

# Evaluating and Managing Cumulative Effects: Process and Constraints

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**ABSTRACT** / Cumulative effects (CEs) result from the combined effect of multiple activities over space or time. This implies a persistence through time and often a transmittal mechanism through space. Environmental legislation often requires a broader CE assessment in addition to the more direct, project-specific impacts. Current efforts to evaluate and manage CEs are hampered by the conceptual problems of defining the key issues, specifying the appropriate spatial and temporal scales, and determining the numerous interactions and indirect effects. These problems can be greatly alleviated by following an explicit process. The process proposed in this paper includes a scoping phase, an analysis phase, and a planning and management phase, with each phase consisting of two to five discrete but interrelated tasks.

Numerous approaches have been developed to assess CEs, and these range from simple checklists to complex, physically based models. The utility of each approach depends on the resource of concern, relative risk to those resources, information available, and time frame for the evaluation. In nearly all cases the assessment and regulation of CEs is severely hampered by the variability in site conditions and management effects, inability to predict secondary or indirect effects, lack of data on recovery rates, difficulty of validating predictive models, and uncertainty of future events.

Since any proposed activity could contribute to a wide range of potential CEs at different spatial and temporal scales, a tiered or nested approach should be followed to assess CEs. The difficulty of assessing and predicting CEs also suggests that in many cases the most efficient approach is to focus on minimizing on-site impacts. Under some circumstances adaptive management can also be a viable alternative to detailed CE assessments. Regular monitoring and feedback is critical to the successful management and regulation of CEs.

## Types of Cumulative Effects

The basic concept of cumulative effects (CEs) is deceptively simple. A single automobile, a small clear-cut in a forest, or a single septic tank is unlikely to have a measurable effect on air quality, runoff, or water quality, respectively. However, with increasing numbers of cars, clear-cuts, or septic tanks, the likelihood of a detectable change increases. Because this change results from multiple activities at different locations (Figure 1a), this type of CE occurs either off-site or across a large area, such as an airshed.

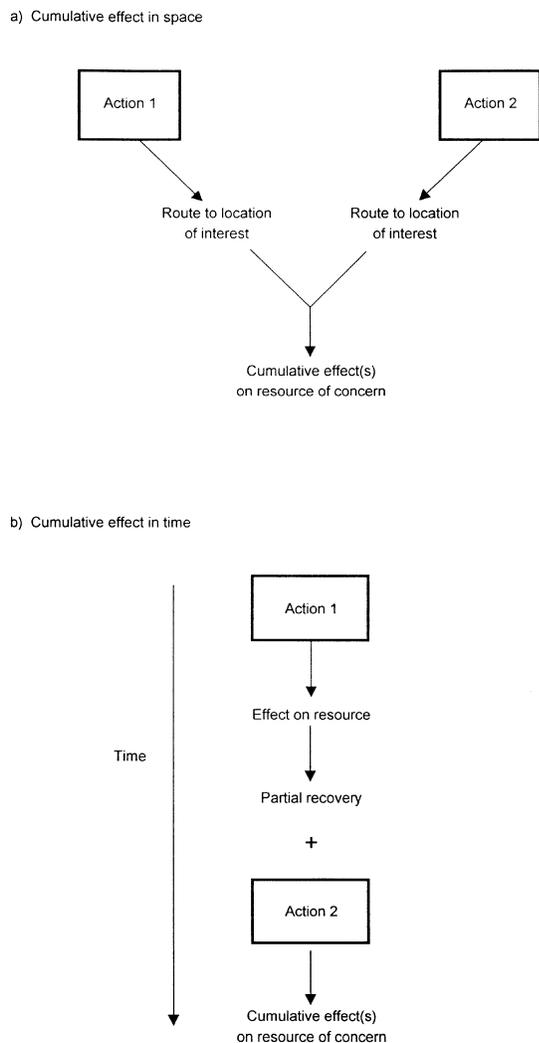
A CE can also be generated by sequential activities on the same site if the initial effect persists and interacts with subsequent activities (Figure 1b). Examples of the latter include soil compaction by repeated activities on the same plot of land (Geist and others 1989, Shepperd 1993), or the successive application of road salt killing the immediately adjacent vegetation (Hoffstra and Hall 1971). In most cases, however, a CE results from multiple activities in space that also persist in time.

CEs can be additive or synergistic, although the

former is far more common (Cada and Hunsaker 1990). To be synergistic the combined effect has to be greater than the sum of the individual effects, and in most physical systems this will rarely be the case. Synergism is more likely when different chemicals can interact or when changes in water temperature or hardness alter the form and toxicity of heavy metals (Abel 1996). In other cases a set of management activities might generate a nonlinear response, and some might consider this a type of synergism. For example, unpaved roads, forest harvest, ski areas, and grazing may each increase the amount of sediment delivered to a particular stream. Though much of the sediment may never be delivered downstream (Walling 1983), the resultant increase in sediment is an additive effect that could disproportionately reduce fish spawning or rearing success (Waters 1995). If fish are specified as the primary issue of concern, this nonlinear biological response to an additive change in sediment loads might be considered as a synergistic CE.

Additional complexity is derived from secondary or indirect effects. For example, an increased sediment load is likely to reduce the cross-sectional area of the stream channel, thereby leading to an increase in bank erosion and a widening of the stream (Figure 2) (e.g.,

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**Figure 1.** Possible combination of activities over space (a) or time (b) that will lead to a cumulative effect.

Ryan and Grant 1991). In addition to further increasing sediment inputs into the stream, the bank erosion will remove some of the riparian vegetation and increase stream temperatures (Grant 1988). These changes may further affect the fish and other components of the aquatic ecosystem (Figure 2).

The great challenge in analysing CEs is recognizing and predicting the numerous interactions and indirect effects (Dixon and Montz 1995); at some level the prediction of CEs is an intractable problem (Stakhiv 1988). However, this paper will argue that much of the confusion in classifying, defining, assessing, and managing CEs is due to poorly defining the resources of concern and the spatial and temporal scales of the analysis. Hence, this paper will present an explicit process for assessing and managing CEs, and this will be followed by a discussion of the current methods and

inherent limitations to assessing CEs. The final two sections discuss alternative approaches for managing CEs and the need to tier or nest CE assessments.

## Legal Background

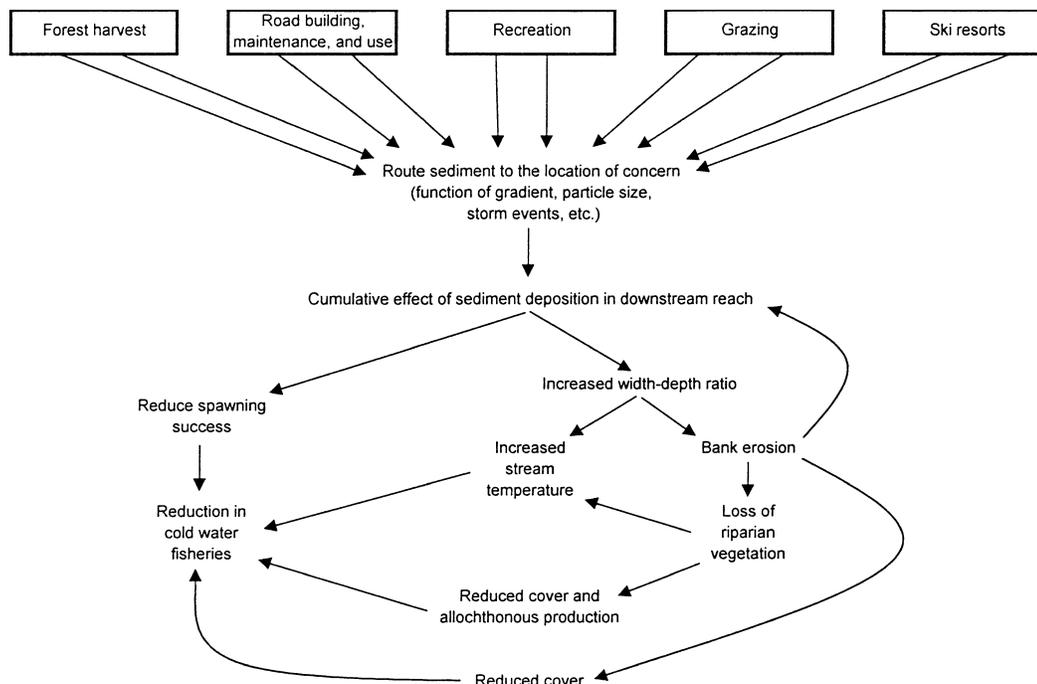
The assessment of CEs generally stems from the more general requirement to assess the environmental impact of any proposed project. In the United States the 1969 National Environmental Policy Act (NEPA) requires an assessment of environmental impacts, including the “results of all past, present, and reasonably foreseeable future actions,” regardless of who is responsible for each action (Thatcher 1990). A series of court cases helped transform this general principle into explicit regulations to consider the cumulative impact of a proposed action (Thatcher 1990, Herson and Bogdan 1991). Regulations promulgated by the U.S. Council on Environmental Quality (CEQ) define a cumulative impact as:

the impact on the environment which results from the incremental impact of the action when added to other past, present, and reasonably foreseeable future actions regardless of what agency (Federal or non-Federal) undertakes such other actions. Cumulative impacts can result from individually minor but collectively significant actions taking place over a period of time. (40 C.F.R. 1508.7)

Similar regulations have been developed at the state and local level in the United States as well as in other countries (e.g., Sonntag and others 1987, Dixon and Montz 1995).

Although NEPA provided the initial impetus for assessing CEs in the United States, other environmental legislation has led to analogous requirements. For example, the Clean Water Act regulates point sources through a permit system and attempts to reduce pollution from nonpoint sources through the promulgation of best management practices (BMPs). If a water body still does not meet water quality standards after the application of technology-based or other required controls, the offending pollutant has to be regulated on a catchment scale by establishing a total maximum daily load (TMDL) (US EPA 1991). The first steps in this procedure are to determine the assimilative capacity of a water body for the pollutant of concern and the natural loading. Since the TMDL is defined as the allowable pollution from all point sources (“wasteload allocation”), all anthropogenic nonpoint sources (“load allocation”), and a margin of safety (US EPA 1991), this process is effectively a CE assessment. In the case of a TMDL, the assessment is also used to determine the allowable loading from each source.

Given the limitations in data, understanding, and



**Figure 2.** Example of how multiple actions can generate a series of secondary and interacting effects. The direct and indirect effects can then result in a nonlinear biological response.

personnel, regulatory agencies have been reluctant to initiate the TMDL process for each water body not meeting water quality standards. Recent lawsuits are forcing TMDLs to be rapidly implemented on a large number of basins throughout the United States. The problem is that for many pollutants, such as clean sediment, there is a severe lack of methodologies to determine the assimilative capacity, set a margin of safety, and accurately predict the contribution and delivery of each source of pollution (US EPA/USFS 1993, Benda 1999).

Similar problems have arisen in the realm of wetland regulations (Stakhiv 1988). In the U.S. Section 404 of the Clean Water Act established a permit system for dredging and filling wetlands, and in 1995 approximately 62,000 permit applications were received by the U.S. Army Corps of Engineers (Abbruzzese and Leibowitz 1997). Public concern over the cumulative loss of wetlands, when combined with the requirements of NEPA, means that the Corps of Engineers is having to develop procedures to assess the CEs associated with changes to individual wetlands (Johnston 1994).

Larger-scale CEs are also a concern in hydropower development. Between 1979 and 1985 there were over 6100 hydropower applications in the United States, and this is more than double the number submitted in the previous 60 years (Cada and Hunsaker 1990). Again, this has forced the Federal Energy Regulatory Commis-

sion to initiate studies and develop procedures to evaluate CEs (McCold and Saulsbury 1996).

## Objectives

Although there is a strong legal mandate to assess CEs, there is little consensus about how this might be done. Recent publications indicate that there has been relatively little cross-fertilization between the various disciplines concerned with CEs (e.g., Williamson and Hamilton 1989, Reid 1993, Johnston 1994). Thus the objectives of this paper are to: (1) set out a conceptual procedure for assessing and managing CEs; (2) define the critical issues and current limitations in assessing and managing CEs; (3) discuss alternative approaches to manage potential CEs; and (4) discuss the key trade-offs and public policy issues inherent in efforts to assess and regulate CEs.

This paper will emphasize cumulative watershed effects, as these have been the most extensively studied (see Reid 1993) and continue to be a flashpoint of concern. However, most of the issues and discussion will be more broadly applicable. This paper will focus on the analytical aspects of assessing and managing CEs in natural resource systems and not delve into the broader sociopolitical issues of resource allocation and decision making (Smit and Spaling 1995). By drawing on the literature from different fields, this paper should help

develop more rational procedures to assess CEs, clarify key issues for public debate, and ultimately help ensure that the limited resources for environmental regulation are utilized as efficiently as possible.

### Process for Assessing and Managing Cumulative Effects

The assessment of CEs is particularly problematic because of the large number of affected resources, the numerous pathways by which these resources can be affected, and the uncertainty over the appropriate spatial and temporal scale(s) for the analysis. In many environmental assessments CEs are not even considered (McCold and Holman 1995), and this may be due in part to the lack of an explicit process for analyzing CEs. Often a CE assessment is nothing more than a cursory checklist covering a smorgasbord of issues. Such an approach may satisfy the legal requirements, but it typically provides minimal guidance to managers or decision makers. A recent U.S. government report noted that “federal agencies have struggled with preparing CE analyses since CEQ issued its regulations in 1978” (CEQ 1997, p. 4).

Both experience and a review of the literature suggests that a more focused and directed process for assessing cumulative effects is needed. A recent guide to assessing cumulative effects was a significant step forward (CEQ 1997), but this downplays the more difficult decisions and limitations in assessing cumulative effects. In particular, the CEQ guide does not explicitly address the need to: (1) define recovery rates, (2) evaluate the role of natural processes, and (3) determine the effect of temporal and spatial variability on both the predictability and the detectability of CEs. Each of these factors greatly affect the likelihood of a CE, and thereby the level of effort that should be devoted to a given assessment.

The underlying conceptual similarities in the generation and evaluation of CEs suggest that a generic, explicit process can be defined to guide the assessment and management of CEs. The process presented in this paper was developed from an extensive literature review, practical experience, and U.S. case law. The process can be divided into three phases, and these are the scoping phase, analysis phase, and implementation and management phase (Figure 3). Within each phase there are two to five interrelated steps. Although the process in Figure 3 is linear, in practice the assessment is almost always an iterative process and may not proceed in sequence. The point is that each of the steps in the process is critical to the overall assessment, and the omission of a step will typically lead to an incomplete or

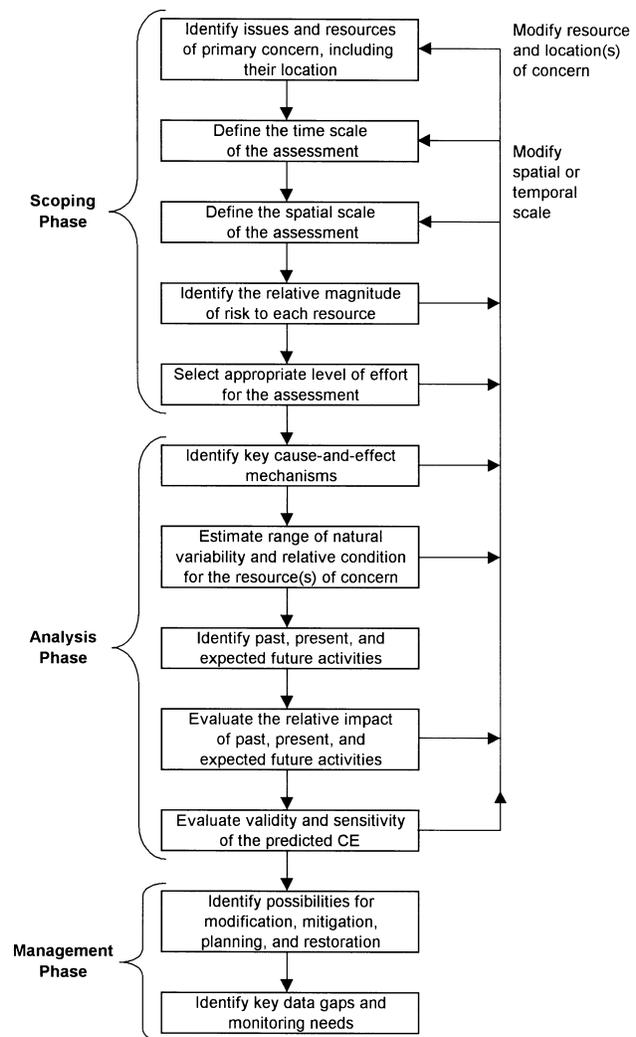


Figure 3. Conceptual process for assessing cumulative effects.

flawed analysis. The following paragraphs briefly explain the purpose and key issues associated with each step.

#### Scoping Phase

*1. Identify the issues and resources of concern, including their location.* The first step is to clearly identify the issues and resource(s) of concern, as this will largely determine the scope and type of CE assessment (Harr 1981, Reid 1993). As an example, forest harvest, roads, and other management activities will generally increase both the size of peak flows and the amount of sediment reaching the stream channel (Reid 1993). Both of these increases are usually considered detrimental to downstream aquatic resources (MacDonald and others 1991). The problem is that different processes control the respective changes in runoff and sediment production, each of these processes recovers at a different rate, and

the different effects will extend across different spatial and temporal scales (i.e., changes in runoff will be more quickly transmitted downstream and dissipated than an input of coarse sediment).

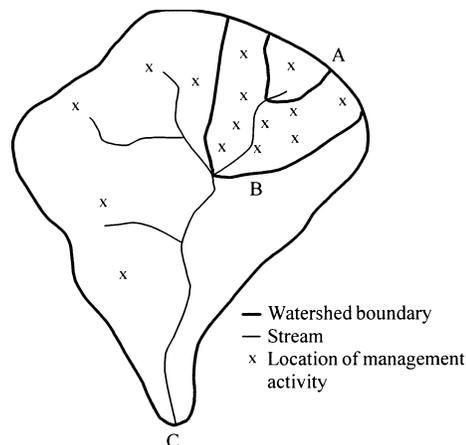
Similarly, if the basic problem is the cumulative loss of wetlands, a different analysis and different analytic units must be used according to the primary concern (Preston and Bedford 1988). If the primary concern is the cumulative loss of flood storage, the analysis has to estimate the amount of flood storage associated with different types of wetlands and the likely effect of the proposed project. If the primary concern is the loss of shorebird habitat, the hydrologic aspects are not as important and the CE assessment must develop metrics that reflect habitat quantity and quality.

These types of differences in the controlling processes means that separate models must be used to assess the potential impact of a proposed project on different resources. Similarly, a single index of disturbance cannot be used to predict separate effects over different spatial and temporal scales. This means that one has to explicitly identify the issues of concern a priori, and then develop the appropriate analytical models. Each analytical model may draw on the same database (e.g., landscape characteristics and a land use history), and there may be some interactions between the models used to analyse different issues (Figure 2), but agencies have to stop using one model to predict different types of CEs (e.g., Cobourn 1989).

In addition to identifying the issues of concern, this initial step also has to explicitly identify the location(s) of concern. This is critical to specifying the spatial scale of the assessment (see step 3 below) and keeping the assessment as specific as possible. If the locations of concern are not clearly identified, the assessment loses focus and can become impossibly large.

**2. Define the time scale of the assessment.** The second step in the scoping phase is to define the time scale(s) to be considered in the assessment. The present landscape is the result of natural and anthropogenic processes that span a wide range of time scales (James 1991, Schumm 1991, Benda 1999), and the assessment must take these into account (see step 7). Similarly, current and reasonably foreseeable future management activities may directly or indirectly affect future conditions for decades (e.g., Ziemer and others 1991). Since any change will have a nearly infinite number of potential repercussions and secondary effects, one must set a specific limit on the type and time range of future effects. This means that specifying the temporal scale for a CE assessment is somewhat arbitrary and depends on the assumed recovery rates.

A focus on shorter-term effects will generally simplify



**Figure 4.** Effect of changing spatial scale on the magnitude of a predicted CE. At the largest scale (outlet of watershed C), the relative magnitude and hence the detectability of a CE will generally be smaller than at the outlet of watershed B. At the smallest scale (outlet of watershed A), one can only observe the effect of a single activity.

the analysis and reduce the uncertainty, and assessments that extend further into the future will generally require more historic data and have greater uncertainty. In many cases the prediction of future CEs will require a stochastic approach in order to account for the uncertainty and variability of future events (e.g., Benda and Dunne 1997a).

**3. Define the spatial scale of the assessment.** Another key step in the scoping process is defining the spatial scale of concern, as the scale will affect the complexity of the assessment and can alter the magnitude of a projected CE (Benda 1999). In general, expanding the spatial scale of the CE assessment diminishes the incremental effect of a proposed activity, and thereby the likelihood of a significant CE (Figure 4) (Bunte and MacDonald 1999). Conversely, focusing on the smallest scale possible will tend to maximize the change caused by multiple activities (Harr 1989). Any further reductions in scale will tend to eliminate the cumulative effect and highlight the direct effect of an individual action or project (Figure 4). To minimize bias, the spatial scale of the assessment should be defined by the spatial scale of the processes that control the resources of concern as specified in step 1. However, some resources, such as migratory birds or air quality, can be affected by activities well outside the geographic zone where the immediate management decisions are being made. In such cases one generally can't consider all anthropogenic impacts and therefore has to limit the geographic scope of the analysis. Specifying explicit geographic boundaries then helps to define the approach and complexity of the assessment.

In the United States the courts have helped define the appropriate spatial scale of a CE assessment (Herson and Bogdan 1991). For example, the Corps of Engineers need not consider the effect of filling a 2-ha wetland in Oregon on the total amount of waterfowl habitat in the Pacific flyway, nor does the U.S. Forest Service need to evaluate how a new road in an upstream tributary might affect downstream reaches in the Mississippi River (Thatcher 1990).

Though these examples may be scientifically defensible, they also illustrate an inherent contradiction in any effort to limit the spatial scale of a CE assessment. By definition a CE assessment is supposed to evaluate impacts over larger spatial and temporal scales, and limiting the spatial scale can exclude some larger-scale effects. From society's perspective, is it cost-effective to conduct larger-scale, comprehensive assessments for every proposed action or project despite the uncertainty in the predictions and our limited ability to control all the causal factors? Or is it better to conduct more limited assessments and use the time and funds to modify projects, mitigate adverse effects, redress past problems, and monitor key resources?

The next step of the procedure addresses these trade-offs, and the last section of this paper suggests that a nested or tiered approach may be the most effective way to address CEs at different spatial scales. In general, larger-scale assessments are useful for policy decisions and to provide a context for specific management decisions; smaller-scale assessments provide more specific guidance for a proposed action or project.

*4. Identify the relative magnitude of risk to each resource, and adjust the scope of the assessment according to the likely cost of a wrong answer.* As noted above, there is always a trade-off between a more comprehensive and costly assessment versus a more limited approach. There are several criteria that can be used to address this trade-off, and these should be explicitly considered as part of the overall scoping process. The first criterion is simply to compare the relative risk to each resource of concern on an initial, largely qualitative basis. Here risk is defined as the value of a resource times the probability of an adverse effect. A more detailed assessment is justified when the resource is a highly endangered species or particularly unique and sensitive (e.g., an oligotrophic lake in a national park). Similarly, a more detailed and rigorous assessment may be justified if, on the basis of best professional judgment, large adverse effects are likely or irreversible harm may result from a hasty decision.

On the other hand, there have to be some limits with respect to the consideration of indirect effects, as every activity can be directly or indirectly linked to many

potential resources of concern. Court decisions in the United States have been somewhat contradictory regarding the evaluation of secondary effects and connected actions (Thatcher 1990, Herson and Bogdan 1991). The U.S. Forest Service could not separate building a road from the timber harvest activities that were likely to follow, but the U.S. Army Corps of Engineers did not need to consider the development that might result from port improvements in Hawaii (Thatcher 1990). Ultimately it is a matter of judgment as to whether the potential adverse environmental effects are significant enough to justify a more rigorous analysis. In most developed countries there are legislative and judicial mechanisms that help ensure these trade-offs are generally consistent with the values of that society.

*5. Select the appropriate level of effort for the assessment.*

The four previous steps should provide an initial estimate of the effort to be devoted to the CE assessment. By defining the issues and locations of concern, the spatial and temporal scales of the assessment, and the relative risk, it should be apparent whether the assessment will require several person years and complex, quantitative models, or whether a simpler, qualitative assessment will suffice (Berg and others 1996).

Although selecting the level of the assessment is defined here as a technical decision, in reality the level of the CE assessment will also be determined by the degree of public concern and the financial and human resources available for the analysis. In less controversial cases an organization may not be able to devote large amounts of time and effort to assessing CEs. Thus the scope and quality of a CE assessment is often determined by these political and institutional considerations. Interested parties then have to resort to administrative appeals or the judicial system to force a change in the scope or approach of a particular CE assessment.

Analysis Phase

*6. Identify key cause-and-effect mechanisms.* Once the scoping phase has been completed, the analysis phase can be broken into five additional steps (Figure 3). The first of these steps is to identify the key cause-and-effect processes. Again the purpose of this step is to focus the assessment on the most important mechanisms, and avoid getting lost in the infinitely large universe of indirect effects and interactions. For example, we know that forest harvest, roads, fire, recreation, and urbanization are all likely to increase the amount of sediment in the stream network, and this sediment might be derived from surface erosion, mass movements, and in-channel sources. In most cases it will not be possible to rigorously quantify all of these sources and mechanisms, so one has to focus on the most important activities and

processes (NCASI 1992). Inevitably this selection will be based on qualitative factors such as experience and professional judgment, and thus the identification of the key mechanisms needs to be justified by reference to current knowledge and understanding.

Several iterations may be necessary to complete the identification of key cause-and-effect mechanisms, as additional processes or interactions may become apparent during the first phase of the analysis, and this may alter the scope of the assessment. A basic principle in modeling is to start simply and only add complexity as needed (Woolhiser 1982). The same philosophy should be applied to the assessment of CEs, as a CE assessment is inevitably a simplified representation of an infinitely complex, and hence ultimately unknowable, system (Stakhiv 1988). The involvement of several people in this step is important to minimize bias and help ensure that all of the key mechanisms are represented.

7. *Estimate the range of natural variability and relative condition for the resource(s) of concern.* A critical step in any CE assessment is to compare the current condition and projected changes with the natural variability in the resources and processes of concern (Cada and Hunsaker 1990, Contant and Wiggins 1991). Often this will be difficult given the lack of long-term data and the long history of disturbance in most areas (Benda 1999). Nevertheless, there are several means by which one can try to put the current condition and predicted change into historical context.

The first step is to understand the basic processes that drive the system of interest. If the resources of concern are subject to infrequent large disturbances, such as large floods or debris flows, the range of natural variability will be much larger than systems driven by more chronic processes, such as soil creep or snowmelt floods (e.g., Caine and Swanson 1989, Grant and Wolff 1991). This understanding of the controlling processes is critical to assessing the adequacy of current data.

In many cases only a limited amount of data will be available for key processes and resources, but longer-term records from other systems can be used to estimate the likely natural variability over longer time scales. For example, short-term records of annual water yield or sediment load might be correlated with a longer record nearby. Alternatively, tree rings or the thickness of sediment deposits in a lake may be correlated with annual sediment loads, and this can provide a longer-term context for the proposed activities. Even if the two systems are not highly correlated, similarities in the physical environment might allow one to at least assume that the magnitude of the variability should be similar.

Historical information can also be used to provide a temporal context. Old aerial or ground-based photos

can be used to compare and analyse longer-term changes. Newspaper articles may report more extreme events, and historic journals, letters, or other accounts can indicate the past status of a resource. Paleoflood analyses, cesium-137, bathymetric surveys, and other techniques can be used to indicate the magnitude of past events or average rates over time (Jarrett 1991, Loaiciga and others 1993, Walling 1995).

In the absence of data on the temporal variability or predisturbance condition of a resource, the usual approach is to compare the system of interest to other, less disturbed systems. The implicit assumption is that the systems are part of the same population and therefore directly comparable. The corollary is that any differences are presumed to result from past disturbances.

There are several problems with this approach that must be recognized in a CE analysis. First, there may not be an analogous undisturbed system, either because the resource is unique, or all the comparable systems have also been significantly disturbed. Second, there is often a reason why a particular system has been left undisturbed, and this may limit its usefulness as a valid comparison. A forested basin may have been left uncut because it was extensively burned 20 or 50 years ago, and this may still be affecting the pattern of vegetation, amount of large woody debris in the stream, amount and type of fish habitat, or type and number of fish. Finally, the presumption that the systems being compared are similar may not be valid. The inherent variability and complexity of most natural systems, together with the differences in anthropogenic activities, limits the comparability of systems over time and hence the certainty of a CE assessment (Bunte and MacDonald 1995, Bolstad and Swank 1997).

When assessing CEs, there is a tendency to minimize differences between sites as well as the natural variability over time. People tend to regard the conditions they observe as "typical," but this characterization is usually based on only a very limited personal or institutional memory. Nearly all long-term data sets exhibit much greater variability than might be apparent from shorter periods (e.g., 5–15 years). Moreover, completely different trends can be identified depending on which time period is selected (e.g., Figure 5) (Schumm 1991). The tendency to underestimate differences between sites and the natural variability over time will lead to an overestimation of the significance of possible CEs (Harr 1981). In systems with very high levels of natural variability it also is much more difficult to detect adverse CEs (Bunte and MacDonald 1999). The tendency to overestimate the magnitude, likelihood, and detectability of CEs then has implications for the amount of effort that should be devoted to a CE assessment. The extent



**Figure 5.** Variation in water levels in the Great Salt Lake, 1850–1993.

to which a proposed project has to mitigate or compensate for the adverse effects of past activities raises questions of fairness and values that are not easily resolved.

*8. Identify past, present, and expected future activities in the area of concern.* An inventory of past, present, and expected future activities is the basis for evaluating CEs on the resource of concern. The amount of effort that needs to be devoted to this task is highly dependent on the spatial and temporal scales defined in the scoping phase. The difficulty of this step should not be underestimated, as acquiring information from a wide range of public and private sources is a very difficult task. Unfortunately the difficulties increase as one attempts to extend the temporal scale back in time or gain a more complete picture of “reasonably foreseeable future actions.” Ideally the inventory of activities from each CE assessment could be entered into a common geographic information system (GIS), as an increasingly complete database would greatly facilitate other CE assessments.

The use of GIS databases would also allow a more explicit evaluation of the spatial linkages between each management activity and the resource of concern (Swift and others 1995, Eedy 1995). This could then lead to much more realistic models than the lumped models commonly used for evaluating cumulative watershed effects (e.g., Cobourn 1989, McGurk and Fong 1995). GISs are particularly good at displaying complex spatial data, and this can facilitate environmental decision making (Eedy 1995).

*9. Evaluate the relative impact of past, present, and expected future activities.* This step combines the past, present, and planned disturbances with the cause-and-effect processes to estimate the cumulative impact(s) on the resources of concern. Depending on the relative risk rating and the resources available, this evaluation can range from relatively simple qualitative assessments

to complex, physically based models. For example, the amount of forest harvest within a given basin is sometimes constrained by a simple limit on the percent of a catchment that can be cut within a given time period. Slightly more sophisticated models may calculate an index of disturbance based on the type of harvest and the estimated recovery rate, but without any specific consideration of the location of the activities relative to the stream network. A third, more rigorous level of analysis is to combine the land use history data with the site characteristics to drive spatially explicit models of runoff, erosion, and sediment transport. The rapid increase in GIS and computing power makes the latter an increasingly feasible alternative, but there has not been any rigorous evaluation of the net improvement in accuracy with increasingly sophisticated methods of CE assessments.

The lack of validation is a major weakness of most procedures to evaluate CEs. The inherent uncertainty and complexity of most natural systems means that predictions will generally be more valid on a relative than an absolute scale (e.g., Potyondy and others 1991). In other words, most assessment procedures are more useful to compare alternative scenarios than to accurately predict future condition (Renard and others 1997). To put the results in context, one must then compare the predicted change from a proposed action to the magnitude and variability of the natural processes affecting the resource of concern (step 7) (Megahan and others 1991). The next two sections of this paper discuss in greater detail the strengths and limitations of the different approaches for assessing CEs.

*10. Evaluate the validity and sensitivity of the predicted CEs.* As in any modeling or predictive exercise, several steps should be taken to help ensure the validity of the predicted CEs. The first and most obvious step is to test past predictions against measured data. Such a test can provide strong evidence for the validity of a model, but a close match does not necessarily validate the model (Oreskes and others 1994). Model predictions might match actual data under one set of conditions, but the same model may be inaccurate under different conditions. Alternatively, the model might produce the right answer for the wrong reasons (Oreskes and others 1994). Since even a single  $30 \times 30$  m pixel is a lumped representation of reality (Beven 1989), it is not possible to fully validate most CE models. Nevertheless, there is an urgent need to test existing approaches, and this has rarely been done (King 1989, Reid 1993, MacDonald and others 1997).

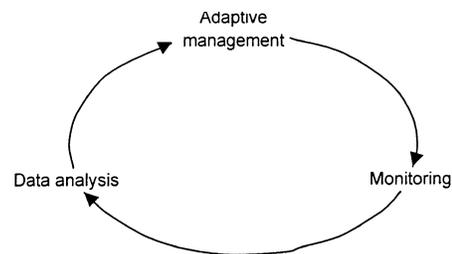
If measured data cannot be obtained, a sensitivity analysis can at least indicate the uncertainty in model predictions as a function of the uncertainty in the

different input data (Abbruzzese and Leibowitz 1997). A sensitivity analysis can also help identify the most important variables and processes, and thereby help focus the analysis on the key cause-and-effect mechanisms (McCuen 1973, Oreskes and others 1994). External and peer reviews are an important means to ensure that the assumptions, relationships, and results of the analysis are consistent with current knowledge.

#### Implementation and Management Phase

**11. Identify possibilities for modification, mitigation, planning, and restoration.** The last two steps bridge the gap from assessment to management and implementation. The basic idea is that a CE assessment is designed to evaluate the impact of a proposed action within a much broader context, and it follows that the additional data and information required for a CE assessment should be put to additional use. For example, a CE assessment might identify which past and present activities are having the greatest environmental impact, and the locations of greatest concern. Though this information is necessary to assess the incremental CE associated with a proposed action, it should be even more useful in determining where mitigation or other changes in management might be most effective in minimizing CEs. In many cases past or ongoing activities may be the primary cause of a CE, and further modification of the proposed action may not be nearly as cost-effective as mitigating an existing problem. Thus, the information generated by a CE assessment can help set priorities for restoration. The improved understanding of the causes of current conditions can also be used to adjust existing policies and improve planning, which should then help minimize future CEs. This integration of cumulative impact assessment with planning can be one of the best ways to achieve the overall goals of environmental protection and sustainable development, and thus serve as a primary justification for continuing efforts to analyze CEs (CEQ 1997).

**12. Identify key data gaps and monitoring needs.** This final step goes beyond the requirements of laws such as NEPA, but stopping the assessment process after step 11 is inefficient and does not optimize environmental protection and management. As suggested in the previous step, a CE assessment should lead to an improved understanding of past, present, and reasonably foreseeable future impacts. Thus the process of assessing CEs should help to identify critical data gaps and monitoring needs (CEQ 1997). If the same issues and resources are likely to be a primary concern in future analyses, it follows that the responsible management or regulatory agency should initiate or require a data collection and monitoring program. In most cases the agency initiat-



**Figure 6.** Cycle of adaptive management.

ing the assessment will be reluctant to incur the costs of monitoring, even though this type of feedback is essential for effective resource management (Figure 6) (MacDonald 1993).

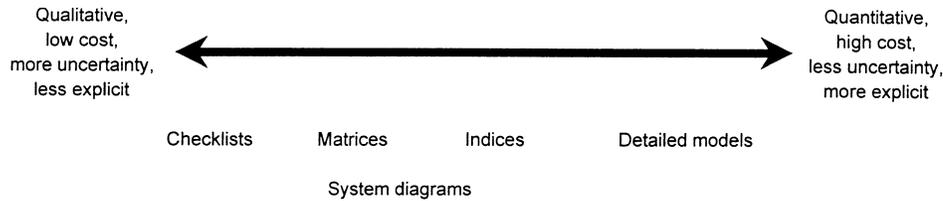
The rationale for not collecting data can also be more subtle, as less data generally will provide more leeway for decision making. Some agencies may also be reluctant to explicitly identify the limitations of existing data and analytical procedures, as this might undermine the validity of their management decisions and thus their own credibility. The problem is that one cannot claim to be managing a resource unless there is some regular feedback loop that links management decisions to resource condition (Figure 6).

#### Methods for Assessing Cumulative Effects

The conceptual process developed in the previous section provides a generic approach to assessing CEs, but one still has to decide which methodology should be used in a given situation. A 1992 review found that only 35 of 89 environmental assessments mentioned CEs, and nearly half of the 35 that mentioned CEs did not provide adequate support for their conclusions (McCold and Holman 1995).

Preston and Bedford (1988) noted that CEs in wetlands are intuitively appealing and tangible, but lack “operational formulation.” In view of our limited analytical ability, their suggested approach was to evaluate CEs with respect to individual wetland functions. Preston and Bedford (1988) also recognized that some analyses will necessarily remain qualitative and that a given analysis may be too limited to affect management decisions.

Euphrat and Warkentin (1994) also recognized the limitations of analyzing CEs, and they concluded that a combination of inventories and expert opinion is the most feasible and realistic approach. Reiter and Beschta (1995) conducted a comprehensive review of the CEs of forest practices on discharge, water quality, and stream channels. They concluded that: “cumulative hydrologic effects models may have limited utility, are extremely



**Figure 7.** Continuum of approaches for assessing cumulative effects.

difficult to construct, and, in their present state of development, may be of relatively limited utility to land managers” (p. 162). Similarly, Reid (1993) noted that: “When methods originate from management agencies, they tend to be simple, incomplete, theoretically unsound, unvalidated, implementable by field personnel, and heavily used. Methods developed by researchers are more likely to be complex, incomplete, theoretically sound, validated, require expert operators, and unused” (p. 35).

Thus, current methods for evaluating CEs range from qualitative, low-cost, and less explicit procedures to quantitative, high-cost, and more explicit models (Figure 7) (Smit and Spaling 1995). At least in theory, the qualitative procedures should have greater uncertainty and be less defensible.

The most common example of the qualitative approach is the checklist, and these can range from the few years or no questions required by the California Department of Forestry (Berg and others 1996) to the much more comprehensive checklist suggested by Canter and Kamath (1995). Checklist-type methods are most useful for: (1) identifying which issues, if any, should be investigated in more detail; (2) helping ensure that a range of issues are considered; and (3) providing a simple means to address the issue of CEs. Disadvantages include the qualitative nature of the assessments, the lack of repeatability, and the lack of documentation. Reliability can be increased by using a Delphi or group-consensus approach (e.g., Jourdannais and others 1990, Lull and others 1995). Several authors have suggested that the checklist approach can be extended to more complex systems and second-order effects by using different sets or layers of matrices (e.g., Sonntag and others 1987, Dixon and Montz 1995).

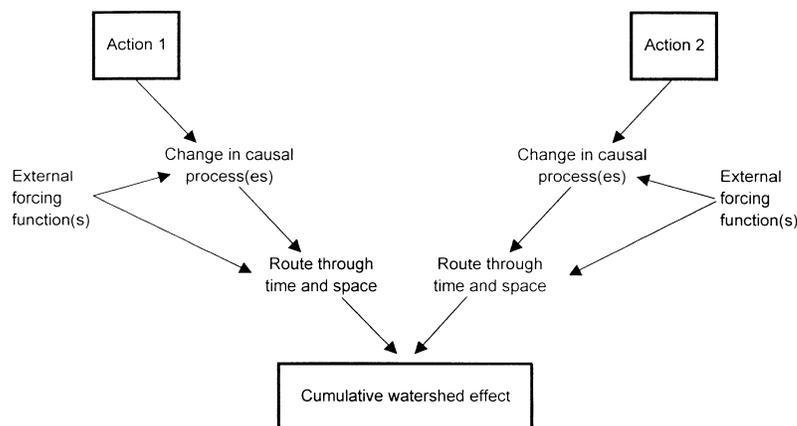
System diagrams and indices of disturbance represent a second level of analysis. System diagrams can help identify the cause-and-effect relationships that generate CEs (e.g., Figure 2), but these are often difficult to quantify and compare (CEQ 1997). GIS overlays or an index of management or disturbance can also provide a broad-scale assessment of potential cumulative impacts. In the state of Washington, for example, a series of simple GIS models were developed to rank polygons or

basins with respect to the likely changes in peak flows, slope stability, and the presence or absence of fisheries. These values were then aggregated for each basin and combined into an overall ranking. Basins with higher values had a higher priority for conducting a more comprehensive analysis (Green and others 1993).

A conceptually similar approach has been developed for evaluating cumulative impacts to wetlands (Leibowitz and others 1992, Abbruzzese and Leibowitz 1997). This process uses relatively simple landscape indicators, such as the amount of hydric soils and current land use, to estimate relative wetland loss in different basins. These landscape indices can then be combined with other information to estimate the loss in terms of a given wetland function, such as flood storage or wetland habitat. As a compromise between the need for rigorous results and the available resources for evaluating CEs, this approach is most useful when quantitative cause-and-effect data are not available, the cost of a wrong answer is low, the cost of improving existing information is high, and the objective is a broader-scale comparison (Abbruzzese and Leibowitz 1997).

Spatially explicit, physically based models represent the third and most detailed set of methods for assessing CEs. The minimum conceptual model for such efforts is shown in Figure 8, and this includes the causal actions, external forcing functions, estimated on-site changes, routing the estimated changes through time and space, and finally the impact on the resource of concern at selected locations.

The difficulty of defining all the landscape characteristics and functional relationships means that most current models are usually intermediate between the second and third levels of analysis. For example, many national forests in the United States calculate the total amount of disturbance using a watershed-scale index (e.g., equivalent clearcut area [ECA] or equivalent roaded area [ERA]) (e.g., Athman and McCammon 1989, Cobourn 1989, 1994, McGurk and Fong 1995). The idea is that all management activities can be converted to the amount of disturbance represented by a unit clearcut area or unit road area (USFS 1974). If the aggregated watershed-scale ECA or ERA exceeds a specified threshold of concern, downstream resources



**Figure 8.** Conceptual flow chart for predicting an off-site cumulative effect.

are presumed to be at risk. A basic limitation of these models is that they use a single index to account for changes in both flow and sediment loads (see step 1 in the previous section). Recovery over time is accounted for, but these indices don't consider the relative location of each activity or route materials through space or time. Most important, these models generally do not relate the predicted physical changes to an impact on coldwater fisheries, domestic water supply, or other designated beneficial use. As is the case for most of the CE models, only a few studies have attempted to validate this approach by comparing the calculated ECA or ERA values to the observed changes in discharge (Belt 1980, Wilson and others 1982, King 1989), sediment yield, or stream channel condition (see step 10) (MacDonald and others 1997, Schnackenberg and MacDonald 1998).

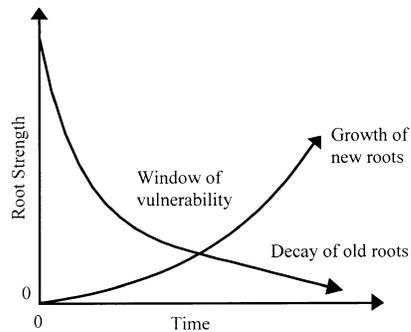
This lack of validation is generally the most serious limitation to the use of models in assessing CEs. In many cases the empirical relationships within the models have been derived from research (e.g., USFS 1981, Potyondy and others 1991), but these relationships have rarely been tested over the full range of site conditions and management activities where they are being applied. The large number of coefficients means that overparameterization is a serious concern, and prediction errors generally cannot be related back to a particular function or coefficient (Reid 1993, Oreskes and others 1994).

In summary, there is a wide range of approaches for evaluating CEs (Smit and Spaling 1995). In the United States the judicial system has generally focused on whether federal agencies have met the requirements of NEPA, not on the technical validity of the CE analyses (Thatcher 1990, Sample 1991). Thus, the courts require an appropriate scoping and a good faith effort. On the basis of these criteria, most current CE models can probably survive a court challenge, and agencies can choose from the full range of approaches in Figure 7.

### Inherent Limitations to Assessing Cumulative Effects

Previous sections of this paper have identified some of the key data gaps and issues that limit the accuracy and usefulness of a CE assessment. Recent guidance documents have largely ignored these issues (Irwin and Rodes 1992, CEQ 1997). A better understanding of the uncertainties and limitations in a CE analysis is critical to eliminate unrealistic expectations and determine what level of analysis is appropriate for a particular situation. Although not all-inclusive, the key limitations are: (1) the variability and uncertainty in quantifying management effects; (2) the inability to predict secondary or indirect effects; (3) the difficulty of defining recovery rates; (4) the difficulty of validation; and (5) the uncertainty of future events.

The first problem—quantifying management effects—stems from several sources. First, the effect of a given management activity is partly a function of the site conditions, but it is not possible to specify completely all the physical and biological conditions of a site (Coats and Miller 1981, Beven 1989). Second, it is not possible to precisely specify all the causal relationships and their interactions, even if all the site conditions are known (e.g., Renard and others 1997). Third, the effect of a given activity will be affected by unpredictable external forces, such as the frequency and magnitude of future storm events. An exceptionally large or intense storm event can cause more erosion than might normally occur in any number of average years (Haan and others 1994), and we can only predict the likelihood of such an event, not whether it will actually occur. Fourth, there always is some variability in how a given activity was carried out, and hence some uncertainty in predicting its contribution to a CE (Harr 1981, Reiter and Beschta 1995). Taken together, these issues limit our ability to extrapolate from different sites or forward in time.



**Figure 9.** The window of vulnerability for slope failure after timber harvest is largely a function of the rate of root regrowth relative to the decay of the old roots. The occurrence of a large storm in the middle of this window will have a more severe effect than an identical storm occurring either earlier or later.

Exactly the same constraints apply to the second limitation, which is the prediction and analysis of indirect and secondary effects. However, the task is made even more difficult because of the need to characterize each of the relationships that contribute to a particular indirect effect (e.g., Figure 2). The uncertainty associated with each step will accumulate through the analysis, and this leads to a greater uncertainty in predicting indirect effects than direct effects. In some cases, unforeseen secondary effects turn out to be more important than the predicted first-order changes. For example, the accumulation of selenium in the Kesterson Wildlife Refuge in central California was of far greater significance than the expected accumulation of nutrients, salts, or pesticides (Harris 1995).

The prediction of indirect CEs is further complicated by time lags between the causal action and the resource response. For example, forest harvest will reduce root strength, but the lag in root decay means that most landslides occur from two to fifteen years after harvest (Figure 9) (Gray and Megahan 1981, Sidle 1991). Similarly, it may take years or decades for the sediment from upstream areas to reach sensitive downstream areas (Gilbert 1917, James 1991, Benda and Dunne 1997b), or for a toxic chemical to accumulate. These types of lags can make it very difficult to quantitatively determine or predict cause and effect. The implication is that we need a relatively complete knowledge of the physical and biological relationships within the assessment area before we can reliably predict indirect CEs, and this will rarely be the case.

The third limitation to assessing CEs is the need to define recovery rates. In most cases the effect of a given management activity will diminish over time, and a more rapid recovery rate will reduce both the likelihood and the magnitude of a CE. The problem is that

there are relatively few data on recovery rates for different processes and resources, and one may need to define multiple recovery rates to accurately assess the impact of various management activities. Recovery rates will also vary with site characteristics and extrinsic factors, such as climate, and this uncertainty directly limits the accuracy of our predictions.

The fourth limitation in assessing CEs is the problem of validating a predictive model. As noted in the previous section, a perfect match between the predicted and observed output does not validate the structure, algorithms, or equations embedded within a model. The problem of validating model components, when combined with the limitations of characterizing sites, means that we probably will never be able to fully validate a CE model.

The final limitation is simply the uncertainty with respect to future events. A severe storm at the time of minimum root strength may cause a landslide at a given location, yet the same slope might not fail if the storm was a few years earlier or later (Figure 7). If we wish to account for the uncertainty of future events, we will have to incorporate a stochastic component into our models for predicting CEs (Ziemer and others 1991). This means that predictions will have to be expressed in terms of risk. To the extent that we can quantify the likelihood of different scenarios through Monte Carlo or other procedures, land managers and society will have to decide what level of risk is acceptable.

It is unlikely, however, that society can achieve a clear consensus on the acceptable level of risk (Rees 1995). Consistent rules for determining the acceptable level of risk will be difficult to establish, as the acceptable risk will vary among different segments of society according to the perceived value of the resource versus the cost of protecting that resource.

These limitations in prediction and validation do not necessarily mean that efforts to rigorously analyze CEs should be abandoned. The point is that there will always be some uncertainty in the assessment and prediction of CEs, and this must be explicitly considered when determining what level of analysis is appropriate. These limitations must also be recognized when the results of the analysis are presented to decision makers or the public. As noted in CEQ's recent guidance document (CEQ 1997), a CE analysis should lead to a better decision, but an analysis can never ensure a perfect decision.

#### Alternative Approaches to Cumulative Effects Assessment

In many cases there is a reluctance to initiate a CE analysis, and this is commonly due to the cost of the

analysis as well as the problems of uncertainty and complexity. While these are valid concerns, they do not obviate the need for a CE assessment (CEQ 1997). There are also at least two complementary approaches that may assist in managing CEs, and these are: (1) adaptive management, and (2) minimizing the effects of individual actions. The use and limitations of these two alternative approaches are briefly discussed below.

#### Adaptive Management

Adaptive management is an iterative process where the current condition is used to determine subsequent actions (Walters 1986). To be effective, adaptive management requires regular monitoring and the rapid feedback of this information into management decisions (Figure 9).

A major limitation of adaptive management is that a detectable change in the resource(s) of concern has to occur before there can be a change in management. If the resource of concern exhibits large temporal variability, as shown in Figure 4, a sustained and relatively large amount of degradation must occur before management adjusts. If the resource is also slow to recover, the cumulative degradation could be substantial. Similarly, if there is a lag between management activities and resource response (Benda and Dunne 1997b), the resource may continue to degrade even after a change in management. In each of these situations adaptive management may not provide an adequate level of resource protection.

These problems can be partially alleviated by using a weaker level of significance, as this will usually increase the statistical power (i.e., the ability to detect a difference when in fact there is a difference). The trade-off for this more sensitive detection limit is a higher likelihood of Type I error (i.e., that an observed difference is in fact due to chance) (Gilbert 1987).

In view of these concerns, the adaptive management approach will be most useful when the resource of concern is: (1) relatively responsive to management activities over short time scales; (2) exhibits little temporal variability; and (3) can be accurately monitored. The trade-off between attempting to predict CEs and using adaptive management has to be evaluated on a case-by-case basis.

#### Minimize On-Site Changes

The second alternative approach is to minimize on-site changes, with the implicit assumption that this will largely eliminate off-site CEs. Although this logic may not apply in all cases, the effects of many activities do diminish with increasing temporal and spatial scale (Harr 1989, Bunte and MacDonald 1999). The largest

effect of a given action is usually local and immediate, and this means that the effect of a given action can be most easily detected at smaller spatial and temporal scales. It follows that if one can minimize adverse effects at the local scale, there should be a reduced potential for CEs at a larger scale. This logic is largely why Oregon has declined to require a CE assessment for forest management activities. In contrast, the neighboring states of California and Washington have adopted a simple checklist and a comprehensive watershed analysis procedure, respectively (Berg and others 1996, Washington Forest Practices Board 1995).

Because minimizing on-site effects will reduce, but not eliminate the possibility of off-site effects, this second approach also needs to be combined with a focused monitoring program. For example, best management practices (BMPs) have generally been shown to be relatively effective in protecting water quality and aquatic habitat (Megahan and others 1991, Binkley and MacDonald 1993, MacDonald and others 1997, Arthur and others 1998). Typically the most severe water quality problems occur when BMPs are simply not applied or maintained (e.g., CDF 1999). Thus, a two-pronged program that monitors the implementation of BMPs and selected key resources may be an effective means to minimize adverse CEs. The longer-term monitoring of key resources should also improve our understanding of the sensitivity and variability of the resource(s) being monitored (Contant and Wiggins 1991).

The combination of BMP implementation and off-site resource monitoring does not mean that CEs won't occur or that the possibility of CEs can be dismissed in an environmental assessment. In some cases off-site resources can be more susceptible than the resources immediately adjacent to the proposed project (e.g., Montgomery and Buffington 1993). Alternatively, there can be a nonlinear biological response to a given environmental change. Nevertheless, the resources for environmental analysis and protection are limited. Thus the intensive CE analyses recommended by some authors (e.g., McCold and Saulsbury 1996) may not be justified given the inherent limitations in assessing and predicting CEs.

On the other hand, there are some problems, such as global warming, that are exclusively due to CEs. The challenge is to identify these broader problems before they become intractable or demand emergency action. This is particularly important for situations when the CE will persist well beyond the causal actions.

#### Need for a Tiered Approach

The previous sections have emphasized that both the type and magnitude of a CE will vary with the temporal

and spatial scale of the assessment. This suggests that a tiering or hierarchy of CE assessments is needed to address fully the potential range of CEs. From society's perspective it is not efficient for all proposed actions to conduct an assessment of CEs at all scales, as the larger-scale problems will be common to all projects. Repetitious or excessively large-scale CE assessments will reduce the resources available for other assessments or environmental protection activities. The trade-off between assessments and other environmental protection activities has not been explicitly addressed in the literature, and it would be useful to evaluate the relative costs and benefits of conducting CE assessments at different spatial and temporal scales.

In an ideal world CE assessments would be tiered according to the responsibilities at each organizational level. Thus a local management agency would be responsible for assessing the CEs of a proposed activity at the project or local scale, while a regional or state agency would be responsible for putting that analysis in a larger context and assessing regional-scale effects. Similarly, the national government would provide more programmatic or national-scale assessments, and international agencies such as the Mekong River Commission or the United Nations should take the lead on international or global-scale issues. For this tiered approach to be effective, there must be both a clear delineation of responsibilities and good communication between each level.

The explicit adoption of a tiered approach to CEs should largely resolve the problem of defining the scope and scale of a CE analysis. Thus an assessment of CEs in a large river basin would be separated from—but still provide the context for—an assessment at the sub-basin or project scale. Assessments at the sub-basin or project scale would provide the necessary input for larger-scale assessments. Effectiveness and efficiency will be maximized when these different levels of assessment are nested and cross-referenced, and one does not attempt to address all of the larger-scale issues in each project-level assessment.

## Conclusions

Cumulative effects can be important and have to be considered in environmental planning and management. In most cases our ability to assess CEs is limited by the lack of data on past management activities, our inability to route effects through time and space, and the uncertainty of future events. Secondary and interacting effects add another layer of complexity and uncertainty to assessing current and potential CEs.

Given the potential breadth and complexity of CEs, a CE assessment should focus on the issues of greatest concern. Following an explicit process will help ensure that a CE analysis is well focussed and conducted at the appropriate scales. The methodology for assessing each potential CE must be selected in accordance with our level of understanding and knowledge, the level of risk to the resource(s) of concern, and the perceived value of those resource(s). The use of geographic information systems should improve the quality of our assessments, but there is a limit to which such models can be parameterized and validated.

Our limited capability to accurately assess CEs suggests that proportionally more effort should be devoted to minimizing on-site effects. Focused monitoring of key resources can provide an early warning of CEs, and in some cases adaptive management may be the best approach. A nested series of separate but cross-referenced project- and policy-scale assessments are the most effective means to address CEs over the full range of spatial and temporal scales.

Uncertainty is a hallmark of all CE assessments, and this must be recognized by managers, regulators, and the public. The problems of scope, scale, and predictability are based in science, but their resolution is a question of values and will therefore be a continuing source of controversy.

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