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Development and validation of an environmental fragility index (EFI) for the neotropical savannah biome

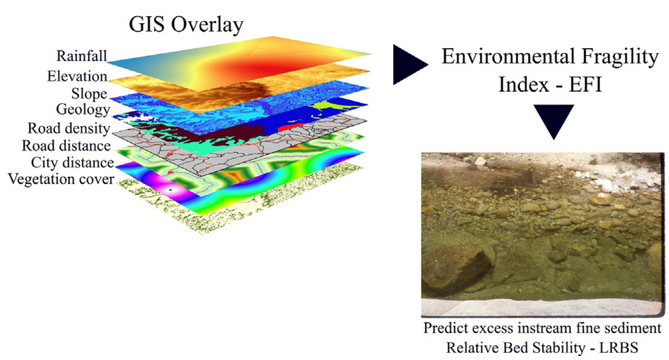
Diego R. Macedo [a,b,*], Robert M. Hughes [c], Philip R. Kaufmann [d], Marcos Callisto [b]

[a] Departamento de Geografia, Instituto de Geociências, Universidade Federal de Minas Gerais, Av. Antônio Carlos 6627, CEP 31270-901, Belo Horizonte, MG, Brazil
[b] Laboratório de Ecologia de Bentos, Departamento de Biologia Geral, Instituto de Ciências Biológicas, Universidade Federal de Minas Gerais, Av. Antônio Carlos 6627, CP 486, CEP 31270-901, Belo Horizonte, MG, Brazil
[c] Amnis Opes Institute and Department of Fisheries and Wildlife, Oregon State University, 200 SW 35th Street, Corvallis, OR 97333, USA
[d] Western Ecology Division, National Health and Environmental Effects Laboratory, Office of Research and Development, United States Environmental Protection Agency, 200 SW 35th Street, Corvallis, OR 97333, USA

HIGHLIGHTS

- We developed an environmental fragility index (EFI) through GIS procedures.
- We tested its ability to predict excess fine sediment in 148 sample streams.
- Natural cover and anthropogenic pressures comprised 70% of metrics in the EFI.
- Model-based expansion estimated high to extreme EFI in 65% of the study catchments.
- Contributing land area in the São Francisco Basin is less fragile than the Para Basin.

GRAPHICAL ABSTRACT



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ABSTRACT

Augmented production and transport of fine sediments resulting from increased human activities are major threats to freshwater ecosystems, including reservoirs and their ecosystem services. To support large scale assessment of the likelihood of soil erosion and reservoir sedimentation, we developed and validated an environmental fragility index (EFI) for the Brazilian neotropical savannah. The EFI was derived from measured geoclimatic controls on sediment production (rainfall, variation of elevation and slope, geology) and anthropogenic pressures (natural cover, road density, distance from roads and urban centers) in 111 catchments upstream of four large hydroelectric reservoirs. We evaluated the effectiveness of the EFI by regressing it against a relative bed stability index (LRBS) that assesses the degree to which stream sites draining into the reservoirs are affected by excess fine sediments. We developed the EFI on 111 of these sites and validated our model on the remaining 37 independent sites. We also compared the effectiveness of the EFI in predicting LRBS with that of a multiple linear regression model (via best-subset procedure) using 7 independent variables. The EFI was significantly correlated with the LRBS, with regression R^2 values of 0.32 and 0.40, respectively, in development and validation sites. Although the EFI and multiple regression explained similar amounts of variability ($R^2 = 0.32$ vs 0.36), the EFI had a higher F-ratio (51.6 vs 8.5) and better AICc value (333 vs 338). Because the sites were randomly selected and well-distributed across geoclimatic controlling factors, we were able to calculate spatially-explicit EFI values for all hydrologic units within the study area (~38,500 km^2). This model-

* Corresponding author.
E-mail address: diegorm@ufmg.br. (D.R. Macedo).

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based inference showed that over 65% of those units had high or extreme fragility. This methodology has great potential for application in the management, recovery, and preservation of hydroelectric reservoirs and streams in tropical river basins.

1. Introduction

Studies quantifying environmental properties, such as fragility, conservation status, resource exploitation, and rehabilitation needs are essential for decision makers to make rational management decisions and conserve natural resources (Villa and McLeod, 2002). In a more direct sense, the environmental fragility of the landscape, here understood as the susceptibility of the environment to suffer major disruptions in its dynamic equilibrium, mainly in increased erosion and sedimentation rates (Burcher et al., 2007), should be studied to improve landscape management (Manfré et al., 2013). Further, it is necessary to develop analytical approaches that integrate interactions between anthropogenic and biogeophysical aspects of landscapes to support those studies (Angelstam et al., 2013) because anthropogenic pressures determine impacts, and some of those impacts are irreversible (Pavlickova and Vyskupova, 2015).

Current natural resource management tools can quantify the economic benefits of human appropriations of nature, or ecosystem services (Costanza et al., 1997), in which there is great emphasis on freshwater ecosystems (Egoh et al., 2007; Ringold et al., 2013; Vihervaara et al., 2010). Hydroelectric power generation is a key ecosystem service provided by water runoff in drainage networks, but it requires reservoirs for water storage. Although 16% of the global electrical energy matrix is hydroelectric, it is about 70% in Brazil (Von Sperling, 2012). In addition to power generation, lakes and reservoirs provide other ecosystem services such as recreation, flood control, water supply, navigation, nutrient cycling, and food production (Baron et al., 2002; Von Sperling, 2012). Thus, it is necessary that these facilities operate as efficiently and as long as possible, to maximize their ecosystem services (Arias et al., 2011) and benefit/cost ratios. On the other hand, the ecological impacts of hydroelectric reservoirs can be very large, for example, altered flow regimes, siltation, habitat simplification, alien species presence, and reduced biodiversity (Becker et al., 2016; Callisto et al., 2014; Morais et al., 2017; Sanches et al., 2016; Uehara et al., 2015). Conservation actions that increase the useful life of the reservoir can have conservation benefits at larger scales, by reducing the necessity for constructing new reservoirs.

Like all lacustrine ecosystems, reservoirs accumulate multiple influences from their drainage basins (Baron et al., 2002) and sediment deposition is a major threat to the operation and service life of reservoirs (Arias et al., 2011; Von Sperling, 2012). Because of reservoir construction and operation costs, there are currently financial subsidies for conserving the surrounding vegetation and discouraging other uses (e.g., agriculture, pasture, or urbanization) to help maintain reservoir quality (Arias et al., 2011). Further, reducing the anthropogenic sources of upland soil erosion and resultant siltation in streams and reservoirs is important for biodiversity conservation, because anthropogenic increases in fine sediment are identified as a major cause of biodiversity loss in streams (Bryce et al., 2010; Ferreira et al., 2014; Kaufmann et al., 2009) and reservoirs (Lenhardt et al., 2008; Molozzi et al., 2013).

In recent years, geographical information systems (GIS) have become powerful tools in environmental management, helping decision makers to rapidly and efficiently analyze spatial and temporal patterns in ecosystem services and potential project impacts on natural resources (Nemec and Raudsepp-Hearne, 2012). GIS spatial analysis tools are essential in studies where multiple spatial variables can be integrated to aid decision making (Malczewski, 2006). GIS, mathematical models, and ecological indicators are used increasingly to aid decision makers in evaluating the costs and benefits of large projects, such as hydroelectrical and industrial plants, major housing and transport systems (e.g. Pavlickova and Vyskupova, 2015) or in large-scale fragility studies (Manfré et al., 2013; Toro et al., 2012; Xiaodan et al., 2010). Those tools are especially useful for prioritizing the use of financial resources in an efficient and rational manner (Villa and McLeod, 2002). The novelty of our study was that we tested the environmental fragility index that we developed with strict analytical procedures (Villa and McLeod, 2002); to our knowledge this had not previously been done. Moreover, we validated our approach through comparison of model results with field data collected using state-of-science regional survey methods. Thus, the aim of our study was to construct and validate an environmental fragility index (EFI) for basins with hydroelectric reservoirs in the neotropical savannah biome.

2. Methods

2.1. Study area

For the construction and testing of the EFI we sampled 148 wadeable streams of Strahler order 1–3 on 1:100000 scale maps (Strahler, 1957) between September 2010 and September 2013. We used 111 sites (75%) to develop the index and retained 37 independent sites (25%) as a well-distributed set of validation sites across the study area to test the effectiveness of the EFI. The sites were located within 35 km upstream of the reservoirs formed by four hydropower dams (Nova Ponte, Três Marias, Volta Grande, São Simão), considered as hydrologic units (sensu Seaber et al., 1987), in a total study area of 38,500 km^2. Those hydrologic units are situated in two major Brazilian river basins: São Francisco and Paraná (Fig. 1). The São Francisco and Paraná River Basins have great hydroelectric potential, and four of the largest hydroelectric power plants operating in Brazil today are located in the Paraná (e.g., Itaipu, Ilha Solteira) and São Francisco (e.g., Paulo Afonso, Xingó) Basins (ANEEL, 2018; Von Sperling, 2012). The sampling network was defined following an approach developed by the USEPA-EMAP Wadeable Stream Assessment (Olsen and Peck, 2008), using a spatially balanced generalized random tessellation stratified (GRTS) design, which allowed extrapolating the EFI results to the entire study area (Olsen and Peck, 2008; Silva et al., 2017; Whittier et al., 2007).

2.2. Environmental fragility index (EFI)

Rates of sediment production in drainages are influenced by complex interactions among rainfall regime, soil and substrate properties, lithology, slope and terrain characteristics, vegetation cover, and land use and management (Guerra et al., 2017). Thus, useful EFI scores should express the fragility of drainages as the likelihood of fine sediment generation and transport. We defined fragility based on the interaction between vulnerability (the intrinsic or natural potential of catchments to yield fine sediment when disturbed) and anthropogenic influences that act on that vulnerability to augment or moderate erosion rates. We assessed vulnerability using natural landscape data from GIS coverages, and assessed influences that act on that vulnerability by vegetation cover and anthropogenic pressures as also indicated by GIS-derived data.

2.2.1. Acquisition of natural landscape data

We used existing digital maps to determine catchment characteristics. The 148 catchments were determined from the terrain model of the Shuttle Radar Topographic Mission – SRTM (~30 m spatial resolution; USGS, 2015). The rainfall data were calculated using the time

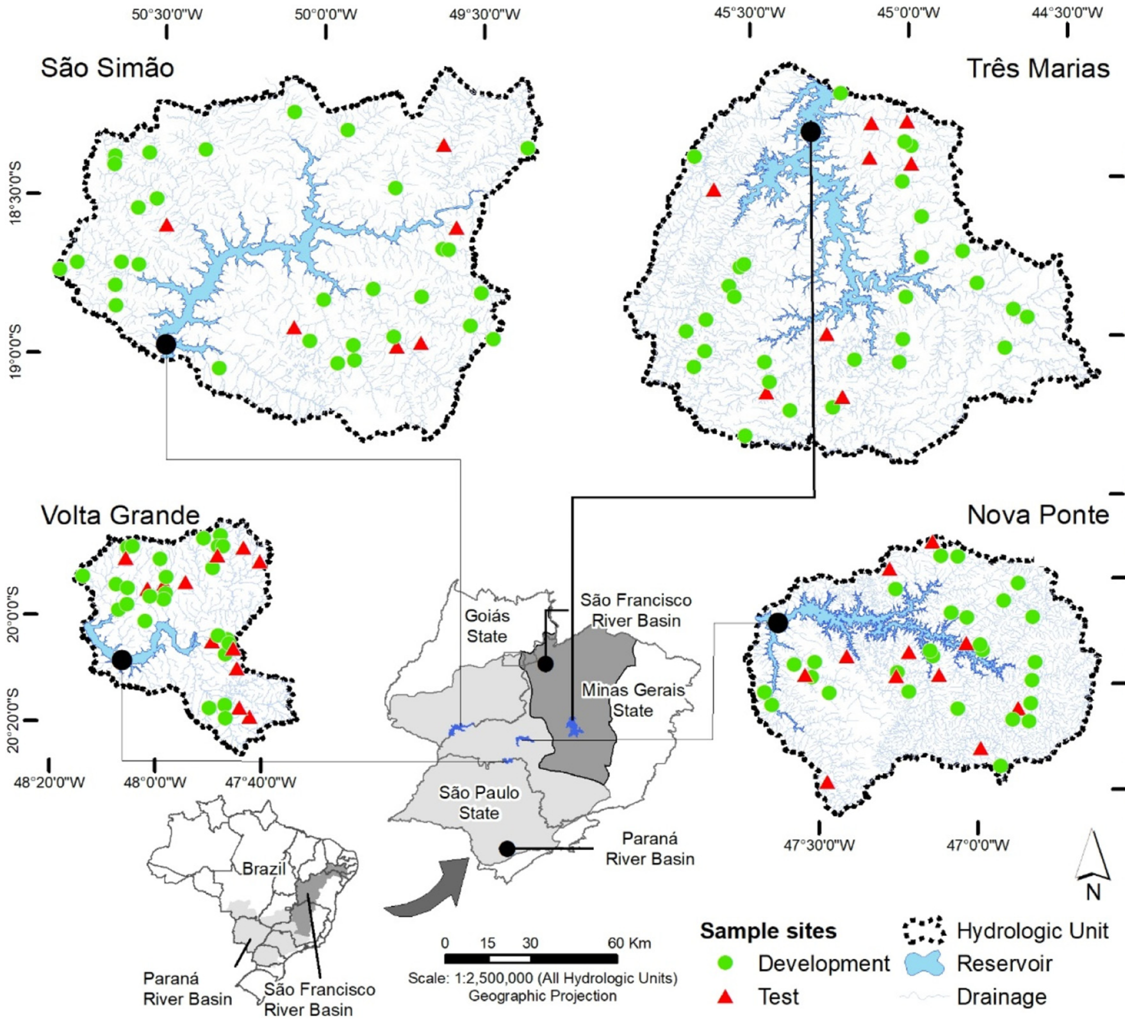

Fig. 1. Locations of the studied hydrologic units and sampling sites.

series provided by the Brazilian National Agency of Waters (ANA, 2017a, 2017b) and the Brazilian National Institute of Meteorology (INMET, 2017). Sixty-seven rainfall stations had rainfall extracted, georeferenced, and interpolated using the thin plate spline procedure (Fick and Hijmans, 2017). We calculated mean annual rainfall for the year before, and the two years before each stream site was sampled to accommodate inter-annual differences in rainfall and lag effects between catchment erosion and stream sedimentation. We also calculated the rainfall intensity index, the ratio of the total annual precipitation (mm) and the duration of the rainy season (months), at the same temporal scale. For both data types, we calculated the mean of cell values for each of the 148 catchments. Morphometric variables for each of the 148 catchments also were extracted from the SRTM terrain model. To represent the degree of terrain dissection, we calculated the range, standard deviation and coefficient of variation of both elevation and slope in catchments. All the morphometric variables were calculated for every cell within each catchment. Finally, the prevalence of each geologic unit was extracted for each of the 148 catchments by extracting lithology data from the 1:250,000 Brazilian Geological Map (IBGE, 2003). Each lithology type was assigned a vulnerability value according to Crepani et al. (2008), thereby converting a set of class variables into a single, continuous numerical variable that was more useful analytically.

2.2.2. Acquisition of anthropogenic pressures data

The mapping of land use and cover in the 148 catchments was performed by interpretation of multispectral images from the Landsat TM satellite sensor, aided by Google Earth fine resolution images (Macedo et al., 2014). The Landsat images used were acquired for the months of sampling and fine resolution images were used to support the interpretation. The Google images clearly show the shape and texture of the elements, whereas the Landsat images have different spectral responses for the targets, enabling high mapping accuracy. Areas were identified with natural vegetation cover (woodland savanna, grassy-wood savanna, parkland savanna, palm swamp) and four anthropogenic uses (pasture, agriculture, urban, and eucalyptus reforestation). We measured the percentage of vegetation cover and agriculture within each catchment and used them as single variables, but also calculated a catchment disturbance index (CDI) based on weighted percentages of anthropogenic land use in the catchment. That is, urban areas were weighted more highly than agricultural areas, which were in turn weighted more highly than pasture areas (Ligeiro et al., 2013). To further characterize anthropogenic presence and influence, we calculated house and population densities within each catchment using 2010 Brazilian Census data (IBGE, 2015). Catchment road density and road distance from each site were calculated from Open Street Map data (OpenStreetMap Foundation, 2017). The Euclidean distance between each sample site and the nearest cities provided an additional proxy for the intensity of human pressure.

2.2.3. Acquisition of local fragility impact response

To assess its validity and effectiveness, we compared catchment EFI scores to the site streambed stability. This impact was measured by assessing the physical characteristics of each site through use of physical habitat protocols (Peck et al., 2006). The sediment geometric mean diameter was determined by visual assessment of the diameter size class (e.g. silt, sand, gravel, cobble) of 105 particles observed at 5 points spread over each of 21 systematic cross sections in the wetted channel (Peck et al., 2006). The mean sediment critical diameter was calculated from the slope, bankfull flow, and the hydraulic roughness represented by the residual pools and the volume of wood in the site (Kaufmann et al., 2009).

The above field data were used to calculate a relative bed stability index (LRBS; Kaufmann et al., 2008) score for each site. The LRBS is the $\log_{10}$ of the ratio of bed surface geometric mean particle diameter divided by the estimated critical diameter at bankfull flow. Decreases in LRBS result when fine sediment supplies exceed the ability of streams to transport that sediment. Negative LRBS values indicate unstable beds, high sediment transport rates, and bed textural fining (Kaufmann et al., 2008). Unstable streambeds result from bedload deposition during low flow periods. Low LRBS values cannot persist in a stream without augmented sediment supply, so this index has been used to assess the effects of siltation produced by anthropogenic landscape pressures and the resulting stream responses (Benoy et al., 2012; Jessup et al., 2014; Kaufmann et al., 2009). Positive LRBS values indicate stable beds or sediment coarsening. Consequently, the LRBS is an indicator of increases or decreases in sediment supply relative to the ability of the stream to transport bedload sediments (Kaufmann et al., 2009). The LRBS has been used to evaluate regional patterns in bed stability and sedimentation and their general relationship to human disturbances and natural and anthropogenic landscape variables (Faustini et al., 2009; Kaufmann and Hughes, 2006; Kaufmann et al., 2009; Leal et al., 2016; Leitão et al., 2018).

2.2.4. Environmental fragility index (EFI)

The EFI development involved six steps: variable selection, standardization, weighting, consistency, index calculation, and an efficiency test. We used only the development dataset of 111 sites to build the EFI. First, we analyzed Pearson correlations among 20 candidate variables to identify those that were highly correlated ($r > |0.6|$). Next, the natural landscape variables and anthropogenic pressures were standardized in the same numerical scale to aid aggregation. The values shown for each variable were converted to ordinal scores of 1 to 5, taking into account the variability encountered through visual inspection of scatterplots about EFI component variables versus LRBS.

To analyze the environmental fragility of an area, one must recognize that different variables have different importance in the index (Malczewski, 2006), so we needed to weight them. To do so, we used the Analytical Hierarchy Process method (AHP; Saaty, 1977), widely used in environmental analysis developments (e.g., Nguyen et al., 2016; Sahoo et al., 2016; Tian et al., 2013; Wang et al., 2008; Xiaodan et al., 2010). In this analytical method, we compared the variables in pairs, so that each interaction was given a weight, which aids organizing each variable in a hierarchy. The variables were arranged in a matrix and the relative importance of each variable relative to the other was assessed. For example, when variable A was more important than variable B, it received a score of x, and similarly, variable B received its reciprocal $1/x$. To calculate the scores, we ran ordinary least squares (OLS) regressions using all variables versus LRBS, two independent variables at a time, to compute the relative importance of each variable. We used the standardized β values to evaluate the proportional importance (ratio among lowest and highest β) of variables in each OLS regression. To determine significant OLS results ($p > 0.05$), we considered both variables equally important. To convert the β ratio (continuous values between 0 and 0.99) to degree of importance (ordinal scale 1 to 9), we divided them into 9 equal classes (Table 1). The final weight was the sum of the relative importance of each variable, and the sum of weights had a value of 1.0.

Table 1
Degree of importance of relationships between environmental variables. Correspondence between degree of importance and β ratios in parentheses.

Degree of importance (β ratio)	Definition
1 (0.88–0.99)	Equal importance of two elements
2 (0.77–0.88)	Intermediate value between equal and weak importance
3 (0.66–0.77)	Weak importance of an element in comparison to the other
4 (0.55–0.66)	Intermediate value between weak and strong importance
5 (0.44–0.55)	Strong importance of an element in comparison to the other
6 (0.33–0.44)	Intermediate value between strong and certified importance
7 (0.22–0.33)	Certified importance of an element in comparison to the other
8 (0.11–0.22)	Intermediate value between certified and absolute importance
9 (0.00–0.11)	Absolute importance of an element in comparison to the other
1/2, 1/3, 1/4, 1/5, 1/6, 1/7, 1/8, 1/9	Reciprocal values of the previous factors

To check whether the matrix was consistent (e.g., if A was more important than B, and B more important than C, then C cannot be more important than A), we used the consistency ratio (CR; Eq. (1)).

$$\mathrm{CR} = \mathrm{CI}/\mathrm{RI} \tag{1}$$

where RI is the value given, referring to the size of the array and defined by Saaty (1977) and CI is the consistency index determined from Eq. (2).

$$\mathrm{CI} = (\lambda_{\max} - n)/(1 - n) \tag{2}$$

where λmax is the largest or principal eigenvalue of the matrix of variables, and n is the order of the matrix. A CR of 0.10 or less shows that the relationship among the variables is consistent (Saaty, 1977).

To calculate the environmental fragility index (EFI), we summed the variables (A_i), each weighted by its importance value (W_j), according to Eq. (3).

$$\mathrm{EFI} = \sum A_i W_j \; (i = 1, 2 \ldots n) \tag{3}$$

The EFI was divided into 6 fragility classes for map depiction: slight (< 2.0), light (2.0 < 2.5), moderate (2.5 < 3.0), high (3.0 < 3.5), very high (3.5 < 4.0), and extreme (>4.0).

We evaluated the validity of the EFI by comparing results of a linear regression predicting LRBS in the 111 development sites with those from the 37 validation sites not used in developing the EFI. We evaluated EFI model efficiency by comparing LRBS prediction performance with that for an ordinary least squares (OLS) regression model derived via the best-subset procedure using 7 independent variables. We compared the coefficient of determination (R^2), Fisher distribution ratio (F-ratio) and Akaike information criterion, corrected (AICc) values of all three models.

2.2.5. EFI expansion for the entire reservoir hydrologic units

Because the EFI was developed and tested on a spatially-balanced probabilistic sampling network, it can be expanded by model-based inference to all the catchments of the study area (Olsen and Peck, 2008; Silva et al., 2017; Whittier et al., 2007). We used Ottobasin classification (Pfafstetter, 1989) to expand it in a manner easily applied by GIS procedures (Fontes and Pejon, 2008) and compatible with current Brazilian environmental law (Brazil, 2002a). Ottobasins are size-binned hierarchical classifications of all drainage areas and adjacent interstices, similar to the hydrologic unit (HU) classification used by U.S. Geological survey in the USA (Omernik et al., 2017). Although they are not true catchments, they still have some characteristics of catchments, in which the area upstream of a given point influences it to some degree (Fig. 2), making them useful units for water resource analyses and which are used in other Brazilian studies (e.g., Fontes and Pejon, 2008; Moraes et al., 2013; Venticinque et al., 2016). For our model-based expansion, we used level 7 of the Ottobasin hierarchy, extracted from

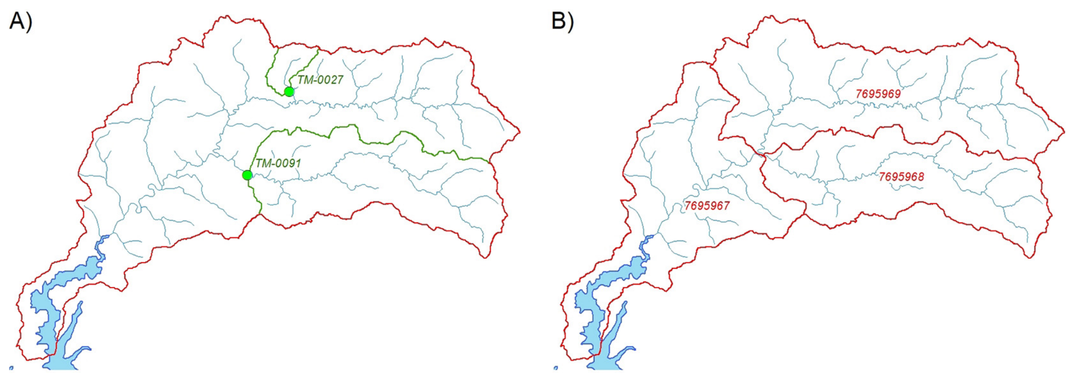

Fig. 2. Example of (A) catchment sampled sites and (B) level 7 Ottobasin classification. Extrema Grande stream watershed, Três Marias Hydrologic Unit.

the Brazilian Water Agency Ottobasin digital map (ANA, 2017a, 2017b). Like HU's, Ottobasins split upstream and downstream portions of basins; depending on the size of these HU's, they ignore upstream conditions about half the time (Omernik et al., 2017). Consequently, they typically do not predict instream conditions as well as true catchments (Omernik et al., 2017; Thornbrugh et al., 2018). Despite their limitations, we used Ottobasins instead of true catchments to expand our EFI results to the entire study area for three reasons. 1) Unless one chooses a subset of the actual catchments along a stream network, the number of nested catchments that can be defined along a stream network is infinite. 2) One might choose a subset of catchments contributing to the nodes defined at the intersections of streams, but in Brazil, like many regions, the resolution of stream mapping is not consistent among regions. 3) Any subset of catchments draining to stream channels excludes the land areas draining directly into the reservoir, so would not allow assessment of the entire study area. Consequently, to assess how well our 148 model-development sample catchments represented the variables used to model EFI, we compared distributions of the set of EFI controlling factors in our sample catchments with those in the level 7 Ottobasins that make up the entire study area.

We calculated the EFI for the entire study area using the same variables used in Eq. (3) but employed two data replacements. First, we used stable mean annual rainfall (last 30 years at the same meteorological stations) instead of the 2 last years mean rainfall, and we also used a vegetation cover map that covers the entire study area (Hansen et al., 2013). Those modifications were necessary to have a consistent, and easily replicated map that is not dependent on short term precipitation patterns. The expansion also used the 6 fragility classes.

3. Results

3.1. Characteristics of the 148 catchments

Topography, geoclimatic characteristics, and anthropogenic pressures generally known to affect erosion and sedimentation differed to varying degrees among the 148 catchments (Table 2). The range of elevation within the 148 catchments was predominantly between 100 and 200 m with coefficients of variation predominately <15%; average slopes were also low but with coefficients of variation as high as 30%. The mean rainfall of the two years prior to stream sampling was generally greater than that in the preceding year; rainfall intensity was moderate (<250 mm per rainy months) in both periods. Over 25% of the catchments occur on volcanic rocks (dacite), just under 50% on sedimentary rocks (mudstones, sandstones, arkose, shale); and about 19% occur on metamorphic rocks (schists, phyllites). In addition, the catchments have great variation in anthropogenic pressures, showing natural cover and housing density (mostly farm homes) on average < 30% and agriculture use near 50%. The catchments are generally far from urban centers, but relatively near to both paved and unpaved roads. Generally, LRBS was negative, indicating that at bankfull conditions, the study streams were actively transporting bedload sediments delivered to the stream channel, and deposited during low flow periods. There are low but significant correlations between half of the predictor variables and LRBS.

3.2. EFI development

Our variable screening and selection process yielded 8 variables, half in each category (natural landscape and anthropogenic pressures; Supplementary Material 1). The variables representing the intrinsic or natural potential of catchments to yield fine sediment when disturbed were rainfall erosivity (Rainfall_2y), terrain dissection (Elevation_cv, Slope_cv), and the erodibility of bedrock underlying soils (Geology_w). The anthropogenic pressure variables were degree of soil protection (% Natural_cover) and human infrastructure (road_den, road_dist, city dist). Visual inspection of scatterplots of EFI component variables versus LRBS in the 111 model development sites (Supplementary Material 2) facilitated standardization of those factors in an ordinal scale between 1 and 5 (Table 3).

The results of ordinary least squares regressions, two independent variables by turn with LRBS, showed significant relationships in most of them, with R^2 predominantly between 10% and 20% (Table 4). The results showed that natural cover was the key variable, because it was the best predictor in most regressions. The final weights, varying between 0.025 and 0.26, highlight anthropogenic pressure factors (~70% of weights; Table 5; Supplementary Material 3). Our analyses indicated an adequate consistency of importance rankings, with a consistency ratio (CR) of 0.049 (Table 5; Supplementary Material 3).

The EFI showed that about 35% of the model development sites had slight to moderate degrees of fragility, and 65% had high to extreme values. There was a clear fragility gradient in the study area (Fig. 3).

3.3. EFI efficiency

The three regression models produced similar results. The linear regression model based on the 111 developmental sites indicated significant correlation between the EFI and the LRBS ($R^2 = 0.322$, $p < 0.001$). The regression of the EFI versus the LRBS based on the 37 validation sites was similarly correlated ($R^2 = 0.40$) and had a slope similar to that for the developmental regression ($\beta = -0.57$ and -0.63; Fig. 4). The best subset regression model explaining variance in LRBS based on 7 variables and 111 sites had a slightly higher R^2 (0.368; $p < 0.001$) than did the developmental regression of EFI versus LRBS, but a much lower F-ratio and a higher AICc (Table 6).

3.4. Expansion of EFI for the entire reservoir hydrologic units

There were no substantive differences in the distributions of the important EFI component variable values between the 148 sampled catchments and those of the entire study area, as represented by Ottobasins

Table 2
Landscape characteristics and relative bed stability of the 148 catchments.

Factors		Mean ± SD	Correlation with LRBS
Natural landscape			
Rainfall_1y	Total rainfall (mm/year) at last hydrologic year	1387.56 ± 323.26	−0.19*
Rainfall_2y	Rainfall mean (mm/year) at 2 last hydrologic years	1483.24 ± 231.49	−0.32*
Rain_Int_1y	Rainfall intensity (mm/rainy months) of last hydrologic year	174.55 ± 37.72	−0.11
Rain_Int_2y	Rainfall intensity (mm/rainy months) of 2 last hydrologic years	184.56 ± 33.93	−0.12
Elev_std	Standard deviation value (m) of all elevation cells inside each catchment	37.88 ± 17.84	−0.05
Elev_range	Range value (m) of all elevation cells inside each catchment	178.64 ± 74.53	−0.02
Elev_cv	Coefficient of variation value (%) of all elevation cells inside each catchment	5.35 ± 2.93	−0.26*
Slope_m	Mean value (%) of all slope cells inside each catchment	8.42 ± 3.69	0.30*
Slope_std	Standard deviation value (%) of all slope cells inside each catchment	5.45 ± 2.51	0.14
Slope_range	Range value (%) of all elevation cells slope each catchment	43.78 ± 20.59	−0.01
Slope_cv	Coefficient of variation value (%) of all slope cells inside each catchment	65.55 ± 17.35	−0.15
Geology_w	Theoretical degree of fragility of predominant geology type in each catchment	2.09 ± 0.68	−0.01
	Geology type predominance (number of catchments):		
	Dacite (39)		
	Schist (20)		
	Phyllite (9)		
	Sandstone (15)		
	Arkose (9)		
	Mudstone (31)		
	Shale (17)		
	Alluvium (8)		
Anthropogenic pressures			
Road_den	Road density (km/km^2) in each catchment	0.66 ± 2.14	−0.13
Pop_den	Population density (inhabitants/km^2) in each catchment	62.96 ± 325.38	−0.11
House_den	Household density (house/km^2) in each catchment	20.8 ± 108.63	−0.11
Road_dist	Site distance (km) from nearest road	3.19 ± 3.84	0.37*
City_dist	Site distance (km) from nearest city	15.33 ± 8.52	0.33*
CDI	Catchment disturbance index	124.37 ± 61.74	−0.32*
%Natural_cover	Natural vegetation cover in each catchment (%)	28.12 ± 21.04	0.41*
%Agriculture	Agriculture use in each catchment (%)	47.78 ± 33.55	−0.20*
Sediment			
LRBS	Relative Bed Stability (Log_{10})	−0.967 ± 1.241	

* $p < 0.05$

(Fig. 5). This finding justifies applying the EFI calculation algorithm developed on the sample sites to the whole reservoir study landscape.

The six fragility classes were distributed across the entire study area, but most frequently in classes 4 and 5 (high and very high fragility), indicating a preponderance of land area with a high probability of abundant fine sediment delivery to stream channels under current conditions (Fig. 6). The spatial distribution of classes allows us to clearly identify hydrologic units most and least likely to contribute fine sediment in sufficient abundance to result in low LRBS in receiving streams and consequent transport of augmented sediment inputs to their downstream reservoirs. Clearly, São Simão and Volta Grande Reservoirs and their streams have greater fragility (greater likelihood of sedimentation) than Nova Ponte and Três Marias Reservoirs and their streams. Both of the latter hydrologic units have more catchments and more Ottobasins with slight, light or moderate fragility than the former two hydrologic units (Fig. 7).

4. Discussion

4.1. Methodological approach

In general, environmental fragility studies have been based only on theoretical aspects of the factors studied, hindering validation of the methodology (Villa and McLeod, 2002). However, we used multi-criteria analysis (AHP) supported by local data relationships and spatial and open access GIS databases to build an environmental fragility index (EFI) at a detailed scale. We used 111 stream sites in four hydrologic units to build the index and 37 validation sites to rigorously test the response of relative bed stability (LRBS) to EFI scores. The relationship between the EFI and LRBS was moderately significant ($R^2 = 0.32$ and 0.40, $p < 0.001$) and supported expansion of the EFI to the entire study area (~38,500 km^2) by model-based inference. Generally, aggregation of spatial data into a single index involves considerable reduction in data

Table 3
Environmental variables and their fragility standardized values in the study area.

Factors	Rating				
	1	2	3	4	5
(1) Rainfall mean 2 last years (mm/year)	<1225	1225–1350	>1350–1475	>1475–1600	>1600
(2) Elevation coefficient of variation (%)	<3	3–6	>6–9	>9–12	>12
(3) Slope coefficient of variation (%)	<50	50–65	>65–80	>80–95	>95
(4) Geology type (degree of fragility[a])	Dacite(1.1) Granite(1.1)	Schist (1.7)	Phyllite(2.1)	Arkose(2.6) Sandstone(2.4)	Alluvium(3.0) Calcarenite(2.9) Shale(2.8) Mudstone(2.7)
(5) Road density (km/km^2)	<0.1	0.1–0.2	>0.2–0.5	>0.5–1	>1
(6) Road proximity (km)	>10	>6–10	>3–6	1–3	<1
(7) City proximity (km)	>25	25-15	<15–10	<10–0	<5
(8) Vegetation cover (%)	>50	50–30	<30–20	0–20	<10

[a] From Crepani et al., 2008.

Table 4
Ordinary least squares (OLS) regressions by pairs of factors.

Factors	r^2	Std β	Ratio	Factors	r^2	Std β	Ratio	Factors	r^2	Std β	Ratio	Factors	r^2	Std β	Ratio
Rainfall_2y	0.129^{***}	−0.268	–												
Elev_cv		−0.182	0.68												
Rainfall_2y	0.105^{**}	−0.294	–	Elev_cv	0.061^{*}	−0.246	–								
Slope_cv		−0.090	0.30	Slope_cv		−0.001	0.004								
Rainfall_2y	0.127^{***}	−0.402	–	Elev_cv	0.061^{*}	−0.247	–	Slope_cv	0.022	−0.150	–				
Geology_w		−0.194	0.48	Geology_w		−0.018	0.07	Geology_w		0.014	–				
Rainfall_2y	0.113^{***}	−0.311	–	Elev_cv	0.097^{***}	−0.274	–	Slope_cv	0.042	−0.160	–	Geology_w	0.018	−0.033	–
Road_den		−0.127	0.40	Road_den		−0.174	0.63	Road_den		−0.143	–	Road_den		−0.136	–
Rainfall_2y	0.191^{***}	−0.227	0.71	Elev_cv	0.212^{***}	−0.262	0.67	Slope_cv	0.151^{***}	−0.089	0.24	Geology_w	0.144^{***}	−0.031	0.08
Road_dist		0.317	–	Road_dist		0.389	–	Road_dist		0.364	–	Road_dist		0.380	–
Rainfall_2y	0.169^{***}	−0.259	0.40	Elev_cv	0.173^{***}	−0.263	0.78	Slope_cv	0.133^{***}	−0.171	0.51	Geology_w	0.11^{**}	−0.078	0.23
City_dist		0.273	–	City_dist		0.336	–	City_dist		0.335	–	City_dist		0.339	–
Rainfall_2y	0.173^{***}	−0.194	0.64	Elev_cv	0.200^{***}	−0.242	0.64	Slope_cv	0.146^{***}	−0.071	0.19	Geology_w	0.157^{***}	−0.128	0.30
% Natural_cover		0.300	–	%Natural_cover		0.373	–	%Natural_cover		0.361	–	%Natural_cover		0.414	–

* $p < 0.05$; ** $p < 0.01$; *** $p < 0.001$

Factors	r^2	Std β	Ratio	Factors	r^2	Std β	Ratio	Factors	r^2	Std β	Ratio
Road_den	0.145^{***}	−0.046	0.12								
Road_dist		0.368	–								
Road_den	0.105^{**}	−0.026	0.80	Road_dist	0.182^{***}	0.300	–				
City_dist		0.314	–	City_dist		0.213	0.71				
Road_den	0.144^{***}	−0.051	0.14	Road_dist	0.193^{***}	0.258	–	Road_dist	0.205^{***}	0.258	0.79
% Natural_cover		0.365	–	%Natural_cover		0.254	0.98	%Natural_cover		0.325	–

* $p < 0.05$; ** $p < 0.01$; *** $p < 0.001$

dimensionality (e.g., multivariate analysis); however, our index performed slightly better than a multiple regression based on 7 predictor variables.

The production and deposition of fine sediments is a major threat to the useful life of a reservoir (Arias et al., 2011; Von Sperling, 2012), but its monitoring can be very expensive because of the need to build and operate monitoring stations (Santos et al., 2014). Recently the Brazilian government installed 57 monitoring stations in the Rio Doce Basin, at a cost of US$ 22,500 each and an estimated annual operating cost of US$ 10,000 per station. However, our assessment was much less expensive in terms of field and analytical costs (approximately US $ 2000 per site), but effective to relate stream-bed sediment state with natural and anthropogenic influences, as has been demonstrated elsewhere in Brazil also (Leal et al., 2016; Leitão et al., 2018), as well as in North America (Griffith et al., 2005; Hughes and Peck, 2008; Kaufmann and Hughes, 2006; Kaufmann et al., 2009; Rowe et al., 2009). We relied on visual stream sediment size classification, which can be very effective if conducted with standardized methods that employ a systematic spatial design (Conroy et al., 2016; Faustini and Kaufmann, 2007; Roper et al., 2010). Our method of field data collection required only simple gear and was quantitative, inexpensive, and relatively fast (about 5 person-hours total for hundreds of physical habitat variables), and has been applied as a standard method throughout the U.S.A (Faustini et al., 2009; USEPA - United States Environmental Protection Agency, 2016), as well as in many state-scale surveys in that country (e.g. Hubler et al., 2016; Rowe et al., 2009). The approach we used is adequate and appropriate for regional assessments (Bryce et al., 2010; Hughes and Peck, 2008; Lisle et al., 2015). The same field methods and gear were used in studies of 83 Amazon stream sites (Leal et al., 2017, 2016; Leitão et al., 2018) as well as 60 Alaskan wilderness sites (Amnis Opes Institute, 2017), further indicating their usefulness as tools for rational environmental management of water resources

Table 5
Relative weights of the study variables.

Factors	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)	Weights
(1) Rainfall mean 2 last years (mm)	–								0.137
(2) Elevation coefficient of variation (%)	1/3	–							0.108
(3) Slope coefficient of variation (%)	1/7	1/9	–						0.026
(4) Geology type (degree of fragility)	1/5	1/9	1	–					0.025
(5) Road density (km/km^2)	1/6	1/4	1	1	–				0.025
(6) Road proximity (km)	3	3	7	9	8	–			0.265
(7) City proximity (km)	1	2	5	7	9	1/3	–		0.146
(8) Vegetation cover (%)	4	4	8	7	8	1	2	–	0.268

Consistency ratio (CR): 0.049.

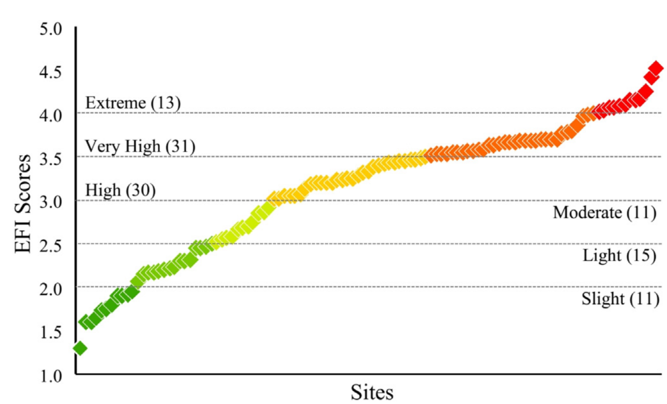

Fig. 3. Site cumulative distribution frequency and classification of environmental fragility index (EFI) scores across 111 catchments. Number of sites in parentheses.

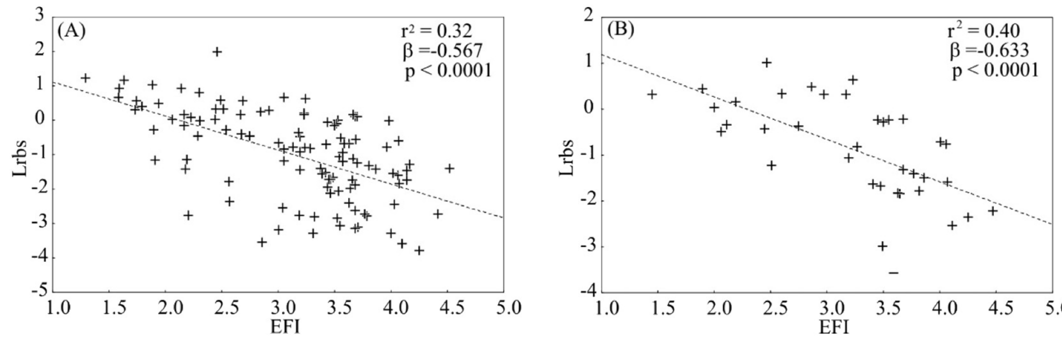

Fig. 4. Regression of the environmental fragility index (EFI) against the log of relative streambed stability (LRBS) for (A) 111 development sites and (B) 37 validation sites.

(Asmus et al., 2009; Hughes and Peck, 2008). We used standardized methods widely used in continental scale surveys to test a simple index developed for environmental assessments of river basins. This integrated approach (GIS index development and in situ tests) facilitated the identification of critical areas of the basin where fragile landscape attributes (e.g., erodible soils, steep slopes) coincide with of human disturbance activities (resulting in excess sediments, siltation and riparian deforestation, and reduced water quality). This knowledge aids decision makers in defining key areas for future rehabilitation or mitigation. The survey of environmental variables for extrapolating EFI scores was also low cost, mainly because of the increased availability of free spatial data, satellite images at different resolutions, and free GIS software (Nemec and Raudsepp-Hearne, 2012).

The EFI expansion to the four hydrologic units was possible because of the spatially-balanced probability sampling design that yielded a set of field sample sites that were well-distributed across gradients of geoclimatic controls and anthropogenic pressures (Olsen and Peck, 2008; Whittier et al., 2007). We strongly recommend that environmental assessments employ probability sampling so that they can infer results to larger areas with known confidence limits and reduce bias in those results. Such water body sampling has proven much more useful than ad hoc sampling in the USA (Hughes et al., 2000). Because of the impossibility of studying 100% of any area, it is necessary that sampling designs be efficient, unbiased, and spatially balanced (Larsen et al., 2008). The network used in this study was based on a spatially balanced sample design, using procedures developed by the USEPA in its National Aquatic Resources Surveys (Olsen and Peck, 2008). Similar probability sampling designs have also been applied successfully in southeast Brazil (Carvalho et al., 2017; Jiménez-Valencia et al., 2014; Silva et al., 2017) and in many states of the USA, including California (Mazor et al., 2016), Oregon (Anlauf et al., 2011), Maryland (Stranko et al., 2012), Washington (Washington State Department of Ecology, 2006), and Alaska (Amnis Opes Institute, 2017).

Table 6
Environmental fragility index (EFI) and multiple linear regression performances versus the log of relative bed stability (LRBS).

EFI vs LRBS (n = 111)	Std β	r^2	F-Ratio	AICc	ΔAICc
Constant	0***	0.322***	51.676	333.26	−5.614
EFI	−0.567***				
All variables (best subsets) vs LRBS (n = 111)					
Constant	0***	0.368***	8.585	338.88	
Rainfall_2y	−0.197*				
Elev_cv	−0.356***				
Slope_cv	0.22*				
Geology_w	−0.29**				
Road_dist	0.167				
City_dist	0.226**				
% Natural_cover	0.301**				

* $p < 0.5$.
** $p < 0.01$.
*** $p < 0.001$.

4.2. Environmental degradation and potential loss of ecosystem services

Our analyses showed that about 65% of the hydrologic units in the study area have high to extreme EFI scores (Figs. 2 and 5) and the most and least fragile areas are easily identified (Fig. 6). Studies in other regions have shown similar results (Toro et al., 2012; Xiaodan et al., 2010), and this highlights the importance of knowing the location of such areas to improve environmental management. Geoclimatic conditions and anthropogenic pressures within the four hydroelectric-dam hydrologic units we studied show a range that encompasses those in many other subtropical river basins. The neotropical savannah biome is currently one of the most threatened in the world, suffering from replacement of natural vegetation with agricultural uses such as crops and pastures (Diniz-Filho et al., 2008; Strassburg et al., 2017). Furthermore, modeling scenarios for the neotropical savanna project deforestation rates of 40,000 km^2 per decade until 2050 (Ferreira et al., 2013), and our results showed that vegetation cover, roads, and proximity to cities are the most important variables controlling the amount of excess fine sediment in streambeds and subsequently transported downstream to reservoirs. Sediment delivered to reservoirs as a result of projected trends of anthropogenic development in the neotropical savannah biome are a substantial threat to its reservoirs and the ecosystem services they support. Because most of the Brazilian hydroelectric matrix outside of Amazonia is located in this biome (Brazil, 2008), anthropogenic sedimentation is a threat to continued hydropower generation. The World Commission on Dams estimated that half the reservoir storage capacity worldwide will be lost to sedimentation in approximately 40 years (WCD, 2000), an outcome that may encourage the construction of new reservoirs to compensate for this loss. The continuation of the current drought regime in southeastern Brazil will further exacerbate lost capacity and decreases in river flow below existing dams.

In addition to generating electricity, reservoirs provide such ecosystem services as fisheries production, water supply, and recreation (Baron et al., 2002). The degradation of the basins upstream of the reservoirs diminishes those services and leads to economic losses. Because quantifying the monetary price of those services is a complex task (see de Groot et al., 2012), a specific study in the Brazilian Cerrado for this purpose is needed. Such a study could be used to compare the costs and benefits of preserving and rehabilitating areas upstream of the reservoirs against the costs and benefits of agricultural commodities having low added value and high dependence on foreign market prices susceptible to economic crises (O'Sullivan and Sheffrin, 2003).

4.3. Environmental management in hydroelectric reservoir basins

The EFI methodology we describe can be used to aid decision-makers to better spend financial and human resources for specific management actions, conserve natural resources, and consequently improve and conserve ecosystem services. Through our analytical approach, one can identify the co-varying natural landscape and anthropogenic

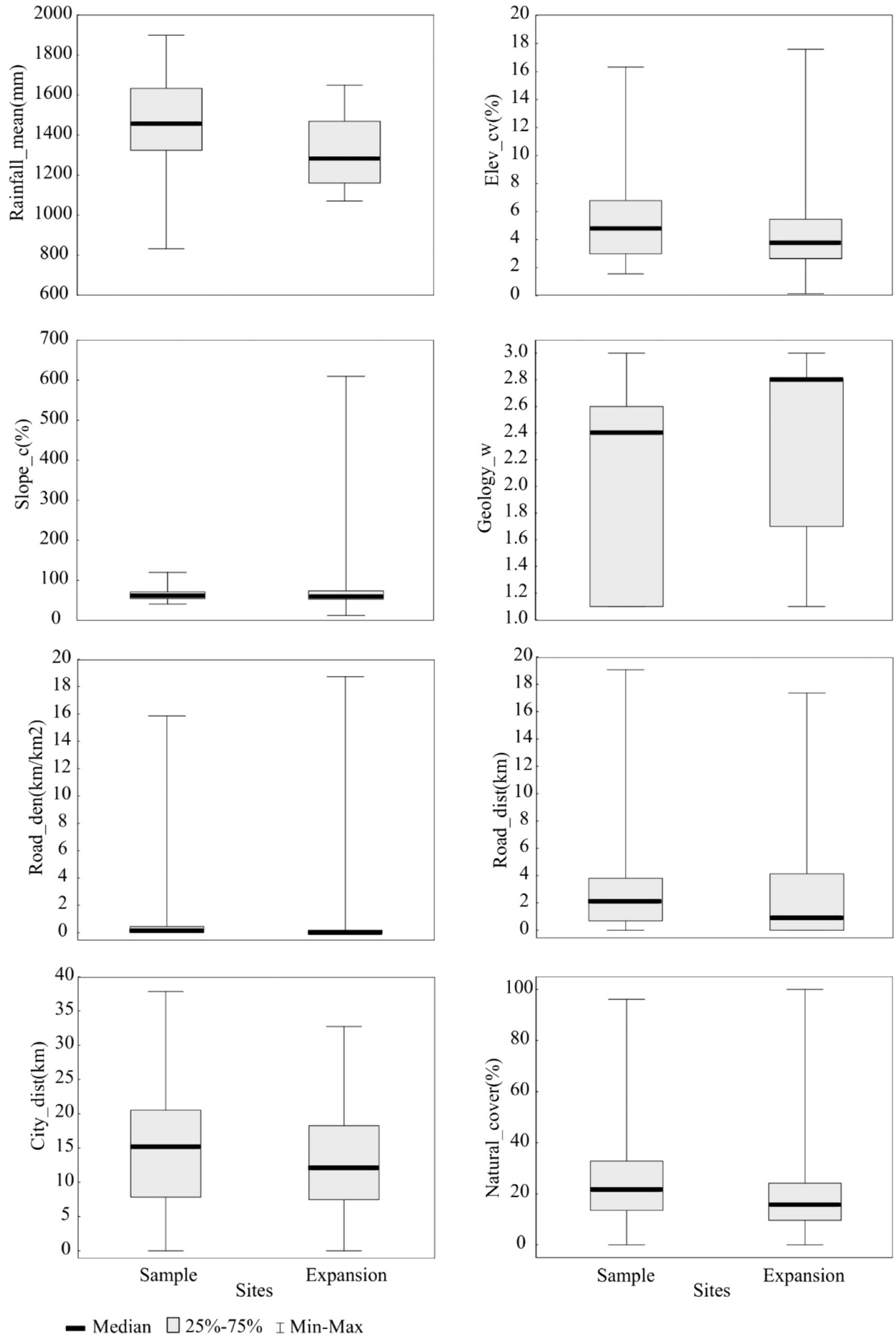

Fig. 5. Comparison of EFI component variable distributions in the 148 sample catchments vs. those in the level 7 Ottobasins used for expansion of EFI model results to the whole study area (Box plots show median, minimum, maximum, and interquartile range).

pressure variables that most strongly control fine sediment delivery to streams in reservoir drainage basins and thereby the usefulness of downstream reservoirs. Although many natural landscape variables (e.g., geology, soils) may not be greatly influenced by human actions (Macedo et al., 2014), one can identify areas where natural landscape characteristics make successful recovery and conservation actions most likely, allowing greater returns on limited financial investments. For example, our results were recently presented to the electrical company that funded much of this research, and to local basin committees and schools. Those results identified specific high-priority hydrologic

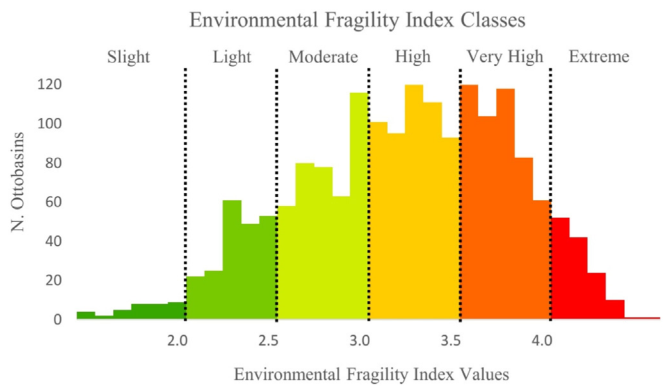

Fig. 6. Histogram of the expanded environmental fragility index (EFI) scores across the 1777 Ottobasins in four hydrologic units.

units for reforestation by managers, thereby illustrating the general importance of effective environmental assessment to adults and children.

Our results corroborate other research emphasizing the importance of maintaining natural vegetation to conserve soil and extend reservoir life in tropical regions and elsewhere (e.g., Arias et al., 2011; Chen et al., 2015; Liu et al., 2014). The analytical perspective of integrating covarying anthropogenic pressures and natural landscape factors in the context of river basins is increasingly applied in water body management (Magalhães Jr, 2007). This approach also has value for biodiversity conservation, because excess fine sediments are associated with biodiversity losses in streams (Bryce et al., 2010; Ferreira et al., 2014; Kaufmann et al., 2009) as well as lakes (Lenhardt et al., 2008; Molozzi et al., 2013). For example, Becker et al. (2016) found markedly higher proportions of non-native species and individuals in Volta Grande, and São Simão Reservoirs than in Três Marias Reservoir.

The current Brazilian legal framework regarding management of water resources recognizes that water is a limited natural resource with great economic value (Brazil, 1997). Therefore, conservation actions for water resources and their ecosystem services must better reflect this legal framework. The management instruments created by legislation (e.g. Brazilian Water Resources Policy (Brazil, 1997) and Environmental Plan of Conservation and Use of Artificial Reservoirs - PACUERA (Brazil, 2002b) can employ the analytical approaches we describe to integrate the interactions between social and geobiophysical conditions and better conserve those water resources.

Many other tropical countries have landscape and economic characteristics similar to those in Brazil regarding the abundance of surface freshwater resources and ecosystem services focused on generating hydroelectricity (WCD, 2000). Therefore, appropriate modifications of our approach are potentially applicable to such countries as India, Indonesia, Bangladesh, China, Democratic Republic of Congo, Kenya,

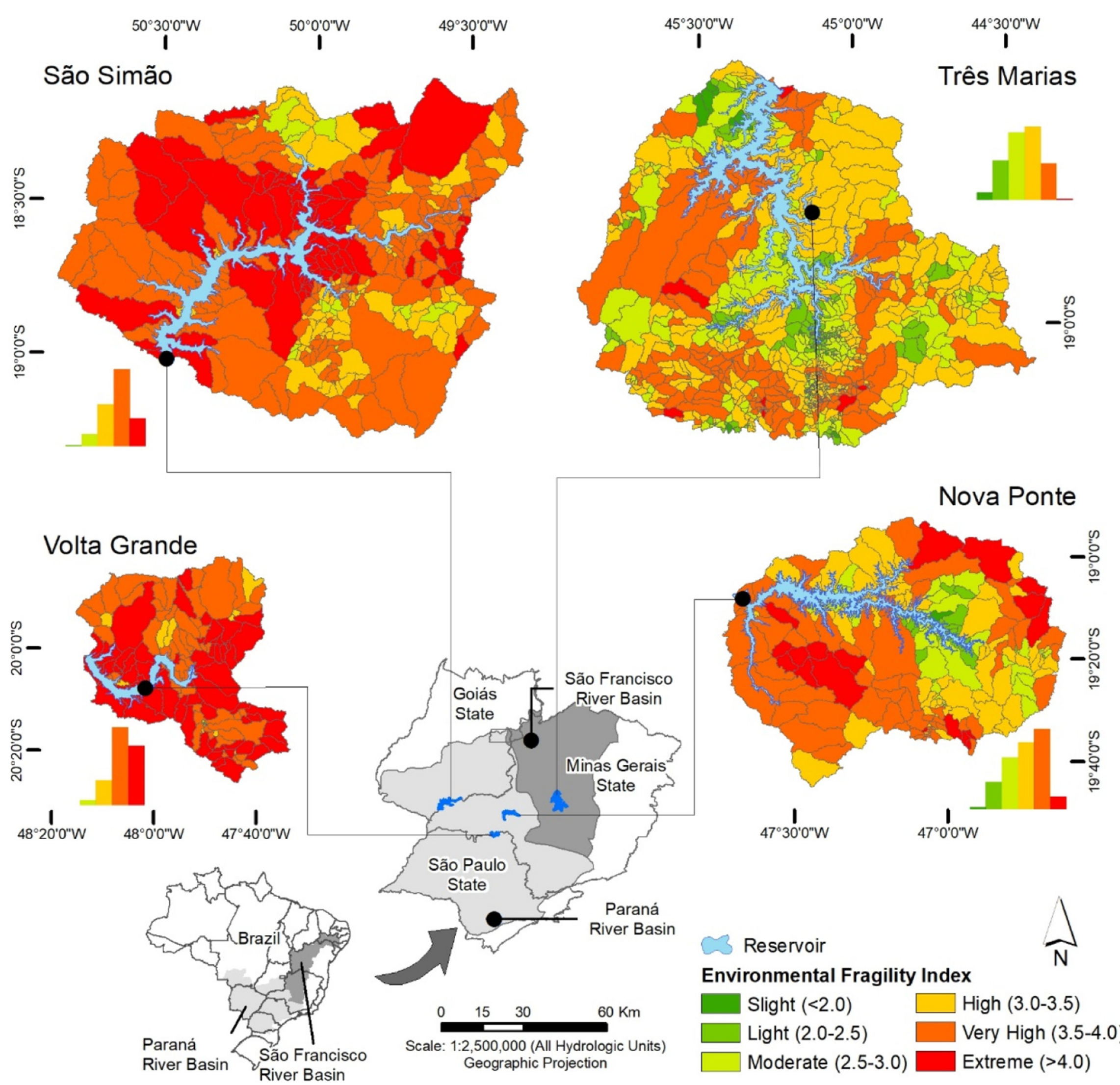

Fig. 7. Environmental fragility index (EFI) classes spatially distributed across 1777 Ottobasins in four hydrologic units. Histograms show EFI score distributions in each Hydrologic Unit.

and Tanzania, which along with Brazil account for over 50% of global freshwater supplies (Gleick, 1998). In this context, India and China stand out, because together they own over half of the largest dams in the world (WCD, 2000). In other words, appropriate modifications of our approach may offer a cost-effective methodology for extending the reservoir lives upon which more than two billion people depend.

4.4. Potential uses and limitations of the methodology

The EFI has many potential applications for neotropical savanna watersheds. The 37-site validation dataset indicated similar performance as the EFI developed using 111 sites, and it opens opportunities to apply the EFI in other reservoir basins, especially in the Brazilian neotropical savanna. If properly modified, it may also offer a methodological approach for better managing freshwater resources in other parts of the world. Appropriately modified environmental fragility indices tailored to other countries have potential because decision methods like the Analytical Hierarchy Process we used are a very useful and convenient tool to support and refine pairs comparisons (Sahoo et al., 2016). Wang et al. (2008) and Xiaodan et al. (2010) both evaluated eco-environmental vulnerability of the Tibetan Plateau using the same AHP development procedures and almost the same types of GIS data as we used in developing the EFI, but their data pairs comparisons differed from ours in four ways: 1) We used linear regressions by pairs as independent variables, whereas the other studies used only the authors' subjective evaluations. 2) We validated the EFI by examining an impact response (LRBS) as the dependent variable, whereas the other studies did not test their indices. 3) We used an independent validation data set; and 4) we made a model-based inference to the entire study area by applying EFI values toto 1777 Ottobasins (hydrologic units). In our study, we used a local site metric (LRBS) to test the response to EFI, but one could use other proxy variables in place of LRBS as indicators of excess sedimentation or environmental stress. Watersheds and hydrologic units similar to the Ottobasins we applied in our study have been used for environmental management in continental areas, such as the USA (Hill et al., 2017; Omernik et al., 2017; Thornbrugh et al., 2018).

The GIS-based EFI approach we describe can be used to rank and group watersheds in priority management groups (e.g., Herrmann et al., 2011; Hill et al., 2017; Thornbrugh et al., 2018) or for selecting potential areas upon which to focus monitoring and assessment. However, there are several limitations to our approach. Sediment indices like the LRBS are better suited for determining regional patterns than monitoring a single river, because of the high covariance of possible predictors that drive it. In general, direct relationships between land use and sedimentation are weak, because of co-varying geoclimatic controls (Guerra et al., 2017; Leal et al., 2017; Lisle et al., 2015). For example, if the EFI underestimates the LRBS, the assessment is conservative, making it more protective, but also unnecessarily expensive for hydropower operators. If the EFI overestimates the LRBS, then areas that require better soil conservation will be missed, potentially affecting both environmental rehabilitation and the long-term commercial viability of the reservoir. Another limitation of the EFI approach is the quality of available GIS databases. Despite recent improvements in digital maps in Brazil that resulted from the efforts of government agencies (e.g. Brazilian Agricultural Research Corporation, Brazilian Institute of Geography and Statistics, Brazilian National Water Agency), most national digital maps are large scale (1: 250,000 or greater), dated (produced before the 1980s), or differ in quality across the nation. Other countries with more accurate geospatial databases should be better able to realize the potential of our approach. For example, the USA has high quality digital maps (e.g. TIGER census database; USDA soil assessment; USGS geological maps, etc.) and also conducts national environmental assessment field surveys that include measures of LRBS for the entire country (Faustini et al., 2009).

Limitations in the accuracy of both the sediment index and the geospatial data likely contribute to the unexplained variation in the prediction of LRBS from the EFI (R^2~30–40%). More specifically, the prediction of excess sediments in our study and their potential application to other regions are constrained by a number of factors. 1) It is very difficult to explain most site-scale streambed sedimentation variability across large landscapes even with relatively large data sets. 2) The predictor-response relationships between landscape variables and stream responses likely differ by geographic location (Macedo et al., 2014), and by differing spatial patterns in drainages within hydrologic units and across differing ecoregions (Omernik et al., 2017). 3) The quality of the GIS database affects the amount of variance in LRBS that can be explained. Nonetheless, despite the moderate predictive capacity of our index, we believe our results make a useful contribution towards understanding how natural landscape patterns and anthropogenic pressures influence streambed sedimentation across large river basins.

5. Conclusions

Our results show that a methodological approach based on local-scale field measurements of excess sediment in streams and catchment-scale landscape data (satellite images, geology maps, terrain model, etc.) is efficient and cost-effective for assessing environmental fragility in streams, reservoir basins, and reservoirs. Furthermore, the resulting index (EFI) can be used to predict which water bodies are most likely to be adversely affected by future landscape erosion and water body sedimentation. Our development and application of the EFI approach in Brazil, where freshwater resources and their ecosystem services are large but threatened by anthropogenic pressures, suggests the potential for cost-effective applications in developing nations and other tropical areas.

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