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Mapping watershed integrity for the conterminous United States

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Abstract

Watershed integrity is the capacity of a watershed to support and maintain the full range of ecological processes and functions essential to sustainability. Using information from EPA's StreamCat dataset, we calculated and mapped an Index of Watershed Integrity (IWI) for 2.6 million watersheds in the conterminous US with first-order approximations of relationships between stressors and six watershed functions: hydrologic regulation, regulation of water chemistry, sediment regulation, hydrologic connectivity, temperature regulation, and habitat provision. Results show high integrity in the western US, intermediate integrity in the southern and eastern US, and the lowest integrity in the temperate plains and lower Mississippi Valley. Correlation between the six functional components was high ($r = 0.85-0.98$). A related Index of Catchment Integrity (ICI) was developed using local drainages of individual stream segments (i.e., excluding upstream information). We evaluated the ability of the IWI and ICI to predict six continuous site-level indicators with regression analyses – three biological indicators and principal components derived from water quality, habitat, and combined water quality and habitat variables – using data from EPA's National Rivers and Streams Assessment. Relationships were highly significant, but the IWI only accounted for 1-12% of the variation in the four biological and habitat variables. The IWI accounted for over 25% of the variation in the water quality and combined principal components nationally, and 32-39% in the Northern and Southern Appalachians. We also used multinomial logistic regression to compare the IWI with the categorical forms of the three biological indicators. Results were consistent: we found positive associations but modest results. We compared how the IWI and ICI predicted the water quality PC relative to agricultural and urban land use. The IWI or ICI are the best predictors of the water quality PC for the CONUS and six of the nine ecoregions, but they only perform marginally better than agriculture in most instances. However, results suggest that agriculture would not be appropriate in all parts of the country, and the index is meant to be responsive to all stressors. The

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IWI in its present form (available through the StreamCat website; <https://www.epa.gov/national-aquatic-resource-surveys/streamcat>) could be useful for management efforts at multiple scales, especially when combined with information on site condition. The IWI could be improved by incorporating empirical or literature-derived relationships between functional components and stressors. However, limitations concerning the absence of data for certain stressors should be considered.

Keywords

Catchment integrity; Sustainable watershed management; Anthropogenic stressors; IWI; ICI; NRSA; StreamCat; Water quality

1. Introduction

Watersheds encompass all biotic and abiotic components – including people – within their boundaries (sensu Likens, 1992), and provide a range of services valued by society (Costanza et al., 1997; Kumar, 2010; Costanza et al., 2014). These include supporting services (e.g., soil formation, nutrients, and primary production), provisioning goods and services (e.g., food, water, wood, fiber, and fuel), regulating services (e.g., climate regulation, flood regulation, and water purification), and cultural services, such as recreation and spiritual activities (de Groot et al., 2002; Millennium Ecosystem Assessment, 2005). Thus, watersheds are a foundation of our cultural, economic, spiritual, and social well-being (Likens et al., 2009) and, as such, should be a critical focus of water resource management. However, the quality and quantity of services generated by watersheds are rapidly declining (Farber et al., 2002) because of accelerating rates of land-use change, water consumption, and climate change inter alia. Some of the greatest threats to freshwater ecosystems globally are modifications caused by anthropogenic activities on aquatic ecosystems and their watersheds (Allan, 2004; Dudgeon et al., 2006; Vörösmarty et al., 2010). The deteriorating state of global freshwater biodiversity reflects these impacts (Dudgeon et al., 2006; Butchart et al., 2010).

Evaluating the overall condition of a watershed represents a societal and a scientific challenge, because watersheds are used and managed for a diverse, interrelated range of activities. Many of these services are directly related to the flow and function of water and its constituents within watersheds. Governments recognize the stress that humans can place on the finite natural resources of watersheds and have progressively implemented policies intended to assure greater sustainability of water-dependent services. Early policies focused on specific impacts (e.g., point-source pollutants), the over-use of certain resources, protection, or habitat restoration (U.S. EPA, 1996). However, these early efforts often failed to address chronic problems that contribute to longer-term declines in the structure and function of watersheds; for example, pollutants associated with non-point runoff from urbanized and agricultural areas. Moreover, the focus on individual resources or habitats usually fails to recognize watersheds as complete, intra-connected systems. The advent of watershed initiatives in the 1990s brought more holistic and interdisciplinary approaches to watershed management (Mitchell, 1990; Cairns and Crawford, 1991; Noss, 1995; Karr,

1996; Kenney, 1997; Hull et al., 2003). These initiatives also raised awareness of the need for methods that could be used to assess the condition, or integrity, of systems at the larger watershed scale.

Existing definitions of ‘integrity’ for river ecosystems have been developed for assessments at the reach or site scale and usually for particular species, such as fish or macroinvertebrates. These scales, however, are inappropriate for defining integrity at the watershed scale. Further, existing assessments of endpoints do not indicate the source of impairment. Lately, the use of the term ‘watershed integrity’ to identify and describe desirable watershed management endpoints has increased (U.S. EPA, 1998; Novotny and Smith, 2004; U.S. EPA, 2012b; Van Abs, 2013). However, ambiguity exists as to what this term actually means. More recently, Flotemersch et al. (2016) defined watershed integrity as the capacity of a watershed to support and maintain the full range of ecological processes and functions essential to the sustainability of biodiversity and of the watershed resources and services provided to society. However, definitions alone do not support the quantification of differences in watershed characteristics between areas, measurement of changes in state over time, or specification of the level at which efforts could be made to preserve or restore integrity (Noss, 1995). Flotemersch et al. (2016) defined an operational Index of Watershed Integrity that used a human health analogy to identify six key watershed functions: (1) hydrologic regulation (HYD), (2) regulation of water chemistry (CHEM), (3) sediment regulation (SED), (4) hydrologic connectivity (CONN), (5) temperature regulation (TEMP), and (6) habitat provision (HABT) (Table 1 and Fig. 1). Furthermore, they identified specific risk factors, or stressors, which impact these six key functions. An advantage of this approach is that the index can be readily deconstructed to identify factors influencing index scores, thereby directly supporting the strategic adaptive management of individual components that contribute to watershed integrity. Moreover, the approach can be iteratively applied and improved as new data and information become available.

Data resulting from a national-scale mapping and assessment of watershed integrity is of value to states and watershed organizations initiating healthy watershed programs (U.S. EPA, 2012a). These programs augment the watershed approach with proactive, holistic aquatic ecosystem conservation and protection designed to conserve critical components of watersheds and, therefore, avoid additional water quality impairments in the future. A national evaluation of watershed integrity also helps identify watersheds with high integrity, as well as those at risk, and help to prioritize conservation and restoration. It will also provide high quality information at a broad spatial scale that will allow ecosystem management to focus on achievable and measureable outcomes underpinned by quality science.

The goal for this research was to calculate an Index of Watershed Integrity (IWI) and map its distribution for the conterminous United States (CONUS), following the conceptual model and definition of watershed integrity by Flotemersch et al. (2016). Our specific objectives were to (1) develop a national map of IWI based on the approach by Flotemersch et al. (2016), (2) develop an associated national map of the Index of Catchment Integrity (ICI) and compare it to the IWI map; and (3) evaluate the performance of the IWI by comparing IWI

results to data obtained from the U.S. Environmental Protection Agency's (EPA) National Rivers and Streams Assessment.

2. Methods

2.1. Index of Watershed Integrity

2.1.1. Index derivation—Flotemersch et al. (2016) suggested that their formal definition of watershed integrity could provide a management tool that addressed whether existing watershed infrastructure supported sustainability goals. However, Flotemersch et al. (2016) understood that such a formal definition, by itself, would not allow for quantitative comparisons of watershed characteristics or changes over time. Instead, they recognized the need for an operational definition that would (1) identify key elements and processes necessary for maintaining production of watershed services, and (2) be capable of evaluating the presence of these key elements using existing datasets that are broadly available. One approach for developing such an assessment would be to compare watershed characteristics to unaltered (i.e., reference) watersheds. However, since all watersheds are altered to some degree – and in many cases significantly – reference condition represents least degraded condition (Stoddard et al., 2006), rather than unaltered condition. As a result, Flotemersch et al. (2016) proposed a conceptual approach that did not require reference watersheds. The approach proposed by Flotemersch et al. (2016) was borrowed from the human health field and identified key functions that watersheds provide and risk factors (or stressors) that have been shown to interfere with these functions. This approach is similar to identifying cardiopulmonary function as a key component of human health and high cholesterol as a risk factor associated with heart disease. Given the six functional components of watersheds, Flotemersch et al. (2016) defined an Index of Watershed Integrity as follows:

$$IWI = IWI_{HYD} \times IWI_{CHEM} \times IWI_{SED} \times IWI_{CONN} \times IWI_{TEMP} \times IWI_{HABT} \quad (1)$$

where IWI represents an Index of Watershed Integrity that ranges from 0 to 1, with higher values having greater integrity, and IWI_i represents the i th functional component. The IWI was defined as a product of the six functional components because each is a critical element of watershed integrity. Therefore, the components are not substitutable; a complete loss of any one component will cause watershed integrity to decline to zero. The integrity of each of these components depends on multiple stressors that can impact their functionality. Given that, $IWI_{i,w}$ is defined as:

$$IWI_{i,w} = \frac{\sum_{j=1}^{n_{i,w}} g_{i,j} \left(\frac{s_{j,w}}{s_{j,max}} \right)}{n_{i,w}} \text{ given } s_k \text{ for all } k \neq j \quad (2)$$

where $IWI_{i,w}$ is the integrity of the i th functional component in watershed w ; $s_{j,w}$ is the observed value of stressor j in watershed w ; $s_{j,max}$ is the maximum value of stressor j across the conterminous US (with $s_{j,w}/s_{j,max}$ ranging from 0 for unaltered to 1 for highest impact); $n_{i,w}$ is the number of stressors affecting function i in watershed w ; and $g_{i,j}$ is a single-

variable mathematical function that describes the relationship between function i and stressor j and that incorporates the conditional effects of all other stressors s_k on the response of IWI_i to s_j . The $g_{i,j}$ function results in a number that ranges from 0 to 1. Examples of $g_{i,j}$ would be a negative linear response, a threshold (tipping point) response, and low or high resiliency responses (Flotemersch et al., 2016). Note that in Eq. 2 $s_{j,w}$ and $s_{j,max}$ represent values at the watershed-scale. In addition, Flotemersch et al. (2016) neglected to include the $n_{i,w}$ divisor in Eq. 2 and the following equation; including it is necessary to ensure that $IWI_{i,w}$ has a 0-1 range.

Recognizing that the specific relationships between function and stressors are largely unknown, Flotemersch et al. (2016) defined an estimator of IWI_i that could serve as a first-order approximation in developing a national-level IWI. This estimator assumed the simplest case scenario where the effects of all human-related stressors on each function were assumed to be a negative linear response, with all stressors treated equally. In the simplest case, where a watershed is subject to multiple stressors j that impact function i , the integrity of the watershed with respect to that function is estimated by

$$\hat{IWI}_{i,w} = \frac{\sum_{j=1}^{n_{i,w}} \left[1 - \left(\frac{s_{j,w}}{s_{j,max}} \right) \right]}{n_{i,w}} \quad (3)$$

where $\hat{IWI}_{i,w}$ is the first-order estimator of $IWI_{i,w}$. Below we describe how this estimator is used to map the IWI for the CONUS.

2.1.2. Index mapping—Flotemersch et al. (2016) proposed stressors and data sources for the calculation of an IWI (Table 1). For our application of mapping the IWI, these human-related landscape stressors were taken from EPA's StreamCat dataset (Hill et al., 2016). StreamCat was built on the framework of the National Hydrography Dataset Plus Version 2 (NHDPlusV2; McKay et al., 2012). NHDPlusV2 is a 1:100,000 national digital network containing more than 2.6 million stream segments in the CONUS; it provides a geospatial framework for integrating landscape data with streams and their associated catchments (McKay et al., 2012; Hill et al., 2016; Moore and Dewald, 2016). StreamCat provides tabular summaries for over 400 anthropogenic and natural landscape variables for all stream segments in the NHDPlusV2 at four spatial scales (Hill et al., 2016): catchments, riparian catchments, watersheds, and riparian watersheds. Here we define *catchment* as the local drainage for an individual stream segment, excluding upstream contributions, and a *watershed* as all of the area draining to a particular stream segment, which consists of the local catchment plus all upstream catchments (McKay et al., 2012; Hill et al., 2016).

The appropriate tabular watershed and riparian watershed values from StreamCat for each of the stressors (Table 1) were used to represent the various $s_{j,w}$ values in Eq. 3. The values for $s_{j,max}$ were defined by Flotemersch et al. (2016) as 100 for stressors that are based on a proportion, like percent urban land cover. For other stressors that have no theoretical limit on their upper bound (e.g., watershed road density), we used a national winsorized (Gilbert, 1987) $s_{j,max}$ value based on the 99.99th percentile. This limited extreme values in the stressor

data to reduce the effect of possibly spurious outliers and allowed for a standardization between zero and one for all stressors in the IWI calculation. Note that the number of stressors included for a particular functional component (n_i) was not constant across all watersheds, since certain stressors were not defined for all watersheds. For example, stressors defined at the riparian scale, such as density of roads in watershed riparian areas (RdDensWsRp100) and percent agricultural land use in watershed riparian areas (PctAg2006WsRp100), are undefined in watersheds that lack riparian areas. This occurs for NHD catchments that do not contain a stream segment (McKay et al., 2012). In those cases, n_i was adjusted accordingly for the watersheds in question.

The stressors originally proposed by Flotemersch et al. (2016; Table 1) included a number of variables within individual functional components (IWI_i) that were correlated. We eliminated highly correlated stressors (correlation coefficients $r > 0.7$) within each of the six functional components (Table S1) so as to avoid giving those variables greater weight. For stressors that were correlated, we retained the variable that we considered to best fit the hypothesized effects for a specific functional component, and dropped the others. For example, density of roads within watershed riparian zones had a correlation > 0.7 with percent urban land use and percent imperviousness in the watershed (Table S1). Only the former was used in the habitat provision component (IWI_{HABT}), because we reasoned that riparian stressors should have more of an effect on local habitat than watershed stressors. Note that the issue of multiple weighting was limited to correlations within an individual functional component. The same variable or a correlated variable could occur as a stressor within multiple functional components if that stressor was considered to have different functional effects. For example, urban land use could impact the habitat provision function through direct loss of habitat via housing and roads, and the hydrologic regulation function through the presence of impervious surface. Since these are different functional effects, variables correlated with urban land use were included with both of these functional components.

To allow for visual inspection of spatial patterns, we produced maps showing the distribution of the IWI within the CONUS. However, watersheds of consecutive stream segments are nested and overlap, so it is not possible to map all watersheds simultaneously. For example, if the IWI for the watershed associated with the mouth of the Mississippi River were mapped, it would obscure all of the component sub-watersheds. We therefore took two approaches to mapping IWI. First, we separately mapped IWI for all watersheds associated with streams of a certain Strahler order (Strahler, 1957); e.g., we produced separate IWI maps for watersheds associated with first and seventh-order streams. However, consecutive stream segments can have the same stream order; for example, a second-order stream occurs below the confluence of a first and second-order stream. Including consecutive watersheds with the same stream order would also result in overlapping watersheds. To avoid this, we only include the furthest downstream watersheds of a particular order. Specifically, a watershed of order n is included only if the order of the adjoining downstream watershed is greater than n . While this approach does not allow all IWI values to be displayed simultaneously, the selected values are mapped onto actual watersheds. Alternatively, we also map IWI values onto the catchment of the particular stream segment that a watershed is associated with. For example, the IWI for the watershed associated with the mouth of the

Mississippi River is mapped onto the catchment at the mouth of the Mississippi. This approach displays all IWI values simultaneously, but the values are not mapped onto watersheds (except in cases of headwater catchments, where the catchment and watershed are equivalent). Such a map is similar in concept to a one-directional moving window, where the value mapped onto a catchment incorporates all upstream influences.

We calculated summary statistics (e.g., minimum, maximum, mean, standard deviation) of the IWI for the CONUS, nine aggregated Omernik Level III ecoregions (U.S. EPA, 2013; hereafter ecoregions), and each state. In addition, we tested for significant differences ($p = 0.05$) in pairwise comparisons of mean IWI values at the state and ecoregion level using the *TukeyHSD* function in R (R Core Team, 2014). While such tests assume normality, they can be used for non-normal distributions in situations like ours with large n (Hogg and Craig, 1995). The ecoregion and state of a watershed were determined based on the terminal stream segment, since each watershed is defined relative to a particular stream segment. Here we used the ecoregion and state in which the constrained centroid of the catchment containing the terminal stream segment was located. The constrained centroid is the same as the centroid if it is situated within the catchment boundary. However, if the centroid occurs outside of the boundary (which can happen, for example, with crescent-shaped catchments), the constrained centroid is forced to occur within the catchment. We also experimented with other ways of determining ecoregion and state of a watershed (for example, assigning a watershed to a region if any or all of it occurred within the region), but found that results (not shown) were insensitive to this.

2.2. Index of Catchment Integrity

Watersheds are integrative and include all of the water draining to a particular stream segment. Thus they are the proper unit to represent the domain of stressors that could possibly affect stream segments that receive water from upstream catchments (i.e., non-headwater catchments). However, there are certain applications where it is desirable to limit this domain to the catchment scale. For example, consider a small stream that does not appear on the 1:100,000 scale NHDPlusV2 maps and which originates in a catchment and merges with the mainstem stream segment. Such a stream is influenced by the local characteristics of the catchment, not the watershed. Given such applications, we also developed the following Index of Catchment Integrity, along with its estimator:

$$ICI_{i,c} = \frac{\sum_{j=1}^{n_{i,c}} g_{i,j} \left(\frac{s_{j,c}}{s_{j,max}} \right)}{n_{i,c}} \text{ given } s_k \text{ for all } k \neq j \quad (4)$$

$$\hat{ICI}_{i,c} = \frac{\sum_{j=1}^{n_{i,c}} \left[1 - \left(\frac{s_{j,c}}{s_{j,max}} \right) \right]}{n_{i,c}} \quad (5)$$

where $ICI_{i,c}$ and $\hat{ICI}_{i,c}$ represent the Index of Catchment Integrity and its estimator, respectively, for the i th functional component and catchment c , and $s_{j,c}$ is the observed value for stressor j in catchment c . While $s_{j,max}$ represents maximum CONUS values at the watershed-scale for the IWI (Eqs. 2 and 3), for the ICI it represents maximum CONUS catchment-scale stressor values (Eqs. 4 and 5). These are evaluated in $\hat{ICI}_{i,c}$ using the appropriate tabular catchment and riparian catchment values from StreamCat. Since catchments are not nested, there is no issue of overlapping, and all ICI values can be displayed by mapping them onto the associated catchments. In addition to mapping ICI values, we also evaluated the ICI by producing basic statistics by CONUS, ecoregion, and state.

2.3. Evaluation of IWI and ICI indices

Watershed integrity is a concept – not a physical property that can be measured. Thus, it is not possible to directly validate the performance of our index. However, we hypothesized that the condition of a stream segment would be related to the integrity of the watershed and/or catchment containing that segment. Given that, we compared values of the IWI and ICI with six site-scale indicators from the EPA's National Rivers and Streams Assessment (NRSA). NRSA is a national survey that uses a spatially-balanced sample of perennial streams to develop national and regional estimates of stream condition (U.S. EPA, 2016). We considered three indicators of biological condition, based on the 2008–2009 NRSA survey: benthic macroinvertebrate multi-metric index (MMI), macroinvertebrate observed-to-expected (O/E) ratio, and fish MMI (U.S. EPA, 2016). We also included three metrics representing water quality and physical habitat.

NRSA developed MMIs to assess biological condition of perennial streams based on 2008–2009 benthic macroinvertebrate and fish assemblage data (Stoddard et al., 2008; U.S. EPA, 2016). An MMI is a sum of scores from a number of individual metrics that summarize assemblage attributes, such as taxonomic composition, tolerance to disturbance, or habitat preference. The particular metrics included in an MMI were selected based on their ability to distinguish between least disturbed and highly disturbed sites within a region (Stoddard et al., 2008; U.S. EPA, 2016). For the 2008–2009 NRSA a separate invertebrate MMI was developed for each of the nine ecoregions. NRSA developed fish MMI values based on three major climatic regions that were aggregations of the nine ecoregions (see Fig. 10.1 of U.S. EPA, 2016). For purposes of reporting, NRSA used categorical versions of the MMIs. These categories were developed using regional thresholds to define good, fair, and poor condition classes for both benthic macroinvertebrates and fish (Herlihy et al., 2008). However, our analysis used the continuous values of both MMIs that were calculated prior to classification.

Another index of biological condition developed by NRSA is the macroinvertebrate O/E ratio. This compares the observed number of taxa (O) with the expected taxonomic composition (E) of macroinvertebrate assemblages at a site under least disturbed condition. Expected values are calculated using three regional models that estimate reference condition based on natural predictor variables (e.g., elevation and stream size; Hawkins et al., 2000;

U.S. EPA, 2016). The O/E ratio represents the relative proportion of reference taxa found at a site; e.g., an O/E ratio of 0.9 means that 90% of the expected reference taxa were observed. Note that the O/E ratio represents one specific aspect of biological condition (biodiversity loss), while the MMIs combine multiple characteristics (Hawkins et al., 2000; U.S. EPA, 2016). NRSA reported on O/E results by categorizing sites into four classes of macroinvertebrate taxonomic loss: < 10%, 10-20%, 20-50%, and > 50% taxa loss. For this analysis, however, we use the continuous version of the O/E ratio.

We developed metrics to represent water quality and physical habitat at individual sites based on the synthetic disturbance variable approach of Whittier et al. (2007) and Whittier and Van Sickle (2010). The water quality metric was based on total nitrogen ($\mu\text{g/L}$), total phosphorus ($\mu\text{g/L}$), and turbidity (NTU), which were collected from a single grab sample at each NRSA site (U.S. EPA, 2007, 2008). We log10 transformed the variables, after first adding one to each total nitrogen and total phosphorus observation because of the presence of zero values. The transformed variables were then input into a principal components (PC) analysis (SAS® version 9.4, PROC FACTOR; SAS Institute Inc., 2014) using the correlation matrix (so there was no need to standardize the variables to adjust for different measurement scales). The first principal component was then used as our water quality metric.

The habitat metric was constructed using the same approach with the following three variables: presence of fine sediments plus hardpan (measured as a percentage of presence at sampling points along the thalweg of each stream site), the areal cover of riparian woody vegetation in the ground and understory vegetation layers (measured as the mean percent cover observed within 22 riparian plots at each site), and the number of different types of human disturbance (e.g., roads, pipes, riprap, crops, grazing) visible within and adjacent to the riparian zone (measured as the mean number of types observed at 22 riparian plots at each site). These variables represent excess sedimentation, vegetation maturity and complexity, and instream and riparian disturbance, respectively (Whittier et al., 2007; Whittier and Van Sickle, 2010). Kaufmann et al. (1999) provides additional details regarding these physical habitat variables. The habitat metric was then defined based on the first principal component of the three log10 transformed variables. Finally, a third metric was developed using the first principal component from all six transformed water quality and habitat variables combined. This combined metric represented a more integrated measure of aquatic condition, compared to the specific focus of the other site metrics.

To examine the relationship between the six site metrics and the IWI and ICI indices, we determined the watershed and catchment containing each of the NRSA sampling sites, respectively. We then associated the IWI/ICI value for that watershed/catchment with the NRSA sampling site. For each of the six metrics, we calculated summary statistics by CONUS, ecoregion, and state using the R statistical package (R Core Team, 2014). In addition, we conducted simple regression analyses using the *lm* function in R, with the individual site metrics as response variables and the IWI or ICI as independent variables by both CONUS and ecoregion. In addition to the composite IWI, we conducted the same analyses using each of the six functional components of the IWI as independent variables. Where results were significant, we examined adjusted R^2 values to determine strength of relationships.

We also performed a separate analysis using the categorical versions of the three indicators of biological condition. We separately modeled these categorical variables as a function of IWI or ICI using multinomial logistic regression with the *multinom* function in the *nnet* R package (Venables and Ripley, 2002). We tested for model significance using a likelihood ratio test with the *lrtest* function in the *lrmtest* R package (Zeileis and Hothorn, 2002), and also calculated the predictive accuracy of the models.

Finally, we conducted a regression analysis for the continuous response variable having the best fit at the CONUS-scale, with agriculture and urban land use in the watershed and catchment as independent variables. These were compared with results using the IWI and ICI as independent variables to assess whether IWI and ICI added additional information as predictors over agriculture and urban land use.

3. Results

3.1. Index of Watershed Integrity

IWI within first-order streams (Fig. 2a) shows a large portion of the country having high integrity, particularly in the west and the upper northeast. Large areas of lower integrity are found in the temperate plains and in the lower Mississippi Valley. Because it maps much larger watersheds, the IWI map for the furthest downstream seventh-order streams (Fig. 2b) has a blockier pattern that lacks the smoother transitions observed with the first-order stream map (see supplementary materials for high resolution versions of these two maps; Figs. S1–S2). In many areas, the IWI values of the seventh-order watersheds are similar to their component first-order watersheds. For example, IWI values are mostly all high for both first and seventh-order watersheds in much of Arizona and New Mexico. However, there are also differences in mapped values. For example, much of West Virginia has high integrity first-order watersheds, but the map shows a seventh-order watershed with more moderate integrity. The IWI of the larger watershed reflects the IWI values of its component sub-watersheds, which suggests large heterogeneities occur across sub-watersheds of this seventh-order watershed.

Trends in the IWI map are similar to those of the first-order watersheds, with high integrity values in the southwest and western mountain states, lower values in the midwestern US and lower Mississippi Valley, and more moderate values throughout the southeast and into the northeast (Fig. 3; see Fig. S3 for a high resolution version of this map). While the map is dominated by spatially extensive patterns and broad, homogeneous areas, there are also spatially intensive patterns (e.g., hotspots of low integrity in New York City and the Central Valley of California) and areas with fairly high heterogeneity, such as the eastern portion of Texas, southern Florida, and the eastern portion of the mid-Atlantic states.

For the CONUS, the median IWI value is 0.79 (SD = 0.19), with a minimum of 0.05 and a maximum of 1.00 (Table S2). The distribution of CONUS IWI values indicates that they are left-skewed, with the tail having a fairly constant frequency for values between 0.4 and 0.85 (Fig. 4). IWI values of 0.94–0.99 are most common, while values lower than 0.39 are rare. Median IWI values varied substantially by ecoregion (Fig. 5, Table S2), with the Xeric ecoregion having the highest watershed integrity (median = 0.95, SD = 0.15), and the

Temperate Plains having the lowest (median = 0.46, SD = 0.12). Pairwise comparisons of mean ecoregion IWI values indicated that all means were significantly different ($p < 0.05$). However, this was due to the large n , which ranged from 124,000 to 437,000 for the nine ecoregions. The largest variation in the IWI occurred in the Upper Midwest (SD = 0.18), while the smallest was found in the Western Mountains (SD = 0.08). IWI values also varied by state (Fig. 6, Table S2), with Nevada having the highest watershed integrity (median = 0.97, SD = 0.14), while Iowa had the lowest (median = 0.42, SD = 0.08). Over 97% of the pairwise comparisons between state IWI means were significantly different ($p < 0.05$). This was also due in part to large n , which ranged from 1,700 to 140,000. Thirty-two pairwise state comparisons had means that were not significantly different (Table S3). Variation was greatest in Minnesota (SD = 0.21) and least in New Mexico (SD = 0.04).

Our analysis found broad spatial trends across the US and between ecoregions and states (Figs. 2–6 and Table S2). However, it is difficult to discern from these maps the cumulative nature of the IWI, i.e., how it changes as sub-watersheds join due to convergence of stream segments. To illustrate those spatial trends, we also mapped the IWI onto all stream segments having a Strahler order ≥ 5 (Fig. 7). Note that this is solely for the purpose of visualizing these network patterns; the IWI is a watershed value, not a stream segment variable. The figure shows how IWI typically varies smoothly as sub-watersheds converge. However, there are many instances (e.g., the upper and lower Mississippi) where there are sharp changes in integrity due to the merging of watersheds with contrasting integrity values. The combined watershed is a mixture of the integrity of the upstream sub-watersheds. The map demonstrates that the integrity of the combined watershed can reflect either the higher or lower IWI value of the contributing watersheds (Fig. 7).

The NHDPlusV2 dataset contains 2.6 million catchments, and so it is not possible to examine IWI values mapped onto individual catchments when the map is viewed at the national scale (Fig. 3). However, patterns between individual catchments can be examined by zooming in to the regional scale. We provide two examples of this. For the Potomac River Basin in Virginia and Maryland (Fig. 8a), IWI values are fairly heterogeneous in the west, with high, moderate, and moderately-low IWI values occurring in fairly close proximity. Also, somewhat linear bands of alternating integrity occur in the western portion within the ridge and valley province. However, the eastern portion of this basin, surrounding the Potomac River, is much more homogeneous, with lower IWI values mostly absent. The Willamette River Basin in Oregon (Fig. 8b) varies considerably from the Potomac. Here, moderately-high to moderately-low integrity values are observed in the lower elevation, agricultural and urban areas, surrounded by higher and more homogeneous integrity watersheds in the Coast Range Mountains to the west and Cascades to the east.

Catchment-based maps of the six functional components of the IWI (Fig. S4) are plotted using the same color table as for the catchment-based IWI map (Fig. 3). In comparing these different maps, two things are apparent: First, the maps for all six components only contain blue shades (Fig. S4), which represent higher integrity values. Values for the six components range from minimum values of 0.45–0.66 to 1.0, with medians ranging from 0.94–0.97. This is in contrast with the IWI map (Fig. 3), which spans the entire color table and ranges in values from 0.05 to 1.0, with a median of 0.79. The lower values observed for the IWI,

compared with the six components, results from the multiplicative nature of the index (Eq. 1). Second, the six maps are highly correlated, with pairwise Pearson correlation coefficients varying between 0.85 (for both CHEM vs. CONN and CHEM vs. SED) and 0.98 (between TEMP and HABT). Although spatial trends in the component maps are similar, they vary substantially in magnitudes of index values. For example, all six component maps show high integrity values in the southwest and lower integrity values in portions of the midwestern US. However, lower mapped values in the midwestern US (i.e., lighter shades of blue) are more extensive for HYD, CHEM, TEMP, and HABT compared to SED and CONN (Fig. S4). There is a 45% difference between the minimum integrity values for CHEM (0.45) and SED (0.66).

3.2. Index of Catchment Integrity

The CONUS ICI map (Fig. 9; Fig. S5 for high resolution) appears to be very similar to the IWI map (Fig. 3), showing the same overall patterns. Additionally, the Pearson correlation coefficient between the two datasets is 0.85. The major difference is that the ICI map has sharper transitions, since values are not averaged over nested watersheds. These transitions can be seen in the eastern portion of the Central Valley in California (Fig. 9). High resolution difference maps highlight the differences between the IWI and ICI. For both the Potomac and Willamette River Basins, most of the areas are mapped in gray, indicating IWI and ICI values that have differences within -0.05 to 0.05 (Fig. 8c,d). For the Potomac, areas where ICI exceeds IWI occur mostly in the northcentral and eastern portions of the region, while areas where IWI exceeds ICI occur mostly in the west (Fig. 8c). For the Willamette, areas adjacent to the Coast Range and Cascade Mountains mostly have IWI values greater than ICI, with the opposite in the interior areas. Note that while the IWI and ICI maps are highly correlated, this is mostly due to high correlation between small watersheds (Table 2). Watersheds with an area $\leq 10 \text{ km}^2$ (i.e., bins -3 through 1), which account for 55% of all watersheds, have Pearson correlation coefficients > 0.94 . In contrast, the Pearson correlation coefficients for watersheds $> 1,000 \text{ km}^2$ (bins 4-7), which account for less than 10% of all watersheds, are each < 0.45 . When watersheds that only contain streams with Strahler order of two or greater are considered, the Pearson correlation between the IWI and ICI is reduced to 0.72. However, IWI and ICI maps of these second-order and higher watersheds still have very similar spatial patterns (Fig. S6).

Summary statistics for the ICI were similar to the IWI for the CONUS, ecoregions, and states (Table S2). The ICI for the CONUS had a median value of 0.83 (SD = 0.19), compared to 0.79 for the IWI, with a minimum ICI value of 0.02 and a maximum value of 1.00. The ICI values varied regionally (Table S2), with the Xeric ecoregion having the highest ICI (median = 0.96, SD = 0.17), while the Temperate Plains had the lowest (median = 0.49, SD = 0.15). Variation in the ICI was greatest for the Southern Plains (SD = 0.18), while the Western Mountains had the lowest (SD = 0.10). The ICI values also varied by state (Table S2), with Nevada having the highest catchment integrity (median = 0.98, SD = 0.15) and Iowa the lowest (median = 0.45, SD = 0.13). Variation was greatest in Minnesota (SD = 0.21) and least in New Mexico (SD = 0.07).

3.3. Evaluation of IWI and ICI indices

The first principal component of the water quality, habitat, and combined PCs accounted for 70.7, 44.8, and 40.5% of the total variation in their constituent components, respectively (Table S4). For the CONUS, all regression results between the six indicators and IWI were significant at $p < 0.001$ (Table S5). Regression coefficients were also statistically significant for most ecoregions, although IWI results were not significant for 10 of the 54 ecoregion models. The average adjusted R^2 value for the six CONUS indicators was 0.13, and was greatest for the water quality (Fig. 10) and combined PCs ($R^2 = 0.27$ and 0.25 , respectively); the IWI accounted for less than 12% of the variation in each of the remaining four indicators. These adjusted R^2 values varied by ecoregion, with average fit of the six indicators ranging from -0.01 in the Xeric ecoregion to 0.39 for the Northern Appalachians (Table S5). While the adjusted R^2 value for the water quality PC exceeded the combined PC value nationally, the reverse occurred for seven of the nine ecoregion models. Adjusted R^2 values for the combined and water quality PC indicators varied by ecoregion, with the greatest values occurring in the Northern Appalachians ($R^2 = 0.39$ and 0.33 , respectively), followed by the Southern Appalachians ($R^2 = 0.35$ and 0.32 , respectively). Adjusted R^2 values for the Upper Midwest and Coastal Plains were greater than 0.20 . For the remaining five ecoregions, the IWI accounted for less than 20% of the variation in the combined and water quality PC indicators. In most cases, the combined and water quality PCs had the two highest adjusted R^2 values in each ecoregion (Table S5); however, this was not true for the Northern Plains (maximum $R^2 = 0.05$ for the habitat PC), Western Mountains (maximum $R^2 = 0.16$ for the macroinvertebrate MMI), and Xeric (maximum $R^2 = 0.04$ for the macroinvertebrate MMI) ecoregions.

Regression results for the six functional components vs. the water quality PC were all significant at $p < 0.001$ (Table 3). Adjusted R^2 values ranged from 0.22 (CONN and CHEM) to 0.29 (HYD). The adjusted R^2 value for the HYD component exceeded that for the IWI ($R^2 = 0.27$; Table S5). Although it is composed of three water quality variables (total N, total P, and turbidity), the water quality PC had one of the weakest relationships with the CHEM functional component.

For the ICI, results for all six continuous site indicators by CONUS were also significant at $p < 0.001$ (Table S5). At the ecoregion scale, non-significant results occurred in 11 out of the 54 ecoregion models, compared to 10 for the IWI. Average adjusted R^2 values for the six indicators was 0.09 for the CONUS, ranging from -0.01 in the Xeric ecoregion to 0.29 in the Southern Appalachians. Average adjusted R^2 values for the ICI were less than their IWI counterparts in six of the nine ecoregions. Nationally, adjusted R^2 values for the ICI are 0.01 - 0.11 less than their respective IWI values (i.e., explaining 1-11% less of the variation), with the exception of the habitat PC; this indicator has an adjusted R^2 value that is 0.05 more than the IWI value. For ecoregion models, the pattern in the adjusted R^2 values and their ranks between the IWI and ICI regressions were not consistent: Adjusted R^2 values for the IWI exceed their ICI counterparts in two-thirds of the ecoregion models. Also, in 61% of the ecoregions the maximum adjusted R^2 value across the six indicators was greater for the IWI than the ICI. The combined and water quality PCs had the two largest adjusted R^2 values with the ICI for three of the nine ecoregions, compared to six for the IWI.

At the national level, all models using the IWI or ICI to predict the three categorical variables (macroinvertebrate MMI, macroinvertebrate O/E, fish MMI) were significant at the $p = 0.001$ level, except for the fish MMI model, which was significant at $p = 0.01$ for the IWI (Table S6. Accuracies ranged from 0.38 (macroinvertebrate O/E and IWI) to 0.49 (macroinvertebrate MMI and IWI). All of these models performed better than would be expected based on random selection: random selection of the three MMI categories would be expected to result in an accuracy of 0.33, while an accuracy of 0.25 would be expected for the four O/E categories. Of the 54 ecoregion models, more than half (33) were significant at $p = 0.1$ or less. Accuracies for the IWI ecoregion models ranged from 0.38-0.71, 0.35-0.58, and 0.44-0.73 for the macroinvertebrate MMI, macroinvertebrate O/E, and fish MMI, respectively. Ranges for ICI model accuracies were similar: 0.44-0.71, 0.32-0.56, and 0.44-0.73, the same three variables, respectively. Thus all 54 models performed better than would be expected randomly. IWI and ICI performed similarly as predictors, with differences in IWI and ICI accuracy between ecoregion models ranging from -0.09 (ICI 9% more accurate) to 0.06 (IWI 6% more accurate).

Based on adjusted R^2 values, the IWI was the best single predictor of the water quality PC for the CONUS and five of the nine ecoregions (Table S7), with the ICI being the best predictor for one of the ecoregions (Xeric). Agricultural land use in the watershed was the best predictor for the remaining three ecoregions (Coastal Plains, Southern Plains, and Temperate Plains), with IWI being the second best predictor. Agriculture was also the second best predictor nationally and in five of the six ecoregions where IWI or ICI was the best predictor. In the remaining ecoregion (Northern Plains), watershed urban area was the second best predictor. Differences in adjusted R^2 values between the first and second best predictors were small, ranging from 0.00 to 0.03, except for the Northern Appalachian ecoregion; there the difference in adjusted R^2 between the IWI and watershed agriculture was 0.13. In other words, the IWI was able to account for 13% more of the variance in the Northern Appalachian water quality PC than the next better predictor (Table S7).

4. Discussion

The concept of watershed integrity has been proposed for identifying and describing desirable watershed management endpoints (U.S. EPA, 1998; Novotny and Smith, 2004; U.S. EPA, 2012b; Van Abs, 2013). Flotemersch et al. (2016) defined watershed integrity as the capacity of a watershed to support and maintain the full range of ecological processes and functions essential to the sustainability of biodiversity and of the watershed resources and services provided to society. They also provided an operational definition for an IWI based on stressors that impact six key watershed functions. For the current work, we applied this operational definition using watershed information from EPA's StreamCat dataset. This allowed us to calculate an IWI value for all 2.6 million watersheds in the NHDPlusV2 and map their distribution across the CONUS. Further, we calculated an ICI using catchment data from StreamCat, and compared it to the IWI. We evaluated the IWI and ICI using six continuous and three categorical site-level indicators that were developed from NRSA data. Finally, we compared IWI and ICI performance at predicting the water quality PC with agricultural and urban land use.

Each of the approaches we used to map the IWI has an advantage and disadvantage: The first approach maps IWI values onto actual watersheds (e.g., Fig. 2), but only includes a subset of all watersheds. Conversely, the second approach displays all IWI values on a single map (e.g., Fig. 3), but the values are not mapped onto entire watersheds (except for headwater catchments). Which of these mapping approaches is most useful depends on the particular management application. For example, if an environmental manager wanted to restore a particular watershed having poor integrity, they would need to know the boundaries of that watershed to identify the locations of particular stressors that could be targeted for restoration. In this case, the first mapping approach would provide the appropriate information. On the other hand, a water quality manager might be interested in using IWI values to rank and prioritize watersheds for awarding grants, in which case it would be more important to view and comprehensively evaluate all watersheds, rather than know their spatial boundaries. The second mapping approach would be more appropriate for such an application. In the future, we hope to develop an application that would allow any watershed within the NHDPlusV2 to be interactively selected and mapped with its IWI value, along with StreamCat variables.

The IWI and ICI maps represent the distribution of human infrastructure, including extensive activities such as agriculture as well as more intensive impacts, e.g., urban areas and mines. Such maps could allow environmental managers to identify high integrity areas to target for protection as well as areas impacted by human-related landscape stressors that could be targets for restoration and rehabilitation efforts. While the national maps provide a broad perspective that can be useful for state, regional, and national applications, the high resolution of the maps (Fig. 8) can also allow for more local comparisons and analysis. Diefenderfer et al. (2009) and Esselman et al. (2011) provide examples of how watershed and local data can be combined to address management decisions. Hill et al. (2017) discuss a multi-scale approach whereby watershed-scale information from the IWI could be combined with site-level condition information to address management issues.

The IWI and ICI are each composed of six functional components. These six components are all highly correlated (pairwise Pearson correlation coefficients for the six IWI components range between 0.85 and 0.98). From a management perspective, this implies that the landscape stressors included in the IWI would rarely lead to isolated changes in watershed function; rather, these stressors can affect multiple functions simultaneously, much as weight gain can lead to increased risk of heart disease, stroke, and diabetes. From an indicator perspective, the high correlations between the six components mean that, arguably, only one of the functional components could be utilized. Use of the individual components for the water quality PC would result in adjusted R^2 values that are similar to or exceed (in the case of the HYD functional component) results from the full IWI (Tables 2 and S5). Thus the most parsimonious and efficient approach would seem to be to use the HYD functional component to represent overall watershed integrity.

While we may ultimately conclude that the HYD component by itself is the best way to move forward, we believe that such a conclusion is premature at this time, and that the full IWI needs to be considered further. This is because we used $\hat{IWI}_i$ (Eq. 3), rather than IWI_i

(Eq. 2), to calculate the IWI. This first-order estimator treats the effects of all human-related stressors on each function as a negative linear response and gives equal weights to all stressors. The IWI was developed so as to allow iterative improvements by substituting empirical or literature-based relationships (i.e., $g_{i,j}$) for some of the linear relationships that were used as first-order approximations. This $g_{i,j}$ function describes the relationship between functional component i and stressor s (Eq. 2). A hybrid version of Eqs. 2 and 3 was included in Flotemersch et al. (2016) that allows stressors to be represented using $g_{i,j}$ when known, along with first-order, negative linear estimates for stressors whose relationships are unknown. Since $g_{i,j}$ can vary by functional component, we anticipate that incorporating distinct $g_{i,j}$ functions to represent the relationship between a specific stressor and the different functional components will weaken the correlation between components. There are two reasons for pursuing this: First, using relationships derived from the literature or empirical analysis to represent the relationships between stressors and the functional components – rather than negative linear approximations – should result in an overall improvement of the IWI. We expect that this improvement should be apparent in improved correspondence between IWI and instream estimates of condition. Second, one of the useful characteristics of the IWI is that it provides a conceptual model for how different stressors affect different watershed functions and, therefore, watershed integrity. Sensitivity analyses could allow us to identify which particular stressors cause the greatest reductions in integrity in a particular watershed and the functional components that are most impacted. Knowing the specific functional components that are at risk would be useful in planning protection and restoration efforts. Sensitivity analysis can also show how volatile areas are to changing index values, e.g., the effect of excluding a single variable. However, a sensitivity analysis would be of limited value at this time, since the use of negative linear relationships to describe the effects of stressors and the equal weighting of stressors means that each stressor has the same marginal effect on a given function. A sensitivity analysis would have greater utility if replacing negative linear approximations with actual $g_{i,j}$ relationships reduces the correlation between the functional components or leads to some other kind of improvement.

There is limited information on the relationship between individual stressors and our six functional components and on the effects of stressor interactions on these functions. The benefit of using a first-order estimator is that it merges the conceptual foundation provided by Flotemersch et al. (2016) with the power of an extensive, large-scale dataset like StreamCat (Hill et al., 2016). This allows the IWI and ICI to be mapped for the conterminous US at the present time without the need to wait for further research to elucidate the actual functional forms and weights. As a result, the IWI and ICI (which are available through the StreamCat website; <https://www.epa.gov/national-aquatic-resource-surveys/streamcat>) provide additional tools for watershed management and a conceptual approach for them to be iteratively improved. A limitation of this approach is that the negative linear relationship and equal weights built into the analysis are assumed and so lack the level of scientific rigor that is desired. Further, this assumption leads to the previously discussed issue of correlation between the functional components. Ultimately, the question of whether our approach provides useful information will be answered by future research that further evaluates the performance of our indices and also depends on whether the IWI and ICI lead to improved decision-making.

Our analysis indicates that the IWI and ICI are highly correlated (Pearson correlation coefficient = 0.85), and that differences between the two maps at the national scale are subtle. However, high correlations are limited to small watersheds (Table 2). Half of all catchments within the NHDPlusV2 are headwater (first-order) catchments, in which case the watershed and catchment areas are equivalent and the IWI and ICI values are identical. Larger watersheds (i.e., those > 1,000 km²) are less correlated ($r < 0.45$) but less numerous (7.5% of all watersheds). Thus, national patterns of IWI and ICI (Figs. 3, 8, 9) are dominated by headwater and small watersheds. Still, maps of IWI and ICI that are restricted to watersheds around streams with Strahler order ≥ 2 have similar patterns at the national level (Fig. S6).

For state, regional, and national applications, use of the IWI alone should normally suffice. However, for efforts comparing individual areas (e.g., Fig. 8), the ICI may provide useful information. This would especially be true in applications where the mainstem river – and thus accumulation of upstream water – is not the major focus. For example, in evaluating ecosystem services provided by a local area, the ICI may provide better information than the IWI on the overall level of impacts. Hill et al. (2017) use the ICI and IWI, along with predictions of invertebrate MMI condition, to show how restoration efforts could target areas having local impairment (i.e., poor condition and low ICI values) and relatively intact watersheds (i.e., high IWI values). Because of the similarity in results, we focus exclusively on the IWI for the remainder of this section.

There is no direct way to measure watershed integrity, since it is a concept. However, we hypothesized that site-level condition would be related to the integrity of the watersheds in which the sites occur. Although relationships between the IWI and the six continuous site-level indicators were highly significant for the CONUS, the IWI could only account for 2-12% of the variation in the four biological and habitat variables. The poor fit observed for the continuous biological and habitat indicators could be because these variables are unaffected by watershed-scale integrity, or because this version of the IWI does not adequately represent watershed functions and risks that have important effects on these four indicators. We think the former is unlikely, given the many studies showing a relationship between biological endpoints and watershed indicators (e.g., Allan et al., 1997; Van Sickle et al., 2004; Esselman et al., 2013; Schinegger et al., 2013). Regarding the latter, one reason that our index might not adequately represent factors that are related to the three biological indicators is that the IWI is consistently defined nationally, whereas the macroinvertebrate MMI, macroinvertebrate O/E, and fish MMI are defined using regional reference sites. This means that reference sites – which by definition have good condition – will have poorer integrity in highly impacted ecoregions and higher integrity in low impact ecoregions.

Our analysis of the three categorical variables was consistent with the continuous analysis, in that we found positive associations but modest results. In all cases IWI and ICI predictions of the categorical variables were better than would be expected from random selection. In some cases, the models performed well, e.g., accuracies greater than 0.7 for the Coastal Plains macroinvertebrate MMI and Xeric fish MMI models (Table S6). We expect that the same changes that might improve the performance of the IWI and ICI with respect to the continuous models would also improve categorical results. To this end, we are

encouraged by a separate study of ours that found a strong relationship between the macroinvertebrate MMI and StreamCat variables (Hill et al., 2017). The high correspondence between macroinvertebrate MMI and StreamCat variables suggests that better fits with the IWI and instream indicators can be obtained through fine-tuning of the IWI response function (i.e., $g_{i,j}$). We are hopeful that the IWI can be iteratively improved by using $g_{i,j}$ functions, and that this will result in stronger relationships with the four biological and habitat indicators.

The IWI accounted for a quarter of the national variation in the water quality and combined PCs for the CONUS, with fits of 32-39% in the Northern and Southern Appalachians (Table S5). Such results could be considered a poor performance for many model applications. However, the water quality and combined PCs are derived from measurements obtained from a single summer-time grab sample. In contrast, the stressors that are quantified in the IWI should be associated with more of a long-term central tendency. Use of these two different types of data would be expected to produce a noisy relationship, which was what we observed (e.g., Fig. 10). In spite of this noise, our findings confirm that site-level indicators based on a set of water quality or combined water quality and habitat variables are related to integrity that was independently derived at the watershed scale. Further, our results suggest that the IWI adequately quantifies watershed integrity with respect to these two indicators. This is especially encouraging given that this was a first-order approximation of a conceptually-defined index.

We also stress that a good fit between the IWI and site-level metrics is only one way to evaluate the IWI. While we hypothesized that the IWI would be related to instream condition, it is meant to represent the integrity of all aquatic systems in an entire watershed. A focus on purely local-scale stream resources would fail to recognize the watershed as a complete, intra-connected system. Future applications will examine the relationship between the IWI and other, more synthetic indicators (e.g., human health).

It should be recognized that the IWI and ICI only incorporate stressors that are comprehensively available from national maps. As a result, they do not include all the stressors that Flotemersch et al. (2016) conceptually associated with the six watershed functions (Table 1, Fig. 1). For example, although cattle are common in many areas of the western US, we were not able to utilize county-level cattle data at this time because of quality control and data resolution issues. Further, the national datasets that were used could omit stressors that are documented on higher resolution, more thematically detailed datasets that are limited in geographic scope. In either case, these omissions would lead to inflated IWI values. In particular, IWI values for the Xeric ecoregion, which were the largest in the country (Fig. 5), would be reduced if data on cattle density could have been incorporated into our index. This overestimate of watershed integrity would need to be taken into account for any applications, especially in the southwestern US. In spite of the room for improvement, we believe that the IWI maps in their present form (Figs. 2–3, S1–S3) could be useful in environmental decision-making if these limitations are kept in mind, and so we have made these data available through the StreamCat website (<https://www.epa.gov/national-aquatic-resource-surveys/streamcat>).

While the IWI or ICI are the best predictors of the water quality PC for the CONUS and six of the nine ecoregions, they only perform marginally better than agriculture in most instances (Table S7). Given these results, a reasonable question is whether the IWI and ICI is worth the effort, since development of the IWI and ICI required considerable effort and agriculture alone could provide similar national results. We believe this effort is warranted, for three main reasons: First, the IWI performed substantially better than agriculture in the Northern Appalachians (0.33 vs. 0.20), accounting for 13% more of the variance. Beyond the scale of the nine ecoregions, there could be other areas where the IWI performs better than agriculture, and could provide managers with better information. Second, agriculture was not the dominant stressor across the board; urban area was the second-most important predictor of the water quality PC after the IWI in the Northern Plains. This further suggests that agriculture would not be appropriate in all parts of the country. Third, the IWI is meant to assess watershed integrity *per se*, and not site level condition. As we have noted, there is no way to directly test the IWI, since watershed integrity is a concept, and so we used the site level variables as proxies. However, our intent was to develop an index that is responsive to all stressors, and could be used in locations where stressors other than agriculture are present. Finally, just as we have argued that improving the IWI would reduce the correlation between the six functional components, we also believe this will increase the performance of the IWI relative to agriculture. For example, IWI performance might be improved by using a non-linear relationship to describe a non-agricultural stressor or by giving stressors different weights.

Production of the IWI was only possible because of the existence of datasets like StreamCat, which was developed in part for this purpose (Hill et al., 2016). Spatial data for aquatic applications are often derived for local drainage areas; for example, EPA's EnviroAtlas maps data for 12-digit US Geological Survey (USGS) Hydrologic Unit Codes (HUC-12s; Pickard et al., 2015). These summaries do not incorporate information from upstream units, and so are not true watershed-scale variables. Datasets of watershed variables have been produced for the NHDPlusV1 as a part of the National Fish Habitat Action Plan (Wang et al., 2011; Esselman et al., 2013). However, NHDPlusV2 improves upon NHDPlusV1 in a number of ways, e.g., improved network topology, spatial detail, and catchment delineations (McKay et al., 2012). Prior to StreamCat, only a limited number of watershed variables were available for the NHDPlusV2; for example, those distributed with the NHDPLUSV2 or those developed by the USGS NAWQA program (pers. comm., Michael Wieczorek, USGS). To the best of our knowledge, StreamCat is the first published and publically available dataset that contains large numbers of both catchment- and watershed-scale variables for all 2.6 million catchments in the NHDPlusV2, and it now contains these IWI and ICI data.

5. Conclusions

Using watershed data from EPA's StreamCat dataset, we developed national maps of the IWI based on the approach by Flotemersch et al. (2016), which addresses risks to six functional components. This work provides a major application of the Flotemersch et al. (2016) concept and identified several key ways in which the application of this index, and the associated ICI, could be improved. In addition, the application of this definition could provide an important tool for states and other agencies. Interest in indices of watershed

integrity or other features, such as watershed health, have grown in recent years (e.g., U.S. EPA, 1998; Novotny and Smith, 2004; U.S. EPA, 2012b; Van Abs, 2013; Flotemersch et al., 2016) and our application of the IWI can advance these objectives in at least two ways. First, the fact that our application of this definition is nationally-consistent facilitates comparison among regions. This aspect of the IWI differs substantially from, for example, indicators of stream condition that were developed within regions as part of NRSA. Second, our map of IWI can serve as a starting point for states, local agencies, or watershed councils seeking to develop their own applications of watershed integrity. Additionally, local information could be included to further refine estimates of IWI depending on the needs and purpose of local practitioners. Such activities could, in turn, provide insight into how the national IWI could be improved in an iterative way. Specifically, the IWI could be improved by incorporating information on stressors that are currently, to our knowledge, unavailable at the national level. Applications of the IWI that incorporate local information on land use could provide impetus to develop such datasets at scales appropriate for the national IWI. We need to continue to assess the utility of the IWI and ICI, to better evaluate their performance vis-à-vis other landscape predictors, and to improve the data and relationships upon which they are based. Despite the potential for substantial improvement, we believe the IWI in its present form could be useful for evaluating and prioritizing restoration and protection efforts at multiple scales, especially when combined with site-level condition information. However, limitations concerning the absence of nationwide data for certain stressors need to be kept in mind.

Supplementary Material

Refer to Web version on PubMed Central for supplementary material.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found in the online version, at ...

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Fig. 1.
Conceptual model of the calculation of the Index of Watershed Integrity including human activities that produce stress and degrade key functions in watersheds.

Fig. 2. Index of Watershed Integrity for the conterminous United States for watersheds associated with first (a) and seventh (b) Strahler order streams. Higher integrity is shown in bluer colors, moderate integrity in yellows, and lower integrity in redder colors. Because watersheds of consecutive stream segments are nested and overlap, it is not possible to show all watersheds simultaneously, and so this figure is limited to watersheds associated with a single stream order. Further, consecutive stream segments can have the same stream order, so the figure includes only the furthest downstream watersheds of a particular order.

Fig. 3.

Index of Watershed Integrity for the conterminous United States showing higher integrity in bluer colors, moderate integrity in yellows, and lower integrity in redder colors. Because watersheds of consecutive stream segments are nested and overlap, each value is mapped onto the most downstream catchment within each watershed.

Fig. 4.
Distribution of the Index of Watershed Integrity (IWI) in the conterminous United States.

Fig. 5.

Box and whisker plot for Index of Watershed Integrity values for the conterminous United States by aggregated Omernik Level III ecoregions. Line within the box represents the median, and the bottom and top of the box are the 25th and 75th percentiles, respectively. Lower whisker is either the minimum value or 1.5 times the interquartile range from the box, whichever is lesser, and upper whisker is either the maximum value or 1.5 times the interquartile range from the box, whichever is greater. Open circles represent values outside these ranges. Key: XER = Xeric, WMT = Western Mountains, NPL = Northern Plains, NAP = Northern Appalachians, SAP = Southern Appalachians, SPL = Southern Plains, UMW = Upper Midwest, CPL = Coastal Plain, TPL = Temperate Plains.

Fig. 6.

Box and whisker plot for Index of Watershed Integrity values for the conterminous United States by state. Line within the box represents the median, and the bottom and top of the box are the 25th and 75th percentiles, respectively. Lower whisker is either the minimum value or 1.5 times the interquartile range from the box, whichever is lesser, and upper whisker is either the maximum value or 1.5 times the interquartile range from the box, whichever is greater. Open circles represent values outside these ranges.

Fig. 7.

Index of Watershed Integrity mapped to rivers (Strahler order ≥ 5) of the conterminous United States showing higher integrity in bluer colors, moderate integrity in yellows, and lower integrity in redder colors. Because watersheds of consecutive stream segments are nested and overlap, each value is mapped onto the most downstream segment within each watershed.

Fig. 8. High resolution maps showing Index of Watershed Integrity (IWI) results for portions of the Potomac River Basin, Virginia and Maryland (A) and the Willamette River Basin, Oregon (B), and differences between the IWI and Index of Catchment Integrity (ICI) values for the same two areas (C and D; blue values represent $IWI > ICI$, and red values represent $IWI < ICI$). Because watersheds of consecutive stream segments are nested and overlap, each IWI value is mapped onto the most downstream catchment within each watershed.

Fig. 9. Index of Catchment Integrity for the conterminous United States showing higher integrity in bluer colors, moderate integrity in yellows, and lower integrity in redder colors.

Fig. 10. Scatterplot showing regression results of the water quality principal component (PC) at National Rivers and Streams Assessment sample sites versus the IWI at corresponding watersheds for the conterminous United States.

Table 1

Key functions that occur in unaltered watersheds and the major stressors affecting these functions. Data sources that can be used to evaluate the stressors are included parenthetically (see key at bottom of table). Within each function highly correlated stressors (correlation coefficients $r > 0.7$) were eliminated. Adapted from Flotemersch et al. (2016).

Key function	Description	Major stressors	
		Within channel	Outside channel
Hydrologic regulation (HYD)	Maintenance of the natural timing, pattern, supply, and storage of water that flows through the watershed	<ul style="list-style-type: none"> • Presence and volumes of reservoirs (NABD) • Stream channelization and levee construction (NA) 	<ul style="list-style-type: none"> • Percent of the watershed comprising agricultural land use (NLCD) • Total length and density of canals/ditches (NHD) • Percent imperviousness of human-related landscapes (NLCD) • Alteration to and spatial arrangement of riparian vegetation (LANDFIRE) • Boundaries, depths, and flows of aquifers (NA) • Groundwater use (NA) *
Regulation of water chemistry (CHEM)	Maintenance of the natural timing, supply, and storage of the major chemical constituents of freshwaters: nutrients (nitrogen & phosphorus), salinity or conductivity, total dissolved solids, hydrogen ions (pH), and naturally occurring minor constituents (e.g., heavy metals). Human-related alterations can include deviations from naturally occurring concentrations of these constituents or the inclusion of non-naturally occurring constituents, such as pesticides and industrial chemicals.	<ul style="list-style-type: none"> • Presence and volumes of reservoirs (NABD) • Stream channelization and levee construction (NA) 	<ul style="list-style-type: none"> • Atmospheric deposition of anthropogenic sources of nitrogen and acid rain (NADP) • Percent of watershed composed of urban or agricultural land uses (NLCD) • Fertilizer application rates (FERT) • Presence and density of wastewater treatment facilities (NPDES), industrial facilities (TRI), superfund sites (SUPERFUND), or mines (MINES) • Cattle density (NA) * • Alteration to and spatial arrangement of riparian vegetation (LANDFIRE) • Chemical constituents of groundwater (NA)
Sediment regulation (SED)	Maintenance of the volume and size composition of inorganic particles that are stored or transported through the stream or within lakes, wetlands, or estuaries.	<ul style="list-style-type: none"> • Presence and volumes of reservoirs (NABD) • Stream channelization and levee construction (NA) 	<ul style="list-style-type: none"> • Alteration to and spatial arrangement of riparian vegetation (LANDFIRE) • Presence and density of mines (MINES), forest cover loss (GFC), and roads (TIGER) • Agriculture (NLCD) weighted by soil erodibility (CONUS-SOIL)

Key function	Description	Major stressors	
		Within channel	Outside channel
Hydrologic connectivity (CONN)	Presence of hydrologic pathways for the transfer of matter, energy, genes, and organisms within watersheds. Systems can vary naturally in their hydrologic isolation (e.g., desert springs) or connectedness (e.g., the Everglades).	<ul style="list-style-type: none"> • Presence and volumes of reservoirs (NABD) • Stream channelization and levee construction (NA) • Road/stream intersections (TIGER/NHD) weighted by stream reach slope (NHD) 	<ul style="list-style-type: none"> • Alteration to and spatial arrangement of riparian vegetation (LANDFIRE) • Density of ditches/canals (NHD) • Groundwater use (NA) * • Presence and density of wastewater discharge sites (NPDES) • Percent of riparian zone composed of urban or agricultural land uses (NLCD)
Temperature regulation (TEMP)	Maintenance of the full range of natural landscape features (both aquatic and terrestrial) required to maintain temperatures that support the aquatic chemistry and biota.	<ul style="list-style-type: none"> • Presence and volumes of reservoirs (NABD) 	<ul style="list-style-type: none"> • Alteration to and spatial arrangement of riparian vegetation (LANDFIRE) • Percent of watershed composed of agricultural land uses (NLCD) • Percent of watershed composed of urban land uses in the riparian zone (NLCD) • Groundwater use (NA) * • Presence and density of wastewater discharge sites (NPDES)
Habitat provision (HABT)	Presence and maintenance of the full range of natural landscape features (both aquatic and terrestrial) that represent the complete set of conditions that are needed to maintain the natural diversity and abundances of aquatic biota.	<ul style="list-style-type: none"> • Presence and volumes of reservoirs (NABD) 	<ul style="list-style-type: none"> • Alteration to and spatial arrangement of riparian vegetation (LANDFIRE) • Density of housing unit developments within riparian zones (TIGER) • Percent of watershed composed of agricultural land uses (NLCD) • Density of road/stream intersections (TIGER/NHD) • Density of roads within riparian zones (TIGER)

* County data were available for groundwater use and cattle density but were not utilized because of quality control and data resolution issues.

KEY:

CONUS-SOIL – Penn State University soil characteristics dataset, based on STATSGO (http://www.soilinfo.psu.edu/index.cgi?soil_data&conus)

FERT – County-level estimates of N & P from commercial fertilizer (<http://pubs.usgs.gov/sir/2012/5207>)

GFC – University of Maryland Global Forest Change 2000–2013 Dataset (http://earthenginepartners.appspot.com/science-2013-global-forest/download_v1.1.html)

LANDFIRE – USFS and USDOJ LANDFIRE Program (<http://www.landfire.gov>)

MINES – USGS Mines Dataset (<https://www.sciencebase.gov/catalog/folder/4f4e4767e4b07f02db47e0ad>), USGS National Coal Resources Data System (NCRDS), and US Stratigraphy (USTRAT) data of coal mine sites (http://ncrdspublic.er.usgs.gov/ncrds_data)

NA – Not available

NAPD – National Atmospheric Deposition Program National Trends Network (<http://nadp.sws.uiuc.edu/data/ntn>)

NHD – National Hydrography Dataset (http://www.horizon-systems.com/NHDPlus/NHDPlusV2_home.php)

NABD – 2012 National Anthropogenic Barrier Dataset (<https://www.sciencebase.gov/catalog/item/56a7f9dce4b0b28f1184dabd>)

NLCD – National Land Cover Dataset (http://www.mrlc.gov/nlcd06_data.php)

NPDES – USEPA National Pollutant Discharge Elimination System (http://www.epa.gov/enviro/geo_data.html)

SUPERFUND – USEPA Superfund Sites (http://www.epa.gov/enviro/geo_data.html)

TIGER – US Census Bureau TIGER/Line Program (http://www2.census.gov/geo/pdfs/maps-data/data/tiger/tgrshp2013/TGRSHP2013_TechDoc.pdf)

TRI – National Toxic Release Inventory (http://www.epa.gov/enviro/geo_data.htm)

[All websites accessed April 26, 2016, except CONUS-SOIL accessed July 25, 2016 and GFC accessed November 11, 2016.]

Table 2

Pearson correlation coefficients (r) between Index of Watershed Integrity and Index of Catchment Integrity by watershed area.

Log10 of Watershed Area (km ²)	Proportion of All Watersheds	Pearson Correlation Coefficient (r)
-3	0.000	0.997
-2	0.001	0.994
-1	0.006	0.978
0	0.068	0.955
1	0.475	0.938
2	0.263	0.822
3	0.111	0.657
4	0.049	0.427
5	0.021	0.293
6	0.005	0.017
7	0.001	-0.372

Table 3

Regression results (n = 1855) for water quality principal component (response variable) derived from 2008–2009 National River and Streams Assessment (NRSA) data with the six functional components of the Index of Watershed Integrity (independent variables) within the conterminous United States (CONUS). All results significant at $p < 0.001$. Indicators: HYD = hydrologic regulation; CHEM = regulation of water chemistry; SED = sediment regulation; CONN = hydrologic connectivity; TEMP = temperature regulation; and HABT = habitat provision.

Indicator	F Value	Intercept	Slope	Adjusted R ²
HYD	750.823	11.480	−12.066	0.288
CHEM	535.361	10.562	−11.367	0.224
SED	596.543	18.986	−19.712	0.243
CONN	528.531	14.210	−14.781	0.222
TEMP	664.518	10.353	−10.971	0.264
HABT	649.390	11.898	−12.604	0.259