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PROFILE

Wetland Mitigation Banking: A Framework for Crediting and Debiting

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ABSTRACT / Wetland mitigation banking as a resource management tool has gained popular support for its potential to provide an ecologically effective and economically efficient means to fulfill compensatory mitigation requirements for impacts to aquatic resources. Although this management tool has been actively applied within the past 10 years (C. Short, 1988, Mitigation banking, in *Biological Report* 88(41): 1–103), assessment of credits and determination of a compensation ratio that reflects existing and/or potential functional condition in a mitigation bank has been a formidable task. This study presents a framework for a systematic approach for determination of credits and debits and subsequently the compensation ratio. A model for riparian systems is developed based on this framework that evaluates credits and debits for spatial and structural diversity, contiguity of habitats, invasive vegetation, hydrology, topographic complexity, characteristics of flood-prone areas, and biogeochemical processes. The goal of developing this crediting and debiting framework is to provide an alternative to the current methods of determining credits and debits in a mitigation bank and assigning mitigation ratios, such as best professional judgement or use of preset ratios. The purpose of this crediting and debiting framework is to develop a method that (1) can be tailored to evaluate ecological condition based on the target resources of a specific mitigation bank, (2) is flexible enough to be used for evaluation of existing or potential ecologic condition at a mitigation bank, (3) is a structured and systematic way to apply data and professional judgment to the decision-making process, (4) has an ecologically defensible basis, (5) has ease of use such that the level of expertise and time required to employ the method is not a deterrent to its application, and (6) provides a semiquantitative measure of the condition of aquatic resources that can be translated to a mitigation ratio.

Urbanization, land development, agriculture, resource extraction, and infrastructure development are often accompanied by impacts to aquatic resources through either direct fill or secondary and cumulative impacts. Discharge of dredged or fill material affecting aquatic resources, such as lakes, rivers, streams, oceans, or wetlands usually falls under the jurisdiction of Section 404 of the Clean Water Act and is regulated by the U.S. Army Corps of Engineers (Corps) regulatory program. Corps' regulations, guidelines, and Memorandum of Agreement (MOA) allow for compensatory mitigation to be performed to offset the unavoidable impacts associated with permitted activities. The 1990 MOA between the Corps and the U.S. Environmental Protection Agency (US EPA) regarding mitigation expresses a clear preference for on-site, in-kind replacement of wetland functions and values. Consequently, compensatory mitigation is often done at or near the project site and consists of either creation of new habitat, restoration or enhancement of degraded habitat, or, in some cases, preservation of intact habitat.

Within the last 10 years wetland mitigation banking has gained popular support as a resource management tool with the potential to provide an ecologically effective and economically efficient alternative to traditional site specific mitigation as a means to fulfill compensatory mitigation requirements (IWR 1992). Mitigation banking is founded on the premise that large, contiguous wetland parcels can have a greater chance of being biologically and hydrologically viable and can accrue more ecologic functions than small, isolated compensatory mitigation sites (Short 1988, Environmental Law Institute 1993). Wetland mitigation banks strive to establish large, contiguous wetland areas that can be used to mitigate for a number of independent impacts. This allows eligible permittees to purchase compensa-

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tory mitigation functions or credits from another entity that has already produced and banked them, thereby eliminating the need to produce compensatory mitigation areas on site. Mitigation banking can have the added advantage of establishing successful wetland functions in advance of the actual loss of functions associated with a permitted activity (IWR 1992).

Despite the recent rise in popularity and regulatory support for mitigation banks, assessment of credits in a mitigation bank and determination of compensation ratios that reflect existing and/or potential ecologic conditions in a mitigation bank continues to be one of the most problematic yet most essential aspects of mitigation banking (Environmental Law Institute 1993, IWR 1994). The November 1995 Joint Federal Guidance for the Establishment and Use of Mitigation Banks requires that mitigation banks include systems for determining the number of credits needed to compensate the impacts of a given project (i.e., defining the currency of the bank and setting mitigation ratios) (Federal Register 1995). The crediting and debiting methodology is a two-step process where the existing or potential condition of a mitigation bank (credits) and at the impact site (debits) are assessed and translated into a currency such as acreage or habitat units (IWR 1992, Environmental Law Institute 1993). The second step consists of a determination of the number of credits needed to compensate for losses from a project (debits) or the compensation ratio.

Numerous assessment methods have been proposed for the determination of credits and debits in wetland mitigation banks. The majority of wetland mitigation banks to date, however, use best professional judgment or simple indices, such as acreage, to determine the compensation ratio (Tabatabai 1994). The main advantage of simple indices is their lack of complexity and ease of use. These indices can be calculated quickly by project proponents and regulatory staff, often with little or no field work and little expenditure of resources. The disadvantage of simple indices is they ignore the complexities of wetland ecosystems and may not be representative of aquatic resource functions impacted and the existing or potential functions that exist in a mitigation bank (IWR 1994). Using best professional judgment to determine the acreage to compensate for loss of aquatic resources not only is problematic in terms of scientific indefensibility but also poses problems of inconsistency, uncertainty, and irreproducibility. Great caution must be exercised when using best professional judgment or simple indices to protect against wetland losses.

As an alternative to simple indices or best professional judgment, credits and debits can be computed using functional evaluation methods. Numerous techniques developed over the last 20 years attempt to use field indicators as measures of habitat function. These techniques include:

- Biotic indices, such as species density and the Shannon-Weaver index of species diversity. These biotic indices can be multiplied by acreage to yield diversity units.
- Assessments based on species composition or habitat suitability for specific indicator species, such as the Habitat Evaluation Procedure (US FWS 1980), Habitat Evaluation System (Pearsall and others 1986), Biological Evaluation Standardized Technique (Barnett and others 1991), and Index of Biotic Integrity (Karr 1991).
- Surveys of habitat characteristics, such as the Wetland Evaluation Technique (Adamus 1983), Wetland Replacement Evaluation Procedure (Bartoldus and others 1992), and Wetland Evaluation Methodology (WEM) (US ACOE 1988).
- Landscape level assessments using Geographic Information Systems (GIS) and other coarse resolution measures of function in a regional perspective, such as US EPA's Synoptic Approach to Impact Assessment (US EPA 1992).

The most recent and one of the most promising functional assessment techniques is the Hydrogeomorphic Method (HGM) (Smith and others 1995), developed by the U.S. Army Corps of Engineers Waterways Experiment Station (WES). This method uses variables measured in the field to compute functional indices for biotic, hydrologic, and biogeochemical functions. These indices are scaled against locally representative reference sites to account for regional variations in wetland ecosystems. However, development of regional models and reference standards requires considerable time, resources, and technical expertise; to date, few regional reference sets have been developed.

Each evaluation method has strengths and weaknesses, which have been previously discussed by several authors (Margules and Usher 1981, Westman 1985, Lonard and Clairain 1985, Jain and others 1993, Stein 1995). However, because mitigation banks are typically used to compensate for impacts resulting from multiple small projects, methods such as those listed above become cumbersome in terms of personnel resources and inefficient in terms of assessing functions impacted at each site eligible to use the mitigation bank. In addition, regulatory agencies may not have the expertise or resources to apply the functional assessment methods properly; therefore, the designated method may not be used accurately. In his review of functional assessment methods, Smith (1993) concluded that "no single method reviewed meets the requirements of a quick screening technique to determine a broad spectrum of wetland values and functions." It is unlikely that any single method could fully satisfy both the quick screening and the comprehensiveness criteria. However, it is our goal to develop a crediting and debiting framework for wetland mitigation banks that will address some of the limitations posed by other crediting strategies while providing a balance between ease of use and defensible measure of ecologic condition. To achieve this goal, a crediting and debiting framework should meet the following criteria: (1) can be tailored to evaluate ecologic condition based on the target resources of a specific mitigation bank, (2) is flexible enough to be used for evaluation of existing or potential ecologic condition at a mitigation bank, (3) is a structured and systematic way to apply data and professional judgment to the decision-making process, (4) has an ecologically defensible basis, (5) has ease of use such that the level of expertise and time required to employ the method are not deterrents to its application, and (6) provides a semiquantitative measure of the condition of aquatic resources that can be translated to a mitigation ratio.

In this paper we present a crediting and debiting framework for wetland mitigation banks that meets the above criteria. The principles of the framework are applied to develop a model for southern California riparian systems. Use of the riparian model is illustrated for Santa Ana River Mitigation Bank (SARMB), located in Riverside County, CA.

Crediting and Debiting Framework

The crediting and debiting framework is based on assessing changes in structural characteristics at the impact site and the mitigation site. Change is assessed by evaluating conditions before and after alterations to the site. Structural characteristics are used as indicators of ecologic condition of the specific class of aquatic resource.

Credits are determined based on the difference between structural characteristics of the post-restoration condition and pre-restoration (baseline) condition at the bank site. Similarly, debits are assessed by determining the difference between pre-project and post-project structural characteristics at the impact site. Each structural characteristic, or criterion, is evaluated on a linear interval scale and assigned a rating that reflects the relative value of that criterion at a given site. Credits are the sum of net gain of values for all criteria at the bank and debits are the sum of net loss of values for all criteria at the impact site. The mitigation ratio is the ratio of debits over the credits. When the mitigation credits must be calculated (or estimated) before the bank is functionally mature, the mitigation ratio can be based on the maximum expected gain at the bank (i.e., the enhancement potential) (Table 1). We will demonstrate the framework using the structural characteristics developed below for the southern California riparian model.

Table 1. Crediting and debiting framework. Application of the crediting and debiting framework involves three main steps: evaluation of credits, evaluation of debits, and determination of the mitigation ratio

Step 1. Evaluation of credits
Credits = Post-project Rating (or Enhancement Potential Rating) − Pre-project Value (Existing Value)
Step 2. Evaluation of debits
Debits = Pre-project Rating of the Impact Site − Post-project Rating of the Impact Site
Step 3. Determination of the mitigation ratio
Mitigation Ratio = Debits/Credits (or Projected Available Credits)

The framework is a systematic approach designed to balance directly measuring hydrologic and physical characteristics of aquatic resources against ease of use. The intent is not to provide an absolute tool for evaluating functional condition, rather to provide an ecologically based framework to organize best professional judgment and apply it in a systematic manner. The framework is intended to apply to the mitigation bank and the typically small impact sites that normally use a mitigation bank. Assumptions associated with this type of crediting and debiting methodology include equal weights assigned to each criteria and a linear increase in values associated with each interval.

This crediting and debiting framework may be applied to various types of ecosystems. Evaluation criteria will vary based on the type of the system being evaluated and should account for the hydrologic, biologic, biogeochemical, and landscape characteristics of the target aquatic system. Below we provide a sample crediting and debiting system developed for southern California riparian wetlands.

Southern California Riparian Model

Riparian systems in the western United States are typically narrow, linear strips of vegetation along rivers, streams, or lakes and are dependent on perennial or ephemeral surface or subsurface water (Knopf and others 1988, US DOI 1994). Dry climates and porous soils found in arid regions cause streamside soil moisture to decrease more rapidly with distance from the streambank than in humid regions, resulting in narrower riparian zones (Reichenbacher 1984). However, flooding duration, intensity, and timing are the ultimate determinants of riparian succession. Flooding waters bring nutrient-rich sediments to the flood plain, export organic and inorganic material from the flood plain, scour mature woodlands, and help spread propagules laterally into the flood plain (Strahan 1984, Warner and Hendrix 1985, Dickert and Tuttle 1985, Gosselink and others 1990a). Riparian systems form dynamic mosaics of active channels, terraces, flood plains, and alluvial fans. The composition and distribution of these systems is a product of fluvial processes, which erode material from some areas and deposit it in others during flood events, facilitating channel migration (Gregory and others 1991). This combination of degradation and aggradation results in the formation of bars and terraces with different drainage patterns and elevations. These elevational differences result in the extensive vegetative diversity of riparian systems (Strahan 1984). The viability of terraces and flood plains depends on their proximity to groundwater levels, surface emergent aquifers, or hyporheic zones (porous substrate allowing water to flow immediately beneath the surface of streambeds) (Stanford and Ward 1993). Therefore, in the arid west, the width and distribution of the riparian zone is ultimately determined by the vertical gradient between the benchland and the streambed (Szaro 1990).

Although their areal extent is proportionately less than in other parts of the country, western riparian systems have a proportionately greater significance for some functions because of the arid climates in which they occur (US DOI 1994). In the arid southwestern United States, riparian areas serve as linear or single-point habitat islands on which a multitude of native wildlife species are totally dependent for survival (Warner and Hendrix 1985). The US DOI (1994) estimated that although less than 1% of the western portion of the United States is covered by riparian vegetation, between 51% and 82% of all species in the southwestern United States depend on riparian areas for survival.

Evaluation Criteria for Southern California Riparian Systems

Based on the crediting and debiting framework, we developed a model for southern California's riparian systems using the following evaluation criteria: (1) spatial diversity and coverage of habitats; (2) structural diversity of habitats; (3) contiguity of habitats; (4) percent of invasive vegetation; (5) hydrology; (6) topographic complexity; (7) characteristics of flood-prone area; and (8) biogeochemical processing. These criteria reflect the fact that assessment of riverine systems requires examination of the entire riparian zone and consideration of the interaction between geology, hydrology, and organic and inorganic inputs to the system. In recognition of the fact that functional capacity differs between low-order and high-order streams, for some criteria we have provided different indicators for first- and second-order streams versus higher order streams. Because first- and second-order streams do not typically support the same complexity of habitat as higher order systems, they will typically score lower on the habitat criteria. For the purposes of this method, trees are defined as perennial woody dicots greater than 7.5 cm diameter at breast height (DBH). Saplings are defined as perennial woody dicots less than 7.5 cm DBH.

The first two evaluation criteria address structure, composition, and diversity of the site. Scoring of the first criterion, coverage and spatial diversity of habitats, should consider the site as a whole and evaluates both diversity of habitat types (i.e., interspersion) and species diversity within each patch. Scoring of the structural diversity of habitats criterion should focus only on the structure within the riparian patches on the site (as opposed to the site as a whole). Although this criterion partially captures species diversity, it is to a lesser extent than the spatial diversity criterion. The first two criteria should be scored based on the vegetative composition of the site regardless of whether the vegetation is native or non-native. Effects of non-native species on habitat integrity are addressed by a separate criterion. Evaluation of structure regardless of the geographic origin of the species accounts for the fact that increased biomass (regardless of species type) contributes to a site's ability to retain water and retain nutrients and compounds, thereby increasing some hydrologic and biogeochemical functions. This attribute is also directly accounted for by the biogeochemical processes criterion, which is scored based on abundance of biomass, regardless of whether or not it is native.

Coverage and Spatial Diversity of Habitats

Riparian habitats are typically patchy with an interspersion of different habitat types (Faber and Holland 1988). This interspersion allows the activities of animals in dry sites to be more closely coupled to those in wet sites. A mosaic of habitat types provides a richer, more continuous food source for mobile fauna than that of a homogeneous habitat. For example, Doyle (1990) found a strong correlation between the extent of herbaceous and deciduous shrub cover in riparian habitats and the abundance and diversity of small mammals. Habitat mosaics also allow animals to fulfill several life functions at a single site (e.g., foraging, escape, reproduction) (Warner and Hendrix 1985, Gosselink and others 1990b). Alpha diversity (diversity within a site) has been correlated to the ability of a patch to support a complex food chain and allow interior species with specific habitat requirements to thrive in the face of competition from generalists (Klopatek 1984, Harris 1988). Assessment of changes to the spatial diversity of a project site provides information about impacts to a site's capability to support a variety of different faunal species.

The ratings for the coverage and spatial diversity criterion are assigned based on the following scale:

0 = Site permanently converted to land use not able to support native riparian vegetation, such as housing, agriculture, or concrete channel.
0.2 = No existing riparian vegetation (e.g., covered with annual grasses and scrub, bare ground). However, site has the potential for revegetation without extensive structural modification.
0.4 = Patches of monotypic riparian vegetation covering up to 50% of the site, interspersed among herbaceous species or bare ground.
0.6 = Patches of diverse riparian vegetation (e.g., at least three different genera of riparian vegetation present) covering up to 30% of the site, interspersed among grasses, invasive plants, or bare ground; and/or greater than 50% of the site covered with monotypic patch(es) of riparian vegetation, interspersed among herbaceous species or bare ground.
0.8 = Diverse riparian vegetation covering between 30% and 75% of the site, e.g., strips or islands of riparian habitat interspersed in open space.
1.0 = Diverse riparian vegetation (e.g., at least three different genera of riparian vegetation present) covering between 75% and 100% of the site, interspersed in open space or herbaceous plant communities.

Structural Diversity of Habitats

The stratification of vegetation into layers, including shrubs, understory, and canopy, provides a variety of different habitats. This allows a diversity of organisms representing different trophic levels to coexist in a single site, thereby supporting a more complex and resilient food chain (Warner and Hendrix 1985). For example, diverse ground cover provides habitat for many insects which form the base of the food chain and provide important ecosystem functions, such as pollination. This allows higher-trophic-level organisms to utilize understory and canopy habitat that may be present (Erman 1984). Structural diversity within a site has been correlated with faunal diversity, especially for birds (Gosselink and others 1990b). The presence of a floristic structure consisting of three strata indicates that appropriate soil, moisture, and topographic conditions exist to support a "healthy" riparian system (Warner 1984). Structural diversity of the vegetated portions of the project site is used as a surrogate for general habitat suitability for an assortment of common species.

Because riparian habitats are typically patchy (Faber and Holland 1988), the ratings for this criterion are based on only the vegetated portions of each site:

0 = Site permanently converted to land use that will not be able to support native riparian vegetation, such as housing, agriculture, or concrete channel.
0.2 = No existing riparian vegetation (e.g., covered with upland grasses and scrub, bare ground). However, site has the potential for revegetation without extensive structural modification.
0.4 = Vegetated areas of the site contain sparse, scattered, patchy, or remnant riparian vegetation that is immature and/or lacks structural (vertical) diversity.
0.6 = The patches of riparian vegetation on the site contain riparian trees and/or saplings (i.e., perennial dicots), but contain no or poorly developed shrub understory.
0.8 = The patches of riparian vegetation on the site contain riparian trees and saplings, plus a well-developed native shrub understory.
1.0 = The patches of riparian vegetation on the site are structurally diverse. They contain riparian trees, saplings, and seedlings, as well as developed native shrub understory and herbaceous layer.

Contiguity of Habitats

Fragmentation and habitat loss are dominant causes of the decrease in biotic diversity (Harris 1988). The ecological value of disjunct habitat patches can be enhanced if they are connected by strips of protected habitat; these corridors facilitate movement between patches (Diamond 1975, Noss 1987). For animals with a home range exceeding the size of an individual habitat patch, corridors provide a means of moving from one habitat patch to another. Without a system of travel corridors allowing these animals passage from one refuge to another, they will probably not occur in future landscapes (Harris 1988). Even if partially disturbed, riparian corridors are vital to the successful migration of neotropical birds and other organisms (Croonquist and Brooks 1991). In addition, habitat connectivity helps small populations (such as endangered species) maintain demographic and genetic integrity in the face of the isolation caused by habitat fragmentation (Frankel and Soule 1981). Changes to linear contiguity affect not only corridors but also contribute to overall habitat fragmentation and decreases in patch size. This can be detrimental for resident as well as migrant species (Harris 1988).

The ecological value of riparian habitats also depends on their integration as units within the surrounding landscape (Gosselink and others 1990b). Many organisms have complex life histories in which different stages require distinct habitats within a regional landscape in order to meet their life requirements (Harris 1988). Therefore, continuity between riparian and upland habitats increases utilization by fauna and provides safe passage between riparian oasis and adjacent uplands (Gosselink and others 1990c). Furthermore, the greater the edge area between riparian habitat and developed areas, the greater the potential negative impact from adjacent upland land use (Warner and Hendrix 1985). Additionally, many riparian plants require adjacent uplands as a flood plain for establishment of their propagules during flooding events (Scott and others 1993). These flood plains also provide refuge for fauna during flooding (Gosselink and others 1990c).

The continuity criterion includes two components. Linear continuity refers to riparian habitat upstream and/or downstream of the site. Lateral continuity addresses the quality of upland habitat and reflects the connection of the site to the surrounding nonriparian habitat. The ratings for the contiguity criterion are assigned based on the following scale:

First and second order streams.

0 = No linear contiguity or transitional upland habitat; completely surrounded by or isolated within an urban setting or converted to an urban/suburban land use.
0.2 = No linear contiguity upstream or downstream, but isolated within upland open space habitat.
0.4 = Contiguous with comparable habitat on one end of the site (upstream or downstream), but surrounded with urban/suburban or other nonopen space lands adjacent (lateral to) to the site on at least one side.
0.6 = Contiguous with comparable habitat on one end of the site (upstream or downstream) and surrounded by transitional upland habitat which is at least 35 m wide.
0.8 = Contiguous with comparable habitat on both ends of the site (upstream and downstream), but surrounded with urban/suburban or other nonopen space lands adjacent (lateral to) to the site on at least one side.
1.0 = Contiguous with comparable habitat on both ends of the site (upstream and downstream) and surrounded by transitional upland habitat on both sides which is at least 35 m wide.

Higher order streams.

0 = No linear contiguity or transitional upland habitat; completely surrounded by or isolated within an urban setting or converted to an urban/suburban land use.
0.2 = No linear contiguity upstream or downstream, but isolated within upland open space habitat.
0.4 = Contiguous with comparable habitat on one end of the site (upstream or downstream), but surrounded with urban/suburban or other nonopen space lands adjacent (lateral to) to the site on at least one side.
0.6 = Contiguous with comparable habitat on one end of the site (upstream or downstream) and surrounded by transitional upland habitat which is at least twice the width of the riparian zone.
0.8 = Contiguous with comparable habitat on both ends of the site (upstream and downstream), but surrounded with urban/suburban or other nonopen space lands adjacent (lateral to) to the site on at least one side.
1.0 = Contiguous with comparable habitat on both ends of the site (upstream and downstream) and surrounded by transitional upland habitat on both sides which is at least twice the width of the riparian zone.

Percent of Invasive Vegetation

Invasive species often thrive in mesic environments and readily establish following disruption of riparian systems. Many invasive species have few, if any, native pests or diseases and thus grow rapidly. Once established, their proliferation excludes reestablishment of native species following subsequent disturbances, such as floods or fires (Warner and Hendrix 1985). Some invasive vegetation, such as *Arundo donax* and *Tamarix* spp. provide little to no habitat value for wildlife species (Hanes 1981, Bell 1993). Moreover, *A. donax* and *Tamarix* spp. pose a greater problem for flood control than native vegetation due to the morphological characteristics of the long stalks (*Arundo*) and deep taproots (*Tamarix*), which obstruct flood control channels more than native riparian vegetation. Overall, the replacement of native riparian habitat with *A. donax, Tamarix* spp., and other invasive vegetation displaces native fauna, reduces flood conveyance, increases evapotranspirative losses, increases water temperature, and creates fire hazards (Bell 1993). For example, eradication of *A. donax* from the Santa Ana River could reduce annual evapotranspirative water losses by an estimated 4.6×10^7 m^3, resulting in an estimated savings of $12 million annually (Iverson 1993). However, it has been suggested that the increased biomass associated with invasive weed infestation may increase retention times and, therefore, the ability of a site to sequester elements or compounds. The contribution of increased biomass to biogeochemical processes is accounted for in the structural diversity and spatial diversity criteria.

The ratings for the percent of invasive vegetation criterion are assigned based on the following scale:

0 = Site is covered by pure stands of invasive vegetation or lacks any riparian vegetation.
0.2 = Site is covered by 70–99% invasive vegetation.
0.4 = Site is covered by 40–69% invasive vegetation.
0.6 = Site is covered by 10–39% invasive vegetation.
0.8 = Site is covered by 5–9% invasive vegetation.
1.0 = Site is covered by less than 5% invasive vegetation.

Hydrology

Hydrology is the most important factor determining the establishment and maintenance of specific wetland functions (Mitsch and Gosselink 1993). Reviews of past mitigation sites reveal that improper hydrology is the most significant problem with many unsuccessful sites (Mitsch and Wilson 1996, Sudol 1996). Riparian systems rely on appropriate and natural hydrology for long-term self-sustainability and viability. This criterion addresses the source of water supporting the wetlands and the exposure of the site to riparian processes, such as scour and overbank flow. The geomorphic structure of the site is addressed by the topographic complexity and flood-prone area criteria. The ratings for the hydrology criterion are assigned based on the following scale:

0 = No regular supply of water to the site. Site not associated with any water source, surface drainage, impoundment, or groundwater discharge.
0.2 = Water supply to the site is solely from artificial irrigation (e.g., sprinklers, drip irrigation). No natural surface drainage, natural impoundment, groundwater discharge, or other natural hydrologic regime.
0.5 = Site is sustained by natural source of water but is not associated with a stream, river, or other concentrated flow conduit. For example, the site is sustained by groundwater or urban runoff. There is no evidence of riparian processes, such as overbank flow or scour or deposition.
0.7 = Site is within or adjacent to an impoundment on a natural water course which is subject to fluctuations in flow or hydroperiod.
1.0 = Site is within or adjacent to a stream, river, or other concentrated flow conduit that provides the primary source of water to the site. The site contains evidence of riparian processes, such as overbank flow or scour or deposition, or is within the flood-prone area (the channel plus the area defined by a horizontal projection at a height of twice the bankfull thalweg; Rosgen 1994).

Micro- and Macrotopographic Complexity

In riparian systems, fluvial processes that erode material from some areas and deposit it in others during flood events form dynamic mosaics of active channels, terraces, flood plains, and alluvial fans with different drainage patterns and elevations (Gregory and others 1991). These elevational differences result in the extensive vegetative diversity of riparian systems (Strahan 1984). Riparian flora depends on connectivity between active channels and flood plains for seed dispersal and germination and on base flow resulting from percolation into flood plain soils for survival during the dry season (Warner and Hendrix 1985, Harris and Gosselink 1990, Faber 1993). The ratings for the topographic complexity criterion are assigned based on the following scale:

First- and second-order streams.

0 = All flows, including flood flows, are contained in a concrete-lined channel, culvert, etc.
0.2 = Flood-prone area is characterized by a homogenous, flat earthen surface with little to no micro- and macrotopographic features.
0.6 = Flood-prone area contains micro- and/or macrotopographic features such as pits, ponds, hummocks, bars, rills, large boulders, but is predominantly homogeneous or flat surface.
1.0 = Flood-prone area is characterized by micro- and

macrotopographic complexity, such as pits, ponds, hummocks, rills, large boulders, etc.

Higher order streams.

0 = All flows, including flood flows, are contained in a concrete-lined channel, culvert, etc.
0.2 = Flood-prone area is characterized by a homogenous, flat earthen surface with little to no micro- and macrotopographic features.
0.5 = Flood-prone area contains micro- and / or macrotopographic features such as meanders, bars, braiding, secondary channels, backwaters, terraces, pits, ponds, hummocks, but is predominantly homogeneous or flat surface.
0.8 = Flood plain is predominantly heterogeneous, and is characterized by microtopographic features such as pits, ponds, hummocks, bars. However, there are no macrotopographic features, such as braiding, secondary channels, backwaters.
1.0 = Flood-prone area is characterized by micro- and macrotopographic complexity, such as meanders, bars, braiding, secondary channels, backwaters, terraces, pits, ponds, hummocks, etc.

Characteristics of Flood-Prone Area

Riparian systems are defined by the geomorphic structure and fluvial characteristics of the valleys in which they exist (Gregory and others 1991). Development of river flood plains and restriction of channel migrations alters the hydrologic regime of riparian systems and severs the critical link between the aquatic habitat and adjacent upland habitat. Alteration of the flood plain reduces overbank flooding, resulting in less seed dispersal and a reduced ability of riparian vegetation to establish (Harris and Gosselink 1990). Kraemer (1984) reported that loss of riparian flood plain along the Sacramento River led to decreased sediment deposition and energy dissipation, resulting in increased flows and less stable streambeds and banks. Once the flood plain is developed, storms result in more overland flow due to impervious surface, but less percolation (Faber 1993). Furthermore, disconnecting rivers from their flood plains reduces their ability to attenuate flood peaks, limits natural sediment deposition and water quality enhancement, and disrupts downstream successional processes and scour cycles (Warner and Hendrix 1985, Harris and Gosselink 1990, Scott and others 1990). Although specific effects vary, in general channel ''improvements'' cause downstream flood hydrographs to have higher peaks and also cause peaks to occur earlier (DeVries 1980).

The flood-prone area is defined as the bankfull channel plus the area defined by a horizontal projection at a height of twice the bankfull thalweg (Rosgen 1994). This criterion is based on flood-prone area instead of the flood plain because the former represents the area regularly exposed to overbank flow. Although the margins of the flood plain contribute greatly to the ecological function of the riparian system, these areas are often not subject to Corps jurisdiction (in semi-arid systems) and are therefore not the focus of mitigation efforts. The ratings for characteristics of the flood-prone area criterion are assigned based on the following scale.

First- and second-order streams.

0 = All flows, including flood flows, are contained in a concrete-lined channel, culvert, etc.
0.2 = Channel has an earthen bottom; however, it is structurally confined (e.g., riprap or concrete sideslopes) such that the flood-prone area is within the confined channel and flow would only overtop the channel during extreme events (i.e., greater than a 50-year-flood event).
0.4 = Channel has an earthen bottom and earthen sideslopes; however, it is incised or confined such that the channel would only overtop during extreme flow events (i.e., greater than a 50-year-flood event).
0.7 = Channel has an earthen bottom and earthen sideslopes and is mildly incised or confined such that the flood-prone area would be subject to periodic overbank flow (i.e., during a 10-year-flood event).
1.0 = Site is a natural channel with little to no evidence of incision or confinement.

Higher order streams.

0 = All flows, including flood flows, are contained in a concrete-lined channel, culvert, etc.
0.2 = Channel has an earthen bottom; however, it is structurally confined (e.g., riprap or concrete sideslopes) such that the flood-prone area is wholly contained within the channel and there is no opportunity for overbank flow, except in extreme events.
0.3 = Channel has an earthen bottom and earthen sideslopes; however, it is incised or confined such that the flood-prone area is wholly contained within the channel and there is no opportunity for overbank flow, except in extreme events.

0.6 = Site is part of a flood plain, which provides an opportunity for overbank flow during moderate flow events (i.e., during a 2- to 10-year-flood event). However, the flood-prone area is confined by levees, berms, dikes, or other obstructions or barriers such that the area available for overbank flow is less than twice the width of the channel at bankful conditions.

0.8 = Site is part of a flood plain, which provides an opportunity for overbank flow during moderate flow events (i.e., during a 2- to 10-year-flood event). The flood-prone area is confined by levees, berms, dikes, or other obstructions or barriers; however, the area available for overbank flow is equal to or greater than twice the width of the channel at bankfull conditions.

1.0 = Site is part of an unconfined natural floodplain at least twice the width of the channel at bankfull conditions and there is evidence of overbank flow.

Biogeochemical Processes

The location of riparian areas along streams along with the relatively low topography, natural ponding, and ground surface roughness of riparian zones allows them to act as sinks for sediment and nutrient runoff from adjacent uplands and as sources for conversion of detritus to consumable organic matter (Childers and Gosselink 1990, Scott and others 1990). Rising water overtops streambanks, slowing the flow velocity, allowing water and suspended material to access the adjacent flood plain and riparian zones (Gosselink and others 1990a, Scott and others 1990). Microbial action in the root zone removes toxics, nitrogen, and other nutrients from the runoff; thereby improving water quality and helping to reduce the impacts of nonpoint source pollution (Schaefer and Brown 1992). Peterjohn and Correll (1984) reported that each ha (2.47 acres) of riparian forest removed 4.1 mg of particulates, 11 kg of particulate organic nitrogen, 0.83 kg of ammonium-nitrogen, 2.7 kg of nitrate-nitrogen, and 3.0 kg of total particulate phosphorus per year. Gregory and others (1991) reported that up to 65% of the nitrogen and phosphorus can be removed from agricultural runoff by riparian vegetation. Heterotrophic microorganisms that thrive in riparian areas are also responsible for converting detritus from leaf litter and other dead organic matter into consumable organic matter. This organic material forms the base for the riparian food chain and can be released downstream as dissolved organic matter (Gregory and others 1991, Schaefer and Brown 1992). Knight and Bottoroff (1984) reported that up to 1000 $g/m^2/year$ of detritus are produced by aquatic macrophytes in riparian zones, and this provides a food chain base for these ecosystems, promoting their biodiversity.

Biogeochemical processes depend on water flow through the site, availability of surfaces to slow water and provide a platform for microbial activity and chemical reactions and as sources of organic carbon. Water flow and availability of flood plain surfaces are addressed by the criteria discussed above. The ratings for surface roughness and sources of organic carbon are assigned based on the following scales:

First- and second-order streams.

0 = channel is contained in a concrete-lined channel, culvert, etc., with little to no vegetation or detritus.

0.2 = Site can support grasses, forbs, or other herbaceous vegetation, or there is debris, leaf litter, or detritus present in the channel.

0.4 = Channel supports at least 5% relative cover of herbaceous or other vegetation and there is at least 10% relative cover of debris, leaf litter, or detritus in the channel.

0.6 = Site contains between 5% and 20% relative cover of any type of vegetation and between 10% and 25% relative cover with debris, leaf litter, or detritus.

0.8 = Site contains greater than 20% relative cover of any type of vegetation or between 25% and 60% relative cover with debris, leaf litter, or detritus.

1.0 = Site contains greater than 20% relative cover of any type of vegetation and greater than 60% relative cover with debris, leaf litter, or detritus.

Higher order streams.

0 = channel is contained in a concrete-lined channel, culvert, etc., with little to no vegetation or detritus.

0.2 = Site can support grasses, forbs, or other herbaceous vegetation, and there is woody debris, leaf litter, or detritus present in the channel.

0.4 = Channel supports at least 25% relative cover of grasses, forbs, herbaceous, or riparian vegetation, and there is at least 10% relative cover of woody debris, leaf litter, or detritus in the channel.

0.6 = Site contains between 25% and 50% relative cover of any strata of riparian vegetation and between 10% and 40% relative cover with woody debris, leaf litter, or detritus.

0.8 = Site contains between 50% and 75% relative cover of any strata of riparian vegetation and between 40% and 60% relative cover with woody debris, leaf litter, or detritus.

1.0 = Site contains greater than 75% relative cover of any strata of riparian vegetation and greater than 60% relative cover with woody debris, leaf litter, or detritus.

Calculation of Credits and Debits

Credits and debits are calculated based on the number of condition units per ha (CU) gained at the mitigation bank and lost at the impact site. The number of CU per ha is calculated by adding the scores for most of the evaluation criteria. We chose to multiply the scores for the three habitat criteria by the score for the percent of invasive vegetation criterion because infestation with invasive vegetation tends to depress all habitat function in riparian systems. Using the score for percent invasive vegetation as a multiplier precludes the need to specify native versus invasive vegetation under the other habitat criteria. The effect of increased biomass associated with invasive plants on hydrologic and biogeochemical processes is accounted for in the biogeochemical processes criterion. Hydrology is widely recognized as the driving force behind wetland and riparian systems. Therefore, the condition units formula weights the hydrologic regime criterion at three times the importance of other criteria. This reflects the fact that appropriate hydrology is fundamental to overall riparian function, and it devalues sites with artificial or inappropriate hydrology. We recognize there may be some overlap between criteria, for example, the density of vegetation at a site contributes to the rating under both the coverage and spatial diversity and the biogeochemistry criteria; however, we believe this is appropriate because certain characteristics of a site contribute to multiple functions (e.g., habitat and biogeochemical functions). The number of condition units/ha is calculated using the following formula:

$$CU = [(ST + SP + CNT)\ I] + FPA + TC + BR + 3H$$

where ST = Habitat − Structural Diversity; SP = Habitat − Coverage and Spatial Diversity; CNT = Habitat − Contiguity; I = Percent of Invasive Vegetation; FPA = Characteristics of the Flood-prone Area; TC = Topographic Complexity; BR = Biogeochemistry − vegetation roughness and organic carbon; H = Hydrology.

When performing functional assessments for their own sake, for the purposes of impact evaluation or for design or evaluation of mitigation sites, functions should not be combined into overall indices. The practice of combining functions can result in certain functions being masked, thereby underestimating the overall importance of a wetland to watershed ecology and decreasing the resolution of the functional assessment. However, the intent of this method is not to evaluate wetland functions but to provide a tool to calculate mitigation ratios based on the ecologic condition of impact and restoration sites. To accomplish this goal in the context of a mitigation bank, we must generate a single number or index.

Application of the Riparian Model

The Santa Ana River Mitigation Bank (SARMB) provided the earliest application of the southern California's riparian crediting and debiting model. The Santa Ana River, with a watershed of area of 6345 km^2 (2450 $miles^2$), is the largest river system is southern California (Hanes 1981). The riparian habitat along the Santa Ana River is southern riparian scrub consisting of *Salix*, *Populus*, and *Baccharis* species. Of the 5667 ha (14,000 acres) of riparian habitat along the Santa Ana River, approximately 2000 ha (5000 acres) are infested with an invasive species commonly known as the giant reed, *A. donax* (Bell 1993). Replacement of the native riparian vegetation with *A. donax* has not only led to loss of suitable habitat for many wildlife species, including the federally listed endangered least Bell's vireo, *Vireo bellii pusillus*, but has also caused problems with water quality and water conservation (Bell 1993, Iverson 1993).

The Santa Ana River mitigation bank is located in the northern portion of Riverside County in the City of Riverside (Figure 1). The goal of the SARMB is to restore a degraded riparian system by reestablishing the native riparian ecological diversity and other riparian functions, such as flood flow alteration, groundwater recharge, improvement of water quality (temperature and organic matter), reduction of fire hazard, and increased recreational use. Credits were established by removal of invasive vegetation and selective planting to encourage natural recruitment of native riparian vegetation.

The size of the initial mitigation bank area was approximately 22.7 ha (56 acres). Since the initial bank establishment, additional areas have been incorporated in the bank, however, the information provided here only reflect the original 22.7 ha. The potential credits were determined using aerial photographs and field surveys. Review of aerial photographs and field surveys revealed three characteristic regions with varying degrees of invasive vegetation infestation within the bank area (Figure 2, Table 2). To determine the available credits, each region of the mitigation bank was rated

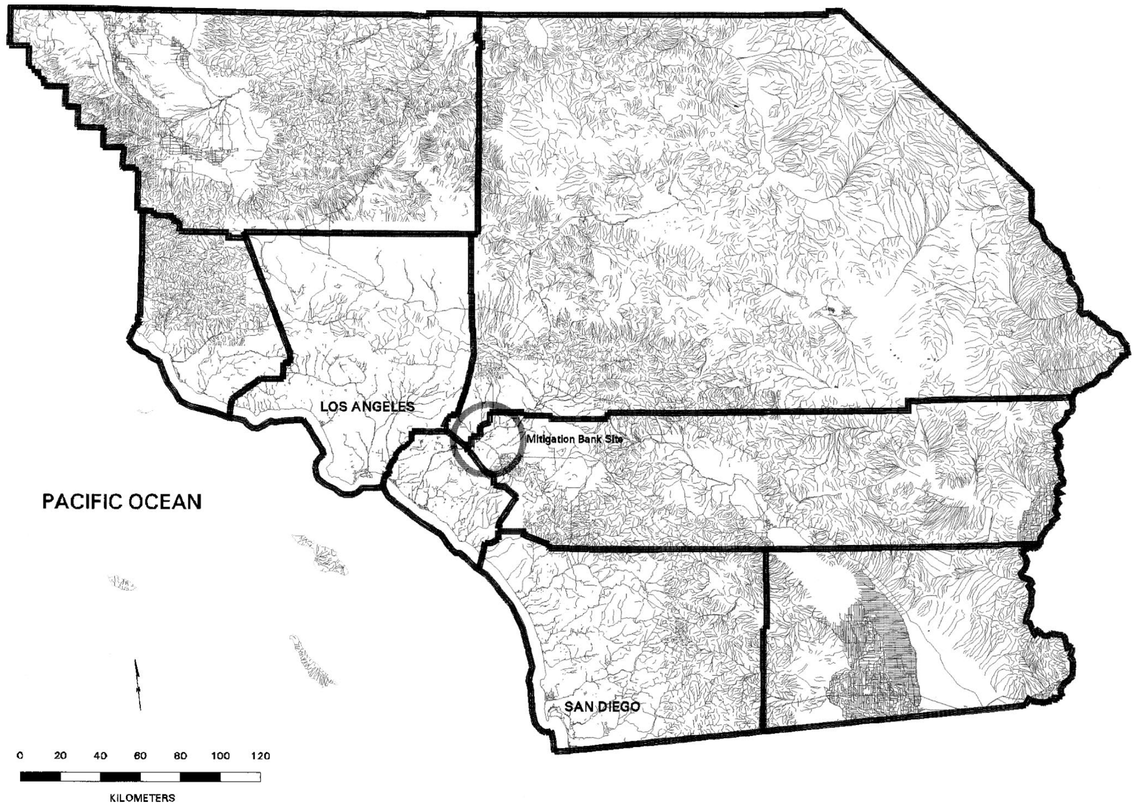

Figure 1. Location of the Santa Ana River mitigation bank. The area circled shows the location on the Santa Ana River where the mitigation bank is located. Thick lines are county boundaries, fine lines are streams.

separately based on homogeneity of the region or subunit. Post-bank scores were assigned based on the maximum possible score a similar uninfested resource in that region could achieve. The cumulative difference between the baseline rating and the predicted post-bank rating is the total available credits in the mitigation bank.

Region A, which is approximately 12.4 ha (30 acres) (62% of the total area), consists of 100% *Arundo* infestation with no native riparian vegetation and is adjacent to a riparian zone with two species of woody riparian plants and poorly developed understory on one half of the site and a structurally and spatially diverse riparian zone on the remaining half. Because of the high degree of *Arundo* infestation, there is low topographic complexity and diversity of detritus in Region A. Region B is approximately 7.6 ha (26 acres) (38% of the total area), and consists of mixed native riparian vegetation with shrub and herbaceous understory interspersed with 20–40% *Arundo*. Region B is connected to structurally and spatially diverse riparian zone on one half and 100% *Arundo*-infested zone on the remaining half. The topographic complexity and density of detritus is relatively greater in Region B than in Region A due to the presence of secondary channels and a native riparian vegetation. Region C is not part of the mitigation bank, however, it will be preserved and will function as a buffer between the bank and adjacent land uses. Region C consists of relatively mature riparian species (10–20 years), with a well-developed canopy, diverse understory, and less than 5% *Arundo* present. All three regions possess natural hydrology and are within a flood-prone area greater than twice the width of the active channel at bankful conditions. The scores for pre- and post-bank conditions are shown in Table 3.

The restoration effort began in 1993, when the local community underwent significant threat of fires fueled by *Arundo*. *Arundo* (reaching height of up to 8 m) is a tall grass native to eastern Asia, introduced to southern California in early 1800s for purpose of erosion control. Due to its high rate of growth (5 cm / day), it outcompetes the native riparian vegetation and soon becomes the dominant species in the riparian zone (Bell 1997). The rhizome, which typically reaches depths of 1 m, quickly stabilizes the stream bank and forms terraces severing the riparian zone from fluvial processes typi-

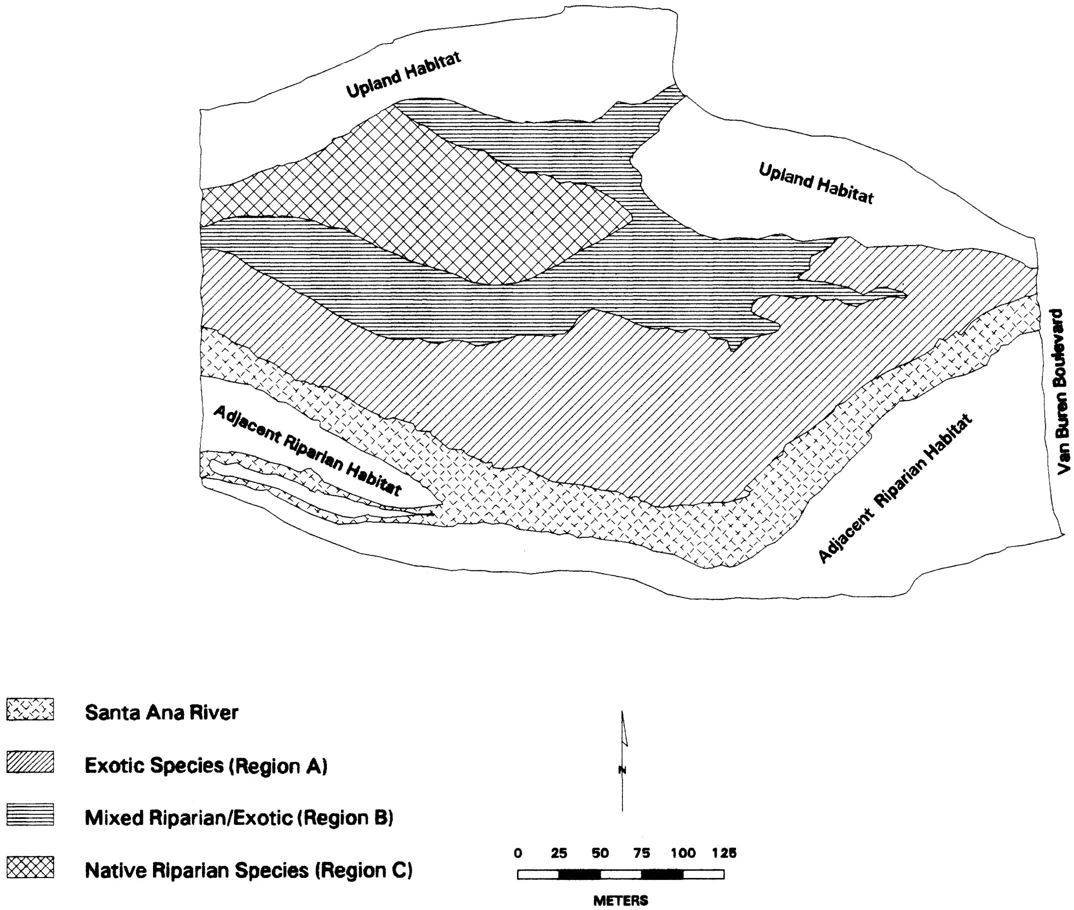

Figure 2. Characteristic regions of the Santa Ana River mitigation bank. The map shows the dominant plant community for each subarea of the mitigation bank prior to initiation of any restoration efforts (e.g., baseline conditions). Mapping was based on aerial photography, dated April 1993. Area C was not included in the mitigation bank because it is existing native riparian habitat.

Table 2. Characteristics of regions A & B of the Santa Ana River mitigation bank. Size, percent of total area, and percent infestation with invasive weeds of the two subareas of the Santa Ana River mitigation bank

Region	Size (ha)	% total area	% invasives
A	12.4	62	100
B	7.6	38	20–40

cally occurring within native vegetation–dominated riparian zones. The reduction of overbank flooding limits aggredation and degradation processes consequently limiting native vegetation propagule dispersal. The high growth rate of *Arundo* combined with its high degree of flammability soon redirects the native riparian community to an *Arundo*-infested riparian zone (Bell 1993). The ecological changes that occur within a riparian zone as a result of *Arundo* infestation include reduction of suitable habitat for native wildlife, highly altered flooding regime, and reduction of biogeochemical processes due to the reduction of surface moisture and presence of noxious chemicals, such as silica, tri-terpines, and sterols (Chanduri and Ghosal 1970, Bell 1997).

Restoration of the Santa Ana River mitigation bank was accomplished using a glyphosate, an EPA-approved herbicide for use in wetlands. The application method varied depending on the extent of *Arundo* infestation and extent of native riparian vegetation present in the treatment areas. The most effective period for applying the herbicide was determined to be during the period when maximum translocation of nutrients to the root is occurring (the period between post-flowering and predormancy) (Bell 1993). Method of application of the herbicide in the SARMB included aerial application (areas >80% *Arundo*), use of all-terrain vehicles (<80% *Arundo*-infested areas easily accessible), and backpack

Table 3. Rating of the two regions in the Santa Ana River mitigation bank. Scores for each criterion for the pre-restoration baseline condition in each subarea. Weighted mean is the average of the criterion score for each subarea adjusted for the subarea's proportion of the total area. Post-project scores reflect the anticipated condition of the site upon maturation of the restoration efforts

Criterion	Region A (62%)	Region B (38%)	Pre-project (weighted mean)	Post-project
ST	0.2	0.6	0.35	1.00
SP	0.2	0.6	0.35	1.00
CNT	0.8	0.8	0.80	0.80
I	0	0.6	0.23	0.80
FPA	0.4	0.4	0.40	0.80
TC	0.2	0.8	0.43	1.0
BR	0.2	0.8	0.43	0.80
H	1.0	1.0	1.0	1.0

Legend: ST = Habitat—Structural Diversity; SP = Habitat—Coverage and Spatial Diversity; CNT = Habitat—Contiguity; I = Invasive Vegetation; FPA = Characteristics and the Flood-prone Area; TC = Topographic Complexity; BR = Biogeochemistry—Vegetation Roughness and Organic Carbon; H = Hydrology.

sprayers (areas difficult to access with vehicles and resprouts). The biomass was cut by hand cutting, chipper, or hydro-ax and removed by hauling to a suitable off-site location or controlled biomass burning. Once the biomass was removed, selective planting was carried out on portions of the bank site to accelerate the natural revegetation process. In the initial 2–3 years treatment or respoutes occurred on a regular basis (2–3 times/year) and continually declined as the native riparian vegetation began to self-recruit and the root mass decomposed. The change in ecological condition of the SARMB became apparent following the third year of treatment as evident by change in characteristics of the flood-prone area, structural diversity of native vegetation, enhancement of topographic complexities (ponds, bars, hummocks, and secondary channels), enhanced biogeochemical processes (due to increase surface moisture and evident by visible microbial activity). The contiguity of the bank area would not be affected as a result of the restoration work, as the site is connected to riparian and upland habitats and no modification is expected to occur in these areas. The SARMB continues to be actively monitored, and it serves as a model for native riparian restoration project throughout California.

Debits at the impact sites are determined by assigning of pre-project and post-project ratings for each criterion. Evaluation of a pre- and post-project at the impact site allows for consideration of any remaining functional characteristics at the impact site following

Table 4. Sample calculation of debits: This table shows a *hypothetical* example of the application of the crediting and debiting framework to an impact site. For this example, the impact would be complete fill of the stream on the project site

Criterion	Pre-project rating	Post-project rating	Net functions lost
ST	0.8	0	0.8
SP	0.8	0	0.8
CNT	0.8	0	0.8
I	0.8	0	0.8
FPA	0.8	0	0.8
TC	1.0	0	1.0
BR	0.8	0	0.8
H	1.0	0	1.0

Legend: ST = Habitat—Structural Diversity; SP = Habitat—Coverage and Spatial Diversity; CNT = Habitat—Contiguity; I = Invasive Vegetation; FPA = Characteristics and the Flood-prone Area; TC = Topographic Complexity; BR = Biogeochemistry—Vegetation Roughness and Organic Carbon; H = Hydrology.

Table 5. Calculation of credits and debits for the Santa Ana River mitigation bank: Sample application of the crediting and debiting framework to determine a mitigation ratio. Credits are determined by using the criteria scores shown in Table 3. Debits are determined by using the hypothetical scores shown in Table 4

Step 1. Evaluation of credits

$CU = \{[(ST + SP + CNT)I] + FPA + TC + BR\} + 3H$

Pre-bank $CU = \{[0.35 + 0.35 + 0.8)0.23] + 0.4 + 0.61 + 0.43\} + 3(1.0) = 4.78$

Post-bank $CU = \{[1.0 + 1.0 + 0.80)0.8] + 0.8 + 1.0 + 0.8\} + 3(1.0) = 7.84$

Projected Credits Available $= 7.84 - 4.78 = 3.06$

Step 2. Evaluation of debits

Debits (Functional Units Lost) $= \{[(0.8 + 0.8 + 0.8)0.8] + 0.8 + 1.0 + 0.8\} + 3(1.0) = 7.52$

Step 3. Determination of mitigation ratio

Mitigation Ratio $= 7.52/3.06 = 2.45$

Legend: ST = Habitat—Structural Diversity; SP = Habitat—Coverage and Spatial Diversity; CNT = Habitat—Contiguity; I = Invasive Vegetation; FPA = Characteristics and the Flood-prone Area; TC = Topographic Complexity; BR = Biogeochemistry—Vegetation Roughness and Organic Carbon; H = Hydrology.

implementation of a project. A hypothetical debiting score scenario to be mitigated at this bank is presented in Table 4, where a project would impact an aquatic resource with a relatively high functional characteristics. The mitigation ratio for the hypothetical debiting scenario for use of the SARMB is calculated to be 2.5:1 (Table 5). This mitigation ratio is based on withdrawal of credits when the credits have reached their expected

Table 6. Credits available during the first 5 years of operation: Discounting of the mitigation ratio (as determined in Table 5) for years 1 through 5. This hypothetical scenario assumes the mitigation site achieves the performance goals by year 5

Year	Total projected credits	Credits available	Mitigation ratio
1	3.06	$3.06 \times 0.2 = 0.61$	$7.52 / 0.61 = 12.3$
2	3.06	$3.06 \times 0.4 = 1.22$	$7.52 / 1.22 = 6.2$
3	3.06	$3.06 \times 0.6 = 1.84$	$7.52 / 1.84 = 4.1$
4	3.06	$3.06 \times 0.8 = 2.45$	$7.52 / 2.45 = 3.1$
5	3.06	$3.06 \times 1.0 = 3.06$	$7.52 / 3.06 = 2.5$

restoration potential, which is estimated to take approximately 5 years.

Although mitigation banks should conceptually reach their predicted functional maturity prior to withdrawal of credits, there may be a need to withdraw credits prior to achievement of functional maturity. It should be noted that in these circumstances financial assurance must be secured by the bank sponsor. This crediting and debiting method gives flexibility in withdrawal of credits prior to full functional establishment of an aquatic resource by allowing adjustment of the mitigation ratio to reflect the existing conditions at the mitigation bank. The actual condition of the mitigation bank may be evaluated at a given time interval and the percentage of total expected credits may become available for withdrawal. A simplified example may be a mitigation bank where at the end of the first year credits have 20% of their maximum potential value, 40% of the total potential value in the second year, 60% of potential value in the third year, and 80% of the predicted value at the end of the fourth year (Table 6). Credits would have their full functional maturity at the end of the fifth year. This would allow sale of credits at a partial value to provide funds for the sponsor within the initial establishment period. In addition, if credits have only partial value in the initial 5 years, the mitigation ratios obtained should be high enough to deter the use of this mitigation bank for projects with impacts to riparian habitats with high functional capacity and consequently should encourage avoidance and minimization of impacts to these habitats. By setting a minimum compensation ratio of 1:1 the crediting and debiting methodology prevents loss of acreage.

Discussion

Despite the existence of numerous methods for assessing functions of aquatic resources, compensation ratios are typically determined based on existing policy and / or best professional judgment of decision makers. Existing mitigation banks typically use either an acreage-based or case-by-case best professional judgment determination of functional characteristics and compensation ratios (IWR 1994, Tabatabai 1994). As mitigation banking gains support from the regulated public, entrepreneurs, and the resource and regulatory agencies, a greater need arises for use of an appropriate crediting and debiting methodology in any mitigation bank. Use of detailed functional assessment methodologies or site-specific evaluation of function for determination of credits and debits is far superior to rapid approaches, such as the one we present in this paper. However, the constraints posed by their application (e.g., time and resources) makes their use impractical in the mitigation banking context. The goal of this framework is to provide a rapid assessment of credits and debits that does not require extensive field data collection and where the assessment of structural components of an aquatic resource could be used by nonwetland scientists. The proposed framework meets the six objectives necessary for a crediting and debiting system to be useful in a regulatory context.

(1) It can be tailored to evaluate ecologic condition based on the target resources of a specific mitigation bank. The example presented in this paper illustrates application of the proposed framework to a mitigation bank where the goal is to restore riparian habitat. This framework is currently being applied to a bank where the goal is restoration of vernal pools; therefore, specific criteria have been developed that reflect the ecologic conditions of depressional wetlands. For example, one of the evaluation criteria addresses duration of ponding and is scaled as follows:

0 = Ponding is transient following storm events and persists for no more than 1 day.
0.2 = Site may pond water for several days following storm events; however, ponding seldom persists beyond 10 days. There may be several ponding events during a season.
0.4 = Ponding duration is on the order of several weeks. There may be several ponding events during a season.
0.6 = Ponding duration is on the order of several months, but less than 6 months. There may be several ponding events during a season.
0.8 = On average, site ponds water for more than 6 months.
1.0 = Site ponds water year-round.

(2) It is flexible enough to be used for evaluation of existing or potential ecologic condition at a mitigation

bank. Because credits are determined based on the difference between structural characteristics of the post-restoration condition and pre-restoration (baseline) condition at the bank site, the framework can be applied in a predictive manner. A CU score can be calculated based on the expected future condition at the bank and used as the ''enhancement potential'' for the purpose of determining mitigation ratios. The success of the restoration can be evaluated by comparing the condition of the resources over time to the expected future condition, and remedial measures can be implemented to ensure the target condition is achieved. This crediting and debiting method also provides the flexibility to account for withdrawal of credits prior to full functional establishment of an aquatic resource by allowing adjustment of the mitigation ratio to reflect the condition of the resources at the mitigation bank at time of purchase of credits.

(3) It is a structured and systematic way to apply data and professional judgment to the decision-making process. The intent of the proposed framework is not to provide an absolute tool for evaluating functional condition. However, it does provide an alternative to subjectively applied best professional judgment by establishing a structure to organize information and apply judgment in an objective manner, based on ecological principles. For example, in instances where a mitigation banking agreement contains limits on the quality of riparian habitat that can use the bank for mitigation, this framework can be used to determine whether the ''quality'' of the resources at the impact site exceeds the stated threshold. Several proposed banks in the Los Angeles District of the Corps involve restoration or enhancement of existing aquatic resources and have stipulations in the banking instruments that only allow impacts to degraded habitats to be mitigated at the bank. In these cases, a proposed project site which receives a pre-project rating of 4 CUs or greater would be precluded from using the bank. This provides an objective way to ensure that the functions gained through the mitigation bank are commensurate with the impacts for which credits were purchased.

(4) It has an ecologically defensible basis. Credits and debits are based on the structural characteristics and landscape setting of the restoration and the impact sites. The evaluation criteria reflect attributes of wetlands shown to be important to their viability and ability to provide a suite of ecologic functions. Many of the criteria presented for riparian systems are similar to one used in established functional evaluation methods, such as HGM and HEP.

(5) It has ease of use such that the level of expertise and time required to employ the method is not a deterrent to its application. The evaluation criteria have been structured so that they can be applied based on information typically provided in biological resource reports that accompany U.S. Army Corps of Engineers permit applications. Because the criteria are generally descriptive, they can be applied with minimal ambiguity based on information supplied by permit applicants. When being applied in the field, scores can be assigned with a reasonable amount of data collection, yet not so intensive as to be a deterrent to its use. In practice, most sites can be evaluated by review of aerial photography and several hours in the field. This is commensurate with the time and resource constraints of the regulatory program.

(6) It provides a semi-quantitative measure of the condition of aquatic resources that can be translated to a mitigation ratio. Scaling of the evaluation criteria is based on a combination of field data collected during development of a regional HGM assessment model (Lee and others 1997), research on the success of past mitigation projects in southern California (Sudol 1996), and professional judgment of scientists familiar with semi-arid riparian systems. The framework organizes this information in a categorical manner and provides a way to translate information about the conditions of a site to a quantitative mitigation ratio. The framework may also be applicable for determination of out-of-kind compensation ratios. Aquatic resources can be evaluated based on criteria developed for each specific wetland type. The relative conditions can then be translated into a common currency or unit of measure and the compensation ratio assigned using the ratio of debits over credits.

The crediting and debiting framework presented in this work is not designed as a functional assessment methodology; rather, it is intended to be a rapid semi-quantitative measure of structural characteristics of an aquatic resource for the purpose of determining compensatory requirements. The framework, as demonstrated with the southern California riparian model, offers an alternative to use of existing functional assessment methodologies or best professional judgment for determination of credits and debits. We encourage more dialogue and debate among scientists, regulators, and the public on the merits of this approach and the details of its applications, such as choice of indicators, scaling of criteria, and architecture of the CU formula. The ultimate goal should be an objective and systematic way to determine ecologically meaningful mitigation requirements that are commensurate with the impacts and result in a net benefit for the resources protected by the Clean Water Act.

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