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ARENA PAPER

A WATERSHED INTEGRITY DEFINITION AND ASSESSMENT APPROACH TO SUPPORT STRATEGIC MANAGEMENT OF WATERSHEDS

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ABSTRACT

Watersheds are spatially explicit landscape units that contain a range of interacting physical, ecological and social attributes. They are social–ecological systems that provide a range of ecosystem services valued by the society. Their ability to provide these services depends, in part, on the degree to which they are impaired by human-related activity. An array of indicators is used by natural resource managers, both private and government, to assess watersheds and their sub-components. Often these assessments are performed in comparison with a reference condition. However, assessments can be hampered because natural settings of many systems differ from those sites used to characterize reference conditions. Additionally, given the ubiquity of human-related alterations across landscapes (e.g. atmospheric deposition of anthropogenically derived nitrogen), truly unaltered conditions for most, if not all, watersheds cannot be described. Definitions of 'integrity' have been developed for river ecosystems, but mainly at the reach or site scale and usually for particular species, such as fish or macroinvertebrates. These scales are inappropriate for defining integrity at the watershed scale. In addition, current assessments of endpoints do not indicate the source of impairment. Our definition of watershed 'integrity' is 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. To operationalize this definition as an assessment tool, we identify key functions of unimpaired watersheds. This approach can then be used to model and map watershed integrity by incorporating risk factors (human-related alterations or stressors) that have been explicitly shown to interfere with and degrade key functions in watersheds. 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. © 2015 The Authors. *River Research and Applications* published by John Wiley & Sons Ltd.

KEY WORDS: index of watershed integrity; IWI; watershed management; scale; sustainability; healthy watersheds

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INTRODUCTION

Watersheds provide a range of ecosystem services that are valued by the society (Costanza *et al*., 1997, 2014; TEEB 2010). These include supporting services (e.g. soil formation, nutrients and primary production), provisioning goods and services (e.g. food, water, wood, fibre 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). Many of these services are directly related to the flow and function of water and its constituents within watersheds. Despite the recognition of the importance of these functions, water in many regions is viewed primarily as an exploitable commodity for human use, that is, a provisioning service. As a result, the wider range of benefits of watershed services has had limited recognition outside of scientific circles. 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*. The deteriorating state of freshwater biodiversity, globally, reflects these impacts (Dudgeon *et al*., 2006; Butchart *et al*., 2010).

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

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focused on specific impacts (e.g. point-source pollutants), the overuse of certain resources, protection or habitat restoration. However, these early efforts often failed to address chronic problems that contribute to longer-term declines in the structure and function of watersheds, such as pollutants associated with non-point run-off from urbanized and agricultural areas (USEPA, 1996). 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). This holistic view of watersheds was critical for several reasons. First, watersheds are hierarchical systems that function at different levels of organization and scale (Thoms *et al*., 2007). A feature of hierarchical systems is that changes at large scales have slower rates of behaviour while changes at small scales occur more quickly (O'Neill and King, 1998). In addition, hierarchical systems exhibit emergent properties, whereby properties and behaviours at higher levels cannot be simply deduced from the collective functioning of their parts (Allen and Starr, 1982). Thus, it may be scientifically inappropriate to use information collected at the site or reach scale to assess watershed-scale attributes (Dollar *et al*., 2007). Second, watersheds are social–ecological systems, and their ability to provide a range of services is dependent on each component functioning and interacting properly (Bunch *et al*., 2011; Walker and Salt, 2012). In other words, society depends on ecological systems to provide ecosystem goods and services (hereafter referred to as ecosystem services; Malinga *et al*., 2015 and references therein), and in turn, exploited ecological systems depend on the society to maintain them in a way that ensures their long-term functioning (Berkes, 2007).

Recently, the use of the term 'watershed integrity' to identify and describe desirable watershed management endpoints has increased (USEPA, 1998, 2012c; Novotny, 2004; Van Abs, 2013). However, ambiguity exists as to what this term actually means. At one extreme, watershed integrity has been used as an extension of biological and ecological integrity, thus positioning it as a benchmark free from human influences (Novotny, 2004; USEPA, 2012c). At the other extreme, watershed integrity has been employed to describe systems that support the sustainable flow of ecosystem services to society, such that the term is not divorced from human influence (USEPA, 1998; Van Abs, 2013).

A clear definition of watershed integrity can advance our understanding and discussion of this concept, especially within an interdisciplinary domain such as watershed science (Noss, 1995; Wicklum and Davies, 1995; Karr, 1996; Hull *et al*., 2003; Bennett *et al*., 2009). However, definitions alone do not support the quantification of differences in watershed character 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). To do this, we must make the term 'watershed integrity' operational—i.e. capable of being measured. Thus we need to identify the individual features and processes that maintain watershed integrity and identify human-related factors that degrade these features and processes.

With these requirements in mind, we cover three objectives in this paper. First, we provide a definition of watershed integrity. To assist in this definition, we also discuss terms that are closely related to watershed integrity;, including biological integrity, ecological integrity, watershed health, healthy watersheds, aquatic condition, watershed resilience and watershed sustainability. Without such context, watershed integrity may be confused with other terms, creating ambiguity and skepticism as to its value and possibly undermining watershed management strategies (Hull *et al*., 2003). Second, we identify key functions that define unimpaired watersheds. Third, we make watershed integrity operational by identifying risk factors (i.e. human-related stressors) that are known to impact these key functions interfering with and degrading the structure, function and feedbacks of watersheds. Indicators for these stressors are then used to construct an index to assess watershed integrity.

DEFINING WATERSHED INTEGRITY

Watershed integrity is similar to many terms common in the fields of ecology and geomorphology (i.e. sustainability, health, biodiversity and ecological integrity) in that it describes activities at the interface between sub-disciplines of environmental science, policy and management (Hull *et al*., 2003). Successful interdisciplinary research requires the 'explicit joining of two or more areas of understanding into a single conceptual-empirical structure' (Pickett *et al*., 1994), which can then generate common and agreed terms (Dollar *et al*., 2007). Watershed integrity is a compound phrase, and to avoid confusion, misinterpretation or relegation to the status of a buzzword, it requires a comprehensive definition (Wicklum and Davies, 1995). This definition should include a statement of the word's meaning, intended use and how it may be measured or expressed quantitatively or operationalized (Noss, 1990; Hull *et al*., 2003; Lackey, 2003). Herein, the two separate words of watershed integrity are examined independently and then collectively.

Watershed

A watershed is the landscape that contributes surface water to a single location, such as a point on a stream or river, or a single wetland, lake or other waterbody (Langbein and Iseri, 1960; Dingman, 2002; Brutsaert, 2005). A watershed comprises a set of physical, chemical and biological

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elements connected by the flow of water. This definition is analogous to the terms 'catchment' and 'drainage basin'. Watersheds are delimited based on topographic divides; thus, they represent areas that collect precipitation that contribute water to a specific location. Commonly, these surficial watershed boundaries are superimposed on aquifer boundaries, and the two often do not align. Thus, the hydrology of a watershed is not just affected by precipitation and surface water; groundwater also plays a major role. This can be seen by considering what distinguishes ephemeral from perennial and intermittent streams: ephemeral streams respond directly to precipitation, while perennial and intermittent streams receive groundwater inputs for all (perennial) or a portion of (intermittent) the year (Mosley and McKerchar, 1993; Winter *et al*., 1998; Rains and Mount, 2002; Rains *et al*., 2006; Winter, 2007). Reconciling the alignment of surficial and groundwater boundaries, understanding their interaction and assessing how humans influence each are major challenges. In this paper, we focus primarily on surficial watersheds, but inclusion of groundwater/surface water interactions is possible within this framework as consistent, nationwide datasets become available.

Watersheds are hierarchically organized (spatially nested) systems, and their delineation depends on the context of the question being asked and the scale of focus (Thoms *et al*., 2007). For example, a watershed may represent the landscape upstream of a single, first-order tributary or the area could be as large as the Mississippi River, which drains about 42% of the continental USA (Brown *et al*., 2005). While the watershed for a given point on a stream network is dictated by the drainage divide, the question of which point to choose is a matter of objectives. Therefore, any watershed characterization must be scale-dependent, contain a description of how the area was defined and include the questions, management objectives or other factors that led to the watershed delineation.

Integrity

Integrity has two general definitions. First is a *value judgement of human character* (e.g. adherence to moral and ethical principles, soundness of moral character and honesty), and second is a *judgement of condition*. It can also be defined as 'the state of being whole, entire, or undiminished' or 'a sound, unimpaired, or perfect condition' (Merriam-Webster, 1993; Dictionary.com, 2014). Of these definitions, 'judgement of condition' is most consistent with how the word is used with respect to the environment.

Aldo Leopold first used the term 'integrity' in an environmental context, writing that 'a thing is right when it tends to preserve the integrity, stability, and beauty of the biotic community. It is wrong when it tends otherwise' (Leopold 1949; p 224–225). Since Leopold's landmark essay, numerous authors have contributed to the evolving definition of integrity, especially biotic integrity (e.g. Frey, 1975; Karr and Dudley, 1981; Karr *et al*., 1986; Angermeier and Karr, 1994). With these works as a foundation, Karr (1996) defined biological integrity as 'the capacity to support and maintain a balanced, integrated, adaptive biological system having the full range of elements (genes, species, assemblages) and processes (mutation, demography, biotic interactions, nutrient and energy dynamics, and metapopulation processes) expected in the natural habitat of a region'. Karr (1981) explicitly linked the degradation of biological integrity to human-related alterations of aquatic ecosystems. Furthermore, he stated that natural variation, including natural disturbance regimes, in these systems is not considered to alter integrity. For example, streams are inherently dynamic and floods or drought of particular magnitude, frequency and duration may represent natural disturbances, but not stressors (Resh *et al*., 1988, Allan and Castillo, 2007). A similar distinction can be made for systems that are susceptible to other natural disturbance regimes, such as fires (Shinneman *et al*., 2013) or hurricanes (Scatena *et al*., 2012). In addition, biota within these systems exhibits a high degree of adaptation to such disturbances (e.g. Shinneman *et al*., 2013, Bae *et al*., 2014). Here, we likewise distinguish natural disturbances from stressors imposed by human-related activity. However, we recognize and consider human activity that alters natural disturbance regimes to be stressors (e.g. natural flooding disrupted by reservoir management). Similarly, human-induced disturbance that emulates a naturally occurring disturbance would also be considered a stressor (e.g. human-initiated forest fires).

Some uses of the term integrity imply a condition with little or no human-related alteration (Angermeier and Karr, 1994; Callicott *et al*., 1999). In reality, there are few, if any, pristine watersheds (Oliveria and Cortes, 2006), and the probability that altered watersheds could return to pre-industrial conditions with the removal of human influence is low. As such, the society must consider what level of diminished integrity and potential loss of biodiversity is acceptable in exchange for the services provided by watersheds. For some, the question narrows to the following: what level of integrity must be maintained in a system to assure the sustainable flow of services desired by the society?

Here, we define *integrity* in an environmental context as the capacity of a system (and its sub-components) to support and maintain the full range of ecosystem processes and functions essential to the long-term sustainability of its it is diversity and natural resources.

Watershed integrity

Using these separate definitions within an environmental context, we define *watershed integrity* as the capacity of a watershed to support and maintain the full range of

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ecological processes and functions essential to the sustainability of biodiversity and of the watershed resources and services provided to society. This definition positions watershed integrity as a management tool that can be used to address whether or not existing watershed infrastructure is capable of supporting sustainability goals. Note that while this definition focuses on surface water, because of how a watershed is defined, the effects of groundwater in contributing to watershed integrity are acknowledged.

RELATED TERMS

To distinguish *watershed integrity* as a management construct and clarify its role in the larger context of watershed management, a discussion of closely associated watershed conservation terms is warranted.

Biological and ecological integrity

Since Karr's seminal work (Karr 1981), biological integrity has often been used interchangeably with ecological integrity (e.g. Noss, 2004). While similar in concept, biological (or biotic) integrity only refers to the integrity of the biotic components of a system. Ecological integrity expands the domain of focus to include both biological (e.g. species abundance and composition) and physical (e.g. geological, chemical, habitat, water and sediments) components (cf. Barbour *et al*., 2000; Novotny, 2004). Initially, the concept of ecological integrity suffered from a degree of vagueness but was clarified by Woodley (1993) who stated that ecological integrity is optimized for its geographic location. A definition put forth by Schofield and Davies (1996) further clarified ecological integrity as the 'ability of aquatic ecosystems to support and maintain key ecological processes and an adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of natural habitats of the same region'. Ecological integrity has therefore emerged as a concept that encompasses the needs of well-functioning landscapes (Fischman, 2004).

The value that the society places on biological or ecological integrity is evidenced by the appearance of these terms in policy. In the USA, biological or ecological integrity have been stated as policy objectives in several national and bi-national laws and agreements, including the U.S. Water Quality Amendments of 1972. The Clean Water Act has the objective to restore and maintain the chemical, physical and biological integrity of the nation's waters (33 USC § 1251). The term biological integrity also appears in the National Wildlife Refuge System Improvement Act of 1997, which mandates that the U.S. Fish and Wildlife Service 'ensure that the biological integrity, diversity, and environmental health of the System are maintained' (16 USC § 668dd).

Aquatic condition

In environmental management, system condition is normally defined as 'the state of the system when compared to one or more benchmarks and usually includes some form of reference to integrity' (Stoddard *et al*., 2006). Monitoring of the condition of aquatic resources within watersheds can help detect trends over time, identify emerging problems, determine the success of watershed management programmes, help direct efforts for stressor control to areas where they are most needed and track the response of systems to emergencies, such as floods and spills (USEPA, 2013). The actual aquatic resource attributes included as part of a condition assessment depend on the type and scale of the question(s) being addressed by the data collection effort (Thorp *et al*., 2013), as well as the resources available. For example, bioassessment and monitoring data for reporting on the biological condition of fluvial systems are generally collected at the reach scale (Flotemersch *et al*., 2006, 2011, Parsons *et al*., in press).

These data are frequently aggregated up to larger scales to facilitate the reporting of average conditions across larger geographies, such as a watershed, state or nation (e.g. USEPA, 2014; European Commission, 2014). Although these data can be collected across the entire watershed, they are still reach-scale data indicating the aquatic condition of reaches throughout the watershed. This is in contrast to true watershed-scale data, which would result from the kinds of indicators discussed later in this paper.

Watershed resilience

The resilience of a system is defined as 'the amount of change a system can undergo (its capacity to absorb disturbance) and retain the same function, structure, and set of feedbacks' (cf. Holling, 1973; Walker and Meyers, 2004; Walker and Salt, 2012). While this concept has general acceptance within the scientific community, it is often described using different terms. For example, Begon *et al*. (2006) defined the concept using the term 'stability', which encompasses both the 'resistance' of a system (i.e. its ability to 'avoid displacement in the first place') and the 'resilience' of a system (i.e. the ability of a system that is influenced by natural and anthropogenic alterations to recover to its previous state, once the disturbances and stressors are removed). Herein, we have elected to use the term 'resilience'.

Watersheds are influenced by natural and anthropogenic stresses, and their ability to recover and adapt to new conditions is dependent on their resilience (Walker and Salt, 2012). Managing a system to maintain resilience includes protecting sensitive areas and minimizing threats (USEPA, 2012b). It also must include protecting those areas most important to the system homeostasis. For watersheds, this includes ensuring that the system has adaptive attributes

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such as river meander belts, riparian wetlands, floodplains, terraces, groundwater recharge and discharge areas, and material contribution areas, all of which function to increase or maintain its natural resilience (USEPA, 2012b). These adaptive attributes, if present in the appropriate amounts for a given system, help absorb disturbance and stresses and maintain the system in its current state (this is a property of stress-adapted systems). For example, alterations to the watershed may lead to changes in the timing, magnitude or duration of discharge that are outside the natural range of variability and predictability. In a resilient watershed, unless large-scale in character, such perturbations are unlikely to result in a permanent change to the system, because riparian areas and floodplain wetlands would help to partially absorb the disturbance and maintain its state. If the resilience is diminished because of loss of these buffering resources, then the watershed may become vulnerable to such perturbations.

Watershed sustainability

While much has been written on environmental sustainability in general, relatively little has been published in the peer-reviewed literature specifically addressing the topic of watershed sustainability. Recently, Sidle *et al*. (2013) concluded that while sustainability has been defined in many ways, all definitions commonly support 'the harmonization of environmental, economic, and social opportunities for the benefit of present and future generations'. This definition is in line with the goals outlined in the discussion of watershed health later in the text.

Watershed health and healthy watersheds

Watershed health is used as an extension of the concept of ecological or ecosystem health at the watershed scale. Karr and Chu (1999) defined ecosystem health as the preferred state of ecosystems that have been modified by human activity (e.g. farm land, urban environments and managed forests). How this 'preferred state' is derived is unclear, but it is assumed to be a consensus-driven social process and within the limits of the prevailing laws. The watershed health approach is consistent with other discussions of ecosystem health that incorporate human activities and consequences (Rapport *et al*., 1998a; Rapport *et al*., 1998b), including the EU Water Framework Directive (European Commission, 2014), as well as those that consider which societal preferences will take precedence (Lackey, 2003). It is also useful given that most ecosystems have been altered to some degree by human activities (Westra, 1994; Flotemersch *et al*., 2006; Oliveria and Cortes, 2006). A system in good health has the ability to provide a sustainable flow of services (Rapport *et al*., 1998a, 1998b) while maintaining functional and structural components at a level deemed acceptable by stakeholders (Mageau *et al*., 1995; Ross *et al*., 1997). Karr and Chu (1999) added that a healthy system should not degrade the integrity of linked resources. In the context of nested watersheds, a healthy watershed does not negatively impact the larger encompassing watershed and ideally improves its condition. This distinction is important because the definition of watershed health includes societal activities and judgments, that is, it does not imply 'natural'. When considering watershed health, measures of watershed integrity may provide information on a given watershed's ability to maintain the ecological processes and functions essential to the long-term sustainability of the resource. In contrast to watershed health, *healthy watersheds* can refer to specific watershed protection programmes (e.g. Boon, 1991; DEFRA, 2003; USEPA, 2012a). For instance, building on the efforts of many states and other organizations, the USEPA initiated the Healthy Watersheds Program that places an emphasis on helping states and others assess, identify and protect high quality or 'healthy watersheds' (USEPA, 2012a). These 'healthy watershed' initiatives seek to be cost-effective, non-regulatory means of protecting watershed resources in contrast to regulatory actions, such as the Clean Water Act of 1972. An additional objective of these programmes is to protect areas that can serve as biological refugia that support the recolonization of restored and reconnected aquatic resources (USEPA 2012b).

Watershed health is a useful framework for discussing the trade-offs made by the society when implementing ecological policy at the watershed level (Calow, 1992; Lackey, 2003). H. T. Odum's (1996) expositions on environmental accounting established that any change in an environmental system is made at the expense of other aspects of the system. More simply, because everything in the environment was already in use prior to any anthropogenic activities (Campbell, 2013), any changes made to the environment impact these pre-existing uses. In watersheds, the consideration of trade-offs must include water scarcity, conversion of natural lands to human use and ecological consequences of introduced species, among other factors (Shrader-Frechette, 1997). An assessment must be made of how such trade-offs impact the ecological processes and functions essential to the long-term sustainability of the resources (i.e. watershed integrity).

The foregoing discussion illustrates that the literature contains a number of related concepts that range from those that are solely related to ecological benchmarks (e.g. the integrity metrics and aquatic condition) to those that incorporate human values and goals (e.g. watershed sustainability and health). All of these concepts have merit and applicability, but their distinct meanings are often blurred because of a lack of standardized definitions. For the remainder of this paper, we focus on our definition of watershed integrity, specifically how to make it operational so that it can be estimated and applied to decision-making.

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MAKING WATERSHED INTEGRITY OPERATIONAL

An explicit definition of watershed integrity facilitates discussion when comparing assessed areas. However, a definition alone does 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). To make the definition of watershed integrity operational, we need to identify the key elements and processes that must be present and intact to maintain watershed production of services, determine the availability of data on those elements and processes and then execute the assessment of watershed integrity. Further, any operational definition must not require large resource investment in methods development or construction of major new datasets. This means that an assessment based on such an approach should be able to utilize existing datasets that are broadly available (e.g. nationally). We fully expect this to be an iterative process; that is, initial efforts based on our approach herein will be improved over time as additional data become available and new research provides better information on the critical relationships that are represented.

Given our definition of watershed integrity as the capacity of a watershed to support and maintain the full range of ecological processes and functions, we would like to be able to assess the status of a given watershed relative to the state required in order to fully provide this support. Such assessments require that we know what the full range of processes and functions is, or should be. One way of attaining this information would be to have some standard for comparison, such as a set of watersheds with unaltered (i.e. reference condition) characteristics and functions. This creates a conundrum, because all watersheds are altered to some degree. In the succeeding text, we discuss approaches to defining reference condition, concluding that they do not describe unaltered conditions. We then propose an approach that obviates the need for reference watersheds and allows estimation of expected values that can be used in assessments of watershed integrity.

For individual aquatic entities, such as lakes, wetlands, streams or estuaries, reference condition is typically estimated using systems that are considered to be as close to natural as possible within a region (Hawkins *et al*., 2010). The characteristics of those reference sites (determined through the collection of monitoring data) are then compared against impacted sites within similar settings (Hughes *et al*., 1986). However, practitioners have long recognized that waterbodies in pristine condition rarely exist, and so what is being described as 'reference' condition is actually 'least degraded' (Stoddard *et al*., 2006). In addition, the degree to which a site represents reference condition also varies regionally. For example, agricultural watersheds can be highly altered, resulting in reference sites that are far from reference condition (Kilgour and Stanfield, 2006). However, even sites with only slight alterations might not reflect natural conditions (e.g. Hill *et al*., 2013), depending on how the watershed responds to human-induced landscape alterations (Figure 1). Whereas programmes such as the USEPA's National Aquatic Resource Surveys (http://water.epa.gov/type/watersheds/monitoring/aquaticsurvey_index.cfm) provide field data to describe reference conditions for various waterbodies/aquatic systems, no such programme provides field-based, aquatic monitoring data at the whole-of-watershed scale. Moreover, aggregating information from these surveys to provide an estimate of watershed status may represent a major challenge.

Researchers have focused on improving assessments by selecting reference sites that maximize representativeness of both unaltered conditions (Sánchez-Montoya *et al*., 2009) and the environments of assessed sites (Johnson, 1999); hindcasting of reference condition in heavily altered landscapes (Kilgour and Stanfield, 2006); and refining modelling approaches to more precisely predict both biological (Moss *et al*., 1999) and environmental reference conditions (see review by Hawkins *et al*., 2010). At the same time, our ability to use deviations from biological and environmental reference conditions to map, quantify and link human-related watershed alterations has improved substantially in recent years (e.g. Falcone *et al*., 2010; Vander Laan *et al*., 2013). Despite these improvements, assessments can suffer from the fact that the natural condition of many systems will differ from reference condition. Additionally, given the ubiquity of human-related alterations across

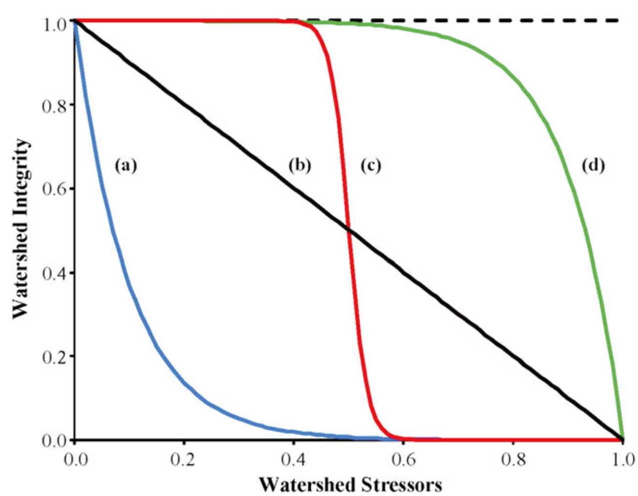

Figure 1. Examples of different responses between watershed integrity and watershed stressors: (a) low resiliency response; (b) negative linear response; (c) threshold (e.g. 'tipping point') response; and (d) high resiliency response. Reference condition is represented by the dashed line. This figure is available in colour online at wileyonlinelibrary.com/journal/rra

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landscapes (e.g. atmospheric deposition of anthropogenically derived nitrogen), we cannot describe truly unaltered conditions for most, if not all, watersheds. Therefore, we need an approach that does not require the use of reference watersheds.

We propose an approach for defining an operational definition of watershed integrity that borrows from the human health field (Van Sickle *et al*., 2006; Van Sickle and Paulsen, 2008). To assess human health, medical practitioners evaluate the condition of different vital systems, for example the circulatory, pulmonary, nervous and endocrine systems. Because the condition of these systems cannot always be directly determined through routine screenings, health practitioners often look for the presence of risk factors as a complementary means of assessment. For example, high cholesterol values, being overweight and inactive, and a history of heart disease are all risk factors related to diseases of the circulatory system. In a similar fashion, we identify key functions that watersheds provide and then identify risk factors (i.e. human-related stressors) that have been explicitly shown to interfere with and degrade these functions. In the succeeding text, we describe both these key watershed features and landscape stressors and then show how they can be used to construct an index of watershed integrity. Note that this approach partitions human stressors (e.g. urban and agricultural land use and density of roads) from natural disturbances (e.g. hurricanes, earthquakes and glaciers). Thus, a watershed that is stressed because of only natural factors could still possess high integrity (Karr, 1981).

Key watershed functions

We have identified six key functions that perform to various degrees in watersheds. A high level of watershed integrity exists when all of these functions are operating at levels that support and maintain the full range of ecological processes and functions essential to the long-term sustainability of biodiversity and ecosystem services. The six key functions are hydrologic regulation, regulation of water chemistry, sediment regulation, hydrologic connectivity, temperature regulation and habitat provision.

Hydrologic regulation (HYD). Water movement is a critical element that defines watersheds. Watersheds function to capture incoming precipitation for varying lengths of time and discharge water as either surface run-off or groundwater (Black, 1997). Below the surface, groundwater and associated constituents move through local, intermediate or regional flowpaths, which can depend on terrain and the specific geologic and hydrologic properties of the underlying soils and aquifers (Winter and LaBaugh, 2003). The details of how these processes operate in a watershed govern the supply, timing and other regime characteristics of water downstream and set the context for the physical, chemical and biological functions of the watershed. Watersheds attenuate the energy of incoming precipitation. Alterations to watershed components and their functions through the construction of impervious surfaces (Shuster *et al*., 2005), agricultural drainage (Naz *et al*., 2009) and irrigation, impoundment (Richter *et al*., 1996), stream channelization and levee construction, wetland and riparian removal (Bruland *et al*., 2003), forest clear cutting, human-caused fires and groundwater pumping all may modify the effectiveness of this attenuation. Such modification results in changes to infiltration, percolation, evapotranspiration, groundwater recharge and surface water export.

Regulation of water chemistry (CHEM). The aquatic chemistry of streams, rivers, lakes and wetlands reflects the processes operating in their watersheds, including physical weathering; plant interception, uptake and release of nutrients; soil formation and transformation; and nutrient retention by wetlands, forests and riparian areas (Grimm *et al*., 1997). The volume, timing and routing of water through a watershed has profound effects on the chemistry of downstream waters (Bormann and Likens, 1979). The retention of nitrogen (Stoddard, 1994; Mayer *et al*., 2007), phosphorus (Richardson, 1985) and ions (i.e. electrical conductivity; USEPA, 2011), in particular, is critical to the downstream water quality and integrity of biota (Minshall and Minshall, 1978). Human activities in the watershed can affect the supply (e.g. fertilizer application and mining), retention (removal of wetlands and/or riparian vegetation and channelization of streams) and processing (sewage and drinking water treatment) of these key chemical elements. In addition, chemical constituents can enter and contaminate groundwater, often with long residence times (Yetiş, 2008).

Sediment regulation (SED). The volume and composition of sediment moving through a watershed depends on a balance between the rates of supply of sediment of various sizes to the stream and the rate at which the flow of water moves them downstream, that is, the stream's sediment transport capacity relative to its sediment supply (Kaufmann *et al*., 2008). The supply rates and sizes of sediment particles delivered to a stream by upslope erosion and mass transport are influenced by basin characteristics, including geology, topography, climate, vegetative cover, run-off characteristics and land disturbances (Buffington *et al*., 2004; Frappier and Eckert, 2007). Changes to sediment regimes following catchment disturbance (either through changed sediment supply, flow modifications or physical barriers) can markedly alter the physical nature of stream channels and consequently their ability to support aquatic organisms (Norris and Thoms, 1999). In addition, fine sediment deposition can smother gravels and disconnect surface water–groundwater interactions within the hyporheic zone of streams (Brunke and Gonser, 1997). Human-related alterations of watersheds can either reduce (e.g. dam building)

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or increase (e.g. agricultural tilling, road building and channelization) sediment transport.

Hydrologic connectivity (CONN). Connectivity is the property that spatially and temporally integrates all of the individual components of a watershed (USEPA 2015). The concept of connectivity builds on classic ideas about how river systems are integrated, such as the river continuum (Vannote *et al.*, 1980), serial discontinuity (Ward and Stanford, 1995) and flood pulse (Junk *et al.*, 1989) concepts. This integration is achieved through various transport mechanisms that function across multiple spatial and temporal scales to deliver materials and energy between watershed components. Connectivity can be described in terms of the frequency, duration, magnitude, timing and rate of change of energy, water, material and biotic fluxes, which are naturally controlled by climate, geology and terrain. A recent EPA report (USEPA 2015) found that streams and wetlands can have strong influences on the integrity of downstream waters through hydrological, chemical or biological connectivity. However, the degree of those connections varies depending on the environment and human activities (Ward and Stanford, 1995).

Water is the dominant transport mechanism for movement of energy and materials within watersheds. Hydrologic connectivity describes the hydrologically mediated transfer of matter, energy and/or organisms within or between elements of the watershed (Pringle, 2003). Aquatic networks are characterized by a high degree of spatial and temporal heterogeneity with mass, energy and organisms flowing in four dimensions (Nadeau and Rains, 2007): longitudinally (i.e. downstream), laterally (i.e. channel to/from riparian and/or floodplain areas), vertically (i.e. to/from groundwater through the hyporheic zone) and over time. Human uses of the watershed can decrease (e.g. by isolating streams and wetlands from hyporheic zones through arroyo cutting; Wallace *et al.*, 1990) or increase (e.g. through ditching or conversion of ephemeral to perennial streams by effluent release) these linkages. Likewise, groundwater pumping directly affects the surface water–groundwater connection (Kirk and Herbert, 2002). Excessive pumping can deplete aquifers and disconnect streams from groundwater sources (Wallace *et al.*, 1990). This disconnection can reduce base flow in surface waters (Kirk and Herbert, 2002), thereby increasing vulnerability to disturbances, such as drought (Scanlon *et al.*, 2012) and climate change (Hill *et al.*, 2014; Leibowitz *et al.*, 2014).

Temperature regulation (TEMP). Temperature is a primary control on the structure and function of ecological communities (Brown *et al.*, 2004) as well as the chemical composition and transformations within a watershed (Demars *et al.*, 2011). The maintenance of coldwater (Epifanio, 2000) and warmwater (Quinn and Kwak, 2003) fisheries, for instance, are both affected by a watershed's ability to regulate water temperature. Watersheds intercept incoming solar radiation and attenuate it through various processes, including riparian shading and evaporative cooling, variations in groundwater infiltration and input, and microclimatic effects (Caissie, 2006). Streamside vegetation removal, dam operations (e.g. hypolimnetic versus epilimnetic flow release), disruption of hydrologic connectivity, effluent release (e.g. from power generation, wastewater or agricultural return flows) and urbanization all have strong influences on water temperature (Poole and Berman, 2001).

Habitat provision (HABT). Natural systems have the ability to convert, transform and organize raw materials and energy into a multitude of habitats that provide for the basic life history requirements of organisms, such as food, shelter and breeding areas. The physical habitat of streams, rivers, wetlands and the near-shore areas of lakes and coastal waters are strongly controlled by watershed structure and function (Allan, 2004). While some aspects of aquatic habitats (e.g. chemical composition and temperature) are included in the previously discussed watershed functions, creating and maintaining the physical structure of habitats are additional critical functions of watersheds. For example, the supply of large woody debris to streams is a key driver of the habitat complexity necessary for native fish species survival (Van Sickle and Gregory, 1990). The presence of dams and reservoirs and impacts to riparian vegetation through roads and urban and agricultural land use are examples of stressors that can affect the habitat provision function.

ASSESSING WATERSHED INTEGRITY: AN OPERATIONAL APPROACH

Given these six functions, we can define an index of watershed integrity as the following:

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

where WI represents the overall integrity of the watershed, scaled between 0 and 1 (with higher values indicating greater integrity), and WI_i represents the integrity with respect to the i-th function, with the subscripts referring to the six previously described functions. Note that we take the product of these six values, rather than the sum, because each of these functions is a critical component of watershed integrity and the functions are not substitutable (a complete loss of any one component would cause watershed condition to decline to zero). This is analogous to human health where, for example the circulatory, pulmonary, nervous and endocrine systems are all essential to health, and the failure of

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any one of these systems can cause death. One result of using a product, and the fact that each quotient is scaled from zero to one, is that it is not necessary to include separate weighting factors for each of the six terms in Equation (1).

Because we cannot describe the expected level of each of these six functions for conditions that no longer exist, our approach is to identify and evaluate human-related alterations of watersheds that are known to modify these key functions in ways that degrade watershed integrity (i.e. stressors; Table I). As stressors—such as impervious surface, land use change, wetland loss and stream channelization—are added to a watershed, its integrity declines. The particular shape of the curve (Figure 1) describing the response of watershed condition to a stressor varies, depending on the sensitivity of the watershed to a specific stressor and the mechanism by which each stressor acts on watershed condition (Allan, 2004). For example, Norris and Thoms (1999) cited both linear responses of stream biota (Marchant *et al*., 1997; Karr and Chu, 1999) and non-linear responses of stream sediments (Thoms, 1987) to stressors to develop a conceptual framework of river health that is similar to our concept of watershed integrity (cf. Figure 1 of Norris and Thoms, 1999). In addition, stressors can interact to exacerbate the responses of stream biota to stressor gradients (e.g. Merovich and Petty, 2007; Townsend *et al*., 2008; Piggott *et al*., 2012; Lange *et al*., 2014; Lawrence *et al*., 2014). It is critical to understand how the presence of multiple stressors influences each of the critical watershed functions, because stressors typically do not occur or act in isolation.

In the simplest case, where a watershed is subject to a single stressor j that impacts a single function i, the condition of the watershed with respect to that function is given by

$$WI_i = f_{i,j}\left(\frac{s_{j,obs}}{s_{j,max}}\right) \tag{2}$$

where $s_{j,obs}$ and $s_{j,max}$ are the observed and maximum values of stressor j, respectively, and $f_{i,j}$ is a single-variable mathematical function describing the relationship between function i and stressor j (e.g. the curves in Figure 1). Note that $s_{j,max}$ is used to scale the x-axis in Figure 1 and that $s_{j,obs}/s_{j,max}$ ranges from 0 (unaltered) to 1 (maximum impact). There is a negative relationship between $f_{i,j}$ and $s_{j,obs}/s_{j,max}$; that is, low stressor values correspond to high WI_i values (Figure 1). Stressor values could include the number of dams, length of roads and watershed imperviousness (see Table I for other examples). For stressors such as impervious surface that are based on a proportion, the value for $s_{j,max}$ would be 1. For other stressors, there is no theoretical limit on the upper number that is possible; for example, there could be an arbitrarily large number of roads per unit area. In such cases, $s_{j,max}$ could be the maximum value observed regionally or nationally. Alternatively, a maximum value could be assigned based on what is considered to be the maximum feasible value, dependent on what is physically or socially practical. Note that for any unit where the observed value is equal to the maximum, then watershed condition will be equal to zero, because $s_{j,obs}/s_{j,max} = 1$, $WI_i = 0$ and therefore $WI = 0$ (from Equation (1)).

Although loss of natural infrastructure, such as wetlands, can act as a stressor on the key watershed functions, we do not include these as such, because we do not know the distribution of natural infrastructure under unaltered conditions. While we could use least degraded reference sites to estimate the natural distribution, this would defeat the purpose of our approach, which was to avoid the use of such sites. For stressors that result from losses of natural infrastructure, we instead link these back to the human actions causing those losses. For example, drainage ditches and tile drainage, urbanization and agricultural conversion are stressors causing wetland loss. Presence of built infrastructure can be measured in an absolute sense because it did not exist in the unaltered watershed. For a situation with n multiple stressors, Equation (2) is expanded as follows:

$$WI_i = \sum_{j=1}^{n} g_{i,j}\left(\frac{s_{j,obs}}{s_{j,max}}\right)\Big| s_{k\forall k \neq j} \tag{3}$$

where $g_{i,j}$ is a mathematical function similar to $f_{i,j}$, except that it incorporates the conditional effects of other stressors on the response of WI_i to s_j; and the last terms mean given ('|') the value of s_k for all ('∀') k not equal to j (i.e. for all other stressors besides s_j). As the response of WI_i to s_j is conditioned by these other stressors, it is not necessary to add weighting factors to each sum. Given this, we can combine Equations (1) and (3) into the following:

$$WI = \prod_{i=1}^{6}\left(\sum_{j=1}^{n} g_{i,j}\left(\frac{s_{j,obs}}{s_{j,max}}\right)\Bigg| s_{k\forall k \neq j}\right) \tag{4}$$

where the product ('$\prod$') is taken over the six watershed functions.

To evaluate WI for watersheds nationally, the major stressors affecting each of the six key functions must be identified, landscape indicators of each stressor must exist from nationally available datasets and the relationship between each function and stressor ($g_{i,j}$) must be known. Table I contains a proposed list of the main stressors affecting each of the six key functions. For each function, stressors are organized by those occurring within and outside of the channel. Included with each stressor is a specific national dataset that could be used to evaluate it. In some cases, no such indicator exists. However, we still include these stressors in Table I, in order to identify missing data layers that need to be developed. Also, explicitly including

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Table I. Key functions that occur in unaltered watersheds and the major stressors affecting these functions

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	• Presence and volumes of reservoirs (NID) • Stream channelization and levee construction (NA)	• Percent of the watershed comprising urban and/or agricultural land use (NLCD) • Total length and density of roads (TIGER) • Total length and density of canals/ditches (NHD) • Percent imperviousness of human-related landscapes (NLCD) • Housing unit density (TIGER) • Wetland and riparian removal* (NHD, NLCD) • Boundaries, depths and flows of aquifers (NA) • Groundwater use (GW)
Regulation of water chemistry (CHEM)	Maintenance of the natural timing, supply and storage of the major chemical constituents of freshwaters: nutrients (nitrogen and 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.	• Presence and volumes of reservoirs (NID) • Stream channelization and levee construction (NA)	• Atmospheric deposition of anthropogenic sources of nitrogen and acid rain (NADP) • Percent of watershed composed of urban and agricultural land uses (NLCD) • Percent imperviousness of human-related landscapes (NLCD) • Fertilizer application rates (FERT) • Presence and density of wastewater treatment facilities (NPDES), industrial facilities (TRI), superfund sites (SUPERFUND) and mines (MINES) • Cattle density (CATTLE) • Wetland and riparian removal* (NHD, NLCD) • 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.	• Presence and volumes of reservoirs (NID) • Stream channelization and levee construction (NA)	• Alteration to and spatial arrangement of riparian vegetation (LANDFIRE) • Presence and density of mines (MINES), logging (FOREST) and roads (TIGER) • Agriculture (NLCD) weighted by soil erodibility (STATSGO), landscape slope (NED) and proximity to waterbodies (NHD)
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).	• Presence and height of reservoirs (NID) •Stream channelization and levee construction (NA) • Road/stream intersections (TIGER/NHD) weighted by channel slope (NHD)	• Alteration to and spatial arrangement of riparian vegetation (LANDFIRE) • Density of ditches/canals (NHD) • Groundwater use (GW) • Presence and density of wastewater discharge sites (NPDES) • Wetland and riparian removal* (NHD, 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.	• Presence and volume of reservoirs (NID)	• Alteration to and spatial arrangement of riparian vegetation (LANDFIRE) • Percent of watershed composed of urban and/or agricultural land uses (NLCD) • Percent imperviousness of human-related landscapes (NLCD) • Groundwater use (GW) • Presence and density of wastewater discharge sites (NPDES)

(*Continues*)

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Table I. (Continued)

Key function	Description	Major stressors	
		Within channel	Outside channel
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.	• Presence, height and volume of reservoirs (NID)	• Alteration to and spatial arrangement of riparian vegetation (LANDFIRE) • Presence/absence of native vegetation types within riparian zones (LANDFIRE) • Density of human populations and housing unit developments within riparian zones (TIGER) • Percent of watershed composed of urban and agricultural land uses (NLCD) • Density of road/stream intersections (TIGER/NHD) • Density of roads within riparian zones (TIGER)

Data sources that can be used to evaluate the stressors are included parenthetically (see key at succeeding note).
*Indicators for wetland and riparian removal are linked back to the built infrastructure causing those losses, because wetland and riparian distribution under unaltered conditions is generally not known.
CATTLE, USDA National Agriculture Statistics Service (http://www.nass.usda.gov/Charts_and_Maps/Cattle); FERT, County-level estimates of N & P from commercial fertilizer (http://pubs.usgs.gov/sir/2012/5207); FOREST, University of Maryland Global Forest Change map 2000–2012 (http://earthenginepartners.appspot.com/science-2013-global-forest/download.html); GW, Total groundwater usage by US county in yr. 2005 (http://water.usgs.gov/watuse/data/2005); LANDFIRE, USFS and USDOI LANDFIRE Program (http://www.landfire.gov); MINES, USGS Mines Dataset (https://www.sciencebase.gov/catalog/folder/4f4e4767e4b07f02db47e0ad); NA, No nationally available data are available for specified stressor to the best of our knowledge; NAPD, National Atmospheric Deposition Program National Trends Network (http://nadp.sws.uiuc.edu/data/ntn); NED, National Elevation Dataset as distributed with NHD (http://www.horizon-systems.com/NHDPlus/NHDPlusV2_home.php); NHD, National Hydrography Dataset (http://www.horizon-systems.com/NHDPlus/NHDPlusV2_home.php); NID, US Army Corps of Engineers National Inventory of Dams (http://geo.usace.army.mil/pgis/f?p=397:1:0:); 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); STATSGO, USGS State Soil Geographic Data (http://datagateway.nrcs.usda.gov); SUPERFUND, USEPA Superfund Sites (http://www.epa.gov/enviro/geo_data.html); TIGER, US Census Bureau TIGER/Line Program (http://www.census.gov/geo/maps-data/data/pdfs/tiger/tgrshp2013/TGRSHP2013_TechDoc.pdf); TRI, National Toxic Release Inventory (http://www.epa.gov/enviro/geo_data.htm); All websites accessed June 18, 2014.

them allows the effect of these missing stressors to be considered.

The specific relationship between the key functions and each stressor is largely unknown, although we may know $g_{i,j}$ for specific functions and stressors in specific regions. For example, Carlisle *et al*. (2009) found a non-linear response between biological alteration and high-density residential land cover within riparian zones of eastern US highlands: there was a strong effect from 0% to 10% high-density residential areas that reached an asymptote at larger values (Figure 2). Note that, because the relationship in Figure 2 is based on a partial dependence plot, the response takes the conditional effect of other stressors into account (Hastie *et al*., 2009), and so could serve as a g function.

Given the reality that $g_{i,j}$ is largely unknown, we define the following estimator of WI:

$$\widehat{WI} = \prod_{i=1}^{6}\left(\sum_{j=1}^{n} 1 - \left(\frac{s_{j,obs}}{s_{j,max}}\right)\right) \tag{5}$$

where $\widehat{WI}$ is our first-order estimate of WI, which assumes a negative linear relationship for $g_{i,j}$ and independence among stressors. This estimator provides us with an operational definition of watershed integrity that can be incrementally improved as either specific $g_{i,j}$ relationships are reported or new datasets become available. Because Equation (5) does not include the conditional effects of other stressors, an alternative approach would be to give stressors different weights:

$$\widehat{WI} = \prod_{i=1}^{6}\left(\sum_{j=1}^{n} \alpha_{i,j}\left[1 - \left(\frac{s_{j,obs}}{s_{j,max}}\right)\right]\right) \tag{6}$$

where $\alpha_{i,j}$ is a weighting factor for stressor j with respect to function i.

If $g_{i,j}$ is known for some, but not all stressors, then a hybrid version could be used that combines $g_{i,j}$ with the first-order, weighted estimators. For example,

$$\widehat{WI}_i = \sum_{j=1}^{m} g_{i,j}\left(\frac{s_{j,obs}}{s_{j,max}}\right)\Bigg|_{s_{k\forall k\neq j}} + \sum_{j=m+1}^{n} \alpha_{i,j}\left[1 - \left(\frac{s_{j,obs}}{s_{j,max}}\right)\right] \tag{7}$$

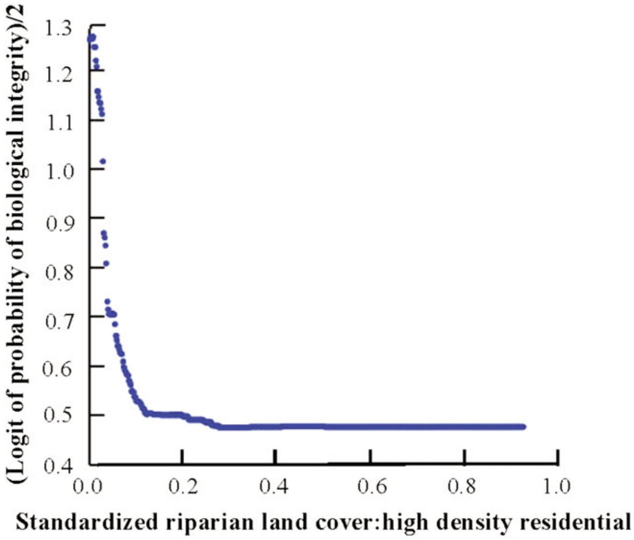

Figure 2. Partial dependence plot showing the relationship between biological integrity and standardized high-density residential land use within riparian zones of the eastern USA. Modified from Carlisle *et al*., 2009. This figure is available in colour online at wileyonlinelibrary.com/journal/rra

where the index m refers to the number of stressors for which $g_{i,j}$ is known.

Finally, we note that there may be instances where the condition of a watershed with respect to a specific key function could be modelled directly. For example, Hill *et al.* (2013) developed a stream temperature model for the USA that, for an individual site, could estimate both the observed temperature and the expected temperature at reference condition. The relationship between these two could then be used as a direct estimate of WI_{TEMP}:

$$WI_{TEMP} = \frac{|TEMP_{exp} - TEMP_{obs}|}{TEMP_{exp}} \tag{8}$$

where $TEMP_{exp}$ and $TEMP_{obs}$ are the expected (i.e. reference condition) and observed temperatures, respectively. Equation (8) assumes that a positive deviation from the expected value has the same effect as a negative deviation. Other formulations are possible; for example, an equation that has separate responses for $TEMP_{obs}$ greater and less than $TEMP_{exp}$ could be used to give greater weight to warmer temperatures. Equation (8) also assumes that the effects of a given change in temperature are dependent on the expected temperature; for example, a 2 °C change has worse effects at warmer expected temperatures than for cooler expected temperatures. Again, alternative formulations could be used; for example, the expected temperatures could be scaled to one to provide uniform results.

An approach such as Equation (8) is appealing, because it reduces the complexity of dealing with multiple stressors and avoids the problem of having to assume that $g_{i,j}$ is a negative linear relationship. The problem is that this particular modelling approach used reference sites that represent least degraded condition (Hill *et al*., 2013), because unaltered sites did not exist. As previously mentioned, these sites do not fully reflect natural condition, depending on the magnitude of the alteration and the watershed response (Figure 1), which would result in biased WI_{TEMP} estimates. Still, results from a model-based approach (e.g. Equation (8)) could be compared with results using the stressor-based approach (use of Equation (3) with the negative linear assumption) to better understand the strengths and weaknesses of each approach.

As a proof-of-concept study, we are conducting a national assessment of watershed integrity based on this approach and using the StreamCat dataset (Hill *et al*., in press), which was developed, in part, for this purpose. StreamCat contains over 160 landscape variables for all 2.6 million National Hydrography Dataset Plus Version 2 (NHDPlusV2; McKay *et al*., 2012) catchments in the contiguous USA and is being augmented to contain all of the available stressor variables in Table I.

SUMMARY AND CONCLUSIONS

Watersheds encompass all biotic and abiotic components—including people—within their boundaries (*sensu* Likens 1992) and provide a range of services valued by the society. Thus, watersheds are a foundation of our cultural, economic, spiritual and social well-being (Likens *et al*., 2009) and, as such, a critical focus of water resource management. Evaluating the integrity of a watershed represents a societal and a scientific challenge, because watersheds are used and managed for a diverse, interrelated range of activities. The concept of watershed integrity is challenging both in terms of defining what it actually is and developing approaches and tools to allow its evaluation. The assessment of watershed integrity also requires approaches that are scale appropriate (Dollar *et al*., 2007) and enable strategic problem-solving. To paraphrase Likens (1992), the ultimate challenge for watershed and river science is to integrate and synthesize watershed and river information available from all levels of inquiry into an understanding that is meaningful and can be used by managers and decision-makers.

There are a number of terms related to this challenge (e.g. watershed integrity, aquatic condition and watershed health) that have separate meanings. However, these meanings are often blurred because of a lack of standardized definitions. Here, we provide distinct definitions for these different terms, which should allow the scientific community to better address the needs of decision-makers. This includes our main focus, watershed integrity, which we define 'as the capacity of a watershed to support and maintain the full

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range of ecological processes and functions essential to the sustainability of biodiversity and of the watershed resources and services provided to society'. We developed an operational index to evaluate the level of integrity. For this index, we used a human health analogy to identify six key watershed functions (viz. hydrologic regulation, regulation of water chemistry, sediment regulation, hydrologic connectivity, temperature regulation and habitat provision) and the specific risk factors, or stressors, which impact them. Examples of national datasets that can be used to evaluate these risk factors were provided. We derived a mathematical expression (Equation (4)) that evaluates watershed integrity by combining the integrity of the six watershed functions, where the integrity of each is based on the relative presence of specific stressors. This expression assumes the quantitative relationships between multiple interacting stressors and the key functions are known. Although such relationships do exist (e.g. Figure 2), the availability of this information is limited and represents an ongoing need for additional research. However, we also show how first-order approximations can be used in the absence of such information, given that many of these relationships are as yet unknown. The approach can be iteratively applied and improved as these relationships become known and/or as new national datasets become available. For example, the version of the index that uses first-order approximations (Equation (5)) could be replaced as information on weightings (Equation (6)) or conditional effects of other factors (Equations (7)) becomes available. A significant aspect of our approach is that it allows us to avoid the use of least degraded sites to describe reference condition.

A characteristic of our index is that it can be readily deconstructed in support of strategic adaptive management; that is, the risk factors of the watershed index score can be examined individually to evaluate how each is influencing the overall index score. This affords managers the opportunity to examine how each factor might be managed differently to better achieve overall objectives. For example, if the index score for a given watershed is below an established threshold, the factors bringing the overall index score down can easily be identified. If multiple factors are negatively influencing the overall score, a cost–benefit analysis of available management options then could be conducted to identify the best course of action to achieve desirable management endpoints.

Previous efforts to assess the status of watersheds on larger scales, though limited in number, have been undertaken in the USA and elsewhere (e.g. Garrido Pérez *et al.*, 2010, for México). For example, in the USA, the U.S. Fish and Wildlife Service, along with the National Marine Fisheries Service and the Association of Fish and Wildlife Agencies, developed and implemented the National Fish Habitat Action Plan (Association of Fish and Wildlife Agencies, 2006). Its objective was to protect, restore and enhance the nation's fish and aquatic communities through partnerships to foster fish habitat conservation. Similarly, the U.S. Forest Service developed the Watershed Condition Framework (USFS, 2011), which employed an integrated, systems-based approach for classifying watershed condition based on an evaluation of underlying ecological, hydrological and geomorphic states. These programmes focus solely on identifying high-quality aquatic resources and make use of locally collected data (e.g. biota and water quality) that are not available nationally. Our intent was to develop a watershed integrity index that can be applied nationally. This approach differs because it focuses on the functional attributes of these systems, rather than simply relying on more easily obtainable state components. It also differs technically through its use of appropriately scaled data that identify risk factors that impact these key functions. Beyond informing on the biogeochemical health of the immediate system, a focus on functional attributes provides information that can be used to explain and predict the effects of watershed discharges on downstream riverine and estuarine condition (Gregory *et al.*, 1991).

Data resulting from a national-scale mapping and assessment of watershed integrity should be of value to states and watershed organizations initiating healthy watershed programmes (USEPA, 2012a). These programmes 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 that uses our approach—which we are currently in the process of conducting, using the StreamCat (Hill *et al.*, submitted) database—would help identify watersheds with high integrity as well as those at risk and help to prioritize conservation and restoration. The index of watershed integrity developed here should provide high-quality information at a broad spatial scale that will help allow ecosystem management to focus on achievable and measureable outcomes underpinned by quality science.

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