The potential and limitations of linking biological monitoring data and restoration needs of urbanized waterways: a case study

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Abstract

The implementation of effective strategies to mitigate the impacts of urbanization on waterways represents a major global challenge. Monitoring data plays an important role in the formulation of these strategies. Using monitoring and historical data compiled from around an urban area (Baltimore, USA), this paper is an assessment of the potential and limitations of the use of fish assemblage monitoring data in watershed restoration. A discriminant analysis between assemblages from urban and reference sites was used to determine components which have been reduced or eliminated from Baltimore area waterways. This analysis produced a strong discrimination between fish assemblages from urban and reference sites. Species primarily associated with reference sites varied taxonomically and ecologically, were generally classified as pollution intolerant, and were native. Species associated with urbanized sites were also native, varied taxonomically and ecologically, and were mixed in pollution tolerance. One factor linking most species associated with reference sites was spawning mode (lithophilic). Spawning habitat limitations may be the mechanism through which these species have been reduced in the urbanized faunas. While this presents a strong general hypothesis, information regarding the specific habitat requirements and responses to urbanization of these species is limited. This represents a major limitation to producing effective restoration strategies based on exact goals and targets. Without these, determining the type and number of restoration activities required to restore ecological communities remains problematic.

Introduction

Urbanization poses an increasing threat to the biodiversity and services provided by aquatic ecosystems worldwide. A majority of global population growth in the future is projected to occur in urban areas; in particular, effects on waterways will be pronounced as impacts integrate in these low lying points on the landscape (Bernhardt and Palmer 2007). Urbanization of watersheds has been shown to have major impacts on aquatic ecosystems (Walsh et al. 2005). Increased runoff is the primary source of impact to aquatic systems, with major impacts on hydrology, stream geomorphology, pollutant load and habitat attributes (reviewed in Walters et al., 2009). Species which are intolerant of these changes can decline or disappear and are replaced by organisms which are tolerant (e.g. Fraker et al. 2002; Helms et al. 2005; Morgan and Cushman 2005). In terms of impact on ecosystems, declines in biodiversity, ecosystem services, and ecosystem complexity are routinely observed pursuant to urbanization (Allan 2004).

Urbanization has had a profound negative impact on aquatic ecosystems. However, ecosystems have often proven to be surprisingly resilient and recent studies have challenged sweeping generalizations and assumptions about its effects (Pickett et al. 2008). There has been widespread improvement in the water quality and biota of many urban waterways since the implementation of the Clean Water Act in the 1970s (e.g. Knopman and Smith 1993; Lyon and Stein 2009). The disproportionately high value of urban ecosystems to large numbers of people has generated much interest in their restoration (Bernhardt and Palmer 2007). Unfortunately, many well- intentioned and expensive stream restoration projects have failed to accomplish the goals of improving ecosystem function and integrity, as shown conclusively by several current reviews (e.g. Bernhardt and Palmer 2011; Violin et al. 2011; Stranko et al. 2012). A process based approach to stream and watershed restoration with distinct targets for habitat traits may be more appropriate in addressing ecological goals (e.g. Beechie et al. 2010).

Palmer et al. (2005) suggested identification of a ‘guiding image’ representing a reasonably attainable level of ecosystem quality and function as a primary need for successful ecosystem restoration. Recently, there have been attempts to create and use guiding images of fish assemblages to assess specific waterway and watershed restoration needs. The Target Fish Community approach (TFC) (Bain and Meixler 2008) is one such method that has been developed and employed in urban waterway restoration (e.g. Blankers and Bain 2010; Meixler 2011). In this method, the expected fish assemblage for an impacted waterway is developed from a survey of fish assemblages of similar waterways meeting a series of criteria (e.g. ecoregion, size, geomorphology, zoogeography), often partially or wholly available from monitoring programs. Comparing this expected assemblage with the observed assemblage from the impacted waterway generates a guiding image of attainable fish assemblages by identifying missing or underrepresented fish taxa. Using life history data for these fish can then suggest causal hypotheses and highlight needs in the habitat which can be addressed through stream restoration, and can be especially useful when incorporated into a modeling framework such as PHABSIM (Waddle et al. 2001; Parasiewicz 2008). This method therefore transcends the typical assessment use of fish assemblage data and has the potential to be useful for meeting ecologically based restoration goals. It is important to note that the successful implementation of this approach depends critically on connecting the underrepresentation of fish taxa with mechanisms responsible for their reduction or elimination, which can then be addressed with stream restoration. While numerous studies have related urbanization and resident fish assemblages, few have confirmed involved mechanisms (Peoples at al. 2011).

An approach is employed here using fish assemblage monitoring data of urbanized and less urbanized streams in Baltimore and its surrounding area, with the goal of assessing the utility of this approach to guide local watershed restoration. Discriminant analysis is used on a large data set of fish assemblages to identify taxa which are underrepresented in Baltimore urban waterways relative to less urbanized reference streams. A review of the attributes of identified species associated with reference and urbanized sites and historical information is conducted to suggest mechanistic causative hypotheses of impact for urbanization of fish assemblages. Limitations of current knowledge to adequately address these hypotheses and provide distinct goals for restoration of habitat are examined.

Methods

Study sites- Watersheds and reaches of waterways included in the study were selected based on criteria that maximized their comparability. A ‘guiding image’ for local fish assemblages was constructed using analysis of monitoring data (Palmer et al. 2005; Bain and Meixler 2008). Watersheds chosen were all from the Baltimore area, were in the same EPA ecoregion (64c- Piedmont Upland), had similar geomorphic properties and aerial extent, and the same stream order near the Fall Line. Urbanized and less urbanized reference sites were divided according to the relative amount of developed and urbanized acreage and the amount of impervious surface (Table 1). This resulted in an urbanized group consisting of the Jones Falls and Gwynns Falls, and a reference group consisting of Little Gunpowder Falls, Winters Run, and Bynum Run.

Data were gathered only on stream reaches satisfying a number of conditions. Stream reaches included in the study were limited to main stem sections, from the Fall line to roughly three miles upstream. Restricting the used data set to these types of reaches ensured that different stream ecosystems would not confound the analysis (e.g. headwater streams, cold water habitats). In addition, using these main stems ensured that conditions at the sites reflected integration of impacts occurring at the watershed scale and not only those pertaining to a particular reach.

Data Collection

Data used in these analyses consisted primarily of fish faunal monitoring surveys conducted by the City of Baltimore’s Water Quality Monitoring Office (WQMO) and the Maryland Department of Natural Resources as part of the Maryland Biological Stream Survey (MBSS) program. In addition, a smaller amount of data was included which was collected directly for the study by University of Baltimore (UB). Restriction of study sites to the above given criteria resulted in 27 separate faunal samples available for comparison, nine of which were in the reference group and 18 in the urbanized group (Figure 2). Breakdown by watershed is as follows: Gwynns Falls (8), Jones Falls (10), Little Gunpowder Falls (2), Bynum Run (3), and Winters Run (4). Data were available from all watersheds from the MBSS program (12 sites total), from Jones and Gwynns Falls from the WQMO program (14 sites total), and the Jones Falls from UB (1 site).

Maryland Biological Stream Survey (MBSS) methods

Data from the Maryland Biological Stream Survey (MBSS) survey included in the study were collected over the years 1996- 2006 as part of a comprehensive monitoring program conducted by Maryland Department of Natural Resources. Data collection techniques have a high degree of standardization to maximize comparability between sites and time periods (Stranko et al., 2010a). For each data collection site, 75 m reaches were measured out and block nets placed at both ends. Double- pass electrofishing was then performed using settings as conditions required. All fish were identified to species, tallied, and released, with specimens only collected for unidentifiable individuals.

Water Quality Management Office (WQMO) survey methods

Monitoring data included in this study collected by the City of Baltimore WQMO spanned the years 2002- 2005 and were limited to sites within the Jones and Gwynns Falls. Methods employed by the WQMO followed those established by the MBSS described above (T. Eucare pers. communication).

University of Baltimore (UB) methods

Data for the UB sampling site was collected over the years 2009- 2011 using a variety of sampling methods. The numbers of species and individuals collected at the UB site were within the range of those collected by MBSS and WQMO at their respective sites. Combination of various sampling methods reduced overall bias present in the individual methods (Hayes 1983; Hubert 1983). Therefore the UB site was valid for comparison with the other sites based on similar sampling effort and quality. For all techniques, fish were identified by species and then released, and kept if the species was not identifiable. Methods used included barrel- style fish traps, nets, electrofishing, and rod and reel. Fish traps were baited and left in situ overnight. Mesh in the fish traps was of both 0.635 cm and 0.318 cm sizes. A small proportion of fish were collected through dip nets, Surber samplers, and seine nets. Rod and reel collection occurred at several points in the lower Jones Falls and was done using both artificial lures and bait on different occasions. Electrofishing consisted of a single pass over a 75 m stretch, during which all individuals were captured, identified and released. We used a Smith- Root electrofisher on settings automatically determined by the unit. All fish captured by netters were temporarily kept in buckets, identified, and quickly released.

Data Analysis

Fish assemblages from urbanized and reference sites were compared for general differences using a multivariate discriminant analysis. The strength and direction of the discriminant scores were used to determine the importance of each fish species to the analysis in delineating urbanized and reference sites. The program used to conduct the analysis was Canonical Analysis of Principal coordinates (CAP), version 12 (Anderson 2004). To prevent excessive attention of the analysis to rare or outlier species, species with fewer than 10 occurrences over all sites were excluded from the analysis. Remaining species counts at each sites were logarithmically transformed ($\log_{10}(x+1)$) and standardized according to species totals over the range 0-1. This combination of data treatments is the same as recommended by Jackson (1993) for being effective for Principal Coordinates Analysis (PCOA), and is appropriate for faunal samples with relatively similar amounts of effort. The discriminant analysis itself was based on a Bray-Curtis distance matrix following an unconstrained PCOA. Strength of the resulting discrimination was assessed using in- program cross validation tests (leave- one- out allocation of sample sites to groups), and the significance of the discrimination was determined using a random permutation test (n= 9999). The test statistic of the latter was given by the squared first canonical correlation (δ^2).

Results

Principal Coordinates Analysis

Substantial separation of groups of sites was observable in the unconstrained PCOA. Unconstrained PCOA identified two axes explaining 62% of the variation in the data set; PCOA axis 1 (35%) and PCOA axis 2 (27%). Visualizing PCOA scores over all sites reveals three fairly well defined groups: reference sites, Jones Falls sites, and Gwynns Falls sites (Figure 1). The primary differentiation for PCOA axis 1 exists between all urban and reference sites, while PCOA axis 2 accounted mostly for differentiation between Jones Falls and Gwynns Falls sites. Sample size was higher for the urbanized site groups (N=18 for urbanized vs. N=9 for reference) and this may account for explicit representation of differences between Jones Falls and Gwynns Falls sites on PCOA axis 2.

Discriminant analysis – Urban vs. Reference streams

A single canonical axis was identified as separating delineated urban and reference groups (λ= 0.97198). Both cross validation and random permutation tests showed the discrimination between urban and reference fish assemblage groups to be strong. Cross validation using leave- one- out allocation resulted in 100% accurate assignment of sites into urban and reference groups (27/27). The random permutation test results showed a highly significant discrimination between urban and reference groups (δ^2= 0.944753, p = 0.0001).

Discriminant scores of sites from this analysis are represented in Figure 2. All sites in the urbanized group had scores which were negative, while those from the reference sites were positive. Within the urbanized group, sites from Jones Falls and Gwynns Falls displayed no apparent pattern with regard to the strength of the discriminant score. For the reference group, the sites with three of the four lowest scores were from Bynum Run samples, while those from Little Gunpowder Falls and Winters Run had the highest scores. This suggests that fish assemblages from the latter two watersheds may be more strongly differentiable in this analysis from urban sites than those from Bynum Run.

Loadings of the derived canonical axis in relation to fish species are shown in Figure 3. A handful of species can be interpreted to be primarily associated with urban sites, another handful are roughly neutral, and a greater number are to varying degrees associated with reference sites. Much greater loadings are observed for species associated with reference sites than those for urban sites, which conversely indicates the importance of underrepresented or absent fish species from urban assemblages in the analysis. Of the species associated with urban sites, only the Yellow bullhead (*Ameiurus natalis*) has a strong relationship.

Life History information review

Generally available information on life history traits for fish species in the upper and lower quartile of all species arranged according to discriminant scores are presented in Table 2. Categorizations of species tolerance are taken from Barbour et al. (1999), those for feeding ecology from Jenkins and Burkhead (1993) and Goldstein and Simon (1999), and for reproductive characteristics from Simon (1999). These and similar distillations of life history information are used almost universally for both assessments of biotic integrity and for the Target Fish Community (TFC) approach (e.g. Karr 1981; Simon 1999; Goldstein and Simon 1999; Meixler 2011). There are a number of discernible patterns in life history traits for fish species associated with reference sites (Table 2a). All species in this group are native, and all but one (Sea lamprey- *Petromyzon marinus*) have different forms of invertivory as part of their trophic classification.

The primary life history trait linking reference associated species was spawning mode. Five of the seven species in the reference associated quartile are classified in the brood hider/ lithophil reproductive guild (Jenkins and Burkhead 1993; Simon 1999). One other species (Northern hogsucker - *Hypentelium nigricans*) is an open substrate lithophilic spawner. The other species (Margined madtom - *Noturus insignis*) is speleophilic (rock cavity nester). The species do not show consistency with general tolerance to pollution, entailing three of intolerant, three of moderate, and one of tolerant character to pollution (Barbour et al. 1999).

Species in the urban associated quartile all show lower discriminant loadings, and a weaker influence on the discriminant analysis than those from the reference sites. All but one species (Largemouth bass - *Micropterus salmoides*) are native. All but one have invertivory as a component of their trophic classification, although there are a number of variations on this general strategy present. There are a number of spawning modes among species in the urban associated quartile. Three of the four species are classified as open substrate spawners, and one (Mummichog - *Fundulus heteroclitus*) is an open substrate spawner phytolithophil. Resistances to pollution vary within this group. Two are classified as being intolerant, three as moderately tolerant, and two are tolerant to pollution (Barbour et al. 1999). The urbanized group had one more tolerant member and one less intolerant member than the reference group.

Historical information review

Historical information from appropriate reaches of the Gwynns Falls is available through the observations of Yingling (1940). Due to its similar size, proximity, and location to the west of the Jones Falls, this information can also be useful in making a rough inference of the fish fauna in the Jones Falls, where little or no historic information on fish faunas is available. Given the purported heavily degraded condition of the lower Jones Falls in the 19th century (Winans 1872; Street 1926) and the impacts already present in the Lower Gwynns Falls (Groffman et al. 2003) at that time, it is somewhat surprising that so many of the species which have been identified as being reduced or eliminated in this study were present or common everywhere over the time period observed by Yingling (1940) (Table 3). Of special note are the Common shiner, Northern hogsucker, and the River chub. These three species are in the top quartile of species associated with reference sites in the present urban vs. reference multivariate analysis, and at the current time are rare or absent in modern fish collections from Gwynns Falls sites. Of these three species, the River chub was experiencing range contraction but the others were common everywhere. Decline in these species in the Gwynns Falls must have therefore taken place during the 20th century. In summary, there is solid evidence that at least some species associated with reference sites in this study were lost from Gwynns Falls, and indirect evidence that they were lost from the Jones Falls at some stage.

Discussion

Differences between urban and reference fish assemblages

Discriminant analysis identified a number of consistent differences between fish assemblages from urban and reference sites. While somewhat sparse, historical accounts provide an additional important context for existing fish assemblages in urbanized streams. The addition of these impacted species to the existing fish communities, coupled with the results from the discriminant analysis, creates a rough ‘guiding image’ (Palmer 2005) analogous to the target fish community (Meixler 2011), indicative of ideal restored conditions in these urban streams.

It is unclear what may have been the cause of the faunal decline observed by Yingling (1940). Many industrial impacts were already present on this waterway in the 19th century (Groffman et al. 2003). Yingling believed that declining water quality and altered hydrology from dams were responsible. Increasing development and expansion of the city of Baltimore into the Gwynns Falls watershed at that time may have accounted for it, and may have represented the beginning of the true impacts of urbanization on the Gwynns Falls ecosystem. An analysis of land use proportions in the Gwynns Falls watershed over the period 1938- 1999 shows a dynamic transition from a very heterogeneous watershed to one dominated by low permeability urban land use (Wehling 2001). In 1938, 2994 ha in the watershed had low permeability urban use. By 1957 this number had risen to 5,412 ha, by 1971 7,592 ha, and in 1999 was 10,735 ha. During this period watershed alteration was driven by the rise of the automobile and expansion of the suburbs (Foresman 2003). The timing of changes in this watershed coincide with the faunal declines documented by Yingling (1940). Available historical information suggests that changes in the Gwynns Falls fish assemblage are due to 20th century urbanization of the watershed.

In contrast, the reference areas in the study differ from the urbanized watersheds in having greater proportions of forested area, and far lower amounts of medium to high density residential areas associated with impervious surface (Table 1). While the amount of agricultural lands is greater in Little Gunpowder Falls and Winters Run, the problems presented by these land use types are somewhat offset by the fact that they are not impervious surfaces. The fact that Bynum Run scored the weakest in site association with reference sites (Figure 2) is consistent with its rank in terms of urbanization. Bynum Run is intermediate in terms of impervious surface, medium- high residential land use, and percent forested area. This last point highlights the likely importance of land use traits in the persistence of sensitive components of the fish fauna in these streams.

Many of the species associated with reference areas in the current study have been identified as being susceptible to urban conditions in previous studies. Pirhalla (2004), using the overall Maryland MBSS data set, found Common shiner, Cutlips minnow (*Exoglossum maxillingua*), and Northern hogsucker to be the most intolerant species of urban conditions in the Eastern Piedmont ecoregion, and additionally River chub in the statewide analysis. The current analysis reflects these patterns partially, although several fish reverse affinities. For example, Longnose dace (*Rhinichthys cataractae*) and satinfin shiner (*Cyprinella analostana*) are considered to be moderately tolerant species and creek chub are considered to be tolerant. However, in the data presented here, Longnose dace and Satinfin shiner are associated with urbanized habitats. Meixler (2011), in her construction of a target fish community for a river in Massachusetts, found the Common shiner to be the primary underrepresented species. In the Southeastern Pennsylvania eastern Piedmont, Horwitz et al. (2008) found that Northern hogsucker and to a lesser extent, Common shiner, were negatively associated with urbanization while the Mummichog was the only species shared by both studies as being associated with urban sites. In parallel with the current study, they also noted that Longnose dace and Satinfin shiners could be found in some urban forested areas. Many species identified as being underrepresented or absent from urban areas in this study have been similarly found to be sensitive to urbanization in other studies. The consistency with which these species are identified across studies and areas suggests that they are 'urban indicator species', which are sensitive specifically to alterations in the habitat in urbanized watersheds.

Urbanization impact mechanisms on fish communities

In this study, consistency of the fauna reduced or eliminated from the urban species assemblages begs the question of why and how these species were impacted, and not others. A review of available life history information results in some speculative hypotheses in this regard. While here the focus is on species that are associated with reference sites, it is important to note that it would also be of interest to determine why intolerant species persisted in urban areas.

Water quality tolerances (Barbour et al. 1999) of both species associated with reference and urban sites were heterogeneous. While the highest quartiles of species associated with these groups were slightly different in pollution tolerance (reference sites had one more intolerant and one less tolerant species than urban sites), both were a mix of intolerant, moderately tolerant, and tolerant species. This is suggestive of the fact that water quality tolerance (*sensu* Barbour et al. 1999) may not be the overriding issue in the difference between urban and reference assemblages. An overview of some water quality issues in Baltimore urban streams may reflect this heterogeneity. On the Jones and Gwynns Falls, continuing impact from degrading sewage infrastructure has resulted in mostly organic pollution, including periodic inputs of raw sewage at various locations, increased subsidy of nutrients, and high levels of coliform bacteria (Kaushal and Belt 2012; S. Kemp and W. Pecher unpubl. data). Apart from the primary impacts of urbanization (i.e. runoff and effects), these represent the main challenges to water quality remaining in these systems. Loss of industry, improved water regulation, and an increasing distance from the area's industrial legacy have all resulted in the improvement of water quality, as is the case for many stream and river systems (e.g. Knopman and Smith 1993; Lyon and Stein 2009). Aspects of these streams' geomorphology may moderate the influence of urbanization pressure. Brown et al. (2009) found that response of fish and macroinvertebrates to urbanization pressure varied strongly by region, even when impacts were severe. Both streams have a moderate to high gradient over the stream reach. Elevated rates of flushing through high, flashy flows may act to prevent accumulation of contaminated sediment and organic waste. In the Jones Falls, dissolved oxygen readings are consistently above 7.0 ppm (MD Dept. of Environment 2009; Kemp unpubl. data). The heterogeneity in the tolerances of fish species found in the urban sites in this study seems to suggest a partial effect at best of water quality on fish assemblages.

Trophic classifications are fundamental metrics used in calculating biotic indicies of impact such as the Index of Biotic Integrity (IBI- Karr 1981; Roth et al. 2000). Trophic metrics are included in IBI calculations because they reflect stable connections in the feeding structure of a diverse ecosystem. A typical symptom of the degradation and simplification of ecosystems is that specialists are replaced by generalist species (e.g. Scott and Helfman 2001). For example, high proportions of insectivorous cyprinids and piscivores and low proportions of omnivorous species are correlated with reference conditions for Midwestern streams (Miller et al. 1988). Numerous regional variants on trophic metrics for the IBI have been developed according to the conditions present in the region (e.g. Hughes and Oberdorff 1999). For Maryland Eastern Piedmont streams, regional development of a fish IBI incorporated percent omnivores, generalists, and invertivores as an indicator of increased anthropogenic stress (Roth et al. 2000). However, in the current study, fish in the upper quartile of association with reference sites with a single exception (Sea lamprey) are at least partial invertivores, and all species similarly associated with urbanized sites are at least partial invertivores. Given the lack of major differences between the two groups it is difficult to speculate on a cause of elimination directly related to dietary or trophic factors.

Lack of suitable spawning habitat or other spawning related reproductive failures would impact the persistence of a population of fish. Species associated with reference sites in this study are consistently classified as lithophilic spawners. Strong impacts of urbanization on lithophilic spawners are well documented and range from loss of suitable substrate, embeddedness of substrate, lack of bed stability and complete loss of nests, and siltation of interstitial spaces leading to lack of flushing of waste and oxygen exchange (Berkman and Rabeni 1987). The general results of this study mirror those of Peoples (2010), who concluded that effects of urbanization on spawning habitat for lithophilic and speleophilic spawners was a primary mechanism in limiting these groups of species in urbanized habitats. The lithophilic spawners in the reference associated group in this study range encompass a variety of specific spawning types. The River chub and the Cutlips minnow construct large pebble mound nests (Jenkins and Burkhead 1993). The Common shiner spawns using pit building or broadcasting on clean gravel and sand but also uses the nests of other species (Johnston 1999), including River chub (Miller 1964). The Northern hogsucker is classified as an open substrate spawner and broadcasts eggs over clean gravel and sand substrate, and is not typically known to use the nests of other species (Raney and Lachner 1946). The Creek chub (*Semotilus atromaculatus*) and the Sea lamprey are both spawning pit constructors (Jenkins and Burkhead, 1993). Presumably all of these species could be impacted by one or more of the mechanisms affecting lithophilic spawners listed above. As noted by Miller (1964), pebble mound nests of River chub are sometimes obliterated by high flows. Increased frequency of high flows is diagnostic of urbanized watersheds with high impervious surface such as the Jones and Gwynns Falls (Table 1; Walsh et al., 2005). Broadcasting species are thought to be some of the most severely affected by urbanization since they neither clean the substrate during spawning nor provide protection to developing embryos (Johnston 1999). Siltation of spawning pit nests would presumably impact the exchange of materials from developing eggs, as it does in trout redds (Wood and Armitage 1997).

Lithophilic spawning behavior places these species at risk for urbanization, and the consistency with which reference associated species are related to this life history trait highlights its validity as a reasonable mechanism for their reduction or loss in urban fish assemblages.

In further support of the hypothesis that lithophilic spawning is a limiting factor is the fact that none of the species associated with urban sites in the analysis are classified as lithophilic spawners. These species range over a variety of pollution tolerances and feeding guilds. This group does contain some (4 out of 7 species) open substrate spawners. While they are not considered to be nest builders (Woolcott and Maurakis 1988), dace in the genus *Rhinchthys* partially clean some substrate in the process of spawning (Jenkins and Burkhead 1993), thus possibly reducing their susceptibility to siltation. The Mummichog has eggs which cling to surroundings, which may help in resisting the effects of heavy urban flows (Murdy et al. 1997). Regardless of the mode of reproduction, however, none of these species in this group are explicitly lithophilic spawners.

Further review of life history information suggests that community- level species interactions in nesting ecology may also play a role in determining urban fish assemblages here. Nest associations between species have been found in about 33 species of cyprinid in N. America, with potential fitness benefits for both host and associate species (Johnston and Page 1992). Benefits to reproductive success include the improvement and cleaning of substrate (Johnston 1999) or shared interspecific parental care of broods (McKaye 1981; Johnston 1994). Large, conspicuous nests of River chub are known to be used by a number of other species, including the Common shiner, Creek chub, Longnose dace, and the Rosyface shiner (*Notropis rubellus* -Cooper 1980; Jenkins and Burkhead 1993). In fact, mound building nests by species such as River chub are occasionally used preferentially by Common shiners (Raney 1940; Miller 1964). Spawning aggregatons of Rosyface shiners have been observed over River chub nests in streams included in this study (Little Gunpowder Falls- pers. Obs.). This species was not found at any site in this study where River chub were absent (unpubl. data). The closely related Bluehead chub (*Nocomis leptocephalus*) may ease urbanization effects on spawning substrate for itself and other associates where present by providing clean pebble mound nests similar to that of the River chub (Peoples et al. 2011). Mound nests constructed by Cutlips minnow are also used by Common shiner and Rosyface shiner (Jenkins and Burkhead 1993). While none of these nest associations are obligate in nature, both Common shiner and Rosyface shiner are otherwise spawners over unmodified substrate, which suffer disproportionately from siltation effects (Johnston 1999). It is therefore possible that the loss of a nesting associate such as a River chub or Cutlips minnow could seriously impact population dynamics of these species through reduced reproductive success.

In summary, the strongest hypothesis regarding the reduction or elimination of reference species in this study is that negative effects of urbanization on spawning habitat for lithophilic, and possibly speleophilic spawners, has prevented these populations from persisting. Neither species feeding guilds nor pollution tolerance can explain the consistent reduction or disappearance of these species. There are however, a number of specific hypotheses which cannot be adequately addressed given the current level of knowledge about life history of eliminated species.

Limitations of species specific information

The capacity to propose valid mechanisms for the elimination or reduction of fish species in urbanized areas is limited by the state of knowledge regarding the life history of these species and lack of knowledge of specific impacts from urbanization. For this reason it is difficult to specify specific habitat requirements for many of the species in this study, which is necessary information for the proper determination of restoration goals.

Thermal tolerances of vulnerable species provide an example of incomplete basic information. In urbanized areas such as Baltimore, several factors at the watershed scale act to raise water temperatures. Removal of vegetation and forest from riparian areas and the watershed results in increases in water temperatures in urban areas (Kaushal et al. 2010). In addition, rapid runoff from heated impervious surfaces entering streams through stormwater systems during warm times of the year can result in acute water temperature increases in stream temperature. Thermal spikes in the Jones Falls can produce stream temperatures approaching 30 C (Kemp unpubl. data). Species associated exclusively with reference areas may be sensitive to this elevated temperature, but beyond general classifications of thermal preferences such as warm, cool, or cold water data are not available for these species. One tack is to assign water temperatures to these general categories (e.g. Nelson et al. 2009), but this does not describe specific thermal tolerances, which are in many cases unknown. Another would be to empirically determine stream temperatures with occurrence (e.g. Rashleigh et al. 2004; Stranko et al. 2005). In this approach however, other factors (spawning substrate, diet, introduced species) which may serve to eliminate particular species cannot be ruled out. One species identified in this study which has some published data on thermal tolerances is the Common shiner. Carlander (1969) reported the fish to have an upper limit dependent on acclimation temperature, though not exceeding 33.5 C. However, neither specific thermal tolerances of egg or juvenile life stages or exposure durations are listed. These results, and for that matter other life history information, were not gathered with the explicit goal of determining the effect of the urban environment on populations of these fish.

One area where there is an extensive and growing body of information for some species concerns spawning behavior. However, there are many items of interest and relevance to practical stream restoration which make this knowledge body incomplete. For example, consider the nesting behavior of the River chub. There are extensive references regarding the important topics of the temperature at which spawning is initiated, size and location of nests within stream habitat, nest associates, and actual behavior of spawning (Greeley 1929; Reighard 1943; Miller 1964; Jenkins and Burkhead 1993; Sabaj et al. 2000). However, there is little information regarding items such as quantitative description of substrate used in nest construction, velocity patterns surrounding pebble nests, and the critical velocity beyond which these nests become unstable or destroyed (although some of this information has been recently reported for the smaller congeneric Bluehead chub– Wisenden et al. 2009). For a pebble mound constructing species such as River chub (and its nest associates), vulnerability of the nests to high flows (Miller 1964; Peoples et al. 2011) may constitute one of the main challenges to its continued existence in an urbanizing watershed. Solid empirical information regarding the effects of urbanization on River chub nesting would theoretically improve chances for success in restoration efforts. Distinct targets could be provided for those involved in watershed and stream restoration, and the amount of restoration required to ensure suitability would be better known. However, the link between restoration in practice and impacts on the ecosystem is weakened by a lack of specific knowledge of habitat requirements and species life history, and the specific mechanisms of how these species are impacted by urbanization. Also, an important point is that chubs of the genus *Nocomis* have been better studied in the urban context than other groups. Data related to the impacts of urbanization on spawning habitat of other species (Margined madtom, Northern hogsucker, Common shiner) are scarce in or absent from the literature. It has been previously noted that a lack of functional understanding exists between implementation of watershed scale restoration and impact on stream ecosystems (Booth and Jackson 1997; Bernhardt and Palmer 2007). Specific life history information and habitat requirements are seen as insufficient in defining habitat for threatened and highly vulnerable species (Stranko et al. 2010b) and for basic reproductive information in habitat integrity assessment (Simon 1999). While robust modeling frameworks for linking restoration activities and fish habitat (e.g. Rashleigh et al. 2004) do exist, data put into these models regarding individual species habitat requirements are often aggregates of empirical associations with watershed attributes.

Evaluation of how species specific information could assist in restoration of habitat for those species should be possible for species where life history information is extremely well categorized and well known. This is true for some species of exceptional recreational and commercial importance such as the Salmonids (e.g. trout, salmon, grayling, Lake whitefish). A recent meta- analysis of restoration of Salmonid habitat by Whiteway et al. (2010) found that a majority of restoration projects for Salmonids resulted in increases in density (73%) and biomass (87%), and this study and Salmonid stream habitat restoration in general was one of very few which showed positive responses to restoration in a recent review (Bernhardt and Palmer, 2011). One important part of closing the gap between restoration practice and its effects on aquatic ecosystems remains improved species specific information of habitat requirements and life history.

Urban stream restoration success rates in terms of restoring ecological goals have been low at best (Bernhardt and Palmer 2011; Violin et al 2011; Sudduth et al. 2011; Stranko et al. 2012). Urban streams may be inherently very difficult to restore and restoration of full ecologic function may not be possible in all cases (Stranko et al. 2012). The degree to which these streams can be restored is limited by practical considerations (Bernhardt and Palmer 2007). Full understanding of these practical considerations with regard to ecological goals depends on improved knowledge of the impact of restoration activities on target species. Only then can a full accounting of the effort needed to reach these goals can be determined.

Restoring urban aquatic systems to the maximum extent possible will disproportionately benefit the greatest number of people (Findlay and Taylor 2006). Therefore it will not be wise to abandon restoration efforts in these streams, even though attaining reference levels of ecosystem integrity is often either impractical or impossible. Improved knowledge of the habitat requirements of common indicators of urbanization impact on fish assemblages can greatly improve the effectiveness of community analysis approaches using monitoring data, and can aid in determining the levels of habitat restoration and investment required to successfully restore urban ecosystems.

Conclusion

In the case of urbanized Baltimore fish assemblages, it is possible to identify particular species which consistently are eliminated or underrepresented in the fish fauna, and to construct a guiding image of a successfully restored fish community. As in the TFC method, their absence or underrepresentation potentially tells much about watershed restoration needs if ecological restoration goals are to be met. However, it is not possible to evaluate the suitability of the habitat, pre- or post- restoration, in a rigorous and quantitative way due to a lack of relevant information regarding these species.

While they fill important and unique ecological roles, species such as the River chub, Common shiner, Northern hogsucker, and Margined madtom do not occupy important recreational and commercial niches and therefore have not been studied to the same degree as other species (e.g. Salmonids). Having identified these species, it would be useful to collect species- specific information both inside and outside of urban contexts. Identification of the mechanism responsible for eliminating these components of the fish assemblage would provide restoration practitioners with quantifiable goals and targets. These would be highly useful in the planning and design process, and would provide estimated levels of funding needed for meeting particular ecological restoration criteria. Availability of the cost of restoring waterways to particular levels would assist in making decisions regarding what level of habitat restoration is possible.

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Table Captions

Table 1: Land use type as a percentage of total land area in watersheds included in study. Total acreage reflects all land within watershed boundaries. All data courtesy of Maryland Department of Planning (Pers. Comm. 2013) except impervious surface estimates (MD DNR, 2013).

Table 2: Life history information of species from upper and lower quartile of all species arranged according to discriminant score.
Table 2a: Species associated with reference sites, highest quartile
Table 2b: Species associated with urbanized sites, lowest quartile

Table 3: Summary of selected fish observed by Yingling (1940) in the lower reaches of the Gwynns Falls over the years 1923- 1940. Trends in abundance noted are indicated, as well as current abundance as suggested by modern fish surveys. Numbers of specimens represent the total number of each species collected in fish surveys in the Gwynns Falls included in the discriminant analysis (N= 8).

Figure Captions

Figure 1: Principal coordinates of all sites by watershed (GF- Gwynns Falls, JF- Jones Falls, LGF – Little Gunpowder Falls, WR – Winters Run, BR- Bynum Run).

Figure 2: Discriminant scores for sampling sites. Urbanized sites are shown in grey and reference sites are shown in white.

Figure 3: Loadings for original variables (fish taxa) used in the discriminant analysis of urbanized vs. reference sites, arranged according to value. Approximate quartiles are indicated by long dashed lines (quartile 2 is shortened in size by one species).

Table 1: (Excel)

Land Use Description	Gwynns Falls	Jones Falls	Winters Run	Little Gunpowder Falls	Bynum Run
Low Density Residential	5%	23%	25%	22%	17%
Medium Density Residential	27%	17%	8%	2%	28%
High Density Residential	20%	13%	4%	0%	8%
Commercial	8%	6%	4%	1%	5%
Industrial	6%	1%	1%	0%	1%
Institutional	7%	7%	2%	1%	3%
Other Developed Land	5%	7%	3%	1%	4%
Very Low Density Residential	1%	5%	9%	9%	4%
Transportation	3%	2%	1%	0%	0%
Total Developed Land	**82%**	**80%**	**55%**	**36%**	**69%**
Agriculture	2%	6%	21%	36%	14%
Forest	16%	13%	23%	28%	17%
Water	0%	0%	0%	0%	0%
Wetlands	0%	0%	0%	1%	0%
Barren Land	0%	0%	0%	0%	0%
Total Undeveloped Land	**18%**	**20%**	**45%**	**64%**	**31%**
Total Acreage	**41,707**	**37,280**	**37,544**	**37,340**	**14,582**
Impervious Surface	**42.20%**	**35.40%**	**12.02%**	**6.10%**	**21.10%**

Table 2:

Table 2a: (Excel)

species	discriminant score	Native?	tolerance	trophic class	reproductive guild	guild description	other
COMMON SHINER (*Luxilus cornutus*)	0.9521	Y	M	Invertivore, benthic and drift	A.2.3	brood hiders/ lithophils	versatile feeder
MARGINED MADTOM (*Noturus insignis*)	0.9303	Y	M	Invertivore, benthic	B.2.7	Nest spawners/ speleophils	
RIVER CHUB (*Nocomis micropogon*)	0.9171	Y	I1	Planktivore/ invertivore	A.2.3	brood hiders/ lithophils	ontogenetic dietary shift
NORTHERN HOGSUCKER (*Hypentelium nigricans*)	0.8265	Y	I	Invertivore/ herbivore	A.1.3	Open substrate/ lthophils	
CUTLIP MINNOW (*Exoglossum maxillingua*)	0.7986	Y	I	Invertivore/ benthic	A.2.3	brood hiders/ lithophils	hunter
SEA LAMPREY (*Petromyzon marinus*)	0.7905	Y	M	Herbivore/ detritivore, filter feeder	A.2.3	brood hiders/ lithophils	spawning adults parasitic
CREEK CHUB (*Semotilus atromaculatus*)	0.784	Y	T	Invertivore/ carnivore	A.2.3	brood hiders/ lithophils	

Table 2b: (Excel)

species	discriminant score	Native?	tolerance	trophic class	reproductive guild	guild description	other
YELLOW BULLHEAD (*Ameiurus natalis*)	-0.7232	Y	T	Invertivore/ carnivore, benthic	B.2.7	Nest spawners/ speleophil	whole body feeder
MUMMICHOG (*Fundulus heteroclitus*)	-0.4561	Y	M	Omnivore	A.1.4	open substrate/ phytolithophil	
LONGNOSE DACE (*Rhinichthys cataractae*)	-0.3439	Y	I	Invertivore, benthic	A.1.2	open substrate spawner	
SATINFIN SHINER (*Cyprinella analostanus*)	-0.1884	Y	I	Invertivore/ planktivore, drift	A.2.4	brood hider, speleophil	opportunistic feeder
SPOTTAIL SHINER (*Notropis hudsonius*)	-0.1124	Y	M	Invertivore/ planktivore	A.1.2	open substrate spawner	
BLACKNOSE DACE (*Rhinichthys atratulus*)	-0.0101	Y	T	Invertivore	A.1.2	open substrate spawner	
LARGEMOUTH BASS (*Micropterus salmoides*)	-0.0037	N	M	Invetivore/ carnivore	B.2.2	Nest spawner/ polyphil	whole body feeder

Table 3: (Excel)

Species	Yingling (1940)	Recent collections
Common Shiner (*Luxilus cornutus*)	common everywhere	rare (2 specimens)
Swallowtail Shiner (*Notropis procne*)	common everywhere	rare (1 specimen)
Margined madtom (*Notorus insignis*)	rare (1 specimens)	rare (7 specimens)
River Chub (*Nocomis micropogon*)	range contracting to upper parts stream	absent
Northern Hogsucker (*Hypentelium nigricans*)	common everyhwere	rare (5 specimens)

Figure 1: (Sigma Plot 11.0)

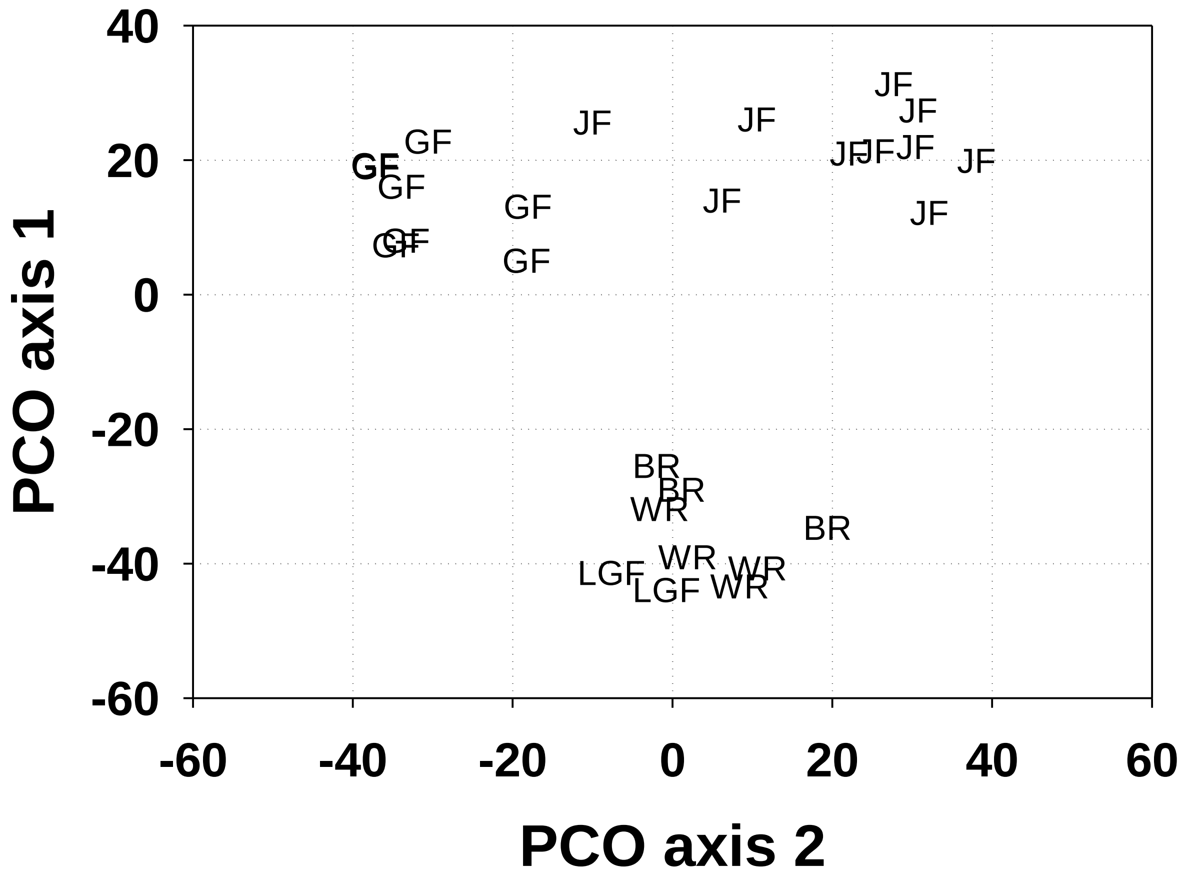

Figure 2: Sigma Plot 11.0.

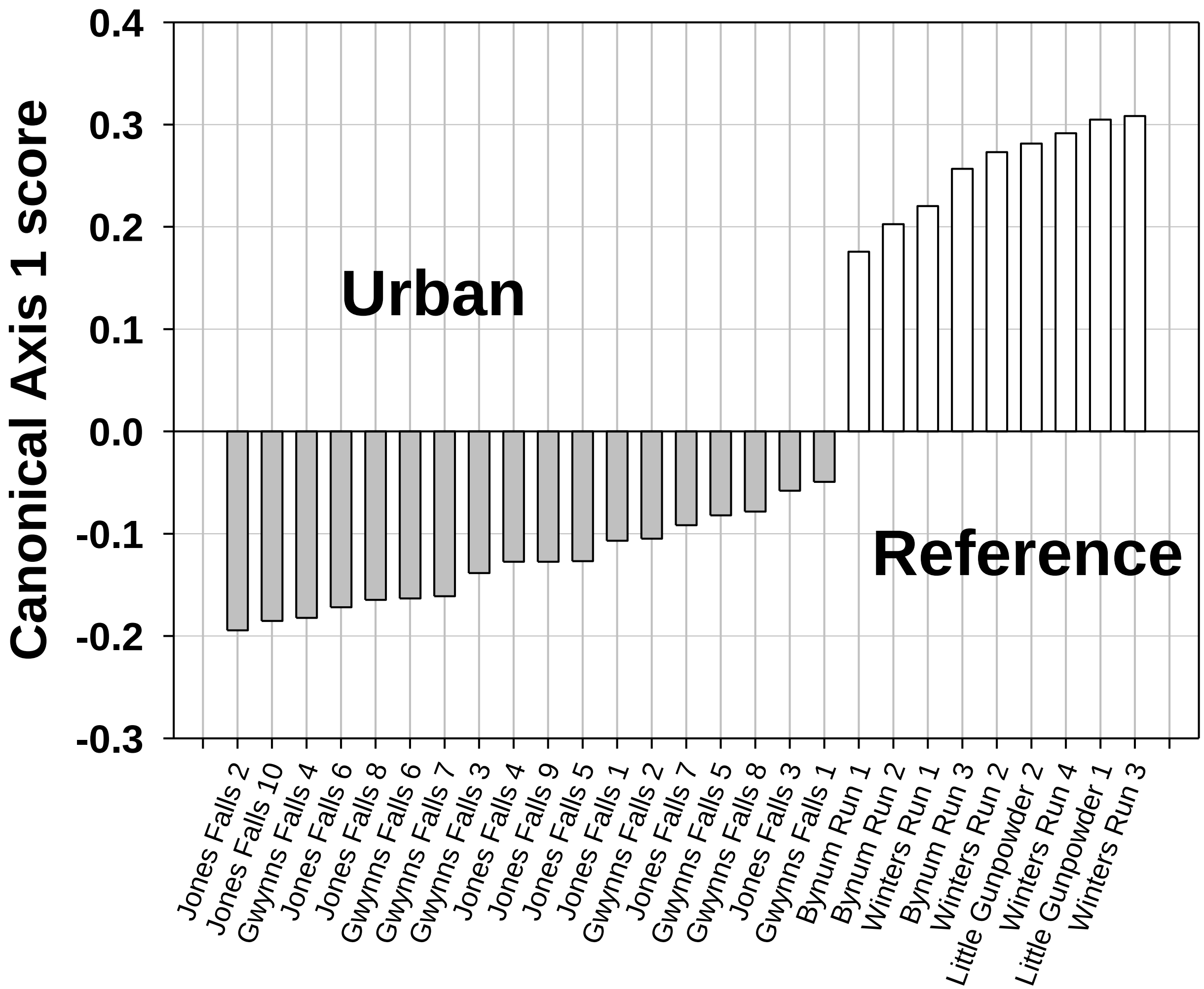

Figure 3: Sigma Plot 11.0.