, $p = 0.016$) best fit a Type 5 function. Increased mean abundance of multivoltine taxa occurred at 77% agriculture in the riparian corridor, whereas FBIRes showed a change in the mean FBI score at 53% cover. A threshold of 53% agriculture in the riparian corridor was also identified for %Clingers, as the mean declined from 32 to 14%.

Discussion

Scale-specific effects of agricultural cover

Our finding that increasing agricultural cover at the catchment scale had no effect on BMI community composition in streams with forested riparian corridors is consistent with a large literature demonstrating that riparian forest is an effective buffer (Sweeney and Newbold 2014). Moreover, our findings suggest that riparian vegetation can protect BMI community conditions even when agricultural cover in upland areas of the catchment exceeds 80%. The continued protection provided by the riparian forest at large levels of catchment scale disturbances surpasses expectations of hypothetical models (e.g., Allan 2004a) and results from a field study by Feld (2013) that predict buffering effects would be overwhelmed at greater levels of catchment disturbance.

Although we cannot exclude the possibility, it does not appear that insensitivity of the BMI community to catchment agriculture was due to riparian buffering effects being overwhelmed at a percentage of catchment cover below that included in our study. Indeed, if such a threshold had occurred, BMI to agriculture associations for Forested and Agricultural Riparian treatments should have been similar. Instead, we observed that taxonomic and trait abundances of communities exposed to similar gradients of catchment agriculture were associated with cover in the Agricultural Riparian treatment but not the Forested Riparian treatment. Differences between results of the Forested and Agricultural Riparian treatments suggest that BMI communities in the Agricultural Riparian sites are responding to the absence of riparian forest cover. Our findings thus provide support for development and implementation of management policies aimed at the conservation and rehabilitation of riparian forests as a strategy for protecting stream communities in agricultural landscapes.

Development of riparian corridors for agriculture can alter stream temperature, modify stream habitat and shift the dominant source of organic matter from allochthonous to autochthonous (Gregory et al. 1991; Naiman and Decamps 1997). Furthermore, agricultural activity in the riparian zone disproportionately contributes nutrients, sediments and pesticides to stream ecosystems relative to agriculture in upland areas (reviewed by Allan 2004b). The numerous mechanisms by which agriculture in the riparian corridor can alter stream ecosystems may explain why BMI communities in the Agricultural Catchment treatment diverged along multiple pathways with increased agriculture in the riparian corridor. Moreover, the specific taxa and traits associated with agricultural cover in the riparian corridor suggest that ecological changes were a result of a loss of riparian forest functions and an increase in stressors associated with agriculture. For example, the association between agricultural cover and %Shredders suggests that the loss of riparian forest reduced delivery of leaf litter to the study sites. In contrast, the increase in the abundance of tolerant taxa suggests increased loading of agricultural stressors, such as nutrients, sediments and pesticides. However, because our study did not measure specific stressors or riparian functions further research is needed to confirm the processes and stressors that linked riparian agriculture with BMI communities.

We hypothesized that BMI communities in streams with extensive agricultural cover in the riparian zone would not be associated with variation in cover at the catchment scale because stressors from agriculture and loss of riparian function would have extirpated sensitive taxa and traits. In support of this hypothesis was our finding that taxonomic and trait richness was not associated with increasing catchment agriculture. This finding is consistent with several past studies that have observed the loss of taxa sensitive to agricultural stressors (e.g., Wagenhoff et al. 2012; Johnson and Angeler 2014) and suggests taxa richness in our study streams was more strongly affected by the loss of riparian forest than the intensity of agriculture in the catchment. This interpretation is supported by evidence from the Agricultural Catchment treatment where community richness was associated with the loss of taxa (e.g., Aeshnidae and Helicopsychidae) considered to be sensitive to agricultural stressors. Overall, our results further demonstrate the important role the riparian corridor plays in maintaining regional biodiversity (Naiman et al. 1993).

Agricultural cover in the catchment was associated with abundance of taxa and trait modalities in streams without forest in the riparian corridor. Variation in agricultural cover among the catchments may have influenced BMI taxa and trait abundance through two mechanisms. First, greater forest cover in catchments with reduced agriculture may have increased upland forest functions, such as nutrient retention and organic matter processing in upland soils (Mulholland 1992; Dosskey and Bertsch 1994), while also intercepting a portion of agricultural runoff from upland activities. Second, watersheds with less agricultural cover may have been associated with reduced stressor loadings to streams providing better water quality to stream biota. Although more research is required to determine the mechanism, our observation of continued importance of catchment land cover in the absence of natural riparian vegetation has implications for regional land management strategies. First, it suggests that taking steps to conserve remaining forested areas in the upland areas could assist in protecting existing amounts of taxonomic and functional diversity where agricultural cover is below identified catchment scale thresholds. Second, identification of independent effects of agricultural land use outside the riparian corridor supports evidence from past studies indicating implementation of best management practices (BMPs) in the upland areas of agricultural catchments may be an effective strategy for improving instream ecological conditions (e.g., Yates et al. 2007; Marshall et al. 2008). Improved catchment management through BMP implementation should be further explored as a means of improving ecological conditions of agricultural streams without reducing the extent of agriculture land use.

Agricultural thresholds for management of ecological condition

Groffman et al. (2006) defined a threshold as “the point at which there is an abrupt change in an ecological quality, property, or phenomenon or where small changes in a driver can produce large responses in the ecosystem”. Our study empirically identified eight metrics that exhibited patterns of ecological change with increasing agricultural cover at the catchment or riparian corridor scales statistically consistent with Groffman et al.'s definition. At the catchment scale, we identified thresholds for two trait modalities, %Clingers and %Multivolt, when the riparian forest had been replaced with agricultural cover. Both trait modalities had thresholds of greater than 70% agricultural cover for the Agricultural Riparian treatment. The threshold values we observed are as much as threefold the amount of catchment agriculture identified in previous studies on stream BMI (e.g., Wang et al. 1997; Feld 2013; Waite 2014), although Utz et al (2009) observed thresholds more similar (60–80%) to ours in agricultural regions of the American Mid-West. The difference in thresholds may be in part due to the lack of differentiation between cover in the riparian corridor and catchment in past studies. Consequently, previous studies may have been observing triggering of thresholds primarily because of loss of riparian vegetation rather than agricultural activity in the catchment. Scale-specific effects could explain why threshold values (approx. 50% for most metrics) observed with the removal of riparian forest (Agricultural Catchment treatment) were closer to past estimates of agricultural thresholds (e.g., 30% agriculture; Feld 2013). These thresholds suggest that moderate amounts of forest cover in the riparian corridor can protect the BMI community.

Thresholds observed in our study indicate initial resilience of stream BMI to agricultural cover followed by either a linear decline (Type 3) or a step function (Type 5) suggesting agricultural cover acts as an extrinsic factor threshold (sensu Groffman et al. 2006). Extrinsic factor thresholds have also been observed in stream ecosystems in association with impervious urban land covers and have been promoted as a means of informing land management targets to protect ecosystem integrity (e.g., Hilderbrand et al. 2010; King and Baker 2010). We propose that the thresholds identified in our study could be similarly useful for setting agricultural cover targets for restoration and conservation activities at both the catchment and riparian corridor scale. Indeed, the identified thresholds suggest that restoration of riparian corridors could be a viable mechanism for enhancing stream ecosystems with minimal impact on agricultural production, as many streams in our study region would require modest replanting efforts in riparian corridors to meet cover targets.

We recognize that caution is required in applying the thresholds observed in our study as management targets for land use planning because of limitations in our study design, taxonomic identifications and uncertainty around threshold values. First, we purposely limited the scope of our study to minimize confounding effects associated with covariance of agricultural activity and physiography, as well as effects of tile drainage and stream channelization. Tile drainage can circumvent buffering effects of riparian zones while channelization of streams can alter stream communities due to impaired habitat quality (Osborne and Kovacic 1993; Lau et al. 2006). Both activities have been linked to ineffectiveness of riparian management (Osborne and Kovacic 1993; Greenwood et al. 2012). Thus, the transferability of the identified thresholds to neighbouring agricultural regions and to streams with managed channel forms should be assessed. Second, we were limited to using a mixture of genus and family level data in our descriptions of the BMI community. Taxonomic limitations are routine in BMI studies because of specimen condition and ability to accurately identify early instars (Lenat and Resh 2001). However, it has been argued that more subtle changes in the community can be masked by use of coarser taxonomic resolution (Lenat and Resh 2001). Thus, it is possible that our study missed, or over-estimated, threshold responses by the BMI community. Third, and finally, we found that 90% confidence intervals around the threshold values ranged from a minimum of 9% to a maximum of 20% agricultural cover. Substantial statistical uncertainty around threshold values is common (see Dodds et al. 2010 for an example) due to measurement errors, variability in the subject being assessed and inflated Type I errors (Andersen et al. 2009; Toms and Villard 2015). Therefore, we recommend management agencies follow past suggestions (e.g., Dodds et al. 2010; Hilderbrand et al. 2010) that the precautionary principle be applied to threshold application, such that systems are not managed to the threshold. We also recommend that managers use the identified thresholds in association with land cover targets specific to other stream management objectives, such achieving total daily maximum loads for nutrients and sediments. This recommendation is in recognition of the possibility that the proportion of forested riparian corridor that conserves ecological communities in intensive agricultural landscapes may not be sufficient to meet all stream management objectives.

Prioritization of riparian corridor management

Isolation of scale-specific effects and thresholds related to agricultural cover in the riparian corridor and catchment areas has provided critical information for conserving stream communities in agricultural landscapes. Specifically, our study demonstrates that stream ecological conditions are impacted by agricultural activities within the riparian corridor and catchment areas. However, our results also show that the state of the riparian corridor supersedes the catchment such that management of catchment activities is likely to provide measurable benefits only when the riparian corridor is intensively used for agricultural activities. Furthermore, our study shows that maintenance of forested riparian corridors protects stream communities in the face of intensive agricultural activity at the catchment scale.

Based on our findings, we propose that managers seeking to protect and enhance ecological conditions of streams in agricultural landscapes adopt a preliminary framework for prioritization and management of riparian corridors. Application of the framework requires completion of a regional inventory of land cover in riparian corridors. Priority locations and associated management strategies could then be set using the inventory to assess corridors in the region using the following priority rankings. Priority 1, protect existing riparian forests along streams with minimal agricultural cover in the riparian corridor. Priority 2, initiate restoration activities in riparian corridors with agricultural cover near thresholds to improve ecological conditions and add resilience to vulnerable communities. Priority 3, implement a combination of riparian corridor restoration and catchment management actions in areas where the riparian corridor has been converted to agricultural uses. We envision this framework encompassing an adaptive management component such that as increased knowledge becomes available prioritizations would be refined and extended. For example, Priority 3 could be split to specify that channel condition be used to prioritize riparian corridor management versus catchment management as increased knowledge of interactive effects between channelization and riparian forest is generated. We also recommend that the proposed framework be used in concert with other conservation plans, such as non-point source nutrient reduction plans, to generate a holistic management framework to protect the health of river networks.

Acknowledgements We thank R. Holmes for assistance with fieldwork and N. Pearce for statistical advice. R. Bailey and two anonymous reviewers provided comments on earlier versions of the manuscript. Funding was provided by a Natural Sciences and Engineering Research Council (NSERC) of Canada Discovery Grant (AGY) and a Canadian Water Network Watershed Consortia Grant (AGY), as well as NSERC Collaborative Research and Training Experience Program (JPG) and Trillium Foundation (EMK) Scholarships.

References

Allan JD (2004a) Influence of land use and landscape setting on the ecological status of rivers. Limnetica 23(3):187–198

Allan JD (2004b) Landscapes and riverscapes: the influence of land use on stream ecosystems. Annu Rev Ecol Evol Syst 35:257–284

Andersen T, Carstensen J, Hernandez-Garcia E, Duarte CM (2009) Ecological thresholds and regime shifts: approaches to identification. Trends Ecol Evol 24(1):49–57

Barbour MT, Gerritsen J, Snyder BD, Stribling JB (1999) Rapid bioassessment protocols for use in streams and wadeable rivers: periphyton, benthic macroinvertebrates and fish, second edition. EPA 841-B-99-002. Office of Water, U.S. Environmental Protection Agency, Washington, DC

Benfield EF (1997) Comparison of litterfall input to streams. J N Am Benthol Soc 16:104–108

Dodds WK, Clements WH, Gido K, Hilderbrand RH, King RS (2010) Thresholds, breakpoints, and nonlinearity in freshwaters as related to management. J N Am Benthol Soc 29(3):988–997

Dosskey MG, Bertsch PM (1994) Forest sources and pathways of organic matter transport to a blackwater stream: a hydrologic approach. Biogeochemistry 24(1):1–19

Environment Canada and Climate Change, ECCC (2016). http://climate.weather.gc.ca/climate_normals/index_e.html. Accessed 15 Aug 2016

Feld CK (2013) Response of three lotic assemblages to riparian and catchment-scale land use: implications for designing catchment monitoring programmes. Freshw Biol 58:715–729

Greenwood MJ, Harding JS, Niyogi DK, McIntosh AR (2012) Improving the effectiveness of riparian management for aquatic invertebrates in a degraded agricultural landscape: stream size and land-use legacies. J Appl Ecol 49(1):213–222

Gregory SV, Swanson FJ, McKee WA, Cummins KW (1991) An ecosystem perspective of riparian zones: focus on links between land and water. Bioscience 41(8):540–551

Groffman PM, Baron JS, Blett T, Gold AJ, Goodman I, Gunderson LH, Levinson B, Palmer M, Paerl H, Peterson G, Poff N, Rejeski D, Reynolds J, Turner M, Weathers K, Wiens J (2006) Ecological thresholds: the key to successful environmental management or an important concept with no practical application? Ecosystems 9(1):1–13

Gurnell AM, Gregory KJ, Petts GE (1995) The role of coarse woody debris in forest aquatic habitats: implications for management. Aquat Conserv Mar Freshw Ecosyst 5:143–166

Hilderbrand RH, Utz RM, Stranko SA, Raesly RL (2010) Applying thresholds to forecast potential biodiversity loss from human development. J N Am Benthol Soc 29(3):1009–1016

Johnson RK, Angeler DG (2014) Effects of agricultural land use on stream assemblages: taxon-specific responses of alpha and beta diversity. Ecol Indic 45:386–393

King RS, Baker ME (2010) Considerations for analyzing ecological community thresholds in response to anthropogenic environmental gradients. J N Am Benthol Soc 29(3):998–1008

Krynak EM, Yates AG (2018) Benthic invertebrate taxonomic and trait associations with land use in an intensively managed watershed: implications for indicator identification. Ecol Indic 93:1050–1059

Lau K, Lauer TE, Weinman ML (2006) Impacts of channelization on stream habitats and associated fish assemblages in East Central Indiana. Am Midl Nat 156:319–330

Lenat DR, Resh VH (2001) Taxonomy and stream ecology—the benefits of genus and species-level identifications. J N Am Benthol Soc 20(2):287–298

Marshall DW, Fayram AH, Panuska JC, Baumann J, Hennessy J (2008) Positive effects of agricultural land use changes on coldwater fish communities in southwest Wisconsin streams. N Am J Fish Manag 28(3):944–953

Marzin A, Verdonschot PFM, Pont D (2013) The relative influence of catchment, riparian corridor and reach-scale anthropogenic pressures on fish and macroinvertebrate assemblages in French rivers. Hydrobiologia 704(1):375–388

Moore R, Spittlehouse DL, Story A (2005) Riparian microclimate and stream temperature response to forest harvesting: a review. J Am Water Resour Assoc 41(4):813–834

Mulholland PJ (1992) Regulation of nutrient concentrations in a temperate forest stream: roles of upland, riparian, and instream processes. Limnol Oceanogr 37(7):1512–1526

Naiman RJ, Decamps H (1997) The ecology of interfaces: riparian zones. Annu Rev Ecol Evol Syst 28(102):621–658

Naiman RJ, Decamps H, Pollock M (1993) The role of riparian corridors in maintaining regional biodiversity. Ecol Appl 3(2):209–212

Oksanen J, Blanchet FG, Friendly M, Kindt R, Legendre P, McGlinn D, Minchin PR, O'Hara RB, Simpson GL, Solymos P, Henry M, Stevens H, Szoecs E, Wagner H (2017) vegan: community ecology package. R package version 2.4-2. https://CRAN.R-project.org/package=vegan

Oosterbaan RJ (2017) SegReg 1.7.0.0. Segmented linear regression with breakpoint and confidence intervals (software). https://www.waterlog.info/segreg.htm. Accessed 19 March 2017

Osborne LL, Kovacic DA (1993) Riparian vegetation buffer strips in water-quality restoration and stream management. Freshw Biol 29(2):243–258

Peterson EE, Sheldon F, Darnell R, Bunn SE, Harch BD (2011) A comparison of spatially explicit landscape representation methods and their relationship to stream condition. Freshw Biol 56(3):590–610

R Core Team (2016) R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna. https://www.R-project.org/

Reynoldson TB, Logan C, Pascoe T, Thompson SP, Strachan S, Mackinlay C, McDermott H, Paull T (2012) Canadian Aquatic Biomonitoring Network field manual wadeable streams 2012. Freshwater Quality Monitoring and Surveillance—Atlantic. Environment Canada, Dartmouth

Sweeney BW, Newbold JD (2014) Streamside forest buffer width needed to protect stream water quality, habitat, and organisms: a literature review. J Am Water Resour Assoc 50(3):560–584

SYSTAT (2015) SYSTAT 13.1. SYSTAT, San Jose

Toms JD, Villard M (2015) Threshold detection: matching statistical methodology to ecological questions and conservation planning objectives. Avian Conserv Ecol 10(1):1–8

Utz RM, Hilderbrand RH, Boward DM (2009) Identifying regional differences in threshold responses of aquatic invertebrates to land cover gradients. Ecol Indic 9(3):556–567

Van Sickle J, Johnson CB (2008) Parametric distance weighting of landscape influence on streams. Landscape Ecol 23(4):427–438

Vidon P, Allan C, Burns D, Duval TP, Gurwick N, Inamdar S, Lowrance R, Okay J, Scott D, Sebestyen S (2010) Hot spots and hot moments in riparian zones: potential for improved water quality management. J Am Water Resour Assoc 46:278–298

Virtanen R, Oksanen J, Oksanen L, Razzhivin VY (2006) Broad-scale vegetation–environment relationships in Eurasian high-latitude areas. J Veg Sci 17:519–528

Vlek HE, Verdonschot PFM, Nijboer RC (2004) Towards a multimetric index for the assessment of Dutch streams using benthic macroinvertebrates. Hydrobiologia 516(1):173–189

Wagenhoff A, Townsend CR, Matthaei CD (2012) Macroinvertebrate responses along broad stressor gradients of deposited fine sediment and dissolved nutrients: a stream mesocosm experiment. J Appl Ecol 49(4):892–902

Waite I (2014) Agricultural disturbance response models for invertebrate and algal metrics from streams at two spatial scales within the U.S. Hydrobiologia 726(1):285–303

Wang L, Lyons J, Kanehl P, Gatti R (1997) Influence of watershed land use on habitat quality and biotic integrity in Wisconsin streams. Fisheries 22(6):7–12

Yates AG, Bailey RC (2010) Improving the description of human activities potential affecting rural stream ecosystems. Landscape Ecol 25(3):371–382

Yates AG, Bailey RC, Schwindt JA (2007) Effectiveness of best management practices in improving stream ecosystem quality. Hydrobiologia 583(1):331–344