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# Biological assessment of western USA sandy bottom rivers based on modeling historical and current fish and macroinvertebrate data

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## Abstract

Biological monitoring is important for assessing the ecological condition of surface waters. However, there are challenges in determining what constitutes reference conditions, what assemblages should be used as indicators, and how assemblage data should be converted into quantitative indicator scores. In this study, we developed and applied biological condition gradient (BCG) modeling to fish and macroinvertebrate data previously collected from large, sandy bottom southwestern USA rivers. Such rivers are particularly vulnerable to altered flow regimes resulting from dams, water withdrawals and climate change. We found that sensitive ubiquitous taxa for both fish and macroinvertebrates had been replaced by more tolerant taxa, but that the condition assessment ratings based on fish and macroinvertebrate assemblages differed. We conclude that the BCG models based on both macroinvertebrate and fish assemblage condition were useful for classifying the condition of southwestern USA sandy bottom rivers. However, our fish BCG model was slightly more sensitive than the macroinvertebrate model to anthropogenic disturbance, presumably because we had historical fish data, and because fish may be more sensitive to dams and altered flow regimes than are macroinvertebrates.

**KEYWORDS**

bioassessment, biological condition gradient, biological integrity, biological monitoring, biomonitoring, flow regime alteration, Rio Grande

## 1 | INTRODUCTION

The objective of the USA Clean Water Act (1972) is "to restore and maintain the chemical, physical, and biological integrity of the Nation's waters." To do so, those waters must be monitored using standardized sampling protocols and appropriate study designs so that effective management actions can be prioritized and implemented. Standardized sampling is needed so that when data from various sources are compared any observed differences can be ascribed to ecological differences versus sampling differences (Bonar, Hubert, & Willis, 2009; Hughes, Paulsen, & Stoddard, 2000). Probability survey designs are useful for ensuring sample representativeness and for inferring results with known confidence intervals beyond the sampled sites themselves (Hughes et al., 2000; Hughes & Peck, 2008). However, there are at least four additional challenges to making rigorous biological assessments and management decisions regarding biological conditions in large rivers. (a) Determining what constitutes reference condition for systems with long and extensive histories of modification and degradation, (b) Determining what assemblages should be used as indicators of biological condition, (c) Determining how assemblage data should be converted into quantitative indicator scores, and (d) Deciding which indicator scores constitute sufficient ecological change to merit protection or mitigation.

Determining naturally occurring reference conditions is particularly problematic. Currently, most freshwater biological assessments are based on comparisons between data from least- or minimally disturbed ecoregional reference sites (Hughes, Larsen, & Omernik, 1986; Stoddard, Larsen, Hawkins, Johnson, & Norris, 2006). However, those reference sites are biased towards present conditions that tend to be altered to varying degrees from natural or pristine conditions (Hughes et al., 1986; Stoddard et al., 2006). This results in a temporal shifting baseline phenomenon (Pauly, 1995), as well as markedly different baseline expectations amongst regions with differing degrees, extents and timelines of degradation (Herlihy, Sifneos, Hughes, Peck, & Mitchell, 2020). Such limitations confound our abilities to evaluate the degree to which ecosystems have been altered, to detect rates of degradation, and to set recovery expectations.

There are many potential indicator assemblages by which to assess lotic ecosystems, from protozoa and algae to riparian birds and woody plants. Globally and in the USA, the most-commonly used assemblages for making lotic ecosystem bioassessments are fish and macroinvertebrates (Feio et al., 2021; Ruaro, Gubiani, Hughes, & Mormul, 2020). Because of their differing sensitivities and life histories, assessments based on both fish and macroinvertebrate assemblages tend to yield somewhat different results (Herlihy et al., 2020; Mulvey, Leferink, & Borisenko, 2009). For example, good macroinvertebrate assemblage condition, estimated from multimetric index (MMI) scores existed in only 22 and 51% of the stream length in USA Xeric and Western Mountains ecoregions, respectively (USEPA, 2020). For fish assemblage MMIs, good conditions existed in only 19% of the Xeric and 26% of the Western Mountains stream length. Similarly, poor fish MMI scores were 10.6 times as likely where poor chloride conditions existed in the Xeric ecoregion, whereas excessive total nitrogen was the major driver for poor macroinvertebrate MMI scores in that ecoregion (Herlihy et al., 2020). Therefore, by assessing both macroinvertebrate and fish assemblages, one can obtain a more comprehensive assessment of biological condition as well as the stressors most strongly linked to those poor conditions.

Macroinvertebrates have at least five advantages as a river indicator assemblage (Hughes, 1993). (a) They are relatively easy to collect in a timely manner and collecting gears tend to be smaller and less expensive than those used for fish, especially in large rivers (Curry, Hughes, McMaster, & Zafft, 2009; Guy, Braaten, Herzog, Pitlo, & Rogers, 2009). (b) Most taxa have limited mobility and annual life cycles, thereby reflecting current and recent environmental conditions. (c) Taxa in macroinvertebrate assemblages represent multiple trophic levels and life history strategies. (d) Many taxa can be identified to species or genus and higher taxonomic levels remain useful indicators of environmental disturbance (Moya et al., 2011; Silva, Herlihy, Hughes, & Callisto, 2017; Whittier & Van Sickle, 2010). (e) Because few taxa are threatened or endangered, collection permits are usually easy to obtain.

Fish have somewhat different advantages than macroinvertebrates as riverine ecosystem indicator assemblages (Hughes, 1993). (a) They are purely aquatic. (b) Many taxa are of substantial interest to anglers. (c) Several taxa are listed as threatened, endangered, or vulnerable (Hughes & Noss, 1992; Jelks et al., 2008). Although threatened or endangered taxa provide useful ecological insights, their potential occurrence can hinder obtaining collection permits. (d) Most species are longer lived and individuals are wider ranging than individual macroinvertebrates; therefore, they are useful for assessing longer trends and larger spatial extents, particularly the effects of migratory barriers. (e) Most adult individuals can be identified to species. (f) Species in riverine fish assemblages typically represent multiple trophic levels and life history strategies.

One popular approach for assessing biological condition is the use of predictive modeling of non-rare taxa (Clarke, Furse, Wright, & Moss, 1996; Hawkins, Norris, Hogue, & Feminella, 2000). The resulting indices calibrate for natural variability and consequently separate natural gradients from anthropogenic disturbance gradients. Briefly, this is accomplished by using the residuals from regression lines as the response variables; however, such indices only assess the observed occurrences of non-rare taxa versus their expected occurrences. Rare, often sensitive, taxa are omitted from such analyses and those taxa are often the first to disappear during the early stages of anthropogenic disturbances (Davies & Jackson, 2006; Leitão et al., 2016; R. T. Martins et al., 2021). Also, some tolerant taxa are more likely to occur in the early stages of anthropogenic disturbance, thereby, resulting in no change or even an increase in taxa richness (Davies & Jackson, 2006; R. T. Martins et al., 2021). Partly because of the shortcomings of species richness indicators, Karr (1981) proposed using multimetric indices (MMIs).

MMIs comprised of several metrics have become the most widely employed quantitative indicators of assemblage condition globally (Feio et al., 2021; Ruaro et al., 2020). However, the metrics selected for inclusion in MMIs and the criteria used to select the metrics vary widely, leading to a plethora of MMIs globally (Ruaro et al., 2020) and across the USA (Stoddard et al., 2008; USEPA, 2016b). This can hinder directly comparing MMI assessment results among states and nations because each of those MMIs is based on different reference and scoring criteria as well as different metrics (also see Stoddard et al., 2008; Table 1). However, predictive modeling of MMI metrics has been increasingly employed to minimize MMI natural variability and facilitate national or continental assessments (Moya et al., 2011; Pont et al., 2006; Pont, Hughes, Whittier, & Schmutz, 2009). Also, I. Martins, Macedo, Hughes, and Callisto (2020) reported that MMIs developed in seven different nations were able to satisfactorily assess macroinvertebrate assemblage condition in a Brazilian river basin. Many MMIs are based on taxonomic metrics; however, trait-based MMIs have been reported recently to be more responsive to anthropogenic pressures and better able to discern the ecological

**TABLE 1** Biological condition gradient levels (Davies & Jackson, 2006; USEPA, 2016c)

| BCG level | Key characteristics |
|---|---|
| 1 | Natural structural, functional and taxonomic integrity is preserved |
| 2 | Structure and function similar to natural community with some changes in taxa and biomass; ecosystem level functions are fully maintained. |
| 3 | Evident changes in structure due to loss of some highly sensitive native taxa and shifts in relative abundance; ecosystem level functions fully maintained. |
| 4 | Moderate changes in structure due to replacement of some sensitive ubiquitous taxa by more tolerant taxa; ecosystem functions largely maintained. |
| 5 | Sensitive taxa markedly diminished; conspicuous unbalanced distribution of major taxonomic groups; ecosystem function shows reduced complexity and redundancy. |
| 6 | Extreme changes in structure and ecosystem function; wholesale changes in taxonomic composition; extreme alterations from normal densities |

mechanisms of those biological responses (Chen et al., 2019; Saito, Siqueira, & Fonseca-Gessner, 2015). Not only have MMIs been effective for determining ecological condition, they also can be significantly correlated with human well-being (Angermeier, Krometis, Stern, & Hemby, 2021).

Regardless of the choice of biological indicator(s), determining the degree of ecological change that merits mitigation action is both an ecological and socioeconomic concern. One approach is based on the degree of difference between sets of least or minimally disturbed reference sites and populations of test sites (Hughes, 1995; Stoddard et al., 2006, 2008). This approach has been applied at national and ecoregional (Herlihy et al., 2020; USEPA, 2016a), as well as basin (Jimenez-Valencia, Kaufmann, Sattamini, Mugnai, & Baptista, 2014; I. Martins et al., 2020; Mulvey et al., 2009; Silva et al., 2017) spatial extents. An alternative approach, employing many of the same metrics used in MMIs, but with a differing concept for reference conditions and assemblage scoring, is the biological condition gradient (BCG) assessment (Davies & Jackson, 2006; USEPA, 2016c).

To date, BCG model-based assessments have been applied to stream and river algal assemblages (Charles, Tuccillo, & Belton, 2019; Danielson et al., 2012; Hausmann, Charles, Gerritsen, & Belton, 2016), stream fish and macroinvertebrate assemblages (Bouchard et al., 2016; Gerritsen, Bouchard Jr., Zheng, Leppo, & Yoder, 2017), stream algal and macroinvertebrate assemblages (Paul et al., 2020), estuarine macrophyte and macroinvertebrate assemblages (Shumchenia et al., 2015), and coral reef fish and stony coral assemblages (Bradley et al., 2020). Our objective in this study was to develop BCG models for macroinvertebrate and fish assemblages based on data sets from New Mexico draining rivers. We then validated those models through use of a separate set of data collected by the USEPA from large sandy bottom rivers in 10 southwestern USA states. Based on the literature and the authors' experience, we predicted that because of their greater sensitivity, the macroinvertebrates and the macroinvertebrate BCG model, would reveal greater sensitivity to degradation of riverine sites than the fish model.

## 2 | METHODS

### 2.1 | Study area

Modeling was focused initially on the Middle Rio Grande (Cochiti Dam to Elephant Butte Reservoir) in New Mexico but expanded to other nearby, large sandy bottom rivers for macroinvertebrates (Figure A1). Models were then validated with data from throughout the southwestern USA (Figure 1). The rivers in the region are mostly low-gradient, slowly flowing, sandy bottomed, and warmwater. Most of the region's climate is classified as cold semi-arid steppe (Koppen Bsk; Geiger, 1954). Air temperatures range from average lows of 2.8°C in winter to average highs of 19.2°C in summer and average annual precipitation is low (30–50 cm) (Fox, Jemison, Potter, Valett, & Watts, 1995). River flows are driven in large part by mountain snowmelt, but occasional summer monsoon storms and freshets occur. Much of the natural flows are diverted for irrigated agriculture and municipal water supplies (Fox et al., 1995).

### 2.2 | The BCG process

Typically, BCG assessments are based on seven steps (Figure 2). (1) Raw data are compiled and biological metrics (measurable components of biological assemblages) are calculated. (2) The expert panels are introduced to the BCG concepts and model development processes. (3) Historical data and ecological concepts are used to determine reference conditions. (4) Sets of raw taxa data and their metrics are used to rate each site into one of six biological condition classes by expert panel members. (5) Panel members discuss how they made their determinations and develop rationales for determining numerical models. (6) Steps 4 and 5 are reiterated and the ratings and models are calibrated. (7) Models are validated through use of independent site data sets and subsequently are available for application elsewhere. A BCG assessment is a highly transparent approach based on multiple experts' consensus judgments on the biological importance of changes in multiple biological attributes.

### 2.3 | Data sources and collection methods

Available macroinvertebrate assemblage data for New Mexico and neighboring states, covering the years of 2008–2019, were obtained from the New Mexico Environment Department (NMED), the New Mexico Museum of Natural History and Science (MNHS), and the USEPA's National Rivers and Streams Assessment (NRSA). Sites were

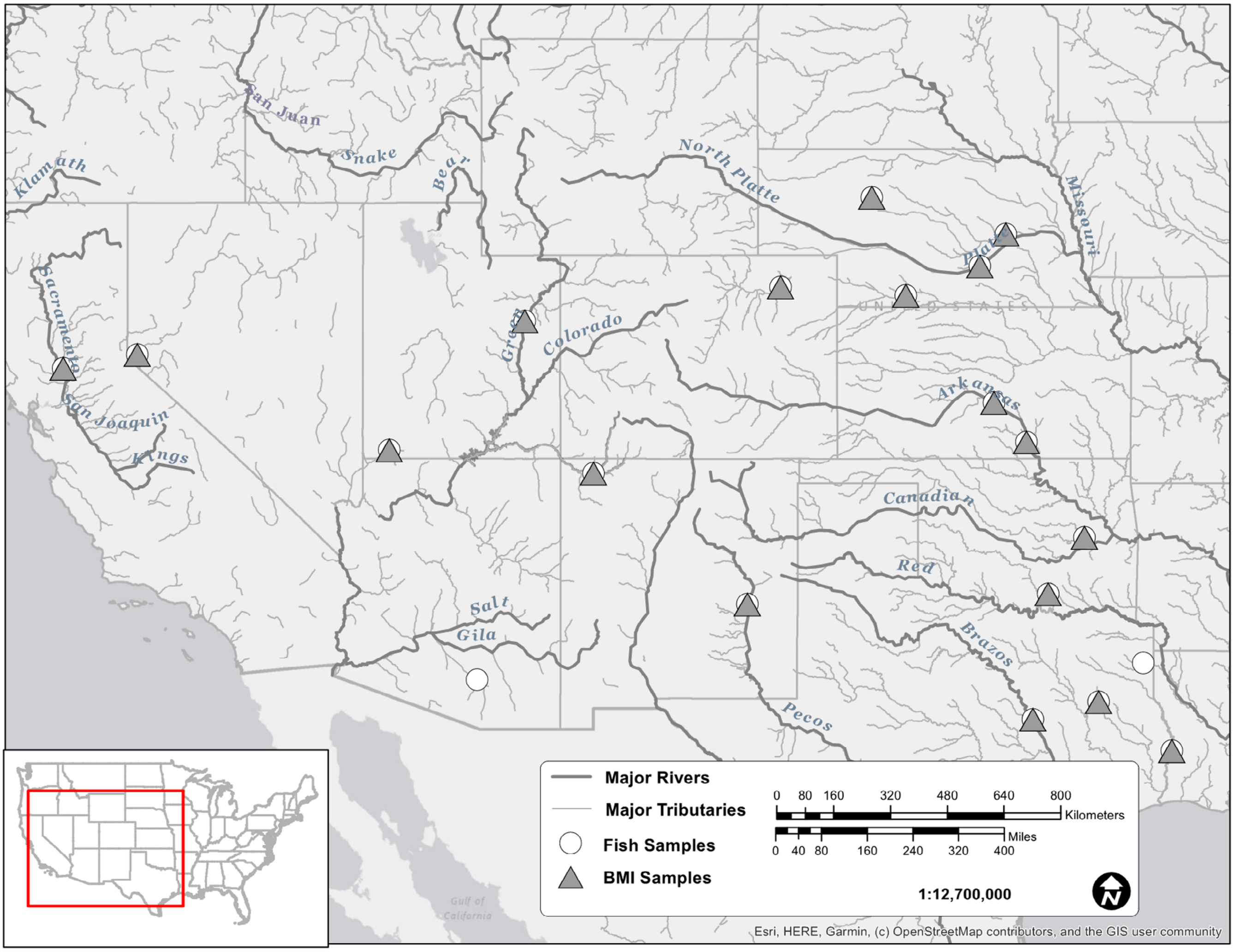


**FIGURE 1** Site locations of fish and benthic macroinvertebrate (BMI) samples used in validating the biological condition gradient for sandy bottom rivers in the western USA

limited to those that were greater than 10-m mean wetted width to better ensure they represented rivers versus small streams, had 70% or more of sandy substrate, and were flowing when sampled. Wadeable river sites were sampled through use of D-frame kick nets at multiple systematically distributed stations throughout the channel; boatable sites were systematically sampled with D-frame kick nets at multiple littoral stations (Hughes & Peck, 2008; Jessup & Bradley, 2020). In both wadeable and boatable sites, the collection stations comprised multiple habitat types. The target number of individuals identified per site was 300 and taxa were identified to the lowest possible level, usually genus. Data from New Mexico and neighboring states were used for model calibration; regional NRSA data were used to validate the models. After screening, the expert panel used data from 44 samples at 39 sites (including five revisit samples) for model development and calibration and 18 sites for model scoring and validation.

Available fish assemblage data were obtained from the NMED, the Museum of Southwestern Biology, the U.S. Fish and Wildlife Service, and the NRSA covering the years of 1925–2019. The collection data of Sublette, Hatch, and Sublett (1990) were of particular importance. Although the Sublette et al. (1990) data only included presence data, much of their data had been collected prior to the construction and operation of major mainstem dams. Fish data were collected by boat, raft, or backpack electrofishing in multiple habitat types throughout the entire site (Hughes & Peck, 2008) or by 18–20 seine hauls in several pools per site (USFWS, 2010). After screening as for macroinvertebrates, 60 samples from 29 Middle Rio Grande sites (including 31 revisit samples) were used for model development and calibration; the data from 19 regional NRSA sites were used for model scoring and validation.

## 2.4 | BCG orientation and webinars

River ecology experts familiar with southwestern USA rivers and fish or macroinvertebrate assessments initially met via webinar. The

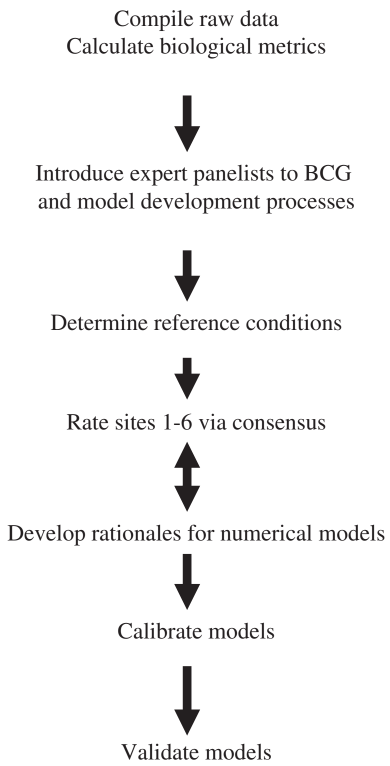


**FIGURE 2** The biological condition gradient process

workshop participants included 8 fish and 12 macroinvertebrate participants in the on-line and in-person workshops; only seven of the former and five of the latter participated in writing this article. The experts included NMED staff plus other state, federal, academic, and private sector researchers. The webinars introduced the BCG concepts and process to the experts (Tables 1 and 2). The experts then confirmed or changed the attribute classification of each taxon (fish or macroinvertebrate) that had been based on published literature and regional tolerance analysis (Tables A1 and A2). This was an essential step, because many taxa tolerances are historically based on small, high-gradient, rocky-bottom streams versus large, slow-flowing, sandy bottom rivers (Whittier, Hughes, Lomnicky, & Peck, 2007; Whittier & Van Sickle, 2010).

## 2.5 | BCG workshop process

During a multi-day workshop, the experts received initial instructions then broke into separate fish and macroinvertebrate workgroups. Each expert viewed taxa data sheets (number and attribute class; Tables A3–A6), classified each site into one of six biological condition levels, and shared their reasoning with the rest of the group. Because of subtle differences in the taxa present as well as expected taxa that were absent when comparing relatively similar sites, the experts sometimes assigned a plus or a minus to integer scores to indicate a bias towards better or worse conditions than the core level (Tables A7 and A8). After scoring the set of calibration sites (44 for

**TABLE 2** Biological condition gradient attributes and their descriptions (USEPA, 2016c)

| Attribute | Description |
|---|---|
| I | Historically documented, sensitive, long-lived, or regionally endemic taxa |
| II | Highly sensitive taxa |
| III | Intermediate sensitive taxa |
| IV | Intermediate tolerant taxa |
| V | Tolerant taxa |
| VI | Non-native or intentionally introduced species |
| VII | Organism condition |
| VIII | Ecosystem function |
| IX | Spatial and temporal extent of detrimental effects |
| X | Ecosystem connectivity |

macroinvertebrates, 60 for fish), then revisiting some sites and discussing rationales for those scores, the expert panels developed a preliminary set of narrative scoring rules. For example, a macroinvertebrate site must meet increasingly higher numbers of total taxa, EPT taxa, shredder taxa, and clinger taxa, versus increasingly lower proportions of non-insect and collector individuals for distinguishing among Levels 2, 3, 4, and 5. For fish, the narrative criteria included increasing numbers of species and individuals classified as sensitive, long-lived and large-bodied, and number of trophic groups versus fewer non-native species and individuals.

## 2.6 | Model development and validation

On the final day of the workshop, the narrative scoring rules were converted into tentative numeric rules. Those tentative rules were applied to subsets of the original data to determine whether the results still represented the group consensus on site level assignments. If not, the rules were adjusted. Such adjustments included altering the required numerical scores up or down, as well as including and/or statements indicating whether all or subsets of the criteria must be met. Each site or sample in the calibration dataset was classified as BCG level 2, 3, 4, 5, or 6 because no level-1 (pristine) sites were identified in the data. The model rule for each BCG level contained an "and" for each variable that had to be met and an "or" for each variable that could be superseded by a different rule (Tables 3 and 4). The tentative numeric rules were converted into fuzzy decision models (Gerritsen et al., 2017; Klir, 2004; Zadeh, 1965, 2008). Such models convert expert consensus to automatic site assessment but indicate degrees of truth between condition classes by depicting borderline classifications or probabilities of class occurrence near class thresholds, versus hard-line classification of class membership obtained from purely statistical analyses.

Following the workshop, the experts were provided with the regional NRSA data for independently validating the numeric model.

**TABLE 3** New Mexico river macroinvertebrate biological condition gradient (BCG) numerical model rules, showing the rule mid-point (and range) at each BCG level

| BCG metric | 2[a] | 3[b] | 4[c] | 5[d] |
|---|---|---|---|---|
| Total taxa richness | ≥30 taxa (25–35) | ≥20 taxa (15–25) | ≥15 taxa (10–20) | ≥10 (5–15) |
| EPT taxa richness | ≥12 taxa (10–14) | ≥7 taxa (5–10) | | |
| Non-Hydropsychidae Trichoptera richness | ≥4 taxa (3–5) | ≥1 taxon (0–1) | | |
| % EPT individuals | ≥40% (35–45) | ≥25% (20–30) | ≥15% (10–20) | >0.5 (0–1) |
| Native mollusk taxa richness | >0 taxa (0–1) | | | |
| % non-insect individuals | ≤20% (15–25) | <20% (15–25) | <20% (15–25) | |
| % Chironomid individuals | | ≤40% (35–45) | | |
| Chironomid taxa richness | | ≥10 taxa (5–15) | | |
| Scraper taxa richness | | ≥2 taxa (1–3) | >0 (0–1) | |
| Shredder taxa richness | ≥6 taxa (5–7) | ≥3 taxa (2–4) | | |
| Clinger taxa richness | ≥11 taxa (10–12) | | ≥2 (1–3) | |
| BCG attribute I, II, III taxa richness | | ≥2 (1–3) | | |
| BCG attribute V % individuals | | ≤7.5% (5–10) | | |
| % collector-gatherer individuals | ≤50% (40–60) | | | |
| Density % of target individuals | ≥50% (40–60) | ≥50% (40–60) | ≥50% (40–60) | |

*Note*: The rule for Level 6 is that the sample does not meet Level 5 rules.
[a]Level 2 rules are conceptual. Level 2 membership is the minimum of all rule memberships, though if shredder taxa >5, use the second lowest rule membership.
[b]Level 3 membership is the second lowest of rule memberships, although if shredder taxa >2, use the third lowest rule membership. Use the maximum of the two chironomid rule memberships before combination with other rule memberships.
[c]Level 4 membership is the second lowest of rule memberships.
[d]Level 5 membership is the minimum of the rule memberships.

**TABLE 4** New Mexico river fish biological condition gradient (BCG) numerical model rules, showing the rule mid-point (and range) at each BCG level

| Metrics | 2 | 3 | 4 | 5 |
|---|---|---|---|---|
| BCG attribute I, II, III % taxa | ≥45% (40–50) | ≥35% (30–40) | ≥15% (10–20) | |
| BCG attribute I, II, III, IV % taxa | | | ≥25% (20–30) | ≥10 (5–15) |
| BCG attribute I, II, III, IV % individuals | ≥60% (50–60) | ≥50% (40–60) | ≥25% (20–30) | |
| Long-lived, native, large-bodied taxa richness | ≥4 (3–5) | ≥1 (0–2) | | |
| Native minnow BCG attribute I, II, III, IV taxa richness | ≥6 (5–7) | ≥3 (2–4) | ≥2 (1–3) | >1 (0–1) |
| Silvery minnow presence | ≥1 (0–1) | | | |
| Broadcast spawner taxa richness | ≥2 (1–3) | ≥1 (0–2) | | |
| Number of trophic categories | ≥4 (3–4) | ≥3 (2–3) | ≥2 (2–3) | |
| % non-native individuals | <1% (0–1) | ≤15% (10–20) | ≤25% (20–30) | ≤75% (65–85) |
| Non-native piscivore taxa richness | <1 (0–1) | <1 (0–1) | | |
| Number of individuals | | ≥100 (80–129) | ≥50 (40–60) | ≥10 (5–15) |
| Combination rules | Must meet all rules | Must meet all rules | Must meet all rules | Must meet all rules |

*Note*: The rule for Level 6 is that the sample does not meet Level 5 rules.

As part of this follow-up classification process, fish and macroinvertebrate experts were separately asked to classify each site from level 2 to 6 and to provide their rationale for doing so. Agreements and disagreements between expert ratings and numeric model predictions were identified and discussed via a subsequent webinar.

## 3 | RESULTS

### 3.1 | Model calibration

Most of the study sites classified by the experts were rated as BCG level 3 (some sensitive-rare taxa lost) to 4 (notable replacement of

**TABLE 5** Macroinvertebrate model performance comparing expert BCG ratings (medians) and numerical model predictions

| Model Prediction | Expert Rating: 2- | 3+ | 3 | 3- | 34tie | 4+ | 4 | 4- | 45tie | 5+ | 5 |
|---|---|---|---|---|---|---|---|---|---|---|---|
| 3 | | 3 | 5 | 5 | 1 | | | | | | |
| 3- | | | | | | | 1 | | | | |
| 34tie | | | | | | 2 | 1 | | | | |
| 4+ | | | | | | 1 | 1 | | | | |
| 4 | | | | | | 1 | 4 | 3 | 2 | | |
| 4- | | | | | | | | 2 | | | |
| 45tie | | | | | | | | | 1 | | |
| 5+ | | | | | | | | 1 | 1 | | 1 |
| 5 | | | | | | | | | 1 | 1 | 2 |
| 5- | | | | | | | | | | | 3 |
| 6+ | | | | | | | | | | | 1 |

Model better than rating (upper right); Model worse than rating (lower left)

**TABLE 6** Fish model performance comparing expert BCG ratings (medians) and numerical model predictions

| Model Prediction | Expert Rating: 4+ | 4 | 4- | 45tie | 5+ | 5 | 5- | 6+ | 6 |
|---|---|---|---|---|---|---|---|---|---|
| 4+ | | | | | | | | | |
| 4 | 1 | 3 | 1 | | | | | | |
| 4- | | 1 | 1 | 1 | 1 | | | | |
| 45tie | | | 1 | | 2 | 3 | | | |
| 5+ | | | 3 | | 1 | | 1 | | |
| 5 | | | 1 | | 8 | 13 | 2 | | |
| 5- | | | | | 1 | 2 | | | |
| 6+ | | | | | | 1 | | | 2 |
| 6 | | | | | | 1 | | 1 | 3 |

Model better than rating (upper right); Model worse than rating (lower left)

sensitive taxa by tolerant taxa) for macroinvertebrate assemblages and 4 to 5 (sensitive species rare; tolerant species dominant) for fish assemblages (Tables 3 and 4). Note, however, that the difference in BCG levels among assemblages result from (a) different sites being assessed for fish than macroinvertebrates and (b) the availability of historical pre-dam data for fish, but not for macroinvertebrates. The review of pre-dam fish data might have better informed the fish experts about a currently unattainable baseline of biological conditions. For the macroinvertebrate assemblage, the expert ratings and numerical model predictions almost always differed by only one BCG level, that is, a model error of three out of 44 cases (6.8%) (Table 5). Similarly for fish assemblages, expert ratings and numerical model predictions almost always differed by only one BCG level and the model error was three out of 55 cases (5.5%) (Table 6).

### 3.2 | Model validation

Macroinvertebrate and fish assemblage samples taken on the same day with standard methods at the same regional NRSA sites were used to validate the numeric models. As with the calibration modeling, site BCG classifications usually differed by less than one level and the model errors were low (3 sites) for both macroinvertebrate and fish assemblages (Tables 7 and 8). When comparing fish versus macroinvertebrate BCG levels from the same sites, however, 10 of the 18 sites with both fish and macroinvertebrate data differed by one or more BCG level (Table 9). Six sites were assigned worse biological conditions for fish than for macroinvertebrates and four were assigned worse conditions for macroinvertebrates.

## 4 | DISCUSSION

One of the major strengths of the BCG approach is its use of historical information for establishing reference conditions against which to evaluate current conditions and to reduce the shifting baseline problem (Hughes, 1995; Pauly, 1995). We used pre-dam (1928–1968) fish data rather than least-disturbed current conditions for modeling six BCG levels, the least-altered three levels of which no longer occurred for fish. Shumchenia et al. (2015) used pre-1850 estuarine seagrass and pre-1650 shellfish information plus stressor-response modeling to produce six BCG levels for estuaries Likewise, their least-altered three levels no longer occurred. A similar perspective is offered when applying Threshold Indicator Taxa Analysis to assessments of pristine tropical forests being transformed by forest removal. For example, sensitive fish species disappeared at 1 and 6% of forest-loss at catchment and riparian spatial extents, respectively (R. T. Martins et al., 2021). Sensitive macroinvertebrates were lost at 1 and 11% of forest loss at catchment and riparian spatial extents, respectively (Brito et al., 2020; R. T. Martins et al., 2021). Thus, more commonly

**TABLE 7** Model validation comparing expert biological condition gradient (BCG) ratings (medians) and model predictions for the benthic macroinvertebrate numerical model

| Model Prediction | Expert Rating: 3 | 3- | 34tie | 4+ | 4 | 4- | 45tie | 5+ | 5 | 5- | 6 |
|---|---|---|---|---|---|---|---|---|---|---|---|
| 3- | | 1 | | | | | | | | | |
| 34tie | 1 | 3 | | | | | | Model better than rating | | | |
| 4+ | | | 1 | 1 | | | | | | | |
| 4 | | | | 2 | | | | | | | |
| 4- | | | | | | | | | | | |
| 45tie | | | | | | 2 | | | | | |
| 5+ | | | | | 1 | | | | | | |
| 5 | | | | | 1 | | 1 | 1 | 1 | | |
| 5- | | | | | | | | | | | |
| 56tie | | | | | | | | | | | |
| 6+ | | Model worse than rating | | | | | | | | | |
| 6 | | | | | | | | | | 1 | 1 |

**TABLE 8** Model validation comparing expert biological condition gradient (BCG) ratings (medians) and model predictions for the fish numerical model

| Model Prediction | Rating Median: 3- | 4+ | 4 | 4- | 5+ | 5 | 5- | 6+ | 6 |
|---|---|---|---|---|---|---|---|---|---|
| 3- | | | | | | | | | |
| 4+ | | | | | | Model better than rating | | | |
| 4 | | 1 | 1 | | | | | | |
| 4- | | | 1 | 2 | | | | | |
| 5+ | | | | 1 | | 1 | | | |
| 5 | | | 1 | | | 7 | | | |
| 5- | | | | | | | | | |
| 6+ | | Model worse than rating | | | | | | 1 | |
| 6 | | | | | | 1 | | 1 | 1 |

**TABLE 9** Comparison of predicted biological condition gradient (BCG) levels between numerical models for fish and macroinvertebrate assemblages, showing numbers of comparable samples

| Fish | Benthic Macroinvertebrates: 3- | 3-4 tie | 4+ | 4 | 4- | 4-5 tie | 5+ | 5 | 5-6 tie | 6 |
|---|---|---|---|---|---|---|---|---|---|---|
| 4 | | | 1 | | | | | 1 | | |
| 4- | | 2 | | | | | | 1 | | |
| 5+ | | | | | | | | | | 1 |
| 5 | 1 | 1 | | 1 | | 1 | 1 | 2 | 1 | 1 |
| 5- | | | | | | | | | | |
| 6+ | | | 1 | | | | | | | |
| 6 | | 1 | | 1 | | | | | | |

implemented biological assessments that are based on current least-disturbed reference sites are biased towards indicating better ecological status than those based on historical perspectives, because in many cases sensitive-rare and ubiquitous species have already been extirpated from the entire region being assessed (Davies & Jackson, 2006; Leitão et al., 2016). On the other hand, macroinvertebrate assemblage evaluation in the BCG exercise relied on expert conceptualization of the undisturbed historic conditions supported by individual knowledge of the taxa and their regional distributions, but they were not explicitly supported by historic examples.

There are several reasons why our biological assessments based on fish and macroinvertebrate assemblages differed. Because historical pre-dam data were not available for macroinvertebrates, it is possible that some sensitive macroinvertebrate taxa had been extirpated by the time the data we used for site assessments were collected, as outlined above (Davies & Jackson, 2006; Leitão et al., 2016). Likewise, the original riparian vegetation has been markedly modified (Finch, Wolters, Yong, & Mund, 1995), which would be expected to fundamentally alter the river physical habitat structure and allochthonous inputs that serve as important sources of food and cover for macroinvertebrates (Gregory, Swanson, McKee, & Cummins, 1991). Again, sensitive macroinvertebrates were lost at 11% of riparian forest loss (Brito et al., 2020; R. T. Martins et al., 2021). Although the altered riparian vegetation presumably affected both fish and macroinvertebrate assemblages, the altered flow regime and channel fragmentation by dams likely affected fish migration and recruitment more than macroinvertebrate movements. For example, our fish rules include broadcast spawners. Dams and altered flow regimes have greatly altered the persistence of those species in the Rio Grande (Archdeacon, Diver-Franssen, Bertrand, & Grant, 2020; Dudley & Platania, 2007) and other southwestern USA rivers (Durham & Wilde, 2006, 2009; Perkin et al., 2015, 2019). Annual flow fluctuations can also dramatically alter BCG scores if they are affected by orders of magnitude differences in short-lived species (Archdeacon et al., 2020).

The native status of fish (determined for each river basin) is recognized in 8 of the 11 fish model rules, including the rules that address native fish in attributes I–IV. Non-native fish are viewed as important biological contaminants and tend to become increasingly dominant as river size, and associated disturbances, increase (Hughes, Rinne, & Calamusso, 2005; Lomnicky, Whittier, Hughes, & Peck, 2007). However, they may be the dominant components of recreational fisheries in sites with poor to good water quality (Lomnicky, Hughes, Peck, & Ringold, 2021). There was no macroinvertebrate rule that incorporated native taxa status.

Fish and macroinvertebrate assemblages vary in their sensitivities to stressors and pressures. Herlihy et al. (2020) reported that poor macroinvertebrate MMI scores were most strongly affected by excess total nitrogen in arid ecoregions of the western U.S., whereas poor fish MMI scores were most strongly affected by high levels of chloride. We found that scores for several fish BCG model rules were affected by dams and altered flow regimes. Others have reported similar effects of dams on fish assemblages (Hughes et al., 2005), including for relatively small barriers (Jumani et al., 2018; Leitão et al., 2018). This is because such dams fragment rivers, precluding fish life history migrations to spawning and rearing sites. Altered flow regimes remove fish migration cues and the durations in which floodplains and off-channel waters remain suitable habitat for larval development (Perkin et al., 2015; Pompeu, Agostinho, & Pelicice, 2012; Schinegger, Trautwein, & Schmutz, 2013). However, macroinvertebrates can fly over dams and reservoirs, and thus are not as affected by river fragmentation and they do not require floodplain inundation for completing larval life histories to the same degree as fish. Furthermore, 55% of the stream network length in the Southwest USA that is mapped as perennial is actually non-perennial or not a stream because of map errors, climate change, or water removals (Colvin et al., 2019; Stoddard et al., 2005). Flow regimes are predicted to be altered further by climate warming, particularly in the southwestern USA (Overpeck & Bonar, 2020).

Finally, one of the limits to a rigorous BCG approach, which focuses on the presence or absence of sensitive-rare taxa as well as non-native taxa, involves implementing sufficient sampling effort to detect such taxa at a site. For example, Hughes, Herlihy, and Peck (2021) determined that 80–210 mean channel widths (MCWs) were needed to collect 90% of the observed fish species in western USA river sites via raft electrofishing. The number of MCWs increased as the proportion of spatially rare species increased. In a study of nine Missouri rivers, Dunn and Paukert (2020) concluded that fish species richness is underestimated whenever any single gear is employed. The sampling effort dilemma is even greater for macroinvertebrate taxa richness. J. Li et al. (2001) reported that 50 to over 60 Surber samples and identification of 3,000 individuals per site were needed to attain stabilized taxa richness counts in western Oregon stream sites. L. Li, Liu, Hughes, Cao, and Wang (2014) found that 15–20 kick net samples and thousands of identified individuals were needed to determine the number of taxa (genus or species) accurately in sand-bottom northeastern China stream sites. Thus, rare-sensitive taxa may be present regionally (Leal et al., 2018; Ligeiro, Melo, & Callisto, 2010), but their detection requires greater sampling effort than is typically expended in biomonitoring programs.

In summary, we found that the BCG process and modeling were useful for classifying the biological condition of southwestern USA sandy bottom rivers based on both macroinvertebrate and fish assemblage condition. The occasional minor differences in BCG levels between the two assemblages indicate their complementary nature and the value of assessing both in rigorous biological assessments (Yoder & Barbour, 2009). Contrary to our prediction, our BCG modeling indicated that the fish model may be slightly more sensitive than the macroinvertebrate model to anthropogenic disturbance in southwestern USA sandy bottom rivers. But this is likely because we had historical data available for fish, but not macroinvertebrates, and because fish may be more sensitive to dams and altered flow regimes than are macroinvertebrates.

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not constitute endorsement or recommendation for use. Statements in this publication reflect the authors' professional views and opinions and should not be construed to represent any determination or policy of the EPA.

## DATA AVAILABILITY STATEMENT

The data that support the findings of this study are available from the corresponding author upon reasonable request.

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**APPENDIX A**

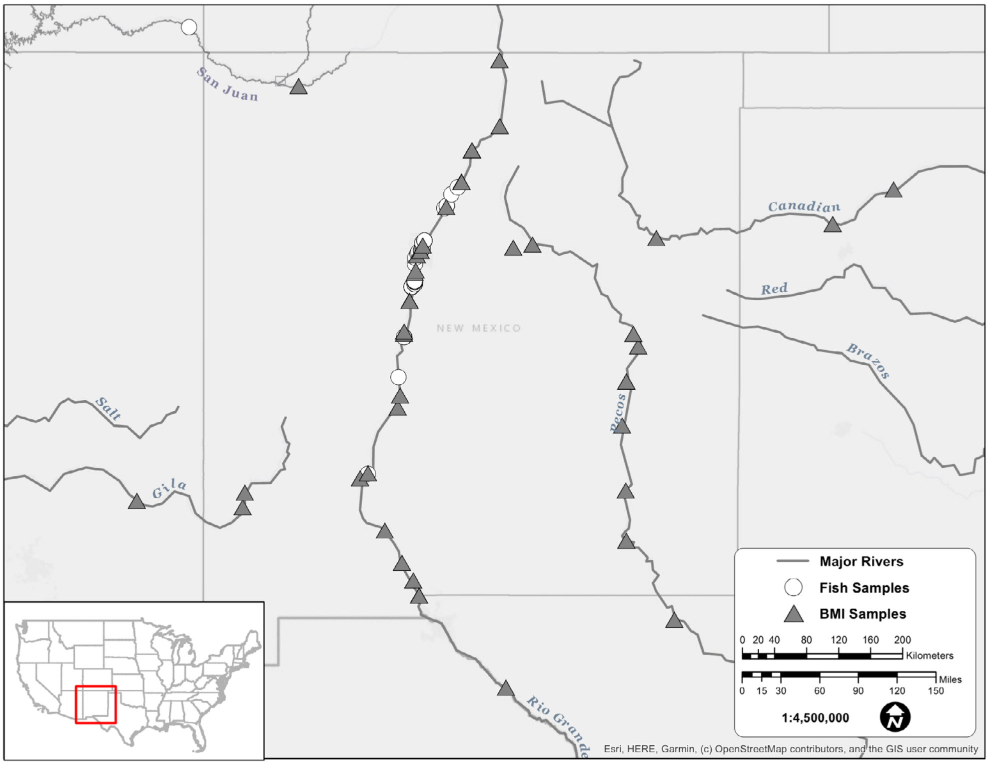


**FIGURE A1** Site locations of fish and benthic macroinvertebrate (BMI) samples used in calibrating the biological condition gradient for sandy bottom rivers in New Mexico [Color figure can be viewed at wileyonlinelibrary.com]

**TABLE A1** Numbers of benthic macroinvertebrate taxa assigned to the BCG attributes

| BCG attribute | # of taxa | Remark/example |
|---|---|---|
| I | 0 | No endemic specialists were identified by the expert panel. |
| II | 13 | All are insects. Several are in the Diptera, Ephemeroptera, and Trichoptera orders. Examples: *Drunella grandis*, *Rhyacophila coloradensis* gr., and *Oligophlebodes*. |
| III | 109 | All are insects (mostly EPT) except for one arachnid (*Testudacarus*). Examples: *Helichus*, *Tvetenia*, *Ephemerella*, *Isoperla*, and *Leucotrichia*. |
| IV | 213 | Includes various classes and orders. Examples: *Stenelmis*, *Baetis*, *Argia*, *Nectopsyche*, *Pisidium*, and *Ferrissia*. |
| V | 60 | Includes all of the oligochaetes and various other classes. Examples: *Naididae*, *Cryptochironomus*, Corixidae, and *Physa*. |
| VI | 4 | Examples: *Orconectes virilis*, *Corbicula*, and *Melanoides*. |
| NA (x) | 82 | The expert panel was unfamiliar with these taxa or the taxa represented a diverse group. |

**TABLE A2** Numbers of fish species assigned to the BCG attributes

| BCG attribute | # of taxa | Remark/example |
|---|---|---|
| I | 5 | Includes ENDEMIC FISH, such as the PHANTOM SHINER, which was historically documented but is now extirpated. |
| II | 3 | Species sensitive to sediment, such as the RIO GRANDE CHUB (which is also found in the Canadian and Pecos Rivers); species sensitive to loss of habitat such as the SHOVELNOSE STURGEON. |
| III | 3 | Moderately sensitive species, such as the LONGNOSE DACE and the RIO GRANDE SILVERY MINNOW |
| IV | 4 | Species commonly found in rivers, such as the RIVER CARPSUCKER and the BLUE CATFISH. |
| V | 5 | Species that are very tolerant, such as the BLUEGILL and GIZZARD SHAD |
| VI | 18 | Non-native species that are moderately tolerant of stressors, such as the GRAY REDHORSE, which is very common in Texas streams. |
| VI-sensitive | 7 | Species that are sensitive to loss of habitat but are not native, such as the DESERT SUCKER. |
| VI-tolerant | 19 | Non-native species that are tolerant, such as the LARGEMOUTH BASS. |
| NA (x) | 7 | Unassigned to an attribute. Examples include BLUEHEAD SUCKER and SUCKERMOUTH MINNOW. |
| X (10) | 1 | THE AMERICAN EEL is a CATADROMOUS FISH that lives in freshwater and migrates to the Sargasso Sea to spawn, indicating ecosystem connectivity. Other fish migrate throughout the rivers, but their potamodromous characteristics are secondary to their sensitivity/ tolerance assignments (attributes II – V). |

**TABLE A3** Site data sheet for a macroinvertebrate assemblage sample: *raw data*

| ExerciseID | Samp0678 | Assigned tier | Reasoning |
|---|---|---|---|
| Collection date | 10/04/2001 | 0 | |
| Collection method | Ben_03 | | |

**Taxa summary**

| BCG attribute | Number of taxa | Count | % taxa | % individuals |
|---|---|---|---|---|
| 1 | 0 | 0 | 0.0% | 0.0% |
| 2 | 1 | 2 | 5.0% | 0.4% |
| 3 | 4 | 15 | 20.0% | 3.3% |
| 4 | 13 | 430 | 65.0% | 95.6% |
| 5 | 2 | 3 | 10.0% | 0.7% |
| 6 | 0 | 0 | 0.0% | 0.0% |
| X | 0 | 0 | 0.0% | 0.0% |
| Total | 20 | 450 | 100% | 100% |

**Taxa list**

| BCG attribute | FinalID | Count | Excluded taxa | Family | Habit | FFG |
|---|---|---|---|---|---|---|
| 5 | *Naididae* | 2 | 0 | Naididae | Collector | Burrower |
| 5 | *Lumbriculidae* | 1 | 0 | Lumbriculidae | Collector | Burrower |
| 4 | *Microcylloepus* | 3 | 0 | Elmidae | Scraper | Clinger |
| 4 | *Cricotopus* | 6 | 0 | Chironomidae | Shredder | Clinger |
| 4 | *Microtendipes* | 5 | 0 | Chironomidae | Collector | Clinger |
| 4 | *Pseudochironomus* | 5 | 0 | Chironomidae | Collector | Burrower |
| 4 | *Simulium* | 14 | 0 | Simuliidae | Collector | Clinger |
| 4 | *Acentrella insignificans* | 3 | 0 | Baetidae | Collector | Swimmer |
| 4 | *Baetis tricaudatus* | 43 | 0 | Baetidae | Collector | Swimmer |
| 3 | *Nixe* | 5 | 0 | Heptageniidae | Scraper | Clinger |
| 4 | *Tricorythodes* | 6 | 0 | Leptohyphidae | Collector | Sprawler |
| 3 | *Petrophilia* | 4 | 0 | Pyralidae | Scraper | Clinger |
| 3 | *Claassenia sabulosa* | 2 | 0 | Perlidae | Predator | Clinger |
| 3 | *Brachycentrus americanus* | 4 | 0 | Brachycentridae | Filterer | |
| 4 | *Cheumatopsyche* | 2 | 0 | Hydropsychidae | Filterer | Clinger |
| 4 | *Hydropsyche* | 333 | 0 | Hydropsychidae | Filterer | Clinger |
| 4 | *Ochrotrichia* | 4 | 0 | Hydroptilidae | Collector | Clinger |
| 2 | *Psychoronia* | 2 | 0 | Limnephilidae | Shredder | Sprawler |
| 4 | *Chimarra* | 5 | 0 | Philopotamidae | Filterer | Clinger |
| 4 | *Turbellaria* | 1 | 0 | | Predator | Sprawler |
| | TOTAL | 450 | | | | |

**TABLE A4** Site data sheet for a macroinvertebrate assemblage sample: *metric data*

| **Site characteristics** | |
|---|---|
| SiteID | 28RGrand624.3 |
| SampleID | NMED_678_R0 |
| Latitude | 36.21361111 |
| Longitude | −105.9344444 |
| Site location | Rio Grande at Embudo Gage |
| Date | 10/04/2001 |
| RG mainstem or other river | RG_abvCochitiDam |
| Site elevation (m) | 1780 |
| State | NM |
| Ecoregion | 22_Xeric |
| | **Metrics** |
| Shannon diversity index | 1.71 |
| Dominant taxon percent individuals | 74 |
| EPT taxa | 11 |
| EPT percent individuals | 90.89 |
| Non-insect percent individuals | 0.89 |
| Chironomidae percent individuals | 3.56 |
| Coleoptera percent individuals | 0.67 |
| Ephemeroptera percent individuals | 12.67 |
| Odonata percent individuals | 0 |
| Collector percent individuals | 18.44 |
| Filterer percent individuals | 76.44 |
| Predator percent individuals | 0.67 |
| Scraper percent individuals | 2.67 |
| Shredder percent individuals | 1.78 |
| Predator taxa | 2 |
| Scraper taxa | 3 |
| Shredder taxa | 2 |
| Burrower percent individuals | 1.78 |
| Clinger percent individuals | 85.11 |
| Sprawler percent individuals | 2 |
| Clinger taxa | 11 |
| Sprawler taxa | 3 |

**TABLE A5** Site data sheet for a fish assemblage sample: *raw data*

| ExerciseID | Sheet3_04238 | Assigned tier | Reasoning |
|---|---|---|---|
| Collection date | 11/5/03 | | |
| Collection method | USGS; USGS Fish NM-TX Border_2; RA | | |

**Taxa summary**

| BCG attribute | Number of taxa | Count | % taxa | % individuals |
|---|---|---|---|---|
| I | 0 | 0 | 0.0% | 0.0% |
| II | 0 | 0 | 0.0% | 0.0% |
| III | 0 | 0 | 0.0% | 0.0% |
| IV | 1 | 5 | 7.7% | 1.1% |
| V | 4 | 129 | 30.8% | 27.2% |
| 6s | 0 | 0 | 0.0% | 0.0% |
| 6 | 4 | 284 | 30.8% | 59.8% |
| 6t | 4 | 57 | 30.8% | 12.0% |
| X | 0 | 0 | 0.0% | 0.0% |
| Total | 13 | 475 | 100% | 100% |

**Taxa list**

| BCG attribute | FinalID | Count | Family | CommonName | Trophic | Habitat |
|---|---|---|---|---|---|---|
| V | *D. cepedianum* | 50 | Clupeidae | Gizzard shad | H | W |
| IV | *Carpiodes carpio* | 5 | Catostomidae | River carpsucker | O | B |
| V | *Cyprinella lutrensis* | 56 | Cyprinidae | Red shiner | O | W |
| 6t | *Cyprinus carpio* | 24 | Cyprinidae | Common carp | O | B |
| 6 | *Pimephales vigilax* | 277 | Cyprinidae | Bullhead minnow | I | W |
| V | *Gambusia affinis* | 22 | Poeciliidae | West. Mosquitofish | I | W |
| 6t | *Lepomis cyanellus* | 8 | Centrarchidae | Green sunfish | I | W |
| V | *L. macrochirus* | 1 | Centrarchidae | Bluegill | I | W |
| 6 | *Lepomis megalotis* | 1 | Centrarchidae | Longear sunfish | I | W |
| 6t | *Pomoxis annularis* | 1 | Centrarchidae | White crappie | P | W |
| 6 | *Morone chrysops* | 1 | Percichthyidae | White bass | P | W |
| 6t | *Ictalurus punctatus* | 24 | Ictaluridae | Channel catfish | I | B |
| 6 | *Pylodictus olivaris* | 5 | Ictaluridae | Flathead catfish | P | B |

**TABLE A6** Site data sheet for a fish assemblage sample: *metric data*

| Site characteristics | |
|---|---|
| SiteID | Sheet3_04238 |
| SampleID | 4,943 |
| Latitude | 31.80450 |
| Longitude | −106.54500 |
| Site location | RIO GRANDE AT EL PASO, TX (Courchesne) |
| Date | 11/5/03 |
| Season | Fall |
| RG mainstem or other river | RG_blwCaballoDam |
| State | NM |
| Strahler order | 8 |
| Watershed SqKm | 75,925 |
| Ecoregion (level 3 or NRSA) | 24_ |
| Number of individuals | 475 |
| Number of taxa | 13 |
| | **Sample metrics** |
| Percent Cyprinidae individuals | 75.16 |
| Number of Cyprinidae taxa | 3.00 |
| Percent insectivore individuals | 70.11 |
| Number of insectivore taxa | 6.00 |
| Percent piscivore individuals | 1.47 |
| Number of piscivore taxa | 3.00 |
| Percent omnivore individuals | 17.89 |
| Number of omnivore taxa | 3.00 |
| Percent herbivore individuals | 10.53 |
| Number of herbivore taxa | 1.00 |
| Percent lithophilic individuals | #ref! |
| Number of lithophilic taxa | #ref! |
| Percent water column individuals | 87.79 |
| Number of water column taxa | 9.00 |
| Percent benthic individuals | 12.21 |
| Number of benthic taxa | 4.00 |

**TABLE A7** Example of expert panel ratings and rationale for a single benthic macroinvertebrate sample with a summary rating of BCG Level 3

| Expert | BCG rating | Rationale |
|---|---|---|
| Panelist #1 | 3+ | EPT high |
| Panelist #2 | 3+ | Good taxa richness (L2-3); good diversity; good % target subsample; good EPT richness (L2); good non_hydro trichop. (L3); good chironomid richness (L3); high %EPT (L2-3); low % non-insects (L2-3); low % chironomids (L3); Sort of high % collector-gatherers but otherwise relatively balanced trophic structure; good scraper richness (L2-3); good shredder richness (L3); good clinger richness (L2-3). |
| Panelist #3 | 3+ | |
| Panelist #4 | 3 | Probably a low elevation UT river, good stonefly diversity, other sensitive taxa |
| Panelist #5 | 3 | Good # and taxa, high EPT, low non-insect, 2 and 3 attributes |
| Panelist #6 | 3− | Good overall diversity; good chironomids and EPT diversity; Plecoptera; FFG % is a bit skewed but not bad; |
| Panelist #7 | 4+ | High EPT, low % chironomids with good diversity, low shredders |

**TABLE A8** Example of expert panel ratings and rationale for a single fish sample with a summary rating of BCG Level 4+

| Expert | BCG rating | Rationale |
|---|---|---|
| Panelist #1 | 4 | Red shiner, channel catfish; like the flathead chub |
| Panelist #2 | 4+ | Healthy variation; att3 and 4 |
| Panelist #3 | 4+ | |
| Panelist #4 | 4 | Lots of att3 individuals (161); would be a 4+ if no channel catfish |
| Panelist #5 | 4+ | A lot of catfish; lots of silvery minnows; if catfish were not there, it would be a 3− |
| Panelist #6 | 4+ | Higher numbers of Att. 3 taxa but significant presence of tolerant non-native |
| Panelist #7 | 4 | Sensitive species are represented; channel catfish bothersome |