Ecological Indicators 40 (2014) 136–146

Contents lists available at ScienceDirect

Ecological Indicators

journal homepage: www.elsevier.com/locate/ecolind

On the scaling behavior of reliability–resilience–vulnerability indices in agricultural watersheds

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ARTICLE INFO

Article history:
Received 28 August 2013
Received in revised form 10 January 2014
Accepted 14 January 2014

Keywords:
Risk indices
Reliability
Resilience
Vulnerability
Watershed
Water quality
Spatial scale
Best management practices

ABSTRACT

Risk indices such as reliability–resilience–vulnerability (R–R–V) have been proposed to assess watershed health. In this study, the spatial scaling behavior of R–R–V indices has been explored for five agricultural watersheds in the midwestern United States. The study was conducted using two different measures of spatial scale: (i) the ratio of contributing upland area to area required for channel initiation (*FA*), and (ii) Strahler stream order. It was found that R–R–V indices do change with spatial scale, but a representative watershed-specific threshold *FA* value exists for these indices to achieve stable values. Scaling with Strahler stream order is feasible if the watershed possesses a tree-like stream network. As an example of anthropogenic influences, this study also examined the role of BMPs placed within an agricultural watershed via a cost-effective optimization scheme on the evolution of R–R–V values with scale. While the placement of BMPs achieved reductions in concentrations and/or loads of constituents, they may not significantly change watershed risk measures, but are likely to cause significant reduction in vulnerability. If primarily upland BMPs are placed in a diffuse manner throughout the watershed, there might not be a significant change in the scaling behavior of R–R–V values.

1. Introduction

Despite the success in achieving notable improvements in cleaning up the Nation's waters, primarily through the implementation of point source controls, a significant number of the water bodies remain impaired and do not meet water quality standards for their designated use (U.S. EPA National Summary of Impaired Waters webpage, 2012). Discharges from unregulated nonpoint sources (NPS) of pollution have not been controlled as successfully as point discharges. Agricultural and urban storm water runoff are diffuse in nature and are major contributors of pollutants such as sediments, nutrients, metals, pesticides, toxics, and many other contaminants that find their way to ground water and receiving water bodies (U.S. EPA, 2000). Unless appropriate management practices are put in place, runoff from natural and anthropogenic sources will continue to degrade the Nation's waters in headwater streams, rivers, lakes and estuaries. In this regard, best management practices (BMPs) are touted as effective management measures for the control of nonpoint source pollution (U.S. EPA, 2001; D'Arcy and Frost, 2001, among others). BMPs such as parallel terraces, riparian corridors and wetlands, in addition to providing biodiversity and habitat benefits (Kadlec and Knight, 1996; Broadmeadow and Nisbet, 2004, etc.), are effective in retaining and/or reducing sediment and nutrient loadings.

1.1. Indicator based approach to watershed health assessment

Assessment of river health, and by extension watershed health, using various indicators has been a popular approach among eco-hydrologists. Biological (e.g. macroinvertebrate population), chemical (e.g. sediment and nutrients data) and/or physical (e.g. pH and temperature) characteristics of riverine and ecological systems serve as indicators with indices being used to summarize the information. Assessment involves comparing index scores for given streams to some reference conditions/streams (Metcalfe, 1989; Fairweather, 1999). Indices include, but are not limited to, Index of Biotic Integrity (IBI, Karr, 1981), Belgian Biotic Index (BBI, De Pauw and Vanhooren, 1983), Family-Level Biotic Index (FBI, Hilsenhoff, 1988), Biological Monitoring Working Party Score (BMWPS, Hawkes, 1998), Index of Stream Condition (ISC, Ladson et al., 1999), Urban Intensity Index (URBI, McMahon and Cuffney, 2000) and River Invertebrate Prediction and Classification System (RIVPACS, Clarke et al., 2003). Assessment methods have also relied on instream flows where hydrologic and hydraulic trends in streams are examined by comparing present day conditions to historical data, as in Indicators of Hydrologic Alteration

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http://dx.doi.org/10.1016/j.ecolind.2014.01.017

(IHA, Richter et al., 1996), Dundee Hydrological Regime Alteration Method (DHRAM, Black et al., 2005) and the Ecological Limits of Hydrologic Alteration (ELOHA, Poff et al., 2010). Such methods have been useful in providing a picture of a given river system or watershed. Some indices are based on indicators that exhibit correlation with each other, resulting in information redundancy (Gao et al., 2009). For stakeholders concerned mostly with stream water quality degradation through nonpoint source pollution, no clear guidelines exist for choice of an appropriate index.

A potential method of assessing watershed health with respect to water quality was proposed by Hoque et al. (2012). We utilized risk-based measures such as reliability, resilience and vulnerability (R–R–V) as a way of developing probabilistic indices that are novel to both watershed hydrologists and ecologists. In broad terms, reliability may be defined as the probability of a system being in a safe, i.e., 'not failed', state. Resilience is the probability of a system recovering from a failed state to a safe state at a given time. Finally, vulnerability is a measure of the severity of the failed state. The usage of these indices has been common in ecological risk assessment. Applications have been documented in Holling (1973), Fiering (1982), Naeem (1998), Leuven and Poudevigne (2002), Petchey and Gaston (2009), etc. In hydrology, the application of R–R–V has thus far been popular in the management of water supply systems, as evidenced in studies as Hashimoto et al. (1982) and Kjeldsen and Rosbjerg (2004). R–R–V indices provide stakeholders with a data-driven method for assessment of watershed health with respect to water quality. The selection of R–R–V was motivated by (i) availability/reconstructability of required time-series data, (ii) familiarity with these indices in the hydrologic community for other applications (as indicated in cited literature), and (iii) the probabilistic framework for reliability and resilience that lends them to risk-based analyses.

1.2. Impact of spatial scale on R–R–V based assessments

Hoque et al. (2012) found that R–R–V estimates varied at different locations within a watershed, but the nature of this variability of R–R–V values was not investigated. Spatial laws have been a topic of considerable interest in watershed hydrology. Early studies into the hierarchical organizational structure of stream networks within watersheds and how their physical attributes may change as a function of spatial scale led to the introduction of stream orders and Horton's scaling laws (Horton, 1945). The concept of stream orders was developed with alternative ordering schemes suggested by Strahler (1952, 1957), Shreve (1966), Hodgkinson et al. (2006), and others. Researchers have noted limitations of stream ordering schemes for hydrologic studies (Kirchner, 1993), and for examining physical, chemical and biological processes such as nutrient spiraling, sediment transport, etc. (Peckham and Gupta, 1999; Gangodagamage et al., 2011). Alternative schemes for spatial scale studies have been proposed by Abrahams (1984), Tarboton et al. (1989), Willgoose et al. (1991), Nikora and Sapozhnikov (1993), Rigon et al. (1994), Maritan et al. (1996), Rigon et al. (1996), Rinaldo and Rodriguez-Iturbe (1998), Betz et al. (2010), Zaliapin et al. (2010) and Gangodagamage et al. (2011). Despite limitations, the Strahler ordering scheme is nevertheless the most widely used and accepted method in hydrology. A multitude of spatial studies have used, and continue to use, Strahler numbers, among them Roth et al. (1996), Boyero and Bailey (2001), Schmera and Eros (2004), King et al. (2005), Vondracek et al. (2005) and Saltman (2009).

Since R–R–V indices are computed from time-series data of water quality constituents (for example sediment, nutrients, pesticides, etc.) along the stream network within a watershed, use of the Strahler scheme as a possible way to address spatial scale is relevant. For instance, could a representative threshold scale exist so that R–R–V values for the watershed would exhibit stable behavior beyond this scale? Two sub-watersheds of similar areas, less than such a threshold, may have different R–R–V values. The smaller drainage areas are likely to be more susceptible to fluctuations in sediment and nutrient loads that would lead to fluctuations in R–R–V values based on these constituents. Hence, separate evaluations of R–R–V values would be needed for such sub-basins within the watershed. However, with increasing drainage areas for nested sub-basins, the sensitivity to disturbances in sediment/nutrient loads would decrease. Any increase in nested drainage area beyond the threshold should not result in significant fluctuations of R–R–V values.

The existence of such a scale would signify that stable R–R–V estimates exist and could be transferred to areas beyond the threshold. This would also be in line with previous studies into the conceptualization of scale-dependant hydrologic processes. For example, Levin (1992) stressed the importance of determining an appropriate scale of investigation when studying riverine ecosystems. Bloschl and Sivapalan (1995) provided an exhaustive review of scaling issues and the role of scaling on defining effective parameters. A threshold area for R–R–V indicator, should it exist, would allow for robust assessment of watershed health.

The performance of BMPs relative to the costs associated with implementing them (monetary or otherwise) have been shown to improve substantially when they are selected and placed through the application of a simulation–optimization framework. Efforts in this regard have been detailed in studies such as Arabi (2005), Whittaker et al. (2009), Maringanti et al. (2011), Bekele et al. (2011) and Lautenbach et al. (2013), among others. However, placement of BMPs also alters the characteristics of both flows and nutrient loads within the watershed, and could thus affect the subsequent nature of R–R–V variability and scaling relationships.

The primary objective of this study is to investigate the effect of spatial scale on risk-based watershed health assessment. A related objective is to examine whether the placement of BMPs within a watershed affects R–R–V values and their scale dependency. To our best knowledge, such an exploration has not been conducted in the literature.

2. Description of study sites

Agricultural watersheds were chosen as they are generally the source of NPS pollution, and all the study sites have had some form of water quality impairment reported in their streams. As is common in the U.S. Midwest, the predominant crops cultivated in the five study watersheds are corn and soy-bean in a rotation system. Also, since the watersheds are located in the same geographical region (Indiana/Ohio/Michigan, see Fig. 1), they share similar climate, topological and hydrological characteristics (NRCS, 2008a,b,c). A brief description of the study sites follow below.

(a) *St. Joseph River Watershed*: The St. Joseph River watershed is an eight digit watershed (USGS HUC: 04100003) that spans over an approximate area of 2825 km^2 and nine counties in three states. The St. Joseph River originates in Michigan and flows through Ohio before forming the Maumee River system along with St. Mary's River near Ft. Wayne, Indiana. About 68% of the watershed is dedicated to farming, being either croplands or pastures. A National Resources Conservation Service (NRCS, 2008a) rapid watershed assessment report concluded that the watershed is sediment and nutrient-impaired, i.e., the streams within the watershed 'did not meet, or were not expected to meet, applicable water quality standards'.

(b) *Cedar Creek Watershed*: Cedar Creek is the largest tributary of the St. Joseph River system, located along its south-western edge (Fig. 1) in north-eastern Indiana. It has an approximate drainage area of 643 km^2. Approximately 72% of its land is used for crop