

Running head: riparian vs. imperviousness influences

Title: Hierarchical multi-taxa models inform riparian vs. hydrologic restoration of urban streams in a permeable landscape

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Abstract. The degradation of streams caused by urbanization tends to follow predictable patterns; however, there is a growing appreciation for heterogeneity in stream response to urbanization due to the local geoclimatic context. Furthermore, there is building evidence that streams in mildly sloped, permeable landscapes respond uncharacteristically to urban stress calling for a more nuanced approach to restoration. We evaluated the relative influence of local-scale riparian characteristics and catchment-scale imperviousness on the macroinvertebrate assemblages of streams in the flat, permeable urban landscape of Perth,

Western Australia. Using a hierarchical multi-taxa model, we predicted the outcomes of

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stylized stream restoration strategies to increase the riparian integrity at the local scale or decrease the influences of imperviousness at the catchment scale. In the urban streams of Perth, we show that local-scale riparian restoration can influence the structure of macroinvertebrate assemblages to a greater degree than managing the influences of catchment-scale imperviousness. We also observed an interaction between the effect of riparian integrity and imperviousness such that the effect of increased riparian integrity was enhanced at lower levels of catchment imperviousness. This study represents one of few conducted in flat, permeable landscapes and the first aimed at informing urban stream restoration in Perth, adding to the growing appreciation for heterogeneity of the Urban Stream Syndrome and its importance for urban stream restoration.

Key words: urban stream syndrome, ecosystem function, macroinvertebrates, riparian zone, restoration, impervious, stormwater, Bayesian hierarchical model

INTRODUCTION

Human populations are becoming increasingly urbanized (PRB 2007, 2015) and aquatic ecosystems conspicuously bear the stress of this urbanization. Urban streams display a suite of maladies collectively known as the Urban Stream Syndrome (Walsh et al. 2005a) with symptoms including increased hydrologic variability, increased water temperatures, elevated nutrients/chemical pollutants, altered energy sources, simplified structural habitat and, as a consequence, reduced native biodiversity (Paul and Meyer 2001, Konrad and Booth 2005, Walsh et al 2005a). To counter the Urban Stream Syndrome, there is increasing policy and public support for restoration. While urban stream restoration is burgeoning worldwide (Bernhardt and Palmer 2007), many restoration projects fail to deliver the anticipated benefits (Bernhardt et al. 2005, Louhi et al. 2011, Violin et al. 2011). Many believe that ineffective

stream restoration frequently occurs because management fails to targets the most influential ecosystem processes (Palmer et al. 2010).

Currently, contention exists about the relative importance of catchment-scale versus local-scale processes for urban stream health. The dominant paradigm is that altered hydrology associated with stormwater runoff over impervious surfaces drives urban stream degradation (Walsh et al. 2007, Schueler et al. 2009, Imberger et al. 2011, Imberger et al. 2014). Supporters of this paradigm argue that restoration that targets catchment-scale hydrology, such as harvesting stormwater, promoting infiltration and disconnecting stormwater transport to streams will be most successful (Walsh et al. 2005b, Walsh et al. 2016). However, the most common form of stream restoration occurs at local spatial scales (Bernhardt and Palmer 2007, Palmer et al. 2014), and includes actions such as channel reconfiguration, instream habitat manipulation and riparian revegetation. While many dismiss these local-scale interventions as ineffective, others present evidence of their benefit. For example, Collier and Clements (2011) and Sheldon et al (2012) found that the health of urban stream ecosystems was influenced more by local character of the riparian corridor than catchment imperviousness. This disagreement over the most influential spatial scale of ecosystem processes can create confusion for managers and lead to the implementation of suboptimal restoration strategies.

There is evidence that disagreement over the importance of catchment-scale versus local-scale processes may arise from the geoclimatic context of urban stream research. The dominant paradigm of urban stream ecology has arisen primarily from research conducted in sloped landscapes with high clay soil content in the eastern USA and southeastern Australia. The effects of storm water runoff on flow is accentuated in these environments. However, heterogeneity in the Urban Stream Syndrome is increasingly recognized (Booth et al. 2016, Roy et al. 2016) and mildly sloped, permeable landscapes are emerging as an exception to the

dominant paradigm. For example, recent studies in North America have found that streams in the mildly sloped, permeable landscapes of the Coastal Plain in mid-Atlantic region of USA experience less physico-chemical and biotic stress relative to adjacent sloped landscapes of the Piedmont region (Utz et al. 2011a, 2011b). In a meta-analysis of urban regions across the USA, Hopkins et al. (2015) also demonstrated that mildly sloped, permeable landscapes buffer urban areas from flow stress because they promote infiltration, reducing the severity and frequency of flow spikes. The recognition that mildly sloped, permeable landscapes respond uncharacteristically to urbanization may provide an opportunity for managers to adapt their restoration strategies to better meet their management objectives.

An Australian example of a city that exists in a mildly sloped, permeable landscape is the city of Perth, Western Australia. Most of the city is situated on the Swan Coastal Plain, where the topography is flat and the soils are permeable (sandy), promoting naturally low runoff and high infiltration (Barron et al. 2013a). These landscape characteristics allow for widespread infiltration of stormwater from impervious surfaces such as roofs and roads through basins and sumps that retain stormwater. The city is undergoing rapid expansion and infill development, placing increasing pressure on streams, with many exhibiting symptoms of the Urban Stream Syndrome. Stream restoration in the region is common and undertaken by community groups and state government agencies. However, there is no published research on the response of stream biota to urbanization in this region, and guidance for restoration activities are based on general ecological knowledge and research conducted largely in the clay sloped landscape of southeastern Australia, which may be suboptimal. Thus, there is a need for local research to determine the most influential processes and spatial scale to maximize return on restoration investment.

The objective of this study was to compare the relative influence of local-scale riparian characteristics and catchment-scale imperviousness on the structure of the macroinvertebrate assemblages of urban streams in Perth. Using a Bayesian hierarchical multi-taxa model of macroinvertebrate abundance, we evaluate patterns in macroinvertebrate abundance and occurrence as relates to riparian integrity and percent catchment imperviousness. To demonstrate the implications of our results for urban stream restoration and management, we use our fitted model to predict the outcomes of four stylized management scenarios that either reduce the influences of imperviousness or improve the integrity of the riparian zone. Our results have direct implications for urban stream management in mildly sloped, permeable landscapes where effects of urbanization and the success of restoration activities have received little attention in the ecological literature.

METHODS

Study area and sampling design

Perth has a Mediterranean climate with cool wet winters and hot dry summers. The majority of rain falls between May and September, with long-term mean annual rainfall between 800-1100mm. The region is flat, sandy and underlain by shallow groundwater, which contributes notably to stream discharge (Barron et al. 2013a, Barron et al. 2013b). Many small streams in the region, particularly those in peri-urban areas, dry during summer when the groundwater level drops below the stream bed (Galvin et al. 2009a). In areas of the metropolitan region, where the depth to groundwater is naturally shallow (< 4m), urbanization causes the water table to rise making some naturally intermittent streams permanent (Bhaskar et al. 2016).

We conducted our study at 21 sites (50-m in length) spanning a rural to urban gradient (Fig. 1). Seven sites occurred on unique tributaries; however, to increase our ability to assess the effect of local-scale riparian integrity, seven tributaries included two sites. These paired sites were selected with and without intact riparian vegetation and were between 0.33 and 1.8-km apart to reduce variation from non-riparian factors. For six of the seven paired sites, the site with higher riparian integrity was downstream, mitigating the potential for downstream riparian influences. To reduce unaccounted for variation in our data, sites were restricted to small (stream order 1-3) permanent streams.

Macroinvertebrate sampling

Three replicate macroinvertebrate samples were taken, during base-flow conditions in May 2015, within the main-channel at each site using a Surber sampler (dimension 20 x 25cm, mesh aperture 250 μ m). The substratum was vigorously disturbed to a depth of 10cm for two minutes. The stream current swept the sediment, organic matter, and macroinvertebrates into the net. An artificial current was generated by sweeping water through the sampler for sites with little flow. Samples were stored in 70% ethanol for processing. In the laboratory, samples were separated into coarse and fine organic matter by washing each sample through stacked 2-mm, 500- μ m, and 250- μ m sieves, respectively. Twenty percent of each of the sieves was sub-sampled using a grid within the sieve by randomly selecting two of the 10 quadrats. For fine sieve samples, all macroinvertebrates were identified using a dissecting microscope. Individuals were identified to family level (Davis 1997) and functional feeding groups were determined (Table S1 in Appendix S1).

Indices of ecosystem components

We evaluated the relative importance of two drivers of the macroinvertebrate assemblage structure at our study sites: (i) percent catchment imperviousness (I), a surrogate for hydrologic alteration and water quality, and (ii) riparian integrity (R), an index of the health of the riparian zone. Imperviousness was derived by visually assessing aerial photographs (2014) of each cadastral parcel contained in the catchment of a site (i.e. the entire area that drains into the stream upstream of the sample site) and assigning a land use category (see Appendix S2). For each land use category, percent imperviousness was calculated by measuring the percentage of the cadastral parcel area that drains directly to piped conveyance systems. The analysis assumed that: (i) bare soil infiltrates water, (ii) building roof areas are directed into subsurface holding tanks that directly infiltrate to groundwater, and (iii) porous pavements are impervious.

We indexed riparian integrity with a metric developed by the West Australian Department of Water River Health Assessment Scheme (Galvin et al. 2009b). The metric synthesized aspects of the riparian zone, including longitudinal continuity, riparian width, structural intactness, canopy cover and native vegetation recruitment to determine an overall index which ranged from 0 (highly degraded) to 10 (intact and healthy). Details of how the metric was calculated can be found in Galvin et al. (2009b) and are presented online in Appendix S3 of the supplementary information.

Model overview

The influence of riparian integrity and catchment imperviousness on taxonomic compositions of macroinvertebrate assemblages was evaluated with a multi-taxa model formulated in a Bayesian hierarchical framework (Gelman and Hill 2007). The model structure links single-taxa models together by assuming that all taxa-specific parameters are

random effects drawn from common community hyper-distributions governed by hyper-parameters. This model structure makes explicit the assumption that taxa-specific parameters are neither equivalent nor independent among taxa, but related according to the community hyper-distributions. There are several benefits of this modeling approach over traditional and more common multi-variate methods (e.g., Olden 2003, Iknayan et al. 2014, Warton et al. 2015a, Warton et al. 2015b). One major benefit is the potential for improved parameter estimates, particularly for taxa with sparse data that are rare or difficult to detect (Link and Sauer 1996, Sauer and Link 2002). This model behavior occurs because the community-level hyper-distributions act as priors for taxa-specific estimates. Taxa with information rich data contribute disproportionately to defining these hyper-distributions. Conversely, parameters for taxa with sparse data will be pulled or “shrunk” towards the mean of the hyper-distributions, thus, ‘borrowing’ information from the community (Gelman et al. 2004). The modeling approach also allows for statistical inference at both the community and taxa levels in one analysis. This contrasts with those methods that model community metrics directly (e.g., Gotelli and Colwell 2001) and thus, lose taxa-specific information and potentially mask community change due to reorganization of the assemblage. Finally, taxa-specific models as opposed to traditional multivariate methods (e.g., Legendre and Legendre 2012) can be used to resolve uncertainty among competing management options through explicit prediction, providing a direct link between ecological research and management (e.g., Beesley et al. 2014, King et al. 2016).

Model structure

Our most basic model assumed that macroinvertebrate counts conform to a Poisson distribution as, $y_{i,j,k} \sim \text{Poisson}(\lambda_{i,j,k})$, where $\lambda_{i,j,k}$ is the mean of the Poisson distribution (i.e., expected count of taxon i at site j in replicate sample k). However, to account for extra-

Poisson variation in our count data and any non-independence in counts due to the nested structure of our spatial sampling design, we considered seven additional distributional assumptions to describe the error structure of our data. These distributional assumptions were made explicit by allowing $\lambda_{i,j,k}$ to vary randomly across different components of our spatial sampling design with three layers of random effects as,

$$\log(\lambda_{i,j,k}) = \mu_{i,j} + \varepsilon_{i,s} + \eta_{i,j} + \varphi_{i,j,k} \quad (1)$$

where $\mu_{i,j}$ is a fixed effect that models the expected count of taxa i at site j ; the parameter $\varepsilon_{i,s}$ is a random effect that models extra-Poisson variation in counts across stream systems (indexed by s) and accounts for potential non-independence among paired sample sites due to spatial auto-correlation; the parameter $\eta_{i,j}$ is a random effect that models extra-Poisson variation in the mean counts among sites; and the parameter $\varphi_{i,j,k}$ is a random effect that models extra-Poisson variation among all samples collected (indexed by k). These random effects were specified as deviates drawn from Normal distributions with a mean of zero and estimated standard deviations (e.g., $\varepsilon_{i,s} \sim \text{Normal}(0, \sigma)$). The model that excluded all of the three random effects (i.e., $\log(\lambda_{i,j,k}) = \mu_{i,j}$) formalizes the assumption that the count data for each taxon conform to a Poisson distribution. The model that included only the random effect across replicate samples (i.e., $\log(\lambda_{i,j,k}) = \mu_{i,j} + \varphi_{i,j,k}$) assumes a Poisson log-Normal error structure (Gelman and Hill 2007), which is similar to modeling counts with a negative binomial distribution. Models that include the other two random effects make explicit the assumption that extra-Poisson variation in counts occurs across the different streams in our study area (i.e., $\varepsilon_{i,s}$) and/or that extra-Poisson variation in counts occurs across all sample locations (i.e., $\eta_{i,j}$). We considered error structures that included all possible combinations of

$\varepsilon_{i,s}$, $\eta_{i,j}$, and $\varphi_{i,j,k}$ (i.e. eight possible combinations) to determine an appropriate error structure for our data with useful predictive properties for all taxa.

We incorporated our ecosystem covariates into the model with a linear model structure as,

$$\mu_{i,j} = \beta_{0,i} + \beta_{1,i}I_j + \beta_{2,i}R_j + \beta_{3,i}I_jR_j \quad (2)$$

where $\beta_{0,i}$ is the taxa-specific intercept of the model representing the average relative abundance of each family. The parameters $\beta_{1,i}$ and $\beta_{2,i}$ represent the potential covariate effects of each ecosystem component on taxa i . We chose to model the potential interaction between the effect of R and I with the parameters $\beta_{3,i}$ because other studies have suggested that local-scale and landscape-scale scale factors can interact to determine the distribution of stream biota (e.g., Violin et al. 2011). All β parameters were formulated as random effects drawn from community-level hyper-distributions, specified as Normal distributions with estimated means and standard deviations. We fit our models in a Bayesian context using a Gibbs sampler implemented in JAGS (Plummer 2003). Description of model fitting methods and code is presented in Appendix S4 of the online supplemental information.

Model evaluation

We performed our model evaluation in two distinct steps. First, we evaluated the fit of our eight general model error structures using a posterior-predictive check (Hooten and Hobbs 2015, Broms et al. 2016). For each site, we simulated count data from the posterior distributions of our model parameters and compared the simulated counts to the observed counts with Pearson's chi-square statistics. The Pearson's chi-squared statistics were

calculated for each sample as $\chi^2_{i,j,k} = (y_{i,j,k} - \lambda_{i,j,k})^2 / \lambda_{i,j,k}$, where $y_{i,j,k}$ is the observed count data and $\lambda_{i,j,k}$ is the expected count. We summed the fit statistics across all replicate samples and sites to provide a measure of model fit for each taxon. To assess whether the observed values of the summed Pearson's chi-squared statistics for each taxon were likely given the assumptions of the models, we used the posterior samples of the simulated data to approximate the sampling distribution of these statistics (i.e., $(y_{i,j,k,z}^{\text{predicted}} - \lambda_{i,j,k,z})^2 / \lambda_{i,j,k,z}$ for each of z MCMC iterations). We reported the corresponding percentile of the fit statistic relative to its predicted distribution as a measure of model fit for each taxon with values > 97.5 or < 2.5 percentile indicating statistically significant lack of fit.

Secondly, we assessed the importance of different covariates for describing variation in counts for the model with the simplest error structure judged to adequately fit the data (i.e., Pearson's chi-squared value between the 2.5 and 97.5 percentile of predicted distribution for all taxa). Information theoretic approaches to variable selection, such as AIC and DIC, are often unreliable when applied to complex Bayesian hierarchical models with multiple random effects (Celeux et al. 2006, Millar 2009, Hooten and Hobbs 2015, Broms et al. 2016); therefore, we applied a simplified procedure based on credible interval overlap with zero that is common in the ecological literature (e.g., White et al. 2013, Beesley et al. 2014, King et al. 2016). Thus, for the simplest error structure with adequate fit, we evaluated support for each covariate by assessing if the 95% Bayesian credible intervals overlap with zero to approximate and $\alpha = 0.05$. Parameters whose 95% credible intervals excluding zero we considered 'statistically significant'.

Example management scenarios

To operationalize our results, we used our fitted model to predict changes in the macroinvertebrate assemblage structure due to changes in R and I . We specifically chose changes to ecosystem components that could broadly represent common urban stream management and restoration strategies and thus highlight the implications of our results to stream management. We chose stylized management scenarios that would result in ecosystem component values that changed from severely impacted to values that approximated healthy systems based on the scientific literature and the observed range in our data. We first defined the characteristics of three general types of stream sites as (1) a severely impacted site with high imperviousness ($I = 33\%$, representing the upper 75th percentile in our data set) and low riparian integrity ($R = 2.5$, representing lower 25th percentile), (2) a site with high imperviousness ($I = 33\%$) and a high riparian integrity ($R = 7.0$, upper 75th percentile in data), and (3) a site with low imperviousness ($I = 5\%$, lower 25th percentile in data) and low riparian integrity ($R = 2.5$). We then simulated two stylized management actions each targeting one of the two ecosystem drivers. The first action was to improve the integrity index (R) of the riparian zone from 2.5 to 7.0. We invoked this increase in R by simulated revegetation of a riparian zone with native trees because this is a common river restoration method in practice in Western Australia (WRC 1999, DoW 2002, SRT 2009). Thus, we chose a management scenario where the proportion of native trees was increased from 10 to 80%, increasing the R score from 2.5 to 7.0. Methods for determining the R increase due to increasing the proportion of native trees is described in Appendix S5. The second hypothetical management action was to reduce the influence of imperviousness at the catchment level by decreasing the I from the value of 33% to 5%, which approximates the expected threshold of impact, beyond which the geomorphology of the stream is substantially altered (Schueler 1994, Stepenuck et al. 2002, Schueler et al. 2009, Vietz et al. 2014). This

could be achieved by management actions such as storm water harvesting (Walsh et al. 2005a, Burns et al. 2012), infiltration and bioretention techniques (Fletcher et al. 2011, Walsh et al. 2012), or the implementation swale drainage (Brabec et al. 2002). Thus, we simulated four management scenarios, the improvement of R for (1) the first and (2) the third stream type and the improvement of I for (3) the first and (4) second stream type (Table 2).

To elucidate the relative performance of each management action, we derived the percent change in the occurrence probability of specific families and the richness of functional feeding groups. Since there are currently no specific restoration objectives for macroinvertebrate communities in Perth, we assumed that our management objective was to maximize the percent change in these metrics with restoration activities. Although general, assuming this restoration objective is broadly applicable as reducing I and increasing R move the system towards the reference condition of no human impact, which is the universal goal implicit in stream restoration. The taxonomic richness of the functional feeding groups was calculated as,

$$S_{group} = \sum_i \psi_{i=group} \quad (3)$$

where ψ_i is the occurrence probability of taxa i , which was calculated as the Poisson probability of obtaining a count > 0 given λ_i as, $\psi_i = 1 - e^{-\lambda_i}$. The term “*group*” indicates the functional feeding group.

RESULTS

The data included in the study totaled 63 macroinvertebrate samples collected at 21 sites located on 14 urban streams. The R score ranged between 0.30 and 8.33 and the I ranged between 1.3% and 55.0%. The R and I had a Pearson correlation coefficient of -0.12. This range of values with low correlation across our sites suggests adequate contrast in the data to separate the potential effects of R and I . We observed 29 families of macroinvertebrates (Table 1), with the most common including Calanoida, Chironomidae, Cyprididae, and Polychaeta, which were all observed in > 50% of the samples (Table 1). The rarest families collected included Ceinidae, Coenagrionidae, Hydridae, and Planorbidae, which were all observed in only a single sample (Table 1). All model structures evaluated achieved convergence for all parameters as indicated by Gelman-Rubin statistics ≤ 1.1 (Gelman et al. 2004). Our posterior-predictive check indicated that all model error structures that accounted for extra-Poisson variation across replicate samples (i.e., included the random effect, $\varphi_{i,j,k}$) resulted in summed Pearson's chi-squared values within the 2.5 and 97.5 percentile (i.e., Table 3) of the predicted distribution for all taxa. Models that excluded this random effect resulted in up to 17 taxa with summed Pearson's chi-squared values outside of the 2.5 and 97.5 percentiles of the predicted distributions, indicating lack of model fit for these taxa (Table 3). The structurally simplest model error structure that fit the count data for all taxa included only the random effect $\varphi_{i,j,k}$ (i.e. error structure 4 in Table 3, $\log(\lambda_{i,j}) = \mu_{i,j} + \varphi_{i,j,k}$) and produced summed Pearson's chi-squared values between the 16th and 67th percentiles of the predicted distributions for all taxa suggesting little under or over dispersion of the data relative to the model predictions. As this was the simplest model with adequate fit, all further analysis was conducted with this model.

Our model indicated that the effect of R was variable among macroinvertebrate families although macroinvertebrates tended to increase in abundance as R increased (Fig. 2a). We observed six macroinvertebrate families that demonstrated a statistically significant positive response to increasing R , including Simuliidae, Parastacidae, Ancyliidae, Leptoceridae, Chydoridae, and Limnysiidae (black dots on Fig. 2a). These families were among the rarer taxa in our samples (collected in ≤ 13 of 63 samples) and represented a mix of functional feeding groups (Table 1). Alternatively, the effect of I on macroinvertebrate abundance tended to be weak and consistently negative across families (Fig. 2b). This suggests that macroinvertebrate abundance should weakly decline with increasing I , on average, although no statistically significant responses were detected for any specific macroinvertebrate family (Fig. 2b).

The interaction effect between R and I was weak and consistently negative across all taxa (Fig. 2c). Two families, Polychaeta and Calanoida, demonstrated a statistically significant interaction (Fig. 2c, black dots), indicating that the effect of R and I will be enhanced when I is low and R is high, respectively. Thus, the predicted increase in the abundance of macroinvertebrates as R increases will be lower when I is high (Fig. 3, upper left panel) and higher when I is low (Fig. 3, bottom left panel).

Our management scenarios elucidated the ability of different management interventions to alter the macroinvertebrate assemblage. For instance, we found considerable inter-family variation, but generally a stronger response to revegetation of the riparian zone (i.e., increased R , Fig. 4a) than decreases in imperviousness at the catchment level (i.e., decreased I , Fig. 4b). This suggests that, of our two forms of restoration, riparian revegetation had a greater potential to alter the macroinvertebrate assemblage structure. Furthermore, the greater ability of R improvements to impact the macroinvertebrate assemblage held true at the functional level. The shredder functional feeding group was most affected by both

hypothetical management interventions, but responded the most to revegetation of the riparian zone (Fig. 5).

We also found that the potential to alter the macroinvertebrate assemblage structure was greatest for both management options when *R* and *I* were at values representing ecologically healthy conditions. For example, revegetating the riparian zone created the largest response by the macroinvertebrate assemblage when *I* in the catchment was low (i.e., grey bars larger than black bars on Fig. 4a and Fig. 5a). Similarly, reduction in *I* created the largest response by the macroinvertebrate assemblage when riparian integrity was high (i.e., grey bars larger than black bars on Fig. 4b and Fig. 5b). This indicates that the strongest macroinvertebrate response to restoration will be achieved by repairing both the *R* and *I* to healthy levels at severely impacted sites. However, if only one management action can be taken, increasing riparian integrity is expected to produce greater results than reducing the influence of imperviousness.

DISCUSSION

Our results indicate that in the flat, permeable landscape of Perth, the greatest macroinvertebrate response will result from combined restoration efforts targeting both riparian integrity at the local scale and the effect of imperviousness at the catchment scale. However, an important finding of this study is that local-scale riparian restoration was predicted to have a stronger influence on the macroinvertebrate community than restoration of catchment-scale imperviousness when applied alone. This result demonstrates a nuance to the widely supported paradigm that stormwater runoff associated with catchment imperviousness is the overwhelming determinant of urban stream health and overshadows local-scale processes (e.g., Roy et al. 2005, Walsh et al. 2007, Imberger et al. 2011). These results add to the growing awareness of heterogeneity in the Urban Stream Syndrome and

suggest that alternative management approaches may be needed in flat, permeable landscapes. Furthermore, our results can be used by restoration practitioners to determine the most effective management actions and prioritize restoration sites in Perth, thus, improving the outcomes of investment in urban stream management.

Our results are likely driven by the geoclimatic character of the Swan Coastal Plain and the common water management practices of Perth. For example, the Swan Coastal Plain's high infiltrative capacity (flat physiography and sandy soils) coupled with the widespread infiltration of roof runoff are likely to attenuate urban hydrologic stress, and increase the importance of local-scale riparian influences. To date, no detailed hydrologic studies have been conducted in Perth to confirm this mechanism; however, a meta-analysis in the USA has found that urban flow stress is reduced in permeable landscapes (Hopkins et al. 2015). Additionally, the unique climate with protracted six-month dry period experienced by urban streams in Perth may also diminish the importance of hydrologic stress and increase the importance of riparian and water quality influences of stream health. Indeed, a recent review by Hale et al. (2016) has emphasized the potential importance of climate to flow stress in urban streams. Studies across broad spatial scales that capture geoclimatic variation may be needed to disentangle the effects of climate, physiography and water management on stream health and determine the relative importance of different ecosystem drivers.

Our study also revealed that while macroinvertebrate families from various functional feeding groups were influenced by local-scale riparian integrity, 'shredders' were the group predicted to increase the most as a result of management intervention. This finding has implications for managing the function of ecosystems, because shredders play an important role in nutrient processing by converting coarse organic matter (e.g., leaves) into finer organic matter, facilitating the assimilation of detritus by other invertebrates (Wallace and Webster 1996). Although we do not know the exact functional relationship between shredder

abundance and nutrient processing, this role is particularly important in streams that are driven predominantly by allochthonous inputs, such as the small streams of our study region (Bunn 1988). Thus, we expect that an intact riparian zone will deliver benefits to shredders on the Swan Coastal Plain by increasing the availability of their food resources. This assertion is supported by unpublished data collected during our macroinvertebrate sampling, which found that sites with higher riparian integrity contained up to an order of magnitude more coarse particulate organic matter than sites with low riparian integrity (Appendix S6). Conversely, the diminishing benefit of riparian restoration to shredders as total catchment imperviousness increases is likely driven by a reduction in the ability of the stream to retain leaves in the face of increasingly “flashy”, scouring flows (Paul et al. 2006, Swan et al. 2008).

Modeling advantages

A possible reason why we were able to detect the macroinvertebrate and shredder group response to increased riparian integrity is that our flexible modeling approach allowed the identification of subtle patterns often missed by standard multivariate techniques. Indeed, many of the families that demonstrated statistically significant responses to riparian management were among the rarer taxa in our samples (Table 1). Because, multi-taxa hierarchical model structures tend to produce more reliable parameter estimates for taxa with sparse data (Link and Sauer 1996, Sauer and Link 2002, Zipkin et al. 2010, Ovaskainen and Soininen 2011), it is likely that these patterns would have been missed with alternative analyses. Indeed, a separate analysis of these data where species were assumed independent produced many parameters estimates with high uncertainty and several extreme parameter estimates for rare species, some of which were statistically significant (unpublished data). Furthermore, analyses that model community summary metrics directly would have likely

missed these patterns. Changes in community metrics that summarize multiple taxa, such as species richness, tend to be disproportionately determined by the most abundant taxa and often lack the detail necessary to inform management decisions (Vollmer et al. 2016). These limitations of community summary metrics have been discussed in the ecological literature, with multi-taxa approaches suggested as a likely solution (Olden 2003, Ovaskainen and Soininen 2011, Gwinn et al. 2016).

Management implications

Although stream restoration at the catchment scale is clearly an efficacious goal, our results suggest advantages of small-scale riparian restoration for flat, permeable landscapes such as the Swan Coastal Plain. Furthermore, there are several other characteristics of riparian restoration that make it a beneficial restoration option. Firstly, local-scale riparian restoration tends to be less costly and logistically simpler than managing stormwater runoff at the catchment scale. For example, current approaches to catchment-scale stormwater management, such as rain gardens, infiltration systems, permeable pavements, green roofs and stormwater harvesting systems require public support and participation, which can be difficult to obtain (Roy et al. 2008, Roy et al. 2014, Brown et al. 2016). Brown et al. (2016) reported 39.5% participation by eligible households in a catchment-scale residential water harvesting program in Melbourne, Australia, which is unlikely to substantially reduce the hydrologic effects of urbanization (Vietz et al. 2014). Roy et al. (2014) found that 30% residential participation in a rain harvesting program in a suburban neighborhood in Cincinnati, Ohio, USA was not enough to improve local stream health and suggested that additional treatment of public roads, apartment buildings, and parking lots may be required, which would require additional participation by government and private industry. Although there have been successful applications of catchment-scale management (Walsh et al. 2015),

these examples illustrate the difficulty in implementing and succeeding at such projects. On the other hand, local-scale riparian restoration is well accepted by the general public and is enacted commonly by community and government organizations worldwide (Bernhardt et al. 2005, Kail et al. 2015). Riparian restoration can also meet objectives beyond typical stream-health goals. For example, restoring riparian forests can benefit many animal and plant groups and species such as birds (Trathnigg and Phillips 2015), bats (Calvert and Neiswenter 2012), and herpetofauna (Banville and Bateman 2012). Furthermore, promoting trees in an urban landscape can provide social, economic and health benefits to humans such as increased property value, reduced heating and cooling costs, and increased sense of wellbeing (Roy et al. 2012). Thus, we see riparian restoration as a practical management option to improve stream integrity that can produce a high return on investment.

Study limitations

As with most data-driven research, our inference was limited by our experimental design in some key ways. Firstly, the urban gradient that we collected our data across spanned between predominantly urban to agricultural land uses. Thus, our most ‘pristine’ sites were in an agricultural setting and influenced by the legacy of this land use (Harding et al. 1998). However, that our results were generated in this urban/agricultural context, does not diminish the integrity of our management predictions because any restoration efforts that occur in a city that developed in an agricultural setting will be equally constrained by this land-use legacy. Most importantly, the issue of land-use legacies adds credence to the growing awareness of the importance of heterogeneity in the Urban Stream Syndrome and appropriate management solutions. Secondly, the complexity of our model and, thus, the questions that we could ask were limited by our small number of samples collected in a single in-stream habitat. Because of our sparse data, we could not statistically evaluate more

complex relationships of macroinvertebrates with the components that make up the riparian integrity score, such as buffer width, continuity and canopy cover. Nor could we statistically evaluate other more complex nonlinear model structures such as threshold effects. These unaccounted-for variables limit the detail with which we can effectively inform management and undoubtedly are a source of ‘nuisance’ variation in our data. However, despite these shortcomings, our analysis could discern general patterns that represent a small but important advancement for urban stream management, particularly for flat, permeable landscapes. However, to inform management in greater detail, a field sampling program of comprehensive spatial coverage at the landscape and site scales would be needed. Programs such as this currently exist in Perth (Department of Water, Government of Western Australian, unpublished data) and analyzing these existing datasets is a clear next step.

CONCLUSIONS

The Urban Stream Syndrome is a global phenomenon that increases homogeneity among urban stream systems (Booth et al. 2016). However, there is a growing appreciation for heterogeneity of the syndrome that can be dependent on the geoclimatic contexts and land-use legacy effects (Booth et al. 2016, Utz et al. 2016, Hale et al. 2016). Acknowledging and understanding this heterogeneity will help stream managers to transition from general management practices to more successful locally tailored management strategies. This study represents one of few in flat, permeable landscapes and the first aimed at informing urban stream restoration in Perth. Our study suggests that local-scale riparian restoration has the potential to generate greater benefits to macroinvertebrates of urban streams in the city of Perth than managing the influences of catchment-scale impervious surfaces, alone; however, much work needs to be done to understand the idiosyncrasies of the Urban Stream Syndrome in flat permeable landscapes such as Western Australia and effectively inform management.

Research that investigates the spatial scales at which land-use influences attenuate and the relative importance of various land-use practices and local features are among the many information gaps encumbering conservation and restoration practices. We want to encourage future research that can directly answer these important management questions and guide future restoration projects.

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Supporting Information

Additional supporting information may be found in the online version of this article at <http://onlinelibrary.wiley.com/doi/10.1002/eap.xxxx/supinfo>

Data Availability

Data available from the Dryad Digital Repository: <https://doi.org/10.5061/dryad.7nm6b>

TABLES

Table 1. Macroinvertebrate families, functional feeding groups and sample sizes. Literature sources for functional-feeding group assignment can be found in Appendix S1 of the online supplementary information.

Family	Functional-feeding group	Total collected	Total samples observed in
Perthiidae	Shredder	11	2
Ceinidae	Shredder	1	1
Ceratopogonidae	Predator	192	16
Simuliidae	Collector/filterer	365	4
Chrysomelidae	Shredder	5	1
Dytiscidae	Predator	6	5
Calanoida	Collector/filterer	1274	32
Parastacidae	Shredder	10	7
Coenagrionidae	Predator	1	1
Aeshnidae	Predator	19	10
Libellulidae	Predator	109	27
Richardsonianidae	Predator	36	8
Hydrophilidae	Shredder	6	2
Ancylidae	Scraper	110	13
Caenidae	Gatherer	192	15
Culicidae	Collector/filterer	66	13
Hydriidae	Collector/filterer	1	1
Chironomidae	Multiple	3728	62
Cyprididae	Collector/filterer	8310	45
Palaemonidae	Gatherer	12	3
Lymnaeidae	Scraper	538	30
Pomatiopsidae	Scraper	38	8
Planorbidae	Scraper	2	1
Ecnomidae	Gatherer	67	8
Leptoceridae	Shredder	29	5
Corixidae	Predator	6	3
Chydoridae	Scraper	318	9
Limnysiidae	Predator	33	11
Polychaeta	Gatherer	2418	50

Table 2. Four stylized management scenarios. R represents the riparian integrity score, where values of 2.5 and 7.0 indicate a highly degraded and a health intact riparian zone, respectively. I represents the percent catchment imperviousness, where values of 33% and 5% indicate high and low levels of catchment imperviousness, respectively. “Site condition” is the condition of the stream site before the management action is applied.

Management scenario	Site condition	Management action
1	$R = 2.5, I = 33\%$ ¹	Increase R from 2.5 to 7.0
2	$R = 2.5, I = 5\%$ ²	Increase R from 2.5 to 7.0
3	$R = 2.5, I = 33\%$ ¹	Decrease I from 33% to 5%
4	$R = 7.0, I = 33\%$ ³	Decrease I from 33% to 5%

¹ first stream type; ² second stream type; ³ third stream type

Table 3. The corresponding percentile of the Pearson's chi-square fit statistics relative to its predicted distribution. Values > 0.975 or < 0.025 indicate statistically significant lack of model fit at $\alpha = 0.05$ (i.e. values in grey). Column labels 1-8 indicate model error structures that either include or exclude random effects modeling extra-Poisson variation across stream systems (i.e. $\varepsilon_{i,s}$), sample sites (i.e. $\eta_{i,j}$), and/or replicate samples (i.e. $\varphi_{i,f,k}$). 1 = null; 2 = $\varepsilon_{i,s}$; 3 = $\eta_{i,j}$; 4 = $\varphi_{i,j,k}$; 5 = $\varepsilon_{i,s} + \varphi_{i,j,k}$; 6 = $\eta_{i,j} + \varphi_{i,j,k}$; 7 = $\varepsilon_{i,s} + \eta_{i,j}$; 8 = $\varepsilon_{i,s} + \eta_{i,j} + \varphi_{i,j,k}$.

Family	Model error structure							
	1	2	3	4	5	6	7	8
Perthiidae	0.39	0.76	0.55	0.31	0.34	0.33	0.55	0.35
Ceinidae	0.54	0.65	0.62	0.60	0.63	0.62	0.63	0.61
Ceratopogonidae	1.00	0.99	0.99	0.16	0.16	0.23	0.99	0.21
Simuliidae	1.00	0.99	0.99	0.24	0.24	0.25	0.99	0.25
Chrysomelidae	0.75	0.71	0.58	0.34	0.32	0.33	0.57	0.32
Dytiscidae	0.87	0.80	0.68	0.67	0.69	0.65	0.68	0.67
Calanoida	1.00	1.00	1.00	0.34	0.45	0.41	1.00	0.44
Parastacidae	0.41	0.35	0.27	0.58	0.50	0.50	0.27	0.48
Coenagrionidae	0.76	0.67	0.58	0.60	0.60	0.60	0.59	0.6
Aeshnidae	0.99	0.84	0.67	0.62	0.58	0.52	0.68	0.53
Libellulidae	1.00	0.98	0.98	0.63	0.64	0.62	0.98	0.64
Richardsonianidae	1.00	0.91	0.67	0.49	0.59	0.53	0.69	0.55
Hydrophilidae	0.89	0.75	0.62	0.43	0.44	0.44	0.63	0.45
Ancylidae	1.00	0.93	0.89	0.41	0.39	0.39	0.88	0.39
Caenidae	1.00	0.95	0.91	0.36	0.40	0.37	0.91	0.40
Culicidae	1.00	0.98	0.96	0.56	0.66	0.68	0.96	0.70
Hydriidae	0.47	0.61	0.55	0.56	0.56	0.56	0.55	0.56
Chironomidae	1.00	1.00	1.00	0.55	0.54	0.55	1.00	0.55
Cyprinidae	1.00	1.00	1.00	0.39	0.42	0.46	1.00	0.46
Palaemonidae	0.95	0.60	0.48	0.33	0.36	0.41	0.48	0.41
Lymnaeidae	1.00	1.00	1.00	0.45	0.40	0.39	1.00	0.40
Pomatiopsidae	0.99	0.88	0.68	0.38	0.33	0.33	0.68	0.31
Planorbidae	0.39	0.64	0.53	0.44	0.44	0.45	0.53	0.45
Ecnomidae	1.00	0.98	0.94	0.28	0.25	0.29	0.94	0.27
Leptoceridae	0.86	0.84	0.69	0.35	0.42	0.39	0.69	0.39
Corixidae	0.91	0.91	0.65	0.48	0.51	0.47	0.66	0.49
Chydoridae	1.00	0.99	0.99	0.21	0.17	0.21	0.99	0.20
Limnesiidae	1.00	0.93	0.89	0.52	0.51	0.49	0.88	0.50
Polychaeta	1.00	1.00	1.00	0.44	0.47	0.50	1.00	0.48

FIGURE LEGENDS

FIG. 1. The location of study sites (numbered) and their catchments on the Swan Coastal Plain, south-western, Australia. Catchment shading indicates the range of imperviousness.

FIG. 2. Posterior summaries of the effect of riparian integrity (R , panel (a)), percent impervious surfaces in the catchment (I , panel (b)), and their interaction ($I: R$, panel (c)) on the abundance of stream macroinvertebrate families. The dots represent the mean of the posterior distributions while the lines represent 95% Bayesian credible intervals. The grey dots and lines indicate parameters that are not statistically different than zero (95% Bayesian credible intervals include zero), while black dots and lines indicate statistical significance. The x-axis is the parameter value; one unit on the x-axis can be interpreted as one unit of abundance on the log scale for every one standard deviation of the covariate.

FIG. 3. Predicted occurrence of macroinvertebrate families as relates to riparian integrity (R , left panels) and percent impervious surfaces in the catchments (I , right panels). The left panels are predictions made when $I = 33\%$ (top panel) and $I = 5\%$ (bottom panel). The right panels are predictions made when $R = 2.5$ (top panel) and $R = 7.0$ (bottom panel). Colored lines indicate families that demonstrated a statistically significant ($\alpha = 0.05$) relationship to R (solid lines) or statistically significant interaction between R and I (dashed lines).

FIG. 4. The predicted percent change in occurrence of macroinvertebrate families between pre- and post-management that either increases riparian integrity (R) or decreases catchment imperviousness (I). Panel (a) represents the effect of riparian zone revegetation that increases native trees from 10% to 85% (R score from 2.5 to 7.0) to a stream site that has 33% catchment imperviousness (Scenario-1) and 5% catchment

imperviousness (Scenario-2). Panel (b) represents the effect of reducing the imperviousness in the catchment from 33% to 5% to a stream site that has an R score of 2.5 (Scenario-3) and 7.0 (Scenario-4).

FIG. 5. The predicted percent change in family richness of functional-feeding groups due to repair of riparian integrity (R , panel a) or repair to percent imperviousness in the catchments (I , panel b). Panel (a) represents the effect of riparian zone revegetation that increases native trees from 10% to 85% (R score from 2.5 to 7.0) to a stream site that has 33% catchment imperviousness (Scenario-1) and 5% catchment imperviousness (Scenario-2). Panel (b) represents the effect of reducing the imperviousness in the catchment from 33% to 5% to a stream site that has an R score of 2.5 (Scenario-3) and 7.0 (Scenario-4).









