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The following is the established format for referencing this article:

Reeves, G. H., and S. L. Duncan. 2009. Ecological history vs. social expectations: managing aquatic ecosystem *Ecology and Society* **14**(2): 8. [online] URL: <http://www.ecologyandsociety.org/vol14/iss2/art8/>

Research, part of Special Feature on Historical and Future Ranges of Variability

Ecological History vs. Social Expectations: Managing Aquatic Ecosystems

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ABSTRACT

The emerging perspective of ecosystems as both non-equilibrium and dynamic fits aquatic ecosystems as well terrestrial systems. It is increasingly recognized that watersheds historically passed through different conditions over time. Habitat conditions varied in quantity and quality, primarily as a function of the time since the last major disturbance and the legacy of that disturbance. Thus, to match the effects of historical processes, we would expect a variety of conditions to exist across the watersheds in a region at any time. Additionally, watersheds have different potentials to provide habitat for given fish species because of variation in physical features. This developing ecological understanding is often preempted by social desires to bring all watersheds to a "healthy" condition, which in turn is reflected in a common regulatory approach mandating a single condition as the long term goal for all watersheds. Matching perceptions and regulations to the way aquatic systems actually change evolve through time will be a major challenge in the future.

Key words: aquatic ecosystems; legacy of disturbance; non-equilibrium ecosystem dynamics

INTRODUCTION

State and federal governments are taking an increasingly rigorous and proactive approach to managing aquatic ecosystems by creating and applying regulatory policies to ostensibly prevent degradation or improve current conditions. In the search for seemingly straightforward regulations, state and federal environmental agencies have often adopted policies and environmental thresholds based on central tendencies (e.g., averages), usually derived from assessments of the historical range of variability (HRV) for watershed processes. Typically, they apply the averages uniformly across the landscape. Unfortunately, use of averages derived from an HRV-based snapshot understanding of past environments further obscures the fact that those environments are constantly in flux. The result is a fundamental incongruity between regulations and on-the-ground reality (Poole et al. 2004): applying regulations based on environmental averages (such as stream temperature, turbidity, suspended sediment, etc.) to dynamic landscapes generates a policy inflexibility that does not allow for changes through time and across different scales, from stream reaches to large landscapes.

From the outset, this problem seriously undermines the effectiveness of aquatic ecosystem regulations, and in process threatens the rational basis for any environmental regulation. In effect, it creates an illusion of failure when a policy appears not to have achieved its benchmarks. It also jeopardizes the necessary relationship between the regulators (federal and state government) and the regulated (private industry) as they jointly pursue land-management activities that will successfully protect and enhance the integrity of aquatic resources.

As a result, current regulatory policy and its application may be part of the problem rather than part of the solution. The significant social challenge here lies in helping both affected communities and policy makers understand that, as much as they might wish for it, having all watersheds in good condition at the same time is outside the historical ecological range of variability.

This paper first reviews three aspects of watershed restoration: (1) the current understanding and management of aquatic ecosystems viewed through the filter of dynamic landscapes; (2) the direct effects of social acceptability of landscapes and how they relate to watershed restoration efforts; and (3) the challenges of management and policy formation in dynamic systems. It then discusses how these three components interact, creating what starts as a problem of conception, tangles with the challenges of communicating science findings, and frequently emerges as a dysfunctional policy.

Background: Historical Range of Variability

Policies that conserve native biodiversity and ecological productivity have become increasingly important in the United States over the last 40 years. Numerous laws, such as the Endangered Species Act (ESA), have been passed with the goal to conserve native species and the ecosystems on which they depend. Major forested areas in the United States, such as our national forests, are managed with conservation of native biodiversity as a central goal. Multiple studies have identified and used HRV in forest structures and processes to help guide management and conservation of biodiversity. These studies have been done for many regions of the western United States including the southwest (Moore et al. 1999, Swetnam et al. 1999), the Oregon Cascades (Cissel et al. 1999), the Sierra Nevada (Millar and Woolfenden 1999), the sequoia forest area of California (Stephenson 1999), and the Oregon Coast Range (Wimberly et al. 2000, Nonaka and Spies 2005).

Landres et al. (1999) provide an overview of the natural variability concepts underlying biodiversity management. They acknowledge that the world is highly modified from the past and that creating static reproductions of past ecosystems is not possible or desirable for most managers. Still, they conclude that natural variability concepts provide a framework for improved understanding of ecological systems as well as for evaluating the consequences of proposed management actions. They further argue that understanding the history of ecological systems helps managers set goals that are more likely to maintain and protect ecological systems and meet social goals, and knowledge of past ecosystem functioning is one of the best means of predicting impacts on ecological systems today. They do caution against using a single a priori time period or spatial extent to define natural variability.

Management of aquatic ecosystems has received a great deal of attention, particularly around salmon (*Oncorhynchus* spp.) conservation and adhering to the ESA. However, because it is not fully understood, the concept of HRV in the management of aquatic ecosystems has had unanticipated consequences. In particular, it has contributed to the illusion of failure in management and recovery programs and policies. At the heart of aquatic regulatory policies lie the narrow or single standards derived from HRV-based assessments. These standards are presumed to be easy to measure, interpret, and enforce: it is assumed that the relationship between an aquatic variable and land management is relatively clearly defined, has well-defined borders, and is generally simple and linear with regard to cause and effect (Holling and Meffe 1996). Moreover, regulatory and policy

standards based on single average values are thought to be based, to a large extent, on the best available science and are applied over broad areas with the expectation of protecting existing favorable conditions and improving less optimal ones.

These expectations and assumptions are, for the most part, mistaken. Measuring and interpreting any in-stream environmental variable is not easy, given the vast complexities involved with transport, mixing, diffusion, and storage of an agent such as water, sediment, or thermal energy. Even with the use of relatively straightforward environmental averages (either based on a time series of flows or on a spatial average of pieces of wood in streams), measurement problems are complicated by the significant temporal and spatial variability encountered in watersheds. Scale considerations are paramount to understanding cumulative effects. The problem is exacerbated when temporal and spatial variability interact in ways that are complex and not well understood (Wallington et al. 2005).

Focusing policies for and management of aquatic ecosystems at the landscape scale presents particular challenges to policy makers, managers, and regulators (Reeves et al. 2002). Although we appear to be grappling with scale and its implications, the understanding of the element of time in aquatic systems is limited in the scientific literature (with some exceptions), and almost nonexistent in public understanding. The interaction of multiple processes operating at multiple spatial and temporal scales is difficult to understand and even more difficult to incorporate into a coherent management strategy. The challenge is to develop a process that not only looks at current aquatic conditions but also the larger watershed context, the historical trajectories of the systems and natural history, and potential threats and expectations.

Changing Scientific Understanding

The study of the intrinsic temporal and spatial variability in the supply and routing of water, sediment, and organic debris in stream systems, and hence in-channel morphology, has gained increasing interest (Swanson et al. 1998; Benda and Dunne 1997; Meyer et al. 2001). In light of recognized natural variability in watershed processes, Pusey et al. (1997) argued that organisms are strongly influenced by the flow regime, that is, the variability in annual flows from extreme floods to low water conditions. Thus, managing the flow regime is more relevant than regulating a mean annual flow. In a similar vein, Poole et al. (2004) and Rieman et al. (2006) challenge the validity of applying metrics developed from studies of individual organisms to whole populations and ecosystems. They contend that such threshold metrics are inappropriate because the inherent variability in aquatic ecosystems is crucial to their long-term productivity.

The focus on single values comes at the expense of recognizing the ecological processes that create and maintain the freshwater habitats of Pacific salmon (Bisson et al. 1997; Beechie and Bolton 1999) and the ecological conditions in which they evolved (Frissell et al. 1997). In recognition of this issue, Holling and Meffe (1996) referred to the use of single threshold values for various environmental parameters as an example of a "command-and-control approach" to natural resource management. They contend this approach often fails when it is applied to situations in which processes are complex, non-linear, and poorly understood, such as in ecosystems containing the freshwater habitat of Pacific salmon and trout (*Oncorhynchus* spp.), and it may lead to further degradation or compromising of the ecosystems and landscapes of interest (Dale et al. 2000; Rieman et al. 2006).

At the same time, the scientific view of the behavior of ecosystems, and the landscapes in which they are embedded, is shifting from an equilibrium perspective to one that recognizes dynamics and non-equilibrium conditions over time (Wallington et al. 2005). The latter perspective views successional processes as much less deterministic than does the former (Pahl-Worstl 1995). In other words, succession does not necessarily occur in an orderly, predictable manner. Instead, it occurs slowly from internal natural changes, as well from large, infrequent events that cause dramatic and rapid change in conditions, and the signature and legacy of these events can influence local conditions for long time periods (Foster et al. 1998). Thus, there may be several ecological states expressed over time, and it is difficult to predict the end point of succession. For this reason, present conditions at any point in time should be viewed as transitory and must be understood in the context of past land use and other human activities, climate, features of the local area, and natural disturbance.

Aquatic ecosystems provide an ideal case for illuminating the problem of management based on equilibrium states and averages. These ecosystems tend to be considered as being in an equilibrium or steady state, and when disturbed they have been expected to return to pre-disturbance conditions relatively quickly (Resch et al. 1988; Swanson et al. 1988). Currently, the range of the conditions exhibited by a particular aquatic ecosystem is assumed to be small, and to lie within a limited number of states. Biological and physical conditions are presumed to be relatively constant through time and to be good (barring human interference) in all systems at the same time. Conditions in aquatic systems with little or no human influences are understood to have the most favorable conditions for fish and other aquatic organisms and are used as references against which the condition of man-

streams (e.g., Index of Biotic Integrity, Karr and Chu 1999) can be assessed. These "pristine" watersheds are used to establish goals for restoration and theoretical threshold conditions for minimizing impacts of management activities.

An example of the mindset regarding aquatic conditions as relatively static, particularly before European settlement, occurs in the Oregon Watershed Enhancement Board's Watershed Assessment Manual. The manual used to assist watershed councils in preparing an assessment in their funding applications for restoration activities. http://www.oregon.gov/OWEB/docs/pubs/wa_manual99/02_history_print.pdf It lists the assumptions behind investigating historical conditions as follows:

Historical accounts provide clues that can be used to develop an understanding of the condition of key watershed resources before settlement. accessed 26 September 2006.

This implies that there is a single condition rather than a range of conditions and assumes that that single condition was the optimum for fish production and should be used as a reference for all restoration.

Another example of static or standardized planning can be found in the requirements of the Clean Water Act. The goal of the Act is to have water that meets a given standard everywhere for various factors (e.g., temperature, turbidity, etc.) at some point in time. It is unlikely that this ever happened in the past or that it will ever happen in the future, despite the best management intentions.

These kinds of assumptions suggest that aquatic conditions were generally stable before European settlement, in terms of both human and presumably biophysical impacts. A subsequent assumption notes that if habitat conditions are suitable for salmonid fish, then they reflect "good" habitat conditions for the watershed, despite copious evidence that unchecked floods and fires routinely create conditions that are far from suitable for salmon habitat, on time frames ranging from years to centuries (Reeves et al. 1995, Liss et al. 2006, Rieman et al. 2006). Specific examples of dynamic changes affecting aquatic ecosystem management include riparian buffers that break down, landslides delivering materials into the stream that subsequently form habitat, floods that are regarded as "catastrophes," and stream reaches in old-growth forest that are regarded as "best" despite considerable evidence to the contrary (Reeves and Bisson 2009).

Changing our Focal Length

Oddly, the terrestrial systems in which aquatic ecosystems are embedded and by which they are presumably strongly influenced are now viewed as dynamic, and as exhibiting a range of conditions. Policy and public understanding are beginning to accept this. Many management approaches and regulations are now aimed at maintaining the dynamic nature of terrestrial systems (Lindenmayer and Franklin 2002), but not of aquatic systems.

We believe that a primary reason for this view is that the major paradigms and classification schemes shaping thinking about aquatic ecosystems do not consider the relevance or influence of time with regard to their behavior or properties, or alternatively we fail to recognize that time is incorporated into the paradigm. The Rosgen channel classification (Rosgen 1994) is an example of the former. It assumes that there is a "stable equilibrium" condition for a given channel type, and that a channel that does not have the features of the equilibrium conditions is considered to be in need of restoration. With regard to the latter, the general interpretation of the "river continuum" (Vannote et al. 1980) is that there is a predictable pattern of organization in the biological and ecological properties of streams as you move through the network (Minshall et al. 1985). The continuum recognizes that there will be replacements in the biological communities over time, but this is not widely recognized or acknowledged when the concepts of the continuum are applied or interpreted. In turn, hierarchical organizations of aquatic ecosystems (Frissell et al. 1988, Fausch et al. 2002) associate appropriate temporal scales with each level of spatial organization. However, they fail to integrate the subsequent influence of time on the behavior and features of the different levels.

Results from some recent research have questioned the static view of aquatic ecosystems. Pristine, or less disturbed, aquatic sites may exhibit a wide range of conditions: Lisle (2002) and Lisle et al. (2007) found that variation in habitat features of pristine watersheds in the Sierra Nevada and northern California, respectively, is much greater than in managed systems. Reeves et al. (1995) found a wide range of physical and biological conditions in three pristine streams in the Oregon Coast Range. Conditions ranged from large amounts of sediment and relatively small amounts of wood in a stream that was relatively recently disturbed to no sediment and large amounts of wood in an old-growth stream that had not been disturbed for more than 250 years. Among other

things, these findings reflect variation in response to disturbance within and among watersheds. These results made them question the validity of using set values from pristine systems to establish benchmarks for management objectives.

Specific life-history adaptations of Pacific salmon to dynamic environments include straying of adults (i.e., fish do not return to the stream in which they were hatched), relatively high fecundity rates (a high number of egg a given body size), and movement of juveniles around the stream network (Reeves et al. 1995). Resident salmonids, such as bull trout (*Salvelinus confluentus*) (Rieman and McIntyre 1995, Dunham and Rieman 1999) exhibit similar adaptations, including multiple life histories, that allow them to persist in dynamic environments. Dunham et al. (2003) cite several examples of the rapid response of fish to wildfire.

Managing Whole Landscapes

Landscape management strives to maintain a variety of ecological states in some desired spatial and temporal distribution. Management at that scale attempts to address the dynamics of individual ecosystems, the external factors that influence the ecosystems that comprise the landscape, and the dynamics of the aggregate ecosystem (Concannon et al. 1999). To achieve this complex goal, landscape management should develop a variety of conditions or states in individual ecosystems at any time, reflecting a highly variable mosaic across the larger landscape (Gosz et al. 1999). The specific features of the ecological states and their temporal and spatial distribution will vary with the objectives for a given landscape (Cissel et al. 1999). A key component of landscape management needs to be recognizing relationships among scales of organization (Caraher et al. 1999). As scale of organization change, the behavior and the relevant principles governing the behavior also change.

Managers and regulators struggle with ecosystem dynamics at these larger scales. One reason for this is that they simply are not accustomed to thinking at large scales or they lack an adequate understanding or tools to do so. Most focus their planning at small spatial scales. Regulators may recognize the need to apply policies and regulations across broad areas but generally default to the application of small-scale reductionist approaches. The struggle is further complicated by the lack of scientifically sound examples of how to operate at large temporal and spatial scales (Johnson and Duncan 2009).

Management and conservation strategies (Holling and Meffe 1996, Dale et al. 2000), including those involving aquatic organisms (National Research Council 1996, Independent Multidisciplinary Scientific Team 1999, Liss et al. 2006), generally encompass large spatial and temporal scales. Many fish populations in the western United States are currently in need of increased legal protection because of declining numbers. This requires moving from the current focus on relatively small spatial scales, which give little or no consideration to the relevance of time, to focus that considers large spatial scales, specifically ecosystems and landscapes, over relatively long time periods (i.e., 10^1 – 10^2 years) (Reeves et al. 1995, Poff and Ward 1990, Poff et al. 1997, Naiman and Latterell 2005). Management and regulatory agencies tend to develop standards from information developed at small spatial scales and assume, generally implicitly, that these standards can be applied across broader areas, merely by "aggregating up." However, this premise is incorrect (Allen et al. 1984, O'Neill et al. 1986); instead, it needs to be recognized that a multi-watershed landscape operates differently through time than does a single watershed. Smaller spatial scales tend to be more variable over time than larger scales (Benda et al. 1998, Wimberley et al. 2000). Thus, conditions in a given watershed can vary widely over time (Reeves et al. 1995, Benda et al. 1998). Failure to recognize the different levels of organization and the potential response of each may lead to serious problems with expectations from the policies and regulations (Caraher et al. 1999) and may incur unintended economic and social costs (Dale et al. 2000), such as repeated investment in the same unlikely restoration outcome.

Background: Social Range of Variability

Beyond any technical questions about the range of variability approach lie the social questions of whether people, policy makers, managers, landowners, and interested citizens—even accept HRV as a guide to managing forests and streams to conserve biodiversity. There does seem to be preliminary acceptance of dynamics as part of terrestrial management. The challenge, then, is to move the broader understanding of landscape dynamics from terrestrial ecosystems into the realm of watershed restoration, where hard realities about ranges of conditions and the importance of time have yet to be faced.

Humans and their activities have always influenced the development of their landscapes to greater and lesser degrees. They have burned vegetation to promote hunting and gathering activities, and they have suppressed fire to promote stand growth and timber profits. They have cleared land for agriculture, then abandoned agricultural lands, they have mined for resources and have restored mined areas, they have channelized and dammed streams.

for flood control and myriad other reasons, they have built villages and cities in floodplains, they have built roads wherever they needed them, they have removed dams and unleashed channels to restore fisheries, wetlands, beaver habitat. Much of our standard of living is based on converting landscapes to conditions outside their historical range to suit our current purposes.

The point of this sample listing is that, in all these human enterprises altering landscapes, which often conflict through time, the chief driver has been social acceptability: when the majority of the population, either by silent consent or participation, has agreed that certain activities are acceptable, the activities have continued, regardless of the impact on the landscape. The dynamic ebb and flow of acceptability amounts to what might be considered the "social range of variability" (SRV), operating alongside the ecological range of variability through time.

We define SRV as the range of an ecological condition that society finds acceptable over a given unit of space and time. The elements in the range can be expressed as the probabilities of conditions that are socially acceptable and differences between the ecological range and SRV can lead society to try to influence ecological variation over time and space (Duncan et al. 2009). By this definition, the SRV reflects social acceptability as it relates to the suite of resource-management options the majority of people will consider acceptable (Shindler and Mallon 2001).

Duncan et al. (2009) developed a framework for considering how to use concepts of variability through time across space that rests on the interaction of ecological and social ranges of variability. Within the framework (redesigned here as Fig. 1), four zones of interaction are implied. In the first, we would find ecological conditions that would occur (without investment or intervention to prevent them) but that do not have social acceptance. This zone would include such events as massive severe wildfires: ecologically within the range of variability, but by no measure socially acceptable. In the second zone, the likelihood of occurrence of any given ecological condition is greater than the likelihood of acceptance but declining, and in the third the likelihood of occurrence is less than the likelihood of acceptance. The fourth zone contains conditions that would not occur (without investment or intervention to enable them) even though a segment of society wants them. This zone represents unrealistic goals for society without investment or other intervention; an example might be having more than two-thirds of the landscape covered by old-growth forests. Social pressure (or negotiation) tends to change the shape of the ecological probabilities curve (as a result of management) and thus creates the range of variability actually experienced. The exact probability distribution can only ever be estimated for the future, as there is always uncertainty about the trend in social preferences. Thus, the SRV and the ecological conditions responding to it should be thought of as potentially highly dynamic.

In the case of watershed restoration, the goal of restoration has been generally taken to mean "in good condition," and, for example in the Pacific Northwest, therefore attractive as salmon habitat. Thus, the socially acceptable efforts of restoration are directed toward returning all watersheds to such a condition. However, as noted above, this has not been the historical ecological reality. The result has been that social pressure to achieve something that lies outside the ecological range of variability foils expectations and creates ongoing illusions of failure. The wrongly placed expectations can subsequently affect design of restoration activities, restoration funding, and attention across the board.

DISCUSSION

The prevailing static focus of aquatic ecosystem management requires little or no understanding of ecological processes (Wallington et al. 2005). In contrast, managing for environmental change requires extensive knowledge of ecological processes and the functional response of species. The dynamic perspective also requires societal understanding and help in establishing goals (Robertson and Hull 2001), and policy makers will need to help the public develop this understanding if they want their policies supported. Furthermore, institutions affected by the policies and regulations founded on dynamics need to be flexible and adaptable and capable of dealing with complexity and uncertainty—no small task (Stankey and Shindler 2006).

There is, so far, an absence of compelling scientific evidence for many of the behaviors of ecosystems suggested by the emerging non-equilibrium, dynamic perspective (Wallington et al. 2005). Unfortunately, because of human impacts there are relatively few places where the current landscape pattern can serve as a reference. References for forested portions of the landscape have been estimated by reconstructing the previous conditions using historical fire regimes (e.g., Cissel et al. 1999) or models (e.g., Wimberly et al. 2000). However, there are essentially no examples for the aquatic portion of landscapes in the peer-reviewed literature. Reeves (unpublished) estimates that 30%–60% of the watersheds (7th-order Hydrologic Units) in the central Oregon Coast Range were in "good condition" on average during the current climatic period. This was based on the results of Reeves et al. (1995),

which showed that the most diverse watersheds, with regard to fish and habitat, were those dominated by 120-160-year-old vegetation. The proportion of watersheds that contained these vegetative ages was determined from the results of Wimberly et al. (2000). Much more work is needed, and this uncertainty makes it difficult for managers and policy makers even to know which theories are important, much less how to apply them in the real world (Hobbs 1998). The overriding objective remains to have a mix of conditions at the broader scale, which requires that individual sites each exhibit a range of conditions over time (Figs. 2 and 3).

Interactions at higher levels of organization are slower than those at lower levels. Consequently, the range and variability in the properties and conditions of the system are relatively wide at lower levels of organization compared with range and variability at higher levels (Wimberly et al. 2000). The range of conditions seen in riparian ecosystems at the site scale might vary from a recently disturbed site, with no or only a few large trees to a site fully stocked with mature trees. The range at the watershed scale is smaller than this because the likelihood that the entire riparian zone would have either no or few trees or all large trees is very remote. At the landscape scale, the range of variation of conditions in riparian zones was even smaller, implying that not all riparian zones within the landscape were in "good" condition at any point in time nor were they all "poor." Consistency at the small scale (site or subwatershed) is determined by the range of variability established at the larger scales (watershed or basin).

The challenge is to develop a process that not only looks at current aquatic conditions but also:

- Looks broadly to determine the large-scale context.
- Looks historically to assess past trajectories of the systems and natural history.
- Looks ahead to identify potential threats and expectations.

This perspective would allow for a more integrated response to basic questions such as: where are we, where we want to go, and how do we get there?

Management practices and regulations should also recognize that there may be natural variation in the potential productivity and diversity of different parts of the landscape (Dale et al. 2000). Recent work by Burnett et al. (2003), working in the Oregon Coast Range, found that potential for watersheds to provide habitat for coho salmon (*Oncorhynchus kisutch*) and steelhead (anadromous *O. mykiss*) varied among watersheds depending on specific geomorphic features. Watersheds with low-gradient channels in wide valleys provide better potential habitat for coho salmon, whereas watersheds with higher-gradient channels in narrow valleys provide better potential habitat for steelhead. The National Oceanic and Atmospheric Administration (NOAA) Fisheries (Agrawal et al. 2005) has identified such areas across the distribution range of coho salmon and steelhead in southern Oregon and northern California. Each of these types of watersheds also experiences a different range of ecological conditions in response to disturbance. The range of conditions in watersheds with higher potential to provide habitat for steelhead is narrower than the range of conditions in watersheds with higher potential to provide habitat for coho salmon.

Applying fixed standards developed for small spatial scales with the expectations of achieving some desired set of constant conditions over large areas is likely to have unintended ecological consequences and create false or unrealistic expectations about the outcomes for policies and regulations (Holling and Meffe 1996, Caraher et al. 1999, Dale et al. 2000, Poole et al. 2004). These standards could compromise or decrease the long-term productivity of ecosystems (Holling and Meffe 1996) and lead to the homogenization of naturally diverse and dynamic ecosystems (Bisson et al. 1997, Poole et al. 2004). More realistic expectations about the conditions of aquatic ecosystems would aid in developing, implementing, and assessing conservation and recovery plans, and generating an understanding of objectives and support from the public and from private landowners.

CONCLUSION

A recent study of the use of HRV in planning biodiversity conservation identified five sites around the United States and conducted workshops designed to elicit input from landowners, interested citizens, scientists, and managers (Johnson and Duncan 2009). The sites were in Georgia, Massachusetts, Colorado, Oregon, and California.

Findings varied significantly across the sites at the level of local context, but one conceptual consistency emerged in general, aquatic ecosystems do not attract as much notice as terrestrial systems when participants discuss restoration and biodiversity conservation issues. It could be stated that aquatic systems are perceived as "given" to a greater extent, and if they need restoration it is to a fixed former state of "health." There was no evidence

willingness to let aquatic systems traverse through multiple conditions through time. One site even had a salmon restoration program for the region despite a lack of data indicating they had ever been native to the area. This of understanding of aquatic dynamics, noted in the literature reviewed above, could reflect a poorer understanding of the real meaning of dynamics in the aquatic context. The public is less likely to support management action based on concepts they do not understand, and HRV concepts are not currently well understood, even among segment of the public that is attentive to forest-management issues (Shindler and Mallon 2006). On the application side, a key issue is how management and restoration approaches can move away from the use of averages that tend to confine the response range of a given stream, ecosystem, or landscape.

In a broader social sense, the aquatics arena can turn the HRV concept into a learning opportunity. Social analysts have noted the need to see science communication as a process as well as a product, to allow information exchange which multiple points of view are recognized, and to integrate science into the political process (e.g., Priest 1999, Pouyat 1999, Weber and Word 2001).

Understanding dynamics in aquatic environments will take focused effort on the part of scientists, educators, and land managers. In the abstract, they will be attempting to reduce the social pressure to manage outside the historical range of variability: i.e., restore all watersheds to "good condition" at once. In the tangible world, spreading the understanding of landscape dynamics into the aquatic environment may trade illusions of failure for conflict over which watersheds are most worth saving based on historical trends. Although writing off some watersheds would not be a socially acceptable solution at the local level, using restoration resources more effectively is socially acceptable at the regional and national scales.

The dynamic nature of aquatic systems—although it is right in front of us with every flood or rain storm—is not a simple concept to convey to the public or to manage for, and understanding that our best restoration efforts may sometimes be bound to fail is a central challenge to our understanding of time. Without a crisis event, science findings tend to move only slowly into public understanding, and likewise into the policy arena. Although the delay is important for verifying and replicating findings, in our case, it can also contribute to depletion of restoration resources by preventing thoughtful prioritization of work, or even stopping work altogether to avoid failure. At the same time, the SRV continues to ignore the variability of past aquatic systems, and restoration efforts continue to aim at a static goal because that is socially acceptable.

It is clear that social acceptability can be a constraining and complex element of aquatic ecosystems management and policy. The acceptability of HRV as a guide in this endeavor has many dimensions, from general questions about how science and scientific results gain acceptability in the policy process to specific questions about the acceptability of different facets of the range of variability. The watershed restoration context clearly raises conjoined issues of acceptability and ecological tolerance that will need further attention for management to proceed successfully. In this, managers may be helped by reevaluating the watershed assessment process, which may offer the most effective communication opportunity: by requiring inquiry into and understanding of past ranges of variability rather than a fixed good condition, funding entities may begin to seek broader understanding and acceptance of how aquatic systems function through time.

RESPONSES TO THIS ARTICLE

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ACKNOWLEDGMENTS

We thank Kathryn Ronnenberg for graphic design and manuscript copy editing. Norm Johnson provided an early review of the document. An anonymous reviewer provided additional comments.

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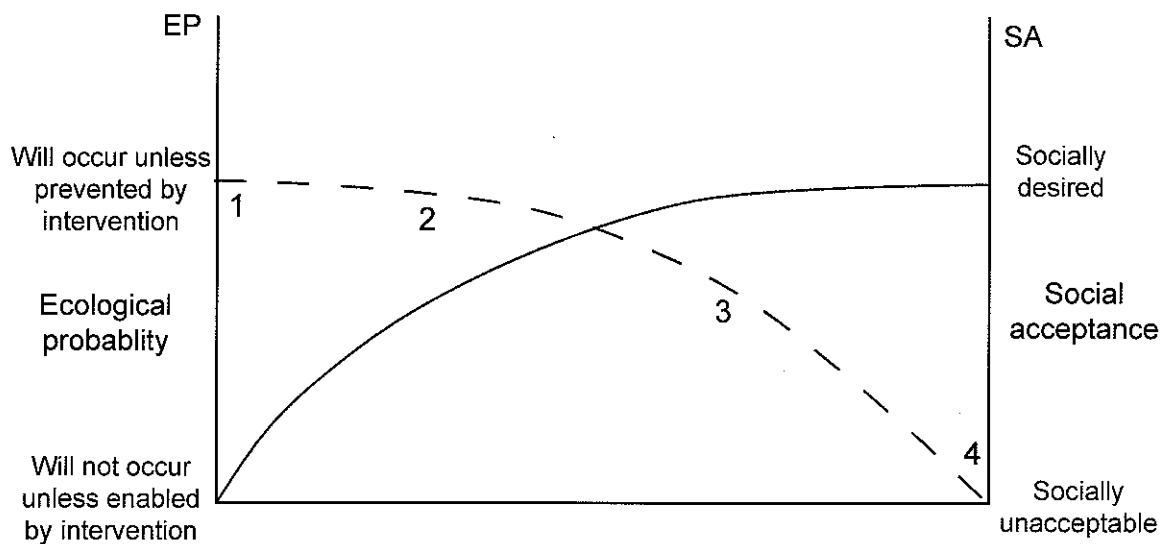
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Fig. 1. Conceptual relationships of ecological probability vs. social acceptability.



Zones:

1. $SA = 0$. Ecologically likely to occur unless prevented by intervention or investment, but not at all socially acceptable.
2. $EP > SA$. Likelihood of occurrence of particular ecological condition is greater than the likelihood of social acceptance, but is declining.
3. $EP < SA$. Likelihood of occurrence is less than the likelihood of acceptance.
4. $EP = 0$. Ecologically unlikely to occur unless enabled by intervention or investment, even though some segment of society desires the condition.

Fig. 2. Schematic illustration of the possible historical mixes of habitat quality in 7th-field watersheds of the Oregon Coast Range over time, with quality largely a function of the time since last major disturbance and the legacy of that disturbance.

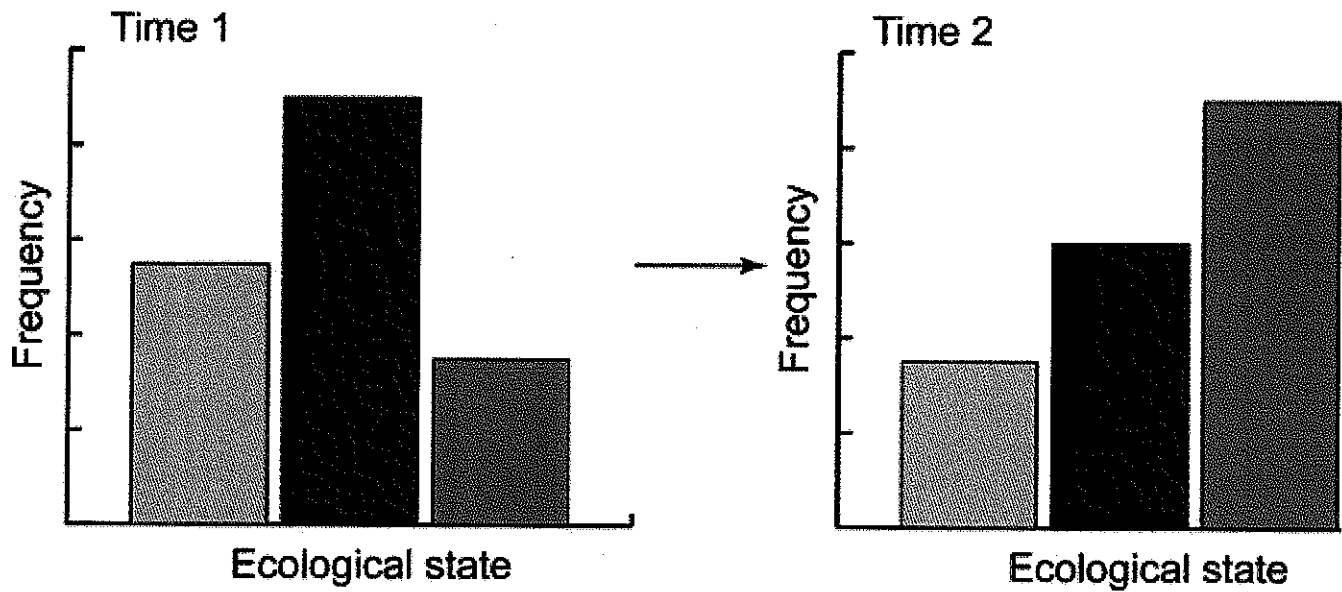


Fig. 3. A schematic of aggregate watershed condition (ecological state) at any time. The map on the left might be the current distribution of ecological condition in 7th-field watersheds and the map on the right might be the target condition. The target condition could be based on the historical distribution of ecological states.

