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How much water does a river need?

BRIAN D. RICHTER*
Biohydrology Program, The Nature Conservancy, PO Box 430, Hayden, Colorado 81639, U.S.A.

JEFFREY V. BAUMGARTNER, ROBERT WIGINGTON
The Nature Conservancy, 2060 Broadway, Suite 230, Boulder, Colorado 80302, U.S.A.

DAVID P. BRAUN
The Nature Conservancy, 1815 N. Lynn St, Arlington, Virginia 22209, U.S.A.

SUMMARY

1. This paper introduces a new approach for setting streamflow-based river ecosystem management targets and this method is called the 'Range of Variability Approach' (RVA). The proposed approach derives from aquatic ecology theory concerning the critical role of hydrological variability, and associated characteristics of timing, frequency, duration, and rates of change, in sustaining aquatic ecosystems. The method is intended for application on rivers wherein the conservation of native aquatic biodiversity and protection of natural ecosystem functions are primary river management objectives.
2. The RVA uses as its starting point either measured or synthesized daily streamflow values from a period during which human perturbations to the hydrological regime were negligible. This streamflow record is then characterized using thirty-two different hydrological parameters, using methods defined in Richter *et al.* (1996). Using the RVA, a range of variation in each of the thirty-two parameters, e.g. the values at ± 1 standard deviation from the mean or the twenty-fifth to seventy-fifth percentile range, are selected as initial flow management targets.
3. The RVA targets are intended to guide the design of river management strategies (e.g. reservoir operations rules, catchment restoration) that will lead to attainment of these targets on an annual basis. The RVA will enable river managers to define and adopt readily interim management targets before conclusive, long-term ecosystem research results are available. The RVA targets and management strategies should be adaptively refined as suggested by research results and as needed to sustain native aquatic ecosystem biodiversity and integrity.

Introduction

The development and management of water resources by humans has altered the natural flow of rivers around the world (e.g. United States: Sparks, 1992; Australia: Walker, Sheldon & Puckridge, 1995; Africa: Petitjean & Davies, 1988; Bruwer & Ashton, 1989; Davies, O'Keeffe & Snaddon, 1993; Mexico: Contreras & Lozano, 1994; Europe: Dynesius & Nilsson, 1994; Asia: Chen & Wu, 1987; Dudgeon, 1992, 1995; global: L'vovitch & White, 1990; Postel, 1995; Abramovitz, 1995), and the impacts of such flow alteration on river biota have been well documented (Ward & Stanford, 1979; Lillehammer & Saltveit, 1984; Petts, 1984; Cushman, 1985; Calow & Petts, 1992). For example, modification in the timing, frequency or duration of floods

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can eliminate spawning or migratory cues for fish, or reduce access to spawning or nursery areas (Junk, Bayley & Sparks, 1989). Increased frequency or duration of high flow levels may displace velocity-sensitive organisms, such as some periphyton, phytoplankton, macrophytes, macroinvertebrates, young fish and deposited eggs (Moog, 1993; Allan, 1995).

A growing need to predict the biological impacts (or recovery) associated with water management activities, and to set water management targets that maintain riverine biota and socially valuable goods and services associated with riverine ecosystems, has spawned what amounts to a new scientific discipline of 'instream flow' modelling and design. The primary application of instream flow–habitat models has been the design of 'environmentally acceptable' flow regimes to guide river management, e.g. to manage reservoir operations and water diversions. Unfortunately, recent advances in understanding the relationships between hydrological variability and river ecosystem integrity (as summarized in Poff & Ward, 1989; NRC, 1992; Stanford *et al.*, 1996) have had minimal influence on the setting of instream flow requirements or on river ecosystem management.

Virtually all models and methods for setting instream flow requirements in common use today have been criticized for their overly simplistic and reductionist treatment of complex ecosystem processes and interactions (Mathur *et al.*, 1985; Orth, 1987; Gore & Nestler, 1988; Arthington & Pusey, 1993; Stanford, 1994; Castleberry *et al.*, 1996; Williams, 1996). Although these methods may be useful for assessing the flow requirements of some individual species, they provide little insight into complex ecosystem dynamics that involve multivariate habitat influences, complex and varied life histories of riverine species, biotic interactions, geomorphic change and other potentially critical factors. The potential use of long-term streamflow data and statistical descriptions of natural flow variability to set ecosystem-based management targets has been underutilized or ignored in the vast majority of river management decisions (NRC, 1992).

In this paper, a new method for developing streamflow-based river management targets is proposed that incorporates the concepts of hydrological variability and river ecosystem integrity. The method, referred to as the 'Range of Variability Approach', or RVA, begins with a comprehensive characterization of ecologically relevant attributes of a flow regime and then translates these attributes into more simple, flow-based management targets. These targets are subsequently used as guidelines for designing a workable management system capable of attaining the desired flow conditions. The RVA will be most useful for setting preliminary or interim flow targets for river reaches with highly altered hydrological regimes, i.e. where one or more annual streamflow characteristics frequently fall outside their historic range(s) of variability. Application of the RVA will be most appropriate when protection of native riverine biodiversity and natural ecosystem functions are primary management objectives. The method readily lends itself to adaptive management. Preliminary flow-based management targets can be identified through use of the RVA; once implemented, these targets subsequently can be refined through site-specific ecosystem research designed to test hypotheses about: (i) the ability of the designed management system to achieve the desired flow conditions, and (ii) biotic and ecosystem dependencies on flow variation (Arthington & Pusey, 1994; Richter *et al.*, 1996). The RVA should be used in lieu of habitat models or other instream flow modelling approaches when conservation of native biota and ecosystem integrity are management objectives.

Before describing the RVA in detail, the ecological underpinnings of the method are summarized and followed by a brief review of a sample of other recently applied river ecosystem management approaches and their shortcomings. After describing the RVA, its application is discussed under different scenarios of availability of historic streamflow records, and its application is illustrated with a case study.

Aquatic ecosystem integrity and the natural flow paradigm

Native riverine species possess life history traits that enable individuals to survive and reproduce within a certain range of environmental variation (Townsend & Hildrew, 1994; Stanford *et al.*, 1996). A myriad of environmental attributes are known to shape the habitat templates (*sensu* Southwood, 1977, 1988) that control aquatic and riparian species distributions, including flow depth and velocity, temperature, substrate size distributions, oxygen content, turbidity, soil moisture/saturation, and other physical and chemical conditions and biotic influences (Allan, 1995). Hydrological variation plays a major part in structuring the biotic diversity within river ecosystems as it controls key habitat conditions within the river channel, the floodplain, and hyporheic (stream-influenced groundwater) zones (Poff & Ward, 1989; Arthington & Pusey, 1994; Townsend & Hildrew, 1994; Richter *et al.*, 1996; Stanford *et al.*, 1996). The often-strong connections between streamflow, floodplain inundation, alluvial ground water movement, and water table fluctuation mediate the exchange of organisms, particulate matter, energy, and dissolved substances along the upstream–downstream, river-floodplain, river-hyporheic, and temporal dimensions of riverine ecosystems (Ward & Stanford, 1983, 1995; Ward, 1989; Sparks *et al.*, 1990; Stanford & Ward, 1992, 1993; Walker *et al.*, 1995).

Because fluvial processes maintain a dynamic mosaic of channel and floodplain habitat structures (Leopold, Wolman & Miller, 1964), creating patchy and shifting distributions of environmental factors that sustain diverse biotic assemblages, hydrological variation is now recognized as a primary driving force within riverine ecosystems (Sparks *et al.*, 1990; Gosselink *et al.*, 1990; Schlosser, 1991; NRC, 1992; DeAngelis & White, 1994; Sparks, 1995; Stanford, *et al.*, 1996). While river ecosystem management or restoration efforts that focus exclusively on flow management are unlikely to succeed, river management objectives related to ecosystem integrity cannot be met without maintaining or restoring hydrological integrity (NRC, 1992). Consequently, perpetuation of native aquatic biodiversity and ecosystem integrity depends on maintaining or restoring some semblance of natural flow variability (e.g. Minckley & Meffe, 1987; Sparks, 1992, 1995; Kingsolving & Bain, 1993; Walker & Thoms, 1993; Walker *et al.*, 1995; Richter *et al.*, 1996; Stanford *et al.*, 1996). The potential for survival of native species and natural communities is reduced if the environment is pushed outside the range of its natural variability (Resh *et al.*, 1988; Swanson *et al.*, 1993).

Accumulated research on the relationship between hydrological variability and river ecosystem integrity overwhelmingly suggests a *natural flow paradigm*, which states: *the full range of natural intra- and interannual variation of hydrological regimes, and associated characteristics of timing, duration, frequency and rate of change, are critical in sustaining the full native biodiversity and integrity of aquatic ecosystems*. Advocates for using natural variability of ecosystems as a guide for ecosystem management (e.g. Swanson *et al.*, 1993; Morgan *et al.*, 1994; Stanford *et al.*, 1996) express the perspective that 'managing an ecosystem within its range of natural variability is an appropriate path to maintaining diverse, resilient, productive, and healthy systems' (Swanson *et al.*, 1993). Thus, if conservation of native biodiversity and ecosystem integrity are objectives of river management, then river management targets must accommodate the natural flow paradigm.

Prescribing flows for river ecosystems

Translating the natural flow paradigm into management targets requires decomposing the temporal complexity inherent in a streamflow regime into ecologically meaningful and manageable parts. Numerous streamflow characteristics are presumably important for the maintenance and regeneration of riverine habitats and biological diversity, including: the seasonal patterning of flow; timing of extreme conditions; the frequency, predictability, and duration of floods, droughts, and intermittent flow; daily, seasonal, and annual flow variability; and rates of change (Resh *et al.*, 1988; Poff & Ward, 1989; Arthington & Pusey, 1994; Walker *et al.*, 1995; Richter *et al.*, 1996).

Streamflow characteristics offer some of the most useful and appropriate indicators for assessing river ecosystem integrity over time, for several reasons. First, as discussed previously, many other abiotic characteristics of riverine ecosystems vary with streamflow conditions, including dissolved oxygen levels, water temperature, suspended and bed-load sediment size distributions, and streambed stability (Ward & Stanford, 1983; Sparks, 1992; Nestler, Schneider & Latka, 1994; Allan, 1995; Richter *et al.*, 1996). * Second, on a larger scale, channel and floodplain morphology is shaped by fluvial processes driven by streamflow, particularly high-flow conditions * (Leopold *et al.*, 1964). Third, in contrast to the comparative paucity, recency and coarse temporal resolution of biological data sets, the availability of long-term daily records of streamflow on many larger (fourth to tenth order) rivers can provide powerful insights into natural variability and the recent history of human perturbations on a river.

There exist numerous methods for setting streamflow-based river management targets, none of which sufficiently addresses the full natural range of variability in hydrological regimes. Here the present study reviews a few of the methods to illustrate the range of approaches and their shortcomings. For a more complete overview, see Gordon, McMahon & Finlayson (1992).

Many instream flow models or methodologies are extremely simplistic, such as the 'Montana Method' (Tennant, 1976), wherein environmental flow regimes are prescribed on the basis of the average daily discharge or the mean annual flow (MAF). In general, 10% of the MAF is recommended as a minimum instantaneous flow to enable most aquatic life to survive; 30% MAF is recommended to sustain good habitat; 60–100% MAF provides excellent habitat; and 200% MAF is recommended for 'flushing flows'. Such approaches have obvious shortcomings, the most serious being the elimination of ecologically important flow extremes and a lack of attention to flow timing.

One of the most technologically sophisticated and widely applied modelling approaches is the Instream Flow Incremental Methodology (IFIM), developed by the U.S. Fish and Wildlife Service (Bovee, 1982). The IFIM is one of a family of approaches that use (across-river) transect-based hydraulic analyses to evaluate basic habitat conditions (e.g. depth, velocity) associated with varying levels of flow. Based upon limited field sampling of fish locations and associated habitat conditions, curves depicting habitat preferences are developed. These curves are then used to predict habitat availability at different flow levels

A variant of the IFIM approach, called the 'Riverine Community Habitat Assessment & Restoration Concept' (RCHARC), has been applied to the Missouri River (U.S.A.) (Nestler *et al.*, 1994). The primary contribution of the RCHARC is the acknowledgment that the spatial distribution and abundance of certain depth and velocity conditions can radically change as a river is morphology changes, particularly under human influences such as damming and channelization. The RCHARC study on the Missouri was used to identify the modern-day flow regime necessary to provide some semblance of pre-dam velocity and depth distributions. All such transect-based models assume stable channels; they characterize habitat in limited terms such as depth and velocity; and they perform better when the habitat requirements of the modelled species at different life stages are known. A recent critique in Williams (1996) further suggests that chance locations of sampling transects can result in meaningless conclusions about the habitat area available.

Hill, Platts & Beschta (1991) suggested that instream flow prescriptions be based on four considerations: instream (base) flows for fisheries, channel maintenance (bankfull) flows, riparian (floodplain inundation) flows, and valley maintenance (> 25 yr flood) flows. They described a variety of strategies for estimating each of these flow levels, which would be cumulatively summed to create a management scheme for instream flows. This approach addresses the fact that river ecosystems are structured by a large range of hydrological variation. However, the authors make no mention of the necessary_duration of high or low flows, nor do they acknowledge the significance of daily or seasonal variation when prescribing flows to sustain aquatic organisms.

Arthington *et al.* (1991) proposed an 'holistic approach' to flow recommendations in Australia, drawing upon features of the natural flow regime (as derived from daily flow records). Four attributes of the natural flow regime are progressively summed to create a recommended, modified flow regime: low flows, the first major wet-season flood, medium-sized floods, and very large floods. The low flow target would presumably be the lowest flow that occurs 'often' (e.g. based upon a specified percentile exceedance flow for each month).

Each of these approaches has inherent shortcomings or challenges to overcome, however, that prevent them from being widely adopted or otherwise make them undesirable for setting comprehensive ecosystem-based management targets:

1 River managers typically demand considerable specificity in flow targets to be met. The methods advocated by Tennant (1976) or by Hill *et al.* (1991) are specific about flow magnitudes, but do not (or only vaguely) specify any particular timing or duration of flow events, or frequencies of occurrence, or rates of change. This lack of specificity may be unacceptable to river managers, and may not always produce desired ecological results. In fact, some of these approaches have been used simply to set instream flow levels at constant annual or monthly minimums.

2 Management decisions that focus on a limited number of features of the hydrological regime are unlikely to sustain or restore all necessary ecological processes and patterns.

3 Management decisions based on information and objectives keyed to a limited number of species and a limited number of their habitat requirements may

Table 1 Summary of hydrological parameters used in the Indicators of Hydrologic Alteration, and their characteristics

IHA Statistics Group	Regime characteristics	Hydrological parameters
Group 1: Magnitude of monthly water conditions	Magnitude Timing	Mean value for each calendar month
Group 2: Magnitude and duration of annual extreme water conditions	Magnitude Duration	Annual minima 1-day means Annual maxima 1-day means Annual minima 3-day means Annual maxima 3-day means Annual minima 7-day means Annual maxima 7-day means Annual minima 30-day means Annual maxima 30-day means Annual minima 90-day means Annual maxima 90-day means
Group 3: Timing of Annual Extreme Water Conditions	Timing	Julian date of each annual 1-day maximum Julian date of each annual 1-day minimum
Group 4: Frequency and Duration of High/Low Pulses	Frequency Duration	No. of high pulses each year No. of low pulses each year Mean duration of high pulses within each year (days) Mean duration of low pulses within each year (days)
Group 5: Rate/Frequency of water condition changes	Rates of change Frequency	Means of all positive differences between consecutive daily values Means of all negative differences between consecutive daily values No. of rises No. of falls

actually result in undesirable effects on the ecosystem as a whole (Sparks, 1992).

4 Research efforts to evaluate interrelationships between flow phenomena and biotic responses are time-consuming (i.e. long-term research). The time scales necessary to attain conclusive research results may be incompatible with the time frames within which management or regulatory decision-making takes place.

5 Research results from one river may not be widely transferable to other river ecosystems.

Given the shortcomings of existing instream flow methods with respect to the natural flow paradigm, *a new approach is needed to quickly define initial, interim river management targets that are based on the natural flow paradigm and that collectively serve as a starting point to begin adaptive management efforts*. Characteristics of such an approach include: (i) management targets can be developed within the river manager's decision-making time frame; (ii) a natural range of variability in timing, duration, frequency and rate of change of natural flow conditions is characterized and incorporated into river management targets; (iii) management targets are translated into a workable set of management rules or a restoration plan; and (iv) both the management actions and flow targets are considered to be hypotheses, which are tested through application and monitoring, and can be refined annually based on monitoring and ecological research results.

Methods: the range of variation approach

In the present study a method was developed, referred to as the 'Range of Variability Approach,' or RVA, that meets these criteria. The RVA identifies annual river management targets based upon a comprehensive statistical characterization of ecologically relevant flow regime characteristics (Richter *et al*., 1996). A set of

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management rules or a management system that will lead to attainment of the targets on an annual basis is then developed. The RVA is adaptive in nature (Walters, 1990; Lee, 1993), in that the ecological effects of applying the management rules are monitored and the monitoring results used to refine management targets and rules.

The RVA has six basic steps for setting, implementing and refining management targets and rules for a specific river or river reach.

Step 1

The natural range of streamflow variation is characterized using a suite of thirty-two ecologically relevant hydrological parameters, using the Indicators of Hydrologic Alteration (IHA) method of Richter et al. *(1996).* Existing long-term (> 20 yrs) daily streamflow records are used to define natural, or less altered, ranges (and other measures) of variability in riverine hydrological regimes. The management team must specify the period of record that best represents natural, historic or undisturbed conditions; alternatively, unaltered daily flow records must be synthesized (described in greater detail later). The IHA method is based upon the statistical derivation of thirty-two ecologically relevant hydrological parameters for each year of streamflow record (Table 1) for the selected reference period or data series. Measures of the central tendency (e.g. mean, median) and dispersion (e.g. range, standard deviation, coefficient of variation) are computed from the annual series for each of the thirty-two parameters and used to characterize interannual variation.

Step 2

Thirty-two management targets, one for each of the thirty-two IHA parameters, are selected. The fundamental concept is that the river should be managed in such a way that the annual value of each IHA parameter falls within the *range* of natural variation for that parameter, as defined by the interannual measure of dispersion derived in step 1. Thus, the management target for any given parameter is expressed as a range of acceptable values. The target may have both upper and lower bounds (e.g. the attained value should fall within ± 1 standard deviation (SD) of the mean), or it may have only a minimum (e.g. attained value ≥ mean – 1 SD) or maximum (e.g. attained value ≤ mean + 1 SD) boundary. The management team must decide on the most appropriate measure of dispersion to use in setting the management targets (e.g. the range, ± 1 or 2 SD from the mean, the twentieth and eightieth percentiles, etc.) and this may vary among the thirty-two parameters.

The management targets should be based, to the extent possible, on *available ecological information*, and should take into account the ecological consequences of excluding extreme events if the target does not include the full range of natural variation. For example, a management target of [attained value ≤ mean + 1 SD] for the annual 1-day maximum streamflow might not achieve ecological disturbance effects necessary for regeneration of certain floodplain plant species. If a particular 1-day maximum streamflow has been shown to be ecologically relevant (e.g. Stromberg, Patten & Richter, 1991), then the target should incorporate that flow level.

In the absence of adequate ecological information, we recommend that the ± 1 standard deviation values be the default for setting initial targets (e.g. Fig. 1). This recommendation is based upon a recognition that adoption of a flow target that corresponds to the minimum or maximum limits of the range of variation in a particular parameter may lead to considerable ecosystem stress over long time periods. On the other hand, the flow targets must allow some management flexibility to accommodate human uses; selection of values near the interannual mean or median as management targets would entirely preclude human water uses in half of the years. But again, the adopted management approach should not entirely preclude the occurrence of infrequent, but ecologically important, extreme occurrences of certain hydrological conditions. Over time, as ecological research and monitoring results illuminate critical flow thresholds for various components of the river ecosystem, flow-based management targets (hereafter, 'RVA targets') should be adjusted in an adaptive fashion.

Step 3

Using the RVA targets as design guidelines, the river management team designs a set of management rules, or a management system, that will enable attainment of the targeted flow conditions in most, if not all, years. It would be extremely difficult, if not impossible, to manage continuously and instantaneously even a fully regu-

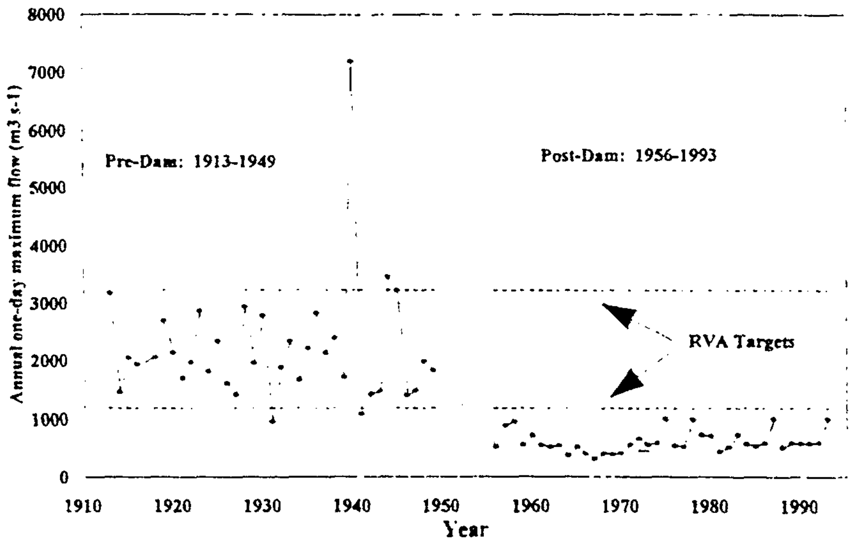

Fig. 1 Application of the IHA method to the Roanoke River in North Carolina reveals the effects of dam construction for flood control in 1956. This graph portrays the values of the 1-day maxima streamflows ($m^3\ s^{-1}$), for each year of record. Horizontal bars denote values of the means and standard deviations for the pre-dam and post-dam periods. An RVA target for this IHA parameter (1-day maxima) could be set at the value of the mean ± 1 SD.

lated river to meet all thirty-two RVA targets independently within each year. Rather, the river management team should design a 'management system' that will enable the RVA targets to be attained, such as a workable set of reservoir operations rules, or maximum allowable river depletions during various seasons, or needed restorative mechanisms such as levee removal, wetland restoration, or adoption of conservation tillage practices within an agricultural catchment. Depending upon the nature of the selected RVA targets, the management system might be designed to achieve targeted flow conditions every year (e.g. if the RVA target has only an upper or lower bound) or in most years (e.g. 68% of years if the RVA target is the mean ± 1 SD).

The design of the management system will likely draw upon available historic data, including streamflow and other climatic data, upon reservoir operations or flow diversion records, and upon other evidence of historic or extant human perturbation, such as historical aerial photographs from which land use can be mapped from different time periods. Such historic data can often be used to identify a historic period during which human land and water uses had not yet pushed hydrological conditions outside of their (RVA) targeted ranges. Alternatively, hydrological simulation models may be used to simulate the hydrological response of a less-altered catchment, or to simulate alternative reservoir operating schemes (Gordon *et al.*, 1992; Maheshwari, Walker & McMahon, 1995).

The proposed management system should be recognized as an hypothesis in itself; that is, the proposed management is hypothesized to be capable of achieving the RVA targets at the specified frequency (e.g. every year, 68% of years). In certain situations, such as for already-regulated rivers, tests of the management system hypothesis can begin in the first year of implementation. Other management systems, such as the restoration of floodplain or wetland storage within a catchment, may need to be implemented and evaluated incrementally.

Step 4

As the management system is implemented, begin (or continue) a monitoring and ecological research programme designed specifically to assess the ecological effects of the (new) management system. The RVA targets are means to achieving biological goals, and are not ends in themselves. The management plan therefore must include a specific statement of measurable biological goals, and must include a monitoring and research programme which evaluates whether the management efforts are achieving these goals. This monitoring and research programme should also include investigations of the hydrological and other abiotic and biotic requirements of key (or indicator) species in the ecosystem. Knowledge gained from these investigations will help clarify whether management targets are appropriate. It will not be possible to adapt the management plan over time in a scientifically sound manner in the absence of a monitoring and research programme.

Additional research may also be necessary in catchments where land use practices have a major or important role in shaping the river's hydrological regime. The effects of modifying land use practices or of implementing hydrological restoration projects across a catchment will not be as predictable as will the effects of modifying a reservoir's operating rules. Monitoring the effects of catchment restoration efforts directly at the restoration locations may thus also be necessary to evaluate whether the management system is achieving the desired results.

Step 5

At the end of each year, actual streamflow variation is characterized using the same thirty-two hydrological parameters, and the values of these parameters are compared with the RVA target values. The annual hydrograph resulting from implementation of the management system over the past year is characterized using the thirty-two IHA parameters, and these values are compared with the respective RVA target values to see which targets were met or not met.

Step 6

Repeat steps 2–5, incorporating the results of the preceding years' management and any new ecological research or monitoring information to revise either the management system or the RVA targets. RVA targets or the management system should be refined incrementally, as warranted, based on the system's performance in meeting the RVA targets over the past year(s), on ecological monitoring and research results, and on other relevant changes in circumstances.

Characterizing the natural range of variation

The process of characterizing the natural range of variation begins with identifying an adequate period of record that adequately represents natural, historic or less-disturbed conditions. Typically, this will require having records that pre-date substantial human perturbation. Less often, a more recent time period may best represent natural or less-disturbed conditions, especially in catchments long perturbed by human influence. For example, improved farming practices and restoration of forested acreage may result in current hydrological variation being more representative of natural or pre-disturbance conditions (e.g. Trimble, Weirich & Hoag, 1987). Regardless of whether the period of record representing relatively unaltered conditions pre-dates or post-dates substantial levels of human perturbation, long-term streamflow data for the representative period will not be available for all rivers or river reaches of interest. Therefore, the RVA has been structured to address three different scenarios of data availability, as described below. Note that the level of uncertainty increases, and the amount of confidence in resulting management targets decreases, as the availability of hydrological data decreases, i.e. from scenario I to scenario III.

Scenario I. Adequate streamflow records exist for the period of record representing natural conditions. At least 20 yrs of record should be used in computing IHA parameter values for characterizing the natural range of variation. We have begun testing the sensitivity of measures of central tendency and dispersion (e.g. means and standard deviations) in the IHA parameters for the thirty-two IHA parameters to differing record length, by repeatedly computing alternative values of these statistical measures for samples of consecutive years spanning increasingly long records. The results of three such tests, developed for three streams representative of different 'stream types' as characterized by Poff (1996), show that the range of estimates of the mean annual 1-day maximum begins to narrow substantially when based on at least 20 yrs of record (Fig. 2). This suggests that the effects of interannual climatic variation on IHA parameter statistics are substantially dampened when at least two decades of data are analysed (but see cautionary note in Walker *et al.*, 1995). We hesitate to suggest a longer period of record as a minimum standard for RVA analyses because the number of sites having the required period of record, and thus to which the RVA can be applied, will decrease as the minimum standard increases.

Scenario II. Inadequate streamflow records exist for the period of record representing natural conditions. If a streamflow record exists, but is less than 20 yrs in length, it may be necessary to extend the existing record using hydrological estimation techniques. Richter *et al.* (1996) briefly describe various approaches for extending hydrological data records using regression relationships between the site of interest and other, less altered or unperturbed stream-gauging site(s) (see also Gordon *et al.*, 1992; Yin & Brook, 1992; Richter & Powell, 1996). Such hydrological estimation

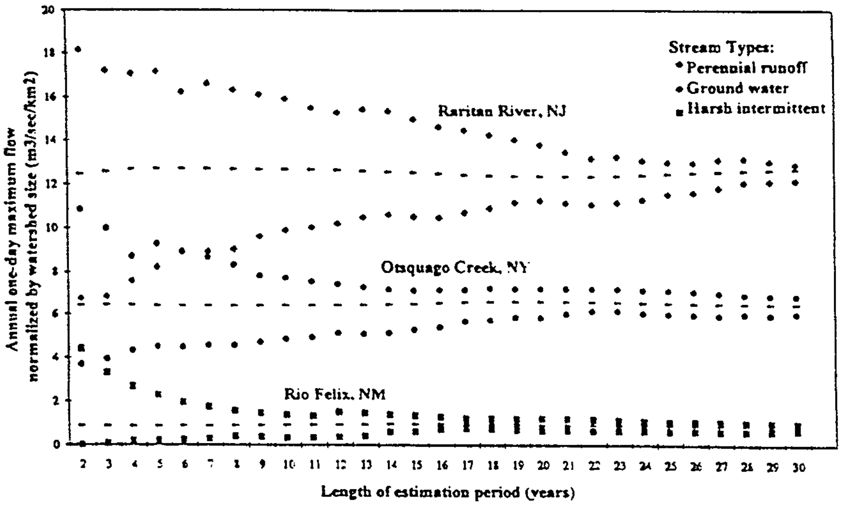

Fig. 2 Average values of the annual 1-day maxima were computed for three different streams, using varying lengths of record from 2 to 30 yrs. Plotted here are *minimum* and *maximum* values of the mean 1-day maxima, derived using each incremental record length, e.g. 2-yr means, 3-yr means, etc. Each of the plotted means have been normalized by catchment area ($m^3 s^{-1} km^{-2}$), to enable comparisons across streams of differing catchment area. Dashed lines represent long-term (30-yr) means. These initial tests suggest that measures of central tendency or dispersion for various IHA parameters may adequately converge around the long-term mean when at least 20 yrs of record are utilized.

techniques depend upon the availability of concurrent data at both the predictor and estimation sites. When selecting predictor site(s) for this purpose, it would be expected that estimation error attributable to human effects would be reduced by selecting *reference catchments* within the same ecoregion, whenever possible (Gordon *et al.*, 1992; Omernik, 1995). The concept of using reference sites to develop expectations of unperturbed or less-altered hydrological (especially water chemistry) conditions representative of their respective ecoregions has been discussed by other authors; the reader is encouraged to refer to Hughes, Larsen & Omernik (1986), Hughes *et al.* (1990) or Gallant *et al.* (1989) for further guidance in selecting appropriate reference catchments.

Alternatively, hydrological simulation models can be used to estimate streamflows under undeveloped conditions (e.g. Maheshwari *et al.*, 1995). Even a few years of streamflow data will greatly aid the calibration of such models, thereby improving their reliability. When streamflow values must be estimated from regression or simulation models, we would recommend against the use of certain IHA parameters in the RVA. In particular, it is expected that the group 5 parameters (rates and frequency of daily hydrograph rises and falls; see Table 1) would be highly sensitive to errors in daily flow estimation.

Scenario III. No streamflow records exist for the period of interest. When no stream-gauge data exist for the catchment of interest, two alternative strategies may be useful: hydrological simulation modelling (discussed under scenario II) or the use of 'normalized' estimates based on data from gauged reference catchments with adequate record lengths, similar conditions of climate, surficial geology and minimal anthropogenic effects. Normalization, as used here, refers to the adjustment of streamflow data or statistical characteristics to account for differences in catchment area or other control variables (e.g. total precipitation). By dividing the reference catchment's daily streamflow data or RVA estimates by either drainage basin area or mean annual flow, the effects of differing catchment areas can be reduced or eliminated (Poff & Ward, 1989). By selecting a reference catchment(s) of comparable size, residual effects of catchment size can be minimized. The normalized RVA targets can then be adjusted for the size of the catchment of interest (e.g. multiply normalized RVA targets by catchment area). Again, we caution against use of these scenario III approaches for the IHA's group 5 parameters, due to expected errors in the estimation of daily flow values. While recognizing fully the potential errors inherent in transferring normalized RVA targets from other catchments, emphasis should be made of the intent of these RVA targets: to serve as initial, interim targets until better hydrological and ecological information becomes available.

Results of case study application

The Roanoke River in North Carolina (U.S.A.) will be used as a case study to illustrate the intended application of the RVA. Dam influences on the Roanoke

River system began in 1950 with the completion of Philpott Lake on the Smith River (in the upper catchment). Kerr Reservoir, completed in 1956, provides flood control in the lower river as well as hydropower-generating capabilities. Two additional hydropower dams were subsequently built downstream of Kerr Reservoir, but they provide little flood storage. Kerr Reservoir thus provides the primary high flow control for the lower river, but the two hydropower facilities downstream of Kerr Reservoir can induce considerable hourly and daily fluctuations in flow. The daily streamflow data for the present analysis were obtained from a stream gauge located just downstream of the hydropower dams at Roanoke Rapids.

The natural range of streamflow variation for the Roanoke River was characterized by generating the thirty-two IHA parameters from a 37-yr pre-dam record (1912–49) taken at Roanoke Rapids, North Carolina (refer to *pre-dam* results in Table 2). Computation of the pre-dam means, standard deviations, and range limits, using the IHA method of Richter *et al.* (1996), constitutes step 1 of the RVA as described earlier.

Selection of RVA targets

Values at ± 1 SD from the mean were selected as the RVA targets for each of the thirty-two IHA parameters (see 'RVA targets' in Table 2). In some instances, due to skewness in the distribution of the pre-dam annual values for certain IHA parameters, the mean − 1 SD values fall outside (below) the pre-dam low range limits. For those parameters (August, September and October means), the pre-dam minima of their range was selected instead. Selection of RVA targets completes step 2 of the RVA.

Design and assessment of the management system

In step 3 of the RVA, the river ecosystem management team is challenged to design a river management system capable of meeting the selected RVA targets on an annual basis. At Kerr Reservoir, this will involve a re-design of reservoir operations rules ('rule curves') that specify desired lake levels and flow releases on a monthly basis.

Reservoir operations during the 38-yr post-dam period have caused many of the annual values of the thirty-two IHA parameters to fluctuate outside the RVA targeted range (e.g. Figs 1 and 3). Table 2 lists the degree of non-attainment (percentage of post-dam years not meeting the RVA target) for each parameter over the 38 post-dam years. Using ± 1 SD as the RVA targets, non-attainment rates of about 32% even under pre-dam conditions would be expected. However, a number of the non-attainment rates for the post-dam period are considerably higher, including the monthly means for March (50% non-attainment) and April (68%); all of the 1-day and multiday maxima (55–100%); the timing of annual minima (97%) and annual maxima (53%); high and low pulse counts and durations (58–97%); numbers of hydrograph falls (97%) and rises (100%); and the hydrograph rise rate (61%).

The results of the present analysis of rise rates were initially surprising; rise rates were expected to be considerably higher in the post-dam period due to rapid releases of water from the hydropower dams. However, further study revealed that under natural, pre-dam conditions the Roanoke experienced frequent and highly flashy runoff events in response to heavy rainstorms, and these pre-dam hydrograph rises commonly exceeded 600 $m^3 s^{-1}$ in a single day. Those frequent, extreme daily rises cause the pre-dam *annual average* rise rates to come out higher than the post-dam annual averages. Furthermore, because the IHA method uses daily mean streamflows for all of its computations (rather than hourly data), the calculated average rise and fall rates from day-to-day do not accurately reflect hour-to-hour rates of change. However, it was found that the computation of rise and fall rates and rise/fall counts in the IHA method does a reasonably good job of detecting hydropower-induced change (see Table 2), even though values of these parameters would be different if computed on an hourly, rather than daily, basis.

Based upon the present RVA analysis, it can be recommended that reservoir operations rules for the Roanoke dams, including the rule curve for Kerr Reservoir, be modified to accomplish five primary objectives: (i) restore high-magnitude flooding; (ii) shift the timing of the largest annual floods back into the spring (February–April) and shift the timing of annual low flow extremes to early autumn (September–October); (iii) decrease the frequencies of high and low pulses and increase their durations; (iv) decrease the frequency of hydrograph reversals (shifts between rising and falling flow levels) attributable to

Table 2 Results of the Indicators of Hydrologic Alteration analysis for Roanoke River at Roanoke Rapids, North Carolina. Basic data used in the analysis were daily mean streamflows, reported here as cubic metres per second

	Pre-dam: 1913–49				Post-dam: 1956–93						
			Range limits				Range limits		RVA targets[1]		
	Means	SD	Low	High	Means	SD	Low	High	Low	High	Rate of non-attainment
IHA group 1									min	1 SD	
October	162	143	27	646	166	120	57	576	27	305	16%
November	156	86	42	419	184	110	56	501	70	242	24%
December	225	138	67	605	211	101	98	520	87	364	13%
January	337	214	83	1094	270	108	100	505	123	551	3%
February	350	139	89	649	293	123	74	554	211	488	42%
March	361	167	166	740	303	170	64	678	194	528	50%
April	314	116	109	596	315	202	72	924	198	430	68%
May	222	94	93	567	296	184	112	899	128	316	34%
June	184	85	83	475	206	99	67	432	99	269	24%
July	195	130	54	689	156	97	73	582	65	325	8%
August	201	192	38	1103	150	59	71	276	38	393	0%
September	164	145	29	632	147	72	62	353	29	309	8%
IHA group 2											
1-day minimum	45	18	13	88	28	6	14	43	28	63	34%
3-day minimum	48	19	14	90	40	11	28	75	29	66	16%
7-day minimum	51	19	15	92	55	16	28	101	32	70	18%
30-day minimum	64	24	25	118	81	25	39	141	40	88	26%
90-day minimum	94	35	31	165	125	38	69	236	58	129	18%
1-day maximum	2208	1021	954	7188	602	217	317	1007	1186	3229	100%
3-day maximum	1938	884	887	6301	592	188	282	1003	1049	2817	100%
7-day maximum	1353	603	617	4114	564	202	228	1000	750	1956	89%
30-day maximum	636	188	313	1181	477	19	133	988	448	824	55%
90-day maximum	424	102	237	819	363	152	109	680	322	527	61%
IHA group 3											
Julian date of annual minimum	264	43	25	308	360	43	2	364	221	307	97%
Julian date of annual maximum	71.9	52	10	342	137.8	96	3	326	20	124	53%
IHA group 4											
25% Low pulse count[2]	11.0	4.6	2	22	36.4	10.6	16	53	6	16	97%
High pulse count[2]	15.7	4.4	7	29	22.7	7.7	6	43	11	20	66%
15% Low pulse duration	7.3	3.0	2.2	15.8	3.2	1.2	1.6	6.1	4	10	74%
High pulse duration	5.9	2.4	3.1	17.3	4.9	2.5	1.5	10.0	4	8	58%
IHA group 5											
Fall rate	–55.2	14.5	–91.9	–29.9	–59.6	13	–29	–91	–70.0	–40.7	32%
Rise rate	89.7	25.6	47.3	152.2	60.2	11	32	84	64.0	115.3	61%
Fall count	68	7.2	57	92	90.9	7	71	103	61	75	97%
Rise count	61.3	8.6	47	79	91.6	6	74	103	53	70	100%

[1]RVA targets are based upon mean +or - 1 sd, except when such targets would fall outside of pre-dam range limits (range limits were then used).
[2]Low pulses are defined as those periods during which daily mean flows drop below the 25th percentile of all pre-dam flows; high pulses are defined as those periods during which the 75th percentile is exceeded.

hydropower generation; and (v) moderate the rate at which flow release rates rise or fall within or between days.

Objectives (i), (ii), and in part (iii) could be accomplished by modifying the rule curve to increase water levels in the Kerr Reservoir during late February through April, and by accommodating the associated reduction in flood storage capacity in the lake by increasing flood release rates. Those strategies would simultaneously serve to increase both the rate and the frequency of high flows and to increase high pulse durations. By adjusting (raising) the rule curve in late February–April, the timing of these annual floods can be managed to occur more frequently during the early spring.

It should be acknowledged that accomplishing the targeted increases in flood magnitude, frequency, and duration will require more than just changing the way that Kerr Reservoir is managed. Downstream roads, houses, and other infrastructure lie in the path of these restored floods. A combination of flood easements, land purchases and relocation of infrastructure will be necessary to accomplish flood restoration on the Roanoke, as in many other river systems.

The attainment of RVA targets associated with the timing of annual minima and the number and duration of low pulses will also require a combination of adjustments to the rule curve during the (natural) low-flow season (September–November), and modifications of hydropower operations. In particular, hydropower releases should not be allowed to drop below the low pulse threshold level (computed as $100\ m^3\ s^{-1}$ for the Roanoke—see low and high pulse definitions in Table 2) in the higher runoff months (e.g. January–May), and the hourly rates of change in hydropower releases should be moderated. These changes in hydropower operations should achieve the benefits of reducing the frequency of low pulses and the frequency of hydrograph rises and falls. However, the role of the Roanoke reservoirs in providing peaking power generation will be affected by changes in the management system, with likely consequences for power revenues.

Implementing a monitoring and research programme

Step 4 of the RVA calls for implementation of hydrological and biological monitoring programmes, and initiation of ecosystem research efforts to track biotic responses to the implementation of the new management system. Changes in the Roanoke's streamflow regime should continue to be monitored at the stream gauge used to develop the RVA targets. However, additional hydrological monitoring will be highly desirable, for example, to enable ecological researchers to link biotic responses to changes in floodplain inundation or water table levels. In Richter *et al.* (1996) various ecosystem components are described, such as littoral zone macroinvertebrates, native fish, and floodplain vegetation communities that should be monitored to track population- and community-level responses to restored flood and drought regimes and moderated streamflow fluctuations.

Striped bass population size and reproduction rates have been monitored along the lower Roanoke since the late 1950s (Zincone & Rulifson, 1991). Based upon analysis of those monitoring data, two flow characteristics are thought to influence strongly striped bass recruitment: daily flow magnitudes and rates of change in flow levels during the 1 April–15 June spawning period. An experimental flow regime was recommended by the Roanoke River Water Flow Committee in 1988 (Rulifson & Manooch, 1993) and implemented beginning in 1989. The flow recommendations were designed to approximate historical, pre-dam conditions by maintaining flows within the twenty-fifth and seventy-fifth percentiles of daily pre-impoundment flows during 1 April–15 June (see Table 3). Additionally, the Flow Committee recommended that the maximum variation in flow rate be restricted to $42\ m^3\ s^{-1}\ h^{-1}$, and preferably less. The close correspondence between the Flow Committee recommendations and three corresponding RVA targets (April, May, June flows; Table 3) is not surprising, given the Committee's use of pre-dam flow conditions and similar measures of dispersion as management targets.

Striped bass recruitment rates in recent years have recovered to their highest post-dam levels since implementation of the Committee's flow recommendations in 1989 (Rulifson & Manooch, 1993). The RVA target for April has been attained in 3 of the 5 yrs since 1989 (Fig. 3), translating into a non-attainment rate of only 40%. Similarly, the May and June targets have been attained in 4 of the 5 years (20% non-attainment). Thus, the April, May and June flow conditions are approaching their expected non-attainment values of 32% under the recently modified management system. Because the response of the striped bass population cannot be compared with replicated control populations, inferences about the effect of partial flow restoration on this population must be carefully qualified. Increased recruitment rates during this time period could be attributed to other factors, such as climatically

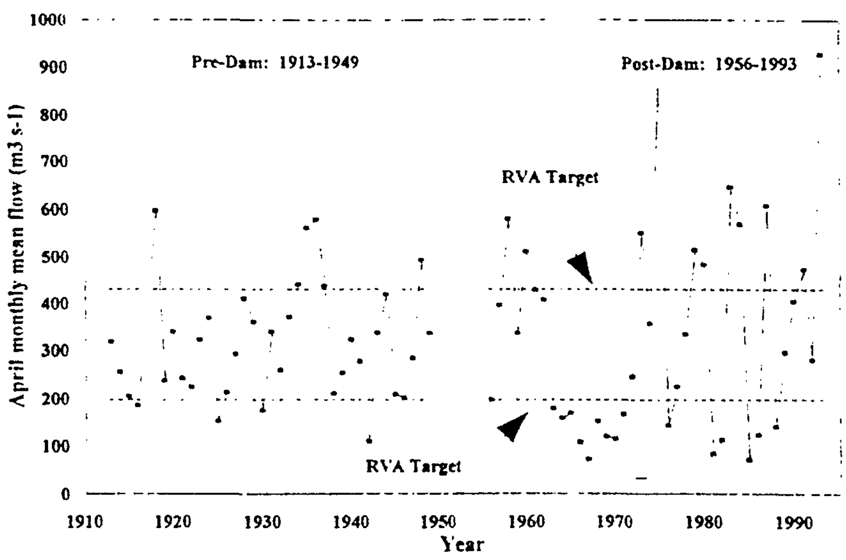

Fig. 3 Monthly means for April are plotted for the Roanoke River. The RVA target for this hydrological parameter can be defined as the range between ± 1 SD from the mean of the pre-dam values. By so doing, 68% (26 yrs) of 38 post-dam years would have failed to meet the targeted conditions.

Table 3 Flow conditions recommended by the Roanoke River Water Flow Committee for striped bass recruitment, and comparison with RVA targets

Dates	Flow Committee lower limit ($m^3 s^{-1}$)	Flow Committee upper limit ($m^3 s^{-1}$)	RVA targets ($m^3 s^{-1}$)
April 1–15	187	388	198–430
April 16–30	164	311	198–430
May 1–15	133	269	128–316
May 16–31	125	269	128–316
June 1–15	113	269	99–269
Rate of change		42 $m^3 s^{-1} h^{-1}$	Falls: 29–68 $m^3 s^{-1} day^{-1}$ Rises: 55–130 $m^3 s^{-1} day^{-1}$

induced differences in water temperature, differences in water chemistry associated with varying effluent discharges along the river, or other unexplainable factors. However, the flow modifications implemented on the Roanoke were based upon considerable knowledge of striped bass ecology and habitat use, and the persistence of high recruitment rates suggests that the restoration of certain flow characteristics is benefiting bass recruitment. The favourable response of striped bass to these management changes illustrates the fact that when flow restoration efforts must occur incrementally, certain components of the riverine ecosystem can benefit prior to attainment of all RVA targets.

Discussion

The RVA is designed to bridge a chasm between applied river management and current theories of aquatic ecology. Virtually all methods currently in widespread use for determining instream flow needs will possibly lead to inadequate protection of ecologically important flow variability, and ultimately to the loss of native riverine biodiversity and ecosystem integrity (Gore & Nestler, 1988; Arthington & Pusey, 1993; Stanford, 1994; Castleberry *et al.*, 1996). Current aquatic ecology theory and empirical observations suggest that a hydrological regime characterized by the full or nearly full range of natural variation is necessary to sustain the full native biodiversity and integrity of aquatic ecosystems. The RVA addresses this paradigm by incorporating into river management targets a suite of ecologically relevant hydrological parameters that comprehensively characterize natural streamflow regimes.

Because the RVA represents a substantial departure from predominant approaches currently being used to prescribe instream flows, we do not expect rapid adoption of the method. Rather, we anticipate considerable debate about the merits of the approach for conserving aquatic biodiversity. The dependence of native aquatic biota on specific values of the hydrological parameters employed in the RVA has not been widely, nor comprehensively, substantiated with statistical rigor. Much of what aquatic and riparian ecologists know or believe about the biotic consequences of flow alteration has been derived from comparisons of dammed *v* undammed rivers (Sklar & Conner, 1979; Bradley & Smith, 1986; Rood & Heinze-Milne, 1989; Copp, 1990; Nilsson *et al.*, 1991; Smith *et al.*, 1991); measured differences in fish or invertebrate communities at increasing distances downstream from dams (invertebrates: Voelz & Ward, 1991; Moog, 1993; fish: Kinsolving & Bain, 1993); correlations developed between long-term ecosystem changes and a limited number of hydrological parameters (e.g. Bren & Gibbs, 1986; Johnson, 1994; Miller *et al.*, 1995); or simply from inferences drawn from (relatively short-term) observations of flow and fluvial processes (Petts, 1979, 1980; Bradley & Smith, 1984; Williams & Wolman, 1984; Johnson, 1992; Lyons, Pucherelli & Clark, 1992), and biotic distributions or growth rates associated with hydrological gradients (Hosner, 1958; Bell, 1974; Johnson, Burgess & Keammerer, 1976; Franz & Bazzaz, 1977; Reily & Johnson, 1982; Pearlstine, McKellar & Kitchens, 1985). Virtually all such studies have statistical weaknesses that limit inferences regarding *causation* between flow and biota (Kinsolving & Bain, 1993; Richter *et al.*, 1996), because flow perturbations cannot be replicated or randomly assigned to experimental units (Hurlbert, 1984; Carpenter, 1989; Carpenter *et al.*, 1989; Stewart-Oaten, Bence & Osenberg, 1992).

While the accumulated evidence in support of the natural flow paradigm is overwhelming, others may be less convinced or ready to use it as a guide in river management. In the present design of the RVA, flexibility in setting specific flow management targets was emphasized, while retaining what could be considered to be the backbone of the approach: the use of natural variability characteristics as ecosystem management guides, accompanied by adaptive refinement of flow targets as ecological research accumulates.

The RVA was designed with a very specific application in mind: setting initial river management targets for river systems in which the hydrological regime has been substantially altered by human activities (e.g. damming, large water diversions, extensive land cover alteration). Substantial alteration will be reflected by near-term annual values of IHA parameters (or the mean for a post-impact period of record) falling outside the range of variation observed for the period of record representing natural or unaltered conditions. Thus, the intent of management targets derived using the RVA is for observed annual IHA parameter values to fall within a natural range of variation.

The RVA was developed to provide explicit adaptive management guidelines that are responsive to the short-term demands of most water management negotiations. The RVA is meant to enable river managers to define and adopt readily interim management targets before conclusive, long-term ecosystem research results are available. The RVA is our response to an urgent need to act in the face of considerable uncertainty. Setting management targets based on a natural range of variation in the thirty-two hydrological parameters does not depend upon extensive ecological information, although such information certainly will help select and refine the targets. An adaptive decision-making process, based upon carefully formulated scientific research and monitoring, holds greatest promise for resolving complex resource management conflicts (Walters, 1990; Lee, 1993). Thus, an adaptive management approach, whereby interim management targets and an associated river management system are prescribed and implemented, the system response is monitored, and management targets and the prescribed flow regime are adjusted based on monitoring results and ecological research, is fundamental to successful application of the RVA. Such an adaptive approach would closely resemble that taken by the 10-Rivers Project in Australia (Arthington & Pusey, 1994), the Kissimmee River restoration effort in Florida (Toth *et al.*, 1995), the modification of hydropower dam operations on the Tallapoosa River in Alabama (Travnichek, Bain & Maceina, 1995), or the approach advocated for the Upper Colorado River Basin Endangered Fish Recovery Program (Stanford, 1994).

The RVA will be redefined as new research on the linkage between hydrological characteristics and aquatic ecosystem integrity becomes available. Clearly, increased funding for this type of applied ecological research is urgently needed (Naiman *et al.*, 1995). The RVA should be modified after further testing of the IHA method (Richter *et al.*, 1996). In particular, it is necessary to define better the minimum streamflow record length needed to characterize adequately the

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influence of climatic variation on IHA parameter values in various geographical regions and different stream types (Poff, 1996). This will help to gain a better sense of the 'expected' (unaltered) values of the IHA parameters (and RVA targets) across ecoregions and stream types. It is hoped that such knowledge will lead to better clarification of recommended strategies for dealing with scenarios I–III as described in this paper, and aid RVA users in the selection of appropriate reference catchments.

A cautionary response to the RVA is expected from professionals experienced in the advanced statistical analysis of stream-gauge records, over the recommended use of ± 1 SD as a default RVA target. The statistically minded will recognize that the frequency distribution of many of the thirty-two IHA parameters are not likely to be normally distributed. Instead, as seen in the Roanoke example, the parameters are likely to exhibit varying degrees of skewness due to the occurrence of occasional extreme values (see also Walker *et al.*, 1995). As has been emphasized and also illustrated for the Roanoke example, however, the RVA calls for a flexible application of the thirty-two parameters, using the ± 1 SD default targets only when ecological or statistical reasons cannot yet be formulated into alternative targets. Where more refined statistical analyses of the IHA parameters for a stream-gauge record suggest more appropriate target values, it would be expected that these alternative targets be used. The present argument focuses on the need to restore or maintain the regime of *natural variability* of the hydrological system, not on the need for any single, inflexible statistical procedure.

Use of the RVA will possibly reduce the flexibility to manage river systems for economic benefits and other human needs, particularly when riverine biodiversity conservation has not been adequately considered in the past. Debate about the values of native riverine biota and river ecosystem functions, and associated trade-offs in management options, will test society's commitment to conserving healthy, functioning, native aquatic ecosystems. It will also help to define what 'sustainable use' of the earth's river systems might look like.

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