A Draft Regional Guidebook for Applying the Hydrogeomorphic Approach to Assessing Wetland Functions of Vernal Pool Depressional Wetlands in Southern California

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ABSTRACT: This Draft Guidebook is an assessment tool that focuses on the functioning of vernal pool wetlands within the Southern Californian eco-region, specifically San Diego County. Its purpose is to provide trained practitioners the means to achieve efficient, reproducible and logical functional assessment results for vernal pool wetlands in San Diego County, California. Results of these assessments can then be used in a variety of ways, such as evaluation of sites for restoration potential, assessment of impacts from existing or proposed projects and monitoring restoration success. Due to the high degree of variability experienced by temporary wetlands in arid climates, we have developed both direct and indirect functional indices for four of the five functions we identified. Direct assessments can only be made when there is sufficient precipitation to elicit the responses that demonstrate function, and we have sought to objectively define "sufficient." Consistent with an HGM approach, use of this Draft Guidebook should be confined to the geographic region and hydrogeomorphic class, subclass and pool types for which it was developed. Use of this methodology outside the boundaries of the reference domain is wholly inappropriate. We are hopeful that our approach can be modified for other pool types within the region, and to vernal pools in other parts of California and Oregon.

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2 Overview of the Hydrogeomorphic Approach

As reviewed in Chapter 1, the HGM approach is a collection of concepts and methods for developing functional indices and subsequently using them to assess a wetland's capacity to perform functions relative to similar wetlands in a region. The HGM approach includes four integral components: (a) the HGM classification, (b) identification of reference wetlands, (c) assessment models/functional indices and (d) assessment protocols. During the development phase of the HGM approach, these four components are integrated in a Regional Guidebook for assessing the functions of a regional wetland subclass. Subsequently, during the application phase, end users assess the functional capacity of selected wetlands following the Regional Guidebook's assessment protocols. This chapter discusses each component of the HGM approach, and the development and application phases. More extensive discussions of the general approach can be found in Brinson (1993) and Smith *et al.* (1995). Guidelines for the development of guidebooks are contained in Clairain (2002), Smith (2001), Smith and Wakeley (2001) and Wakeley and Smith (2001). A comprehensive glossary of terms that are specific to HGM and vernal pool geology, hydrology and biology is provided at the end of this guidebook.

Hydrogeomorphic Classification

Wetland ecosystems share a number of features, including relatively long periods of inundation or saturation, hydrophytic vegetation and hydric soils. In spite of these common attributes, wetlands occur under a wide range of climatic, geologic, and physiographic settings and exhibit a wide variety of physical, chemical, and biological characteristics and processes (Cowardin *et al.* 1979, Ferren *et al.* 1996, Ferren *et al*. 1996ab, Gosselink and Turner 1978, Mitsch and Gosselink 2000). The variability of wetlands makes it challenging to develop assessment methods that are both accurate (*i.e.*, sensitive to significant changes in function) and practical (*i.e.*, can be completed in the relatively short time available for conducting assessments). Existing "generic" methods designed to assess multiple wetland types throughout the United States are relatively rapid, but often lack the resolution necessary to detect significant changes in function for any specific wetland type. The most logical way to achieve an appropriate level of resolution within the available time frame is to reduce the level of variability in the wetlands being considered by focusing on a more restricted set (Smith *et al.* 1995).

The HGM Classification was developed specifically to accomplish this task (Brinson 1993). It identifies groups of wetlands that function similarly using three fundamental criteria: geomorphic setting, water source, and hydrodynamics. Geomorphic setting refers to the landform and position of the wetland in the landscape. Water source refers to the primary water source in the wetland, such as precipitation, overbank floodwater or groundwater. Hydrodynamics refers to the level of energy and the direction that water moves in the wetland. Based on these three classification criteria, any number of "functional" wetland groups can be identified at different spatial or temporal scales. For example, Brinson (1993) identified five hydrogeomorphic wetland classes at a continental scale. These were later expanded to the seven classes described in Table 2.1 (Smith *et al.* 1995). In many cases, the level of variability encompassed by a continent-wide hydrogeomorphic class is still too great for assessment models that are both rapid to apply and sensitive to functional changes relevant to the 404 review process or other assessment purposes. For example, at a continental scale, the depression class includes wetland ecosystems as diverse as vernal pools in California (Solomeshch *et al.* 2007, Witham *et al.* 1998) and in glaciated forests of the Northeast (Colburn 2004, Calhoun and deMaynadier 2008); prairie potholes in North and South Dakota (Hubbard 1988, Kantrud *et al.* 1989); playa lakes in the high plains of Texas (Bolen *et al.* 1989); huecos, springs and tinajas in Utah and west Texas (Joqué *et al*. 2007, MacKay *et al.* 1990, Vinson and Dinger 2008, Wallace *et al.* 2005); and cypress domes in Florida (Kurz and Wagner 1953).

To reduce both inter- and intraregional variability, the three classification criteria are applied at a smaller, regional geographic scale to identify regional wetland subclasses. In many parts of the country, existing wetland classifications can serve as a starting point for identifying these regional subclasses (Ferren *et al.* 1996, Ferren *et al.* 1996ab, Golet and Larson 1974, Ratliff 1982, Rheinhardt and Hollands 2008, Stewart and Kantrud 1971, Wharton *et al.* 1982). Like the continental classes, regional subclasses are distinguished on the basis of geomorphic setting, water source, and hydrodynamics. In addition, certain ecosystem or landscape characteristics may also be useful for distinguishing regional subclasses in certain regions. For example, depressional subclasses might be based on water source (*i.e.*, groundwater versus surface water) or the degree of connection between the wetland and other surface waters (*i.e.*, the flow of surface water in or out of the depression through defined channels). Tidal fringe subclasses might be based on salinity gradients (Shafer and Yozzo 1998). Slope subclasses might be based on the degree of slope, landscape position, or the source of water (*i.e.*, through flow versus groundwater). Riverine subclasses might be based on water source, position in the watershed, stream order, watershed size, channel gradient, or floodplain width. Examples of potential regional subclasses are shown in Table 2.2 (Smith *et al.* 1995, Rheinhardt *et al.* 1997, Hauer *et al.* 2002).

Table 2.1. Hydrogeomorphic Wetland Classes at a Continental Geographic Scale	
HGM Wetland Class	**Definition**
Depression	Depression wetlands occur in topographic depressions (i.e., closed elevation contours) that allow the accumulation of surface water. Depression wetlands may have any combination of inlets and outlets or may be closed basins that lack them completely. The water source may come from one or any combination of the following: precipitation, overland flow, streams, or groundwater/interflow from adjacent uplands. The predominant direction of flow is from the higher elevations toward the center of the depression, but may come from a deep aquifer, or subsurface springs. The predominant hydrodynamics are vertical fluctuations that range from diurnal to seasonal. Depression wetlands may lose water as evapotranspiration, through intermittent or perennial outlets, or as recharge to groundwater. Prairie potholes, playa lakes, vernal pools, and cypress domes are common examples of depression wetlands.
Tidal Fringe	Tidal fringe wetlands occur along coasts and estuaries and are under the influence of sea level. They intergrade landward with riverine wetlands where tidal current diminishes and river flow becomes the dominant water source. Additional water sources may be groundwater discharge and precipitation. The interface between the tidal fringe and riverine classes is where bidirectional flows from tides dominate over unidirectional ones controlled by floodplain slope of riverine wetlands. Because tidal fringe wetlands frequently flood and water table elevations are controlled mainly by sea surface elevation, tidal fringe wetlands seldom dry for significant periods. Tidal fringe wetlands lose water by tidal exchange, by overland flow to tidal creek channels, and by evapotranspiration. Organic matter normally accumulates in higher elevation marsh areas where flooding is less frequent and the wetlands are isolated from shoreline wave erosion by intervening areas of low marsh. Spartina alterniflora salt marshes are a common example of tidal fringe wetlands.
Lacustrine Fringe	Lacustrine fringe wetlands are adjacent to lakes where the water elevation of the lake maintains the water table in the wetland. In some cases, these wetlands consist of a floating mat attached to land. Additional sources of water are precipitation and groundwater discharge, the latter dominating where lacustrine fringe wetlands integrade with uplands or slope wetlands. Surface water flow is bidirectional, usually controlled by water-level fluctuations resulting from wind or seiche. Lacustrine wetlands lose water by flow returning to the lake after flooding and evapotranspiration. Organic matter may accumulate in areas sufficiently protected from shoreline wave erosion. Unimpounded marshes bordering the Great Lakes are an example of lacustrine fringe wetlands.
Slope	Slope wetlands are found in association with the discharge of groundwater to the land surface, or at sites with saturated overland flow with no channel formation. They normally occur on sloping land ranging from very gentle to steep. The predominant source of water is groundwater or interflow discharging to the land surface. Direct precipitation is often a secondary contributing source of water. Hydrodynamics are dominated by downslope unidirectional water flow. Slope wetlands can occur in nearly flat landscapes if groundwater discharge is a dominant source to the wetland surface. Slope wetlands lose water primarily by saturated subsurface flows, surface flows, and by evapotranspiration. Slope wetlands may develop channels, but the channels serve only to convey water away from the slope wetland. Slope wetlands are distinguished from depression wetlands by the lack of a closed topographic depression, and the predominance of the groundwater/interflow water source. Fens are a common example of slope wetlands.

continued

Table 2.1. *(concluded)* **Hydrogeomorphic Wetland Classes at a Continental Geographic Scale**	
HGM Wetland Class	**Definition**
Mineral Soil Flats	Mineral soil flats are most common on interfluves, extensive relic lake bottoms, or large floodplain terraces where the main source of water is precipitation. They receive virtually no groundwater discharge, which distinguishes them from depressions and slopes. Dominant hydrodynamics are vertical fluctuations. Mineral soil flats lose water by evapotranspiration, overland flow, and seepage to underlying groundwater. They are distinguished from flat upland areas by their poor vertical drainage due to impermeable layers (e.g., hardpans), slow lateral drainage, and low hydraulic gradients. Mineral soil flats that accumulate peat can eventually become organic soil flats. They typically occur in relatively humid climates. Pine flatwoods with hydric soils are a common example of mineral soil flat wetlands.
Organic Soil Flats	Organic soil flats, or extensive peatlands, differ from mineral soil flats, in part because their elevation and topography are controlled by vertical accretion of organic matter. They occur commonly on flat interfluves but may also be located where depressions have become filled with peat to form a relatively large flat surface. Water source is dominated by precipitation, while water loss is by overland flow and seepage to underlying groundwater. They occur in relatively humid climates. Raised bogs share many of these characteristics but may be considered a separate class because of their convex upward form and distinct edaphic conditions for plants. Portions of the Everglades and northern Minnesota peatlands are common examples of organic soil flat wetlands.
Riverine	Riverine wetlands occur in floodplains and riparian corridors in association with stream channels. Dominant water sources are overbank flow from the channel or subsurface hydraulic connections between the stream channel and wetlands. Additional water sources may be interflow or occasional overland flow from adjacent uplands, tributary inflow, and precipitation. When overbank flow occurs, surface flows down the floodplain may dominate hydrodynamics. In the headwaters, riverine wetlands often intergrade with slope or depressional wetlands as the channel (bed) and bank disappear, or they may intergrade with poorly drained flats or uplands. Perennial flow is not required. Riverine wetlands lose surface water via the return of floodwater to the channel after flooding and through surface flow to the channel during rainfall events. They lose subsurface water by discharge to the channel, movement to deeper groundwater (for losing streams), and evapotranspiration. Peat may accumulate in off-channel depressions (oxbows) that have become isolated from riverine processes and subjected to long periods of saturation from groundwater sources. Bottomland hardwood floodplains are a common example of riverine wetlands.

Table 2.2. Potential Regional Wetland Subclasses in Relation to Geomorphic Setting, Dominant Water Source and Hydrodynamics				
Geomorphic Setting	**Dominant Water Source**	**Dominant Hydrodynamics**	**Potential Regional Wetland Subclasses**	
			Eastern USA	**Western USA/Alaska**
Depression	Groundwater, precipitation or interflow	Vertical	Prairie potholes, marshes, Carolina bays	Vernal pools
Fringe (tidal)	Ocean	Bidirectional, horizontal	Chesapeake Bay and Gulf of Mexico tidal marshes	San Francisco Bay marshes
Fringe (lacustrine)	Lake	Bidirectional, horizontal	Great Lakes marshes	Flathead Lake
Slope	Groundwater	Unidirectional, horizontal	Fens	Avalanche chutes
Flat (mineral soil)	Precipitation	Vertical	Wet pine flatwoods	Large playas
Flat (organic soil)	Precipitation	Vertical	Peat bogs, portions of Everglades	Peatlands over permafrost
Riverine	Overbank flow from channels	Unidirectional, horizontal	Bottomland hardwood forest	Riparian wetlands

Adapted from Smith *et al*. 1995 and Rheinhardt *et al*. 1997.

Reference Wetlands

Reference wetland sites are selected in the HGM development process to represent the range of variability that occurs in a regional wetland subclass as a result of natural processes and disturbances (*e.g.*, succession, channel migration, fire, erosion and sedimentation), as well as cultural alteration. The reference domain is the geographic area occupied by the reference wetlands (Smith *et al.* 1995). Although the geographic extent of the reference domain should reflect the geographic area encompassed by the regional wetland subclass, this is not always possible because of time and resource constraints.

Reference wetlands serve several purposes. First, they establish a basis for defining what constitutes a characteristic and sustainable level of function across the suite of functions selected for a regional wetland subclass. Second, they establish the range and variability of conditions exhibited by model variables and provide the data necessary for calibrating model variables and assessment models. Finally, they provide a concrete physical representation of wetland ecosystems that can be observed and measured.

Following accepted HGM practice, reference standard wetlands are typically defined to be the subset of reference wetlands that perform a representative suite of functions at a level that is both sustainable and characteristic of the least-human-altered wetland sites in the least-human-altered landscapes. Table 2.3 outlines the terms as commonly used by the HGM approach in the context of reference wetlands.

Table 2.3. Reference Wetland Terms and Definitions (Smith *et al*. 1995)

Term	Definition
Reference Domain	The geographic area from which reference wetlands representing the regional wetland subclass are selected.
Reference Wetlands	A group of wetlands that encompass the known range of variability in the regional wetland subclass resulting from natural processes and disturbance and from human alteration.
Reference Standard Wetlands	The subset of reference wetlands that perform a representative suite of functions at a level that is both sustainable and characteristic of the least human altered wetland sites in the least human altered landscapes. By convention, the functional capacity index for all functions in reference standard wetlands are assigned a 1.0.
Reference Standard Wetland Variable Condition	The range of conditions exhibited by model variables in reference standard wetlands. By convention, reference standard conditions receive a variable subindex score of 1.0.
Site Potential (mitigation project context)	The highest level of function possible given local constraints of disturbance history, land use, orother factors. Site potential may be less than or equal to the levels of function in reference standard wetlands of the regional wetland subclass.
Project Target (mitigation project context)	The level of function identified or negotiated for a restoration or creation project.
Project Standards (mitigation context)	Performance criteria and/or specifications used to guide the restoration or creation activities toward the project target. Project standards should specify reasonable contingency measures if the project target is not being achieved.

By definition, HGM guidebooks assign all functions in reference standard wetlands a Functional Capacity Index (FCI) of 1.0. However, this treatment of "the best wetlands in the reference domain" leads to conflicts between the principles behind the HGM approach, calibration of specific HGM models and unbiased application of HGM to a diversity of sites. Specifically:

1. It is usually impossible to fit a statistical FCI model with actual field data that yields the same fitted FCI as an *a priori* FCI. (In other words, r^2 for the statistical model is < 1.0). Thus, *a priori* FCI values of 1.0 that have been assigned to reference standards in any particular function can only be approximated with real data. This is true for both direct FCI estimates, and indirect FCI estimates, with greater departures generally found with indirect FCIs. Other HGM

guidebooks generally circumvent this problem by using simple FCI models with only a few variables, so that all reference standard wetlands will in fact receive a score of 1.0 for the function. However, the FCI models are not fitted to actual data in many guidebooks. When they are fitted to real data, the wetlands chosen often do not represent the full range of natural variability and anthropogenic disturbance. In this guidebook, we consider the difference between *a priori* FCI scores and fitted FCI scores to be error in the statistical model. We attempted to minimize these errors using standard statistical techniques, but in most cases we chose not to simplify the models to the extent necessary to completely eliminate them. Thus, application of this guidebook is not expected to yield the maximum score of FCI = 1.0 for all undisturbed wetlands in all cases.

2. A deeper philosophical issue arises with regard to the definition of "function." Assigning a value of 1.0 to all functions for all reference standard wetlands implies that the functional capacity is not being estimated on an absolute scale. Consider a shallow vernal pool at the headwaters of a pool network, with a relatively small catchment area. Even in an unaltered landscape, such a pool may only fill during the wettest years. In a typical year, it may store no surface water and very little subsurface water. What *a priori* value should it be assigned for the function "Surface and Sub-surface Water Storage"? There are four options:

a. Assign an undisturbed headwater pool an *a priori* FCI < 1.0 for water storage because, despite its pristine nature, it stores very little water in absolute terms. However, it may be assigned a score of 1.0 for other functions. Redefine the term "reference standard" in a manner that departs from other guidebooks (Table 2.3), and include this pool as a reference standard.

b. Do not classify it as a reference standard, based on the traditional reference standard definition. We suspect that most or all other HGM guidebooks have taken this approach, and omitted undisturbed wetlands from the reference standard class if their hydrology, biota or other attributes are atypical or depauperate.

c. Such a pool could be included as a reference standard and assigned an *a priori* FCI of 1.0 for water storage. This makes model fitting difficult, with at least three options:

i. The FCI can be fitted using a simple model based on *absolute* amount of water stored. However, for undisturbed headwater pools to receive a value of 1.0, the model would need to be so lenient that nearly all pools would receive a value of 1.0.

ii. The FCI can be fitted using a complex and more realistic model based on *absolute* amount of water stored. Reference standard headwater pools would almost certainly receive very low scores in the fitted model, leading to unacceptably high model error.

iii. A preferred route may be to fit the FCI using a realistic model based on the amount of water stored *relative to important covariates*. Examples of such covariates would include maximum depth, landscape position and underlying soil type. In the above example, the fitted FCI would perhaps have different scales for shallow vs. deep pools, or headwater vs. terminal pools in a network. Field application of the model to new pools would not assess "How much water is stored", but rather "How much water is stored, relative to reference standard pools with the same depth and landscape position."

If one accepts the premise that "function" should be estimated in relative terms, rather than absolute, option c) iii) is clearly defensible. However, accurate model fitting would require collecting data from multiple reference pools across the full spectrum of covariates. It would likely require data from many years that span the full range of precipitation events. As mentioned above, we suspect that previous HGM wetland guidebooks have focused exclusively on pools that have typical values for covariates such as hydroperiod, landscape position and depth, excluding those that have more extreme values (Gilbert *et al*. 2006).

d. One could designate shallow headwater pools as unscorable for the water storage function based on insufficient data for model calibration. They could still be retained as reference standards if all other functions that can be scored are given an *a priori* FCI of 1.0. If new data are gathered in the future, the function could be revised using one of the other options.

In this guidebook, we generally strived to incorporate approach c) iii), and opted for approach d) in cases where insufficient field data were available for accurate model calibration.

Assessment Models and Functional Indices

In the HGM approach, an assessment model is a simple representation of a function performed by a wetland ecosystem. It defines the relationship between one or more characteristics or processes of the wetland ecosystem. Functional capacity is simply the ability of a wetland to perform a function compared to the level of performance in reference standard wetlands.

Model variables

Model variables represent the characteristics of the wetland ecosystem and surrounding landscape that influence the capacity of a wetland ecosystem to perform a function. Model variables are ecological quantities that consist of five components (Schneider 1994): (a) a name,

(b) a symbol, (c) a measure of the variable and procedural statements for quantifying or qualifying the measure directly or calculating it from other measures, (d) a set of values (*i.e.*, numbers, categories, or numerical estimates: Leibowitz and Hyman 1997) that are generated by applying the procedural statement, and (e) units on the appropriate measurement scale. Table 2.4 provides several examples.

Table 2.4. Components of a Model Variable

Name (Symbol)	Measure/Procedural Statement	Resulting Values	Units
Basin Depth ($V_{MAXDEPTH}$)	The maximum depth of the pool, as estimated with surveying equipment.	>0	meters
Inlet Modification ($V_{INLETMOD}$)	Discernible modification to the inlet.	0 = no 1 = yes	unitless
Coverage of Basin with Cobbles ($V_{COBBLESBA}$)	The percent cover of the basin surface with angular coarse pebbles or cobbles, as defined in the 1993 USDA Soil Survey Manual.	0 to 100	percent, written as a whole number

Model variables occur in a variety of states or conditions in reference wetlands. For example, percent herbaceous groundcover could be large or small. Based on its condition (*i.e.*, value of the metric), model variables are usually assigned a variable subindex by rescaling them. A variable subindex of 1.0 is often assigned when the condition of a variable is within the range exhibited by reference standard wetlands. As the condition declines from that found in reference standard wetlands, the variable subindex is assigned based on the relationship between model variable condition and functional capacity. In most cases, the rescaling of variables into variable subindices is based on pertinent literature, personal expertise and experience and information from reference wetlands (Smith *et al.* 1995). Lower subindex values reflect decreasing contributions to functional capacity, relative to reference standard wetlands. In some cases, the variable subindex can drop to zero. The rationale for intermediate subindex scores is generally less well defined, although a linear relationship is usually assumed between the original variable's value and the subindex value. For this guidebook, we only assigned subindex scores to the Hydrologic Networks function. For the other three functions for which data were collected, we used statistical analyses to relate the relative contributions each variable made to the function via equation coefficients (see below and Chapter 5). For these functions, the model fitting was accomplished without first scaling each variable to a maximum of 1.0. In the end, the difference between the two approaches is not important, since the score assigned to a particular pool would be the same either way.

In the HGM approach, model variables are combined into an assessment model to produce a Functional Capacity Index (FCI) that ranges from 0.0 to 1.0. Within each function, the variables are usually combined as a simple average. However, we used a statistical model for most functions, in which the coefficient for each variable is derived from a multiple regression or general linear model. The FCI is a measure of the functional capacity of a wetland relative to reference standard wetlands in the reference domain. Wetlands with an FCI of 1.0 perform the function at a level characteristic of reference standard wetlands. As the FCI decreases, it indicates that the capacity of the wetland to perform the function is less than that of reference standard wetlands. In some cases, the FCI may be based on model variables that directly relate to the function of the variable, and can only be assessed under specific field conditions (*e.g*., when the pool is holding water). In this guidebook, we refer to these as Direct FCIs. Alternatively, the FCI may be based on variables that can be measured at any time of the year, correlate well with the level of function, but are *not causally related* to the function. We refer to these as Indirect FCIs.

Conceptual Framework for Computing Direct and Indirect FCIs Using Graphical and Statistical Analyses

For all but the Hydrologic Networks function and Biogeochemical Processes function, we employed both exploratory and formal graphical and statistical analyses to determine how single variables and groups of variables relate to the function. Details of our approach are provided in Chapter 5 in the section titled “Analytical Techniques and Procedures.” We included interactions among variables when they emerged from the analysis and could be explained by known processes. We searched for threshold effects and other nonlinear relationships between the variables and the level of function. Ultimately, we discarded many variables that did not have explanatory power both empirically and logically.

As a first step in developing an FCI based on direct measures of function (*i.e.,* a Direct FCI), we developed guidelines for assigning an *a priori* FCI to each pool. The *a priori* FCI generally describes the overall level of function for a vernal pool based on best expert opinion. The subset of pools deemed to be reference standards (i.e., the most functional representations of natural vernal pools) received an *a priori* FCI of 1.0 for all functions (see extended discussion in Reference Wetlands above). However, we neither expected nor enforced the assumption that non-reference standard pools should have identical FCI scores for all functions. For example, disturbances that severely alter the hydrology of a particular vernal pool may have less impact on its fauna than its water storage capacity. To maintain objectivity, we developed verbal definitions for seven different FCI values ranging between 0.0 and 1.0 (see Appendix D.6). FCI guidelines similar to those in Appendix D.6 have not been made explicit in any other HGM guidebook.

The targets for application of this HGM guidebook are all vernal pools within the reference area, rather than those that provide the maximum *absolute* level of functionality for all functions. Therefore, we addressed the full array of variability encountered in the field, including pools that pond rarely because they are high in the landscape, have small contributing watersheds and/or are shallow. Whenever possible, consideration of these attributes has been included within the Direct and Indirect FCIs. For example, the FCIs for the faunal community cannot be scored for very shallow pools (< 0.07 m) due to an absence of data for calibration, and they include different criteria for moderately shallow pools (0.07 m ≤ max. depth < 0.15 m) vs. deep pools (max. depth ≥ 0.15 m). In all respects, the development of the statistical models (the Direct FCI and the Indirect FCI) was heavily weighted on the reference standard pools and those pools that were the least functional. Scores for the Direct and Indirect FCIs were constrained to be as close as possible to the *a priori* FCI scores for the reference standards (where *a priori* FCI = 1.0) and the least functional pools (where *a priori* FCI ≤ 0.25). This follows the general approach of previous guidebooks, where the intent is to base initial development of the statistical FCI model on the "best" and "worst" pools in the reference area.

For all functions, a general linear model was used to predict the Direct FCI (dependent variable) from a linear combination of categorical and/or continuous variables that clearly relate to the specific function. We would characterize the exploratory data analysis used to arrive at this general linear model as extremely thorough. For each of the functions, all univariate relationships between the field data and the *a priori* FCI were examined both graphically and statistically, and scores of alternative multivariate models were evaluated and compared. After arriving at a single statistical model *(i.e.*, a preliminary Direct FCI), we then validated and calibrated it on the full set of pools that had been sampled for that function (see Table 5.8 for sample sizes).

To accommodate variables that can be measured in the field at any time of year (even when pools are dry), Indirect FCIs were also developed for each function. We calibrated each Indirect FCI on its corresponding Direct FCI using all available pools. Similar to development of the Direct FCIs, we derived Indirect FCIs using exploratory data analysis, examination of all univariate relationships and analysis of a very large number of general linear models with multiple dependent variables. However, development of the Indirect FCI differed in three fundamental ways. First, the dependent variable in the Indirect FCI analyses was the final calibrated Direct FCI. Second, all pools for which field data had been taken were used in the Indirect FCI derivation. (In contrast, the Direct FCI was developed using the *a priori* FCI as the dependent variable and only the most and least functional pools in the first steps.) Third, the set of possible independent variables in the Indirect FCI statistical models was restricted to those that can be measured at any time during the year. (The Direct FCI targeted independent variables that were causally related to the function, even if they could only be measured when a vernal pool is in its wet phase.) By design, the Indirect FCI is a more rapid and convenient way to assess pool function than the Direct FCI, and the Indirect FCI may be calculated at any time of year. However, this convenience comes at the cost of reduced accuracy.

Assessment protocols

The final component of the HGM approach is the assessment protocol. The assessment protocol is a series of tasks, along with specific instructions, that allow the end user to assess the functions of a particular wetland area using the functional indices in the Regional Guidebook. The first task is characterization, which involves describing the wetland ecosystem and the surrounding landscape, describing the proposed project and its potential impacts, and identifying the wetland areas to be assessed. The second task is collecting the data for model variables. The final task is analysis, which involves calculation of functional indices. Chapter 5 provides detailed instructions for site characterization and data collection necessary for development of Direct and Indirect FCIs.

Development Phase

An interdisciplinary team of experts known as the "Assessment Team," or "A Team" ideally carries out the Development Phase of the HGM approach. A team of 5-8 individuals is recommended as sufficiently large to represent critical disciplines and not too large as to be unwieldy (Smith *et al.* 1995). The following disciplines have been recommended for representation on the "A Team": wetland ecology, geomorphology, biogeochemistry, hydrology, soil science, plant ecology and animal ecology.

The product of the Development Phase is a Regional Guidebook for assessing the functions of a specific regional wetland subclass (Figure 2.1). In developing a Regional Guidebook, the A-Team completes the following major tasks. After organization and training, the first task of the A-Team is to classify the wetlands within the region of interest into regional wetland subclasses using the principles and criteria of the HGM Classification (Brinson1993, Smith *et al.* 1995). Next, focusing on the specific regional wetland subclasses selected, the A-Team develops an ecological characterization or functional profile of the subclass. The A-Team then identifies the important wetland functions, conceptualizes assessment models, identifies model variables to represent the characteristics and processes that influence each function, and defines metrics for quantifying model variables. Next, reference wetlands are identified to represent the range of variability exhibited by the regional subclass. Field data are then collected from the reference wetlands and used to calibrate model variables and verify the conceptual assessment models. Finally, the A-Team develops the assessment protocols necessary for regulators, managers, consultants and other end users to apply the indices to the assessment of wetland functions.

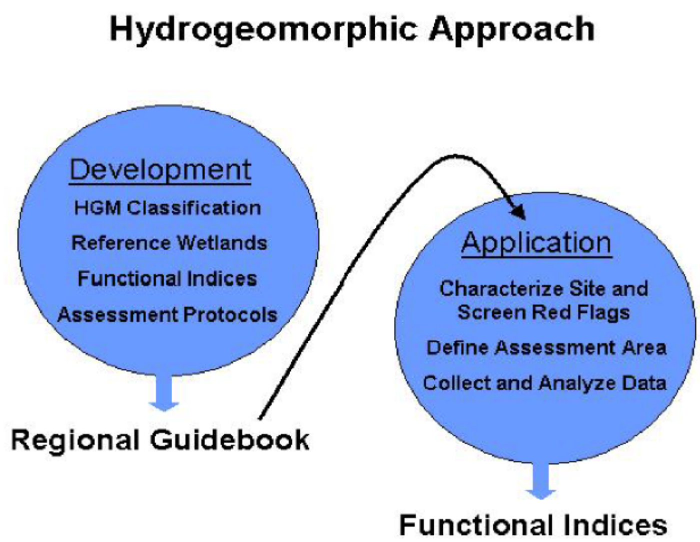

Figure 2.1. Development and application phases of the HGM approach.

The following list provides the detailed steps involved in this general sequence:

Task 1: Organize the A-Team.

a. Identify A-Team members.

b. Train A-Team in the HGM approach.

Task 2: Select and Characterize Regional Wetland Subclasses.

a. Identify/prioritize wetland subclasses.

b. Select regional wetland subclasses and define reference domain.

c. Initiate literature review.

d. Develop preliminary characterization of regional wetland subclasses.

Task 3: Select Model Variables and Metrics and Construct Conceptual Assessment Models.

a. Review existing assessment models.

b. Identify model variables and metrics.

c. Define initial relationship between model variables and functional capacity.

d. Construct conceptual assessment models for deriving FCIs.

e. Complete Precalibrated Draft Regional Guidebook (PDRG).

Task 4: Identify and Collect Data from Reference Wetlands.

a. Identify reference wetland field sites.

b. Collect data from reference wetland field sites.

c. Analyze reference wetland data.

Task 5: Calibrate and Field Test Assessment Models.

a. Calibrate model variables using reference wetland data.

b. Verify and validate (optional) assessment models.

c. Field test assessment models for repeatability and accuracy.

d. Revise PDRG based on calibration, verification, validation (optional), and field-testing results into a Calibrated Draft Regional Guidebook (CDRG).

Task 6: Conduct Peer Review and Field Test of CDRG.

a. Distribute CDRG to peer reviewers.

b. Field test CDRG.

c. Revise CDRG to reflect peer review and field test recommendations.

d. Distribute CDRG to peer reviewers for final comment on revisions.

e. Incorporate peer reviewers' final comments on revisions.

f. Publish Operational Draft Regional Guidebook (ODRG).

Task 7: Technology Transfer.

a. Train end users in the use of the ODRG.

b. Provide continuing technical assistance to end users of the ODRG.

Application Phase

The Application Phase involves two steps. The first is to use the assessment protocols outlined in the Regional Guidebook to carry out the following tasks (Figure 2.1).

a. Define assessment objectives.

b. Characterize the project site.

c. Screen for red flags.

d. Define the Wetland Assessment Area.

e. Collect field data.

f. Analyze field data.

The second step involves applying the results of the assessment (*i.e.*, the FCIs), to the appropriate decision-making process. Although the HGM approach was originally conceived for use in a regulatory context as part of Section 404 of the Clean Water Act, it has a variety of other potential applications. For instance, the HGM assessment models for southern Californian vernal pools were developed primarily for use in ecosystem restoration and preserve management, within an overall planning context. There are several ways in which the HGM approach can be applied as part of an overall planning framework. For instance, in analysis of alternative plans, it can be used to measure variable impacts to existing wetlands, or locate and evaluate potential restoration sites. Because the HGM approach produces numerical values as a measure of various wetland functions, these numbers can be used to quantify and compare impacts and benefits to wetlands due to various alternative proposed plans and actions. It can similarly be used to evaluate the effectiveness of management practices and suggest corrective actions.