James T. Anderson
Craig A. Davis *Editors*

Wetland Techniques

Volume 1
Foundations

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Preface

Wetlands are generically defined as lentic systems that take on characteristics of both terrestrial and aquatic systems where vegetation capable of growing in shallow water proliferates. However, there are many definitions of wetlands in use around the world, including a number that have ecological and legal significance. Even among these definitions, there are numerous subtle nuances that blur the lines between wetlands and either terrestrial or aquatic systems. Despite the confusion and oftentimes contradictory nature of wetland definitions, wetlands are increasingly being recognized as critical ecosystems throughout the world. In particular, we are seeing an increased awareness about the values and benefits derived from the world's wetlands. As this awareness has grown, we have also seen a greater focus on efforts to better manage, conserve, and protect wetlands. Wetland-related research has been and will continue to be critically important in providing guidance to all the efforts to better manage, conserve, and protect wetlands. In fact, there is a plethora of wetland-related literature available to wetland scientists, regulators, and managers, many of which can be found in at least two journals that are dedicated exclusively to wetlands. However, for most wetland professionals, it may be a daunting task to access much of this literature. Additionally, wetland professionals have not had a book available that covers techniques associated with wetland research, management, and regulation.

The lack of such a book has been a major void in the wetland field. In fact, wetland professionals have discussed for some time the need for a book that focused on wetland research and management techniques. We believe the development of a techniques book for a profession is a sign that the profession, in this case wetland science, is maturing. Scientific progress in a field is often advanced by the development of a techniques book because almost all studies and management actions boil down to choosing appropriate techniques, and a book focused on the topic of wetland techniques will provide fledgling scientists and managers a solid foundation for initiating research and management efforts. We have designed this

three volume set for students and professionals interested in wetlands ecology, management, and creation. We are pleased to be a part of the development and progression of our discipline through our involvement with the development of *Wetland Techniques Volume 1: Foundations*, *Volume 2: Organisms*, and *Volume 3: Applications and Management*.

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Acknowledgments

Wetland Techniques is our first attempt at a major book project and it was a wonderful learning opportunity as well as an eye-opening experience in regards to all the effort that goes into creating a series of books of this magnitude. We have new-found admiration for all those before us that have successfully tackled book projects for the benefit of science.

We thank the chapter authors for providing freely of their time and expertise. It has been a pleasure working with the authors and we have learned a lot more about wetlands because of them. We thank all of the chapter referees for giving their time and expertise to improve the quality of this three volume *Wetland Techniques* set through constructive reviews that greatly improved the chapters. We especially thank Rachel Hager, undergraduate student in Wildlife and Fisheries Resources at West Virginia University, for all of her help in formatting and verifying literature citations and performing numerous other tasks to improve the book. We also thank Roseanne Kuzmic, research associate in the Natural Resource Ecology and Management Department at Oklahoma State University, for assistance with verifying literature citations.

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Chapter 1
Study Design and Logistics

David A. Haukos

Abstract Reliable knowledge is critical for management and conservation of wetlands. Essential to the scientific method and achieving reliable knowledge is study design. The primary purpose of study design is the collection of data in an unbiased and precise manner for an accurate representation of a population. Proper study design includes formulation of study questions and objectives, hypotheses to explain an observed pattern or process, conceptual models, appropriate methodology, and a data management plan. Inference of study results and conclusions can be explicitly bounded by defining an appropriate target population. Deductive, Inductive, and Retroductive reasoning are used to infer study results to target populations. Development of multiple competing hypotheses capable of being tested is at the core of the hypothetico-deductive approach that maximizes potential knowledge from a study. Selection of independent and dependent variables to test hypotheses should be done with cost, efficiency, and understanding of the wetland system being studied. Study type (e.g., experimental, observational, and assessment) influences the certainty of results. Randomization and replication are the foundation of any study type. In wetlands, impact studies (e.g., BACI [before-after/control-impact] design) are common and usually follow unforeseen events (e.g., hurricanes, wild fire, floods). Sampling design is dictated by study objectives, target population, and defined study area. A robust sampling effort is essential for accurate data. Reduction in statistical and mechanical errors and data management protocols are overlooked features of study design. In addition to statistical tests, estimation of the magnitude (i.e., effect size) of an effect is crucial to interpretation of study results. When judging the merits of results from a study, investigators should independently assess the hypothesis,

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methodology, study design, statistical approach, and conclusions without regard to how they would have conducted the study. Doing so will facilitate the scientific process.

1.1 Introduction

Unfortunately, the history of wetland science is relatively brief. In the United States, most scientific effort prior to the early 1970s was devoted to justifying draining and filling of wetlands. Information from such study results contributed to the greater than 50 % decline of wetlands in the conterminous United States since European settlement (Lewis 2001; Dahl 2011). It is unlikely that any other ecosystem has suffered such organized willful efforts of alteration, destruction, and obliteration based primarily on misinformation and spurious "facts" than wetlands. Slowly during the past century, acceptance of wetlands as critical components of the natural world has resulted in a multitude of conservation and education efforts to protect wetland ecosystems. One of the rare historical exceptions was the creation of National Wildlife Refuges to protect wetlands vital to migratory birds, primarily waterfowl because of their value to hunters. Since the passage of the Clean Water Act and other legislation since the early 1970s, the ecological values of wetlands have increasingly been recognized by conservation organizations, policy makers, governmental agencies, and society at large. The foundation for these changes in societal values and policies from those factions advocating wetland destruction to a predominance of activities proposed for restoration, enhancement, and protection of wetlands is reliable knowledge of the ecological structure, function, and provision of services by these systems.

Reliable knowledge is the result of accumulation of credible results from wetland investigations conducted using a logical framework – **study design** (Table 1.1). Study design involves more than **experimental design**, which can be defined as a plan for assigning experimental conditions to subjects and the statistical analyses appropriate for the plan (Kirk 1982). Study design is also more than **statistics**, which is a body of knowledge that allows one to make sense of collected data and generalize results from a sample to a population. Both experimental design and statistics are beyond the scope of this chapter; there are a considerable number of available texts on both subjects (e.g., Quinn and Keough 2002; Box et al. 2005; Montgomery 2012). Proper use of study design allows for the development of research goals, objectives, and hypotheses based on observations, previous studies, and ecological theory. Study design includes a declaration of variables to be measured, techniques to be applied, and approaches to analyses of collected data. Furthermore, use of the appropriate study design allows for inference beyond the immediate subjects being studied. Most importantly, this framework allows for acceptance of study results into the overall knowledge of wetland ecosystems for use in conservation efforts, generation of additional research questions, and accumulation of defensible reliable knowledge regarding wetland ecosystems.

Table 1.1 Glossary of common terms used in study design

Accurate	Having low bias and variance; where resulting estimates are repeatable and close to the true value of a population
Alternative hypotheses	Alternative explanations for an observed pattern or process that are usually represented in competing statistical or predictive models
Bias	Difference between long-term average of a sample estimate from true population value
Conceptual models	Abstractions of reality based on observation of an ecological pattern or process envisioned by an investigator but not typically formalized graphically or mathematically
Control	Group of experimental units for which the factor of interest is excluded or otherwise accounted for in study design
Descriptive inference	Using observations from a study to learn about or predict other unobserved facts
Deterministic	Completely predictable, not involving any random components
Effect size	Magnitude of a measurable effect due to a treatment of interest
Empirical models	Models in which data are used to estimate parameters or test predictions
Experiment	A process that imposes a **treatment** on a group of elements or subjects (experimental units) to measure a response and quantify an effect
Experimental design	A plan for assigning experimental conditions to subjects and experimental units
Experimental error	The inherent variation among experimental units treated alike or variation not explained by treatments or other variables
Experimental unit	Subjects (elements) to which individual experimental treatments are applied
Fixed effect	A variable in which levels are not subject to random variation under repetition of the experiment
Fundamental objective	What a decision maker wants to accomplish
Hypothesis	Specific statement of reality that is frequently testable by comparing predictions to data
Independence	Organisms, samples, experimental units, or other objects that can be represented by a statistical distribution one at a time, without dependence on the values of other objects
Independent variables	Those variables hypothesized (including treatments) to contribute to variation in **dependent** or **response variables**, the values of which depend on levels or types of independent variables
Means objective	Intermediate objectives that must be accomplished to achieve or address a fundamental objective
Metapopulation	When the target population of interest is subdivided into discrete patches across a landscape but movement among patches continues
Model	An abstraction or perceived representation of nature
Objectives	Statements of desired achievements by investigators and decision makers
Random effect	Where repetition of the experiment will result in different levels within the analyses unless the same experimental units are used
Randomization	Assignment of treatments to or section of experimental or sampling units at random
Replication	Assignment or selection of multiple experimental units to an individual treatment

(continued)

Table 1.1 (continued)

Sampling	Selection of a subset of potential experimental units from the target population for measurement of variables of interest
Sampling error	The variation among samples (or observations) of a given experimental unit
Sampled population	Population from which samples are taken
Statistical inference	The process of drawing sound and appropriate conclusions from data subject to random variation
Target population	Population for which inference can be made
Theory	A broad, general conjecture about a process that can be tested using study design
Treatment	Something that an investigator imposes on experimental units in some deliberate manner
Unbiased	Long-term average of sample estimates equals population value

A principal goal of study design is to minimize personal bias, values, beliefs, and subjectivity of the scientist so that conclusions can be supported beyond a reasonable doubt. Basic tenets of modern study design are grounded in statistical theory and have been applied for >75 years (Fisher 1935). Exponential increases in computational ability during the past 25 years have allowed for increasingly complex approaches to study design and data analyses. However, failure to adhere to basic components of study design cannot be overcome even with the most complex analytical tools. A well-designed study will allow investigators to focus on current knowledge gaps, provide rigorous tests of information, and enhance efficient use of resources.

1.2 Role of Study Design in the Scientific Method

Study design is a critical part of the scientific method (Fig. 1.1). There are several variations of the scientific method for studying ecological systems, but all include steps of (1) construction of question(s) that address uncertainties in the ecosystem of interest, (2) formation of theories to explain observations or questions based on observation, which leads to multiple hypotheses that have predictions suitable for testing data, (3) design of a study to test primary and alternative hypotheses and their associated predictions, (4) collection and analyses of data, (5) report conclusions and make inference from results, and (6) communicate results through the peer-review process that adds credibility to the findings (Gauch 2003). Typically, the scientific method is referred to as a process because all studies and resultant conclusions are subject to being repeated, typically as a feed-back loop restarting with step (2) by other scientists. However, it would be a mistake for anyone to perceive the scientific method as a defined sequence of steps leading to knowledge, but rather as a framework for creative and productive processes that can be used to accumulate knowledge that leads to truths corresponding with reality of

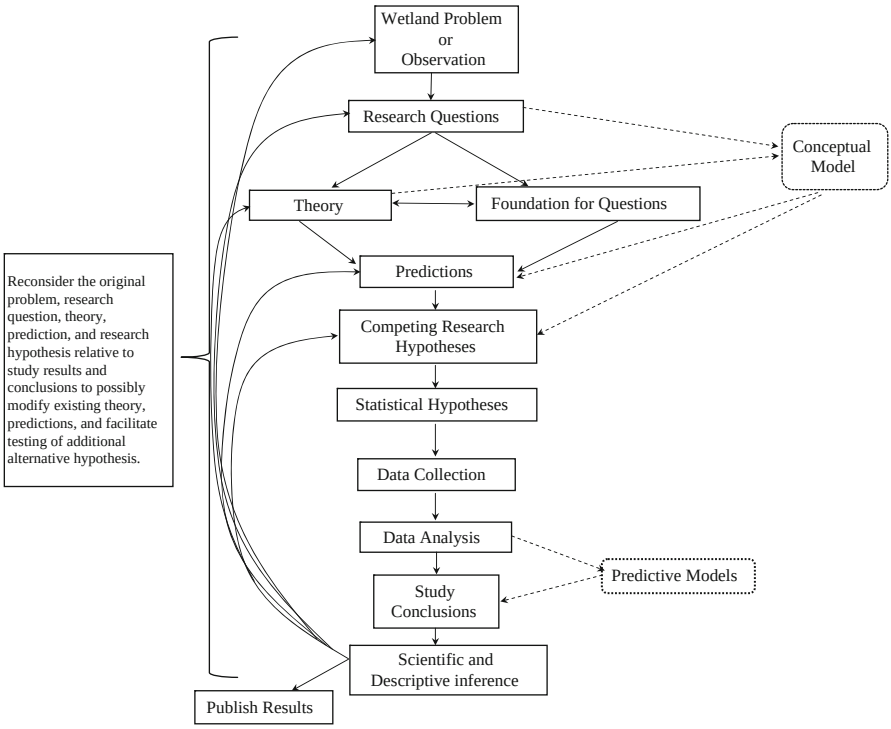

Fig. 1.1 The process of the scientific method to achieve reliable ecological knowledge (Modified from Brown and Guy (2007). Published with kind permission of © American Fisheries Society 2007. All Rights Reserved)

natural systems. Because science is a process, knowledge of natural systems, including wetlands, evolves over time as previous conclusions are subjected to further study, and occasionally rejected due to lack of continued empirical support.

Study design expands on the scientific method and has been presented in an outline or checklist format (Morrison et al. 2001). Although there are multiple texts with step-by-step guides for developing study designs (e.g., Cochran 1983; Cook and Stubbendieck 1986; Martin and Bateson 1993; Lehner 1996), Morrison et al. (2001) characterized the process of study design by 12 discrete steps – (1) question development relative to a natural system, (2) development of >1 hypothesis that may answer the question of interest (testing of competing hypothesis) and associated predictions should the hypothesis be true, (3) determination of a conceptual study design, (4) selection of independent and dependent variables, (5) choose appropriate methods to measure variables of interest, (6) establish acceptable level of precision and accuracy, (7) pilot study or preliminary data collection to test limitations of original design and estimate sample size requirements, (8) establish quality/quantity assurance protocols, (9) conduct data collection, (10) perform data analyses and partition

evidence in support of competing hypotheses, (11) interpret and provide context of results and make appropriate inference of results and conclusions, and (12) publication of results in a peer-reviewed outlet. Individual investigators can insert the details specific to their study within these broad steps to initially design a study.

Without study design, the scientific method would be ineffective as a means to attain reliable knowledge. Investigators must be aware that study design, as well as experimental design, statistical analyses, and data modeling, should not be considered a "cookbook" process devoid of critical thought. Once a wetland-related question has been formulated and competing hypotheses developed, there are usually a number of different approaches to appropriately design a study that can be used to accumulate evidence to test one (or more) hypotheses. Furthermore, development of a study design must be considered in the context of accessibility of study sites, equipment and labor costs, time to collect samples, other time-sensitive constraints (e.g., lab availability, sample storage, and occurrence of measurable dependent variables), and a multitude of other potential considerations. In addition, it is important to realize that it common for changes in the design to occur after a study has been implemented. Hopefully, if one is unfamiliar with the wetland type or study question, limitations of or changes to a study design can be addressed using prior knowledge (e.g., experience of the investigator, literature or previously collected data). Without such insight, the initial study effort is typically defined by determining the limitations of the proposed design and, at times, resulting in a potentially unreliable data set. As these issues may compromise acquisition of reliable knowledge, it is recommended that any proposed study design be reviewed by a biometrician or statistician familiar with the inherent quantitative hurdles associated with natural systems and an investigator familiar with the wetland type or issue proposed for study.

1.3 Development of a Study Hypothesis

Reliance on ecological theory and generation of hypotheses are basic to study design. A **theory** as defined for the purpose of study design is a "broad, general conjecture about a process" (Romesburg 1981: 295) or "a conceptual framework through which the world is observed and facts about the world are discerned" (Williams 1997: 1009). A **hypothesis** is a proposed explanation for a process or phenomenon that creates data. Once an intangible theory has been developed relative to an observed phenomenon, hypotheses can be formulated to describe the natural processes that produced the phenomenon that lead to predictions of outcomes should the hypotheses be true. Development of hypotheses is the key step for a successful study design. Without sound, well-developed, hypotheses, it is difficult to conduct an investigation that will result in clear conclusions and advance our understanding of ecological systems.

Hypotheses follow the question being asked and must be explicitly stated so as to provide predictive power or specific conditions under which the hypothesis is true.

Typically, development of a study question and associated hypotheses is considered a basic and simple task that is hurriedly stated prior to designing a study. In practice, a successful study depends on a thoroughly developed question and sound, conceptual hypotheses that can be represented by competing models and tested with data (e.g., Anderson 2008). However, lacking competing quantitative models, hypotheses can be formulated as existential statements, which include an expression that existence of a phenomenon has identifiable characteristics and causal explanations that exist for each occurrence of the phenomenon. All subsequent steps in a study must refer back to the question and its associated hypotheses, which require prioritization and agreement by everyone involved in the study (including funding sources). Questions can be developed from personal experience, expert opinion, literature, intuition, and guesswork but are usually driven by stated goals.

For complex natural resource issues, Structured Decision Making and Adaptive Resource Management have recently been adopted by many governmental agencies; the approach is focused on developing and prioritizing questions related to a natural resource issue (e.g., Martin et al. 2009). Questions or study objectives under this approach are categorized as fundamental (explicit declarations of core concerns or questions), means (typically methodological and represent an intermediary step in reaching the fundamental objective), process (ground rules for decision processes related to the study), and strategic (fundamental to a broader set of decisions than the one in question) objectives and require considerable effort to distinguish among these (e.g., Keeney 2007; Williams 2012). For example, there may be a controversial issue relative to mitigation of a wetland that is slated to be lost to development. There are likely a number of stakeholders (i.e., developers, local government, natural resources agencies, non-governmental organizations) that have competing views on the type, location, and magnitude of mitigation. Use of the Structured Decision Making Approach facilitates the decision-making process by involving all stakeholders in (1) identifying the problem to be addressed, (2) specifying objectives and tradeoffs, (3) identifying the range of potential decisions, (4) specifying assumptions about resource structures and functions, (5) projecting the consequences of alternative actions, and (6) identifying key uncertainties among other steps (Williams and Brown 2012). This approach explicitly addresses decision uncertainty, which is the typical roadblock preventing sound decisions based on scientific knowledge. One should recognize that uncertainty represents incomplete knowledge, not doubt, when addressing specific wetland issues. It is more desirable to design studies to address fundamental objectives (questions) rather than the other objective types. No single study can address every question, and, if attempted, usually results in mediocrity and an inefficient use of resources.

There are several types of reasoning that can be used to obtain knowledge through generation of theories and hypotheses (Morrison et al. 2001). One can use **induction** to create general conclusions based on a collection of individual facts (i.e., conclusions are drawn based on an association between individual facts); frequently an extrapolation of results from a study to a general situation. As an example of inductive reasoning, an investigator has concluded that the development

of terrestrial buffer strips of 100 m in width filters 90 % of suspended material present in precipitation run-off; therefore, creating buffer strips around wetlands will minimize or eliminate the potential of a wetland filling due to accumulated terrestrial sediments. **Deduction** is reasoning from the general (premise) to a specific event and includes the development of testable explicit predictions under a specific hypothesis to explain observations from a natural system. An example of deductive reasoning would be if avian body mass varied in relation to environmental conditions, then an investigator may predict that average body mass would decline as temperature declines or number of days below freezing increases. The use of **retroduction** involves the relatively subjective attribution of an underlying cause to an observed pattern and is a common occurrence in the discussion section of scientific papers. Conclusions based on retroduction should be considered hypotheses that require additional testing. Most conclusions from scientific studies are the result of retroduction and induction, which Romesburg (1981) lamented does not result in gaining reliable knowledge as one would when using deduction. This **hypothetico-deductive** approach was advocated by Romesburg (1981) as the preferred approach for study of natural systems. Guthery (2008) described the approach as the classical method to test a hypothesis by deducing events or relationships that should be observed under experimentation if the hypothesis was true. Specific, sound hypotheses result in efficient, organized, and goal-oriented studies that minimize uncertainty in results.

Most commonly, hypotheses are stated in terms of treatment effects. Historically, treatment effects were based on potential outcomes of statistical tests. Classical statistical, "null" hypotheses are usually depicted in shorthand as H_o: and generally specify that "no difference" exists among treatments. Whereas the "alternative" hypothesis H_1: can be more specific and takes several forms related to the stated hypotheses but usually is a statement that a difference exists among treatments. Alternative hypotheses may also define a magnitude and direction of a difference. It is important to remember that studies can reject or disprove a null hypothesis, but a hypothesis cannot be considered proven when a null hypothesis is not rejected. For example, a wetland manager can hypothesize that cattails (*Typha* spp.) will be completely eradicated from a wetland due to herbicide treatment, but the presence of a single cattail would cause the hypothesis to be rejected. For the hypothesis to be "accepted," every plant would have to be identified to prove the null hypothesis and it is impossible to be completely certain that all plants have been correctly identified. Therefore, basic to scientific endeavor is the use of study design to formalize the effort to disprove hypotheses rather than prove them (Peirce 1958).

Recently, use of the null statistical hypotheses has been dismissed as uninformative and nonproductive in scientific endeavors because use of a statistical test that only declares whether a difference exists or not exists between treatment means is relatively uninformative (e.g., Johnson 1999; Cherry 1999; Anderson 2008; Guthery 2008). Instead, multiple alternative hypotheses should be developed that are specific and depict cause and effect, measurable predictions, or explicit outcomes that can be tested using data (Platt 1964; Romesburg 1981; Morrison et al. 2001). In a study design, this can be relative to study objectives as long as one

formalizes the question that is being addressed. Popper (1959) and Platt (1964) indicated that science progresses best when hypotheses of natural systems are evaluated empirically by comparing to predicted results to reject hypotheses that are inconsistent with predictions. Further, Anderson (2008) advocated that all plausible alternative hypotheses should be translatable into mathematical models that are subjected to empirical methods to test the relative strength of the evidence for each hypothesis. Chamberlain (1890) urged scientists to conduct studies using the strategy of "multiple working hypotheses". That is, study design should be capable of simultaneously testing multiple plausible hypotheses, eliminating poor hypotheses, and quantifying the relative strength of one hypothesis over the alternatives (Royall 1997). Ultimately, conclusive evidence for a hypothesis (i.e., science answer) can only be possible after all other hypotheses are rejected through study design or additional studies (Williams 1997). Development of hypotheses is a time-consuming, challenging process that is critical to overall rigor of a study. Sound hypotheses are based on (1) familiarity of the system being studied, (2) detailed formulation of the question or observation being studied, and (3) working knowledge of the established literature related to the subject being studied.

In wetland science, development of hypotheses is usually study specific. Hypotheses can range among general statements that explain an observation, specific measureable predictions should a hypothesis be true, directionality of a treatment effect, or support for an ecological theory. Turner (1997) proposed four hypotheses to explain the observation of a high rate of coastal wetland loss in the northern Gulf of Mexico (−0.86 %/year): (1) an extensive dredge canal and spoil bank network; (2) decline in sediments in the Mississippi River during the 1950s; (3) Mississippi River navigation and flood protection levees; and (4) salinity changes. The hypotheses were developed following extensive consideration of all potential factors influencing wetland loss and familiarity with the wetland system being studied. A study was designed to address predictions from each hypothesis. Turner (1997) concluded that, based on his study, dredging man-made channels and forming dredge spoil banks had the greatest impact on wetland hydrology and had the most influence in explaining wetland loss.

Testing ecological theory among ecosystems requires testable predictions for each competing hypotheses. Megonigal et al. (1997) tested two competing hypotheses under the subsidy-stress hypothesis for rate of aboveground net primary production in southeastern floodplain forest. Under the subsidy-stress hypothesis, they hypothesized that periodically flooded forests have higher rates of net primary productivity than upland or continuously flooded forests. As a competing hypothesis, they proposed that effects of periodic inputs of nutrients and water on net primary productivity are diminished or offset by stresses associated with anaerobic soils or drought. Using an experimental field study design, they measured aboveground net primary productivity under three categories of mean growing-season water depth. Megonigal et al. (1997) concluded that extensive flooding caused significant stress on forest productivity, but there was insufficient support for the subsidy-stress hypothesis in the description of patterns of net primary productivity

in flooded forests and suggested testing a more complex interaction between subsidy and stress factors.

Use of predictive hypotheses led Collins and Storfer (2003) to categorize six hypotheses potentially explaining global amphibian declines into two classes. Class I hypotheses were those in which underlying ecological mechanisms affecting amphibians were well known, but the relative magnitude of the effects was uncertain; these were presence of alien species, over-exploitation, and land use change. For Class II hypotheses, there was a poor, but increasing understanding of the relative effects on amphibian populations; these were global changes in UV radiation and climate, contaminants, and emerging infectious diseases. They concluded that additional research using integrated approaches was necessary to understanding all of the complex interacting predictions of the hypotheses.

One can retrospectively test competing hypotheses by compiling results from previous studies. van der Valk (2012) reviewed theories and multiple hypotheses relative to invasive plant species in wetland systems. There are two principal theories for why so many invasive plants are found in wetlands: (1) wetlands are more vulnerable to invasion because they are landscape sinks and susceptible to disturbance and (2) invasive species are superior competitors. Numerous hypotheses and associated predictions have been advanced within the two theories (e.g., enemy release, hybrid vigor, empty niche). Following a review of the evidence, van der Valk (2012) concluded that while there is some support for the superior competitor theory, hypotheses based on landscape sink/disturbance theory had the most support for explaining the presence of invasive plant species in wetlands.

The key to development of a study design is the ability to conclusively reject a hypothesis (or several hypotheses) such that the scientific process can progress. To test the efficient-community hypothesis (all plant species that can become established and survive under the environmental conditions found at a site will eventually be found growing there and/or will be in its seed bank) for restored wetlands, Galatowitsch and van der Valk (1996) compared the floristic composition of natural and restored wetlands in northern Iowa. Although a few similarities were found between natural and restored wetlands, they rejected the efficient-community hypothesis with a conclusion that dispersal ability of plants had a greater influence on recolonization of plants in restored wetlands than site-specific presence.

In wetland science, theories and hypotheses are not restricted to ecological concepts. For example from an economic perspective, Whitten and Bennett (2005: 45) proposed a theoretical concept that "the production of wetland protection outputs is unlikely to be at the level desired by the community," which essentially means that society values wetlands at a level greater than that being provided by conservation efforts. They formulated two basic hypotheses that could be tested by an appropriate study: (1) "an increase in the production of wetland protection outputs would generate a net benefit to the community" and (2) "policies in alternative to those currently in place would reduce the extent of market or government failure in the protection of wetland production outputs" (Whitten and Bennett 2005: 46–47). From an archaeological viewpoint, Kelly and Thomas (2012) outlined competing hypotheses for human presence in the Carson Desert,

Nevada, USA, based on use of a wetland. If humans associated with the wetland were sedentary (i.e., lived year-round in same place), then the human archaeological evidence should be concentrated in and around the wetland with little evidence, except for hunting parties, in surrounding mountains. If humans were nomadic and they used the wetland as a stop-over point, then one would expect to find transient evidence at the wetland and extensive evidence in surrounding mountains as people roamed throughout the region. They then proposed a study, using proper design to test the hypotheses; the final conclusion was that both hypotheses lacked support and thus, additional hypotheses were generated based on the information generated during the study.

While development and testing of competing hypotheses should be the goal of wetland investigations, it should be noted that descriptive research (i.e., natural history), long-term monitoring of ecological systems and their components, estimation of magnitudes of effects, and documentation of changes in status of ecological systems are valid and informative provided that the methodology is not flawed. Results from these types of studies can be used to generate hypotheses for additional testing, document historical or baseline conditions for future comparison, provide input for policy and economic decision makers, and document ecological conditions and responses for future use. Indeed, these types of studies are quite common, but one must realize that conclusions based on these efforts can be considered premature pending a rigorous test of competing hypotheses intended to explain observed patterns, trends, and relationships.

1.4 Study Population

After hypotheses or objectives are explicitly stated, the focus of study design shifts to the process of devising a study to test the hypotheses. Inherent to proper study design is the identification of a **population** to define the biological entity of interest and placement of bounds on the scope of the experiment. A **biological population** is defined as a "group of organisms of the same species occupying a particular space at a particular time" (Krebs 1985: 157). A **metapopulation** is formed when the population of interest is subdivided into discrete patches across a landscape but movement among patches remains (Levins 1969). Often, in wetland science, one is also interested in **communities**, which is "any assemblage of populations of living organisms in a prescribed area" (Krebs 1985: 435).

However, from a statistical and study design perspective, population has a broader meaning than just an organismal definition. It is the total set of elements or membership of a defined class of organisms, objects, or events. For wetland studies, the population of interest may consist of the wetland type, ecological condition of wetlands, organisms depending on the wetland, or a variety of other elements of the system. The statistical or target population is the foundation of study design and subsequent application of results. The **target population** is statistically (i.e., has measurable parameters with true, but usually unknown, values

or distributions that can be estimated) and biologically defined, occupies specific units of time and space, contains measurable attributes, and represents the collection of subjects such as individuals, habitats, communities, or natural systems available to be studied in which results and conclusions would be applicable. Frequently, not every element of the target population is available to be selected for study. Those elements accessible for study form the **sampled population**. For example, if one is conducting aerial waterfowl surveys of wetlands in a defined area, then the target population would be waterfowl on all wetlands in the area; however, if flights are restricted for some reason (e.g., military operations, wind turbines, powerlines), then the sampled population would only be the wetlands available to be surveyed. Under proper study design, one could assume that the sampled population was representative of the target population.

Kentula et al. (1992) indicated that the process of defining the population of wetland elements within a study influences the techniques used in the study, timing of the study, and most aspects of data collection. They highlighted the need to define and record characteristics of the target population early in the planning process. Knowledge of the target element (e.g., wetland, biota) is critical to setting boundaries of the target population. When setting the spatial boundaries of target population, Kentula et al. (1992) emphasized the need that boundaries should be set to include similar hydrologic, climatic, geologic, and other relevant geographic conditions that influence the ecology of wetlands of interest. For example, one would have a much different target population of wetlands if the element of interest is waterfowl compared to amphibians.

All studies have the implicit goal of making **descriptive** or **explanatory inferences** based on empirical data about the natural world (Platt 1964). Both of these inferences have the primary role in study design of allowing the researcher to infer results beyond the immediate data to something broader that is not directly observed (King et al. 1994). Descriptive inference is using observations from a study to learn about other unobserved facts. Statistical inference is the process of drawing sound and appropriate conclusions from data subject to random variation. This includes a formal understanding of the limitations of application of the inference in time and space. That is, one cannot explicitly apply the knowledge to any population outside of the study population or under conditions not experienced during the study. Such limitations must be described during any presentation (verbal or written) of study results.

1.5 Determination of Experimental Unit Variables

Once the target population and all elements within the target population have been defined and identified, then the process of selection of elements for study and variables to measure is initiated. Subjects (elements) to which individual experimental treatments are applied are termed **experimental units**. Experimental units can be individual animals, defined populations of animals, unique ecosystems or

habitats (e.g., isolated wetlands, islands), subdivided units within larger ecosystems or habitats (e.g., management units of a contiguous system, pastures, watersheds), or some measure of time.

Experimental units can be natural features (e.g., wetland, bird, plant) or man-made (e.g., mesocosm, microcosm, greenhouse flat). Balcombe et al. (2005) tested hypotheses that invertebrate family richness, diversity, density, and biomass were similar between mitigation and reference wetlands. As experimental units, they selected 11 mitigation and four reference wetlands across three physiographic regions of West Virginia. Maurer and Zedler (2002) tested hypotheses contributing to the invasion of reed canary grass (*Phalaris arundinacea*) using a parent plant transplanted into a cone-tainer and attached to aluminum troughs to measure tiller growth over time in response to shade and nutrient treatments; each cone-tainer was an experimental unit.

All study designs involve identification, measurements, or estimation of variables considered to affect the hypothesis being tested. There are several classes of variables to consider during study design. Basic to statistical models are independent and dependent variables. **Independent variables** are those hypothesized (including treatments) to contribute to variation in **dependent** or **response variables**, the values of which depend on levels or types of independent variables. In most study designs for wetlands, there is one dependent variable of interest; for example, density of waterbirds, species richness of invertebrates, levels of nutrients in water runoff, and soil moisture. However, there can be several associated independent variables that may be categorical or continuous variables that are hypothesized to influence the variance of the measured dependent variables; for example, wetland type, wetland area, watershed condition, vertebrate sex and age, and time.

Analyses related to a single dependent variable are termed **univariate**, and there is a long history of established methods to test hypotheses involving a single dependent variable for both discrete (i.e., categorical [e.g., chi-square analyses] or factor variables [e.g., analysis of variance]) and continuous (e.g., regression) independent variables. However, simultaneous analyses of greater than one dependent variable are often of interest and use of **multivariate** statistics (e.g., ordination, principal components analysis, path analysis) has greatly increased during the past three decades with advances in computing power necessary to conduct these analyses. Regardless of the approach, the focus of an established study design is to quantify the relationships among dependent and independent variables through some form of data analysis. The goals of data analysis include evaluation of hypotheses, predicting or forecasting an event or response, development of structure of future models, determination of important variables relative to variation of the dependent variable(s), and detection or describing patterns and trends.

Each independent and dependent variable needs to be determined as a fixed or random effect prior to determining the appropriate study design and analyses. A **fixed effect** is a variable in which levels are not subject to random variation under repetition of the experiment (e.g., wetland type, animal age and sex, levels of nutrients applied, number of seedlings planted). A **random effect** is one where repetition of the experiment will result in different levels within the analyses (e.g.,

time [day, month, year] or environmental condition, wetland area, water depth), unless the same experimental units are used. Statistical inference only can be applied to the target population under the actual (fixed) treatment levels that are within the range of random variables being tested. When a study includes both fixed and random variables, it is considered a mixture of effects and requires some additional consideration during analyses (i.e., mixed models). It is important to define variables as fixed or random when describing the methods used in the study. When more than one independent variable is being assessed in a study, the interaction between effects is of great interest. A significant interaction indicates that the magnitude of differences between levels of one effect depends on the level of the other effect. Many times the interaction between effects is more interesting than individual main effects in explaining data observations and results, albeit this is frequently considered more cumbersome to explain than results for simple main effects. However, an investigator must use their knowledge of the system to ensure that significant interactions have biological meaning and are not a spurious result. Spurious (an apparent relationship between noncausal events or variables) results typically result from the presence of a confounding or nuisance variable. At times, further investigation of interactions is necessary to develop confidence that the interaction is meaningful.

In addition to the proper identification of dependent and independent variables, there are many other types of variables that can impact results and should be considered during development of a study design. It is important to categorize all variables that contribute to the variation of dependent variables into those that are of interest and related to the hypotheses being tested and those that are **nuisance** variables, which are assumed to be of little interest but may affect study results. Extraneous or nuisance variables can have disproportionate impacts on results from a study unless accounted for in the study design. Indeed, the failure to control for nuisance variables frequently results in spurious conclusions. Through study design, nuisance variables can be controlled to account for the potential bias associated with the variable. Examples of methods of controlling nuisance variables in study designs using analysis of variance include grouping experimental subjects into **blocks** (use of some common characteristic to group homogeneous subunits of the sampled population) or use of a **covariate** (a random variable of little interest associated with but varies among experimental units) if the variable is categorical or continuous, respectively.

There are a variety of approaches to account for nuisance variables in a study design. For example, in a study of avian response to prescribed fire in wetlands, Brennen et al. (2005) acknowledged that migration timing (changing avian densities over time) and wetland size (species-area relationship) could influence results, yet these variables were not of primary interest in assessing the effect of spring burning of wetlands. Furthermore, the investigators recognized that conducting a wetland study over a large geographic region could be influenced by varying environmental conditions (e.g., differing precipitation patterns) across the target population of wetland. Therefore, because the primary interest was in avian response to a burning treatment, they paired adjacent burned and unburned (control) wetlands across the

geographic range of the study to remove the nuisance effect of varying environmental conditions to ensure that the effect of burning was determined.

Although it is important to control for as many variables as possible in a study design, there is a limit to the number of controllable or measurable variables for most ecological studies. This is true for both field and laboratory studies. Frequently, it is not possible to identify or recognize all of the potential variables affecting a dependent variable. At times, it is difficult, economically unfeasible, or impossible to measure certain variables. Therefore, it is often appropriate to identify and measure proxy (i.e., correlated) variables that can serve as an index to the variable of interest. Finally, there is a statistical limit to the number of variables that can be addressed through study design as well, primarily because sample size relative to the number of variables dictates the potential analyses. Such uncontrollable variables are typically assumed to be random with the same effect across all samples and controlled variables and thus accounted for in the appropriate experimental design.

Proxy variables can take many forms and do not have to be directly related to the variable of interest. For example, Mackay et al. (2007) measured soil moisture, which affects plant root water availability, to use as a proxy variable for detecting water stress. Pfeiffer (2007) used wetland and other surface water area as a proxy variable for the location of spatially clustered wild bird infection in a study of the influence of wild birds and risk from H5N1 highly-pathogenic avian influenza. In a review study, Elser et al. (2007) identified several variables that were correlated (i.e., proxy) with standing biomass of autotrophs including chlorophyll concentration, ash-free dry mass, carbon mass, biovolume, percent cover, and primary production. They then used a meta-analysis combining results from nitrogen and phosphorus enrichment studies measuring standing biomass and proxies to evaluate nutrient limitation in freshwater, marine, and terrestrial ecosystems. Kantrud and Newton (1996) used the amount of cropland in a wetland watershed as a proxy for their quality. Use of proxy variables is common in wetland studies due to the difficultly in measurements of many ecological characteristics; however, one must be somewhat reserved in stating conclusions using proxy variables unless certainty exists relative to the strength of the relationship between a proxy variable and the variable of interest.

1.6 Conceptual Model and Variable Selection

A common approach to determine which variables to measure for testing competing hypotheses is to transform a conceptual hypothesis to a conceptual model that describes the response of a dependent variable to a set of independent variables. The conceptual model forms the basis of an appropriate statistical model with defined components. The statistical model can be formulated as Y (dependent variable) being a function (F) of some fixed and random independent variables or factors. For example, if one is measuring the total nitrogen load in a wetland, Y

could be defined as the amount of total nitrogen in ppt and independent variables of interest could be watershed type (WS), month (MO), precipitation measure (PR), water depth (WD), and vegetation cover (VEG). Such a model could initially be defined conceptually as

$$Y = F[WS, MO, PR, WD, VEG].$$

Such a model is considered indeterminate at this stage because the list of explanatory factors is likely not complete and the response is not predictive (i.e., nature of the relationship of each independent variable with the response variable is not defined). However, such a model is useful in designing a study. Fixed effects (if control of experimental units is possible) could be watershed type, water depth, and vegetation cover. Random effects could be month and precipitation. However, final determination of random and fixed effects depends on the amount of control an investigator has on the system and variables used to measure the effects.

An important principle in conducting experiments is to hold all factors, except the one of interest, constant so that any response to treatment can reliably be attributed to the treatment. Unfortunately, this is rarely possible in wetland studies and thus, investigators must design studies to limit the variation within all variables that are not of interest. In this example, one may be only interested in the relationship between watershed type and total nitrogen while recognizing that factors other than watershed type influence nitrogen levels in wetlands. The proper study design can remove or partition the influence of independent variables in such a way that priority treatment effects can be estimated and evaluated. Ideally, the model will be constructed based on the hypothesis, available data, assumptions, and potential unknown parameters.

Conceptual models can be specific, similar to the above example, or encompass an entire wetland type (Fig. 1.1). A critical step in constructing a conceptual model is identification of the problem or question. Brooks et al. (2005) used existing data to develop a conceptual model of wetland degradation and restoration in an effort to improve scenarios for the use of mitigation wetlands to replace lost wetland area and ecological function. They hypothesized that increasing influence of stressors homogenizes wetland diversity and variability. Devito and Hill (1998) developed a conceptual model of wetland sulfate (SO_4) retention and export based on watershed hydrogeology. An investigator should use conceptual models to develop objectives and competing hypotheses for experimental testing. Ogden (2005) developed a conceptual ecological model for anthropogenic stressors on an Everglades ridge and slough system. He identified five major ecosystem stressors (reduced spatial extent, degraded water quality, reduced water storage capacity, compartmentalization, and exotic species) and made predictions on stressor effects. In addition, he identified a series of biological indicators of wetland restoration success that can be incorporated into future studies. In addition, conceptual models can be informed by or produce predictions from ecological theory. Euliss et al. (2004) developed a conceptual model for prairie pothole wetlands – the wetland continuum. The model allows for simultaneous consideration of climate and hydrologic setting on wetland

biological communities. The model is predicated on the relative position of a wetland on axes of groundwater and atmospheric water gradients for prediction of biological expression. Predictions from the model and hypothesized relationships form hypotheses for future studies.

1.7 Experiment, Treatment, Replication, and Randomization

An **experiment** imposes a **treatment** on a group of elements or subjects (experimental units) to measure a response and quantify an effect. A treatment is something that an investigator imposes on experimental units in some manner. Treatments can be applied at varying amounts or magnitudes that are usually referred to as levels. For example, if one was interested in the effect of the ratio of emergent plant cover to water on bird use of a wetland and applied treatments (i.e., cover to water ratios) of 0, 0.25, 0.50. 0.75. and 1.00, then there would be five levels in the treatment. Frequently in wetland studies, treatments are categorical and selected from nature (i.e., not investigator applied [e.g., watershed type, season, anthropogenic modifications, soil type]).

There are numerous approaches to experimentation, with differences among them related to relative scientific rigor and application of randomization and replication (Skalski and Robson 1992; Morrison et al. 2001). An experiment can also be referred to as manipulative experiments (Hurlbert 1984), comparative experiments (Kempthorne 1966) or randomized experiments (Kirk 1982). A **controlled experiment** is the ideal type of design that essentially isolates the dependent and independent variables of interest while controlling identified nuisance or confounding variables. An experiment produces results and conclusions of greatest rigor and least uncertainty of results. A true experiment requires random allocation of treatments to experimental units and replication of experimental units (Fisher 1935). All identified nuisance variables are controlled or accounted for in the design or through randomization. Frequently, random allocation of treatments to experimental units or strict randomization of selecting experimental units or samples is not feasible. In such instances, the investigator must take precautions to ensure that personal bias or confounding (nuisance variable influencing the treatment effect) effects do not cause doubt in the representativeness of the results. Wetland experiments can be conducted in both laboratory and field settings. Some wetlands types (e.g., prairie potholes, playas, vernal pools) easily serve as replicates in a controlled experiment by their very nature of being small, isolated wetlands with discrete boundaries defined by soil, vegetation, or hydrology. For wetland types (e.g., coastal marsh, peatlands, bottomland hardwoods) that exist over large areas (e.g., 100s–1,000s ha) with no discrete demarcation, it is difficult to identify distinct experimental units and special care is needed to artificially define experimental units to conduct an experiment in these systems to ensure that results are representative.

Unfortunately, true experiments are infrequently conducted in ecological and wetland studies because they are not feasible or would exceed available funding. Frequently in wetland studies, the investigator has no control of the experimental units or treatments (observational study). In these instances, true replication and randomization are compromised in some way. The results from these studies are not as conclusive as for experiments, but they do have merit and inference is possible as long as the context of the study is fully revealed.

Assessment studies usually have a single experiment unit of interest (i.e., a single wetland), where treatments are randomized to test effectiveness in achieving a conservation goal. For example, a manager may be interested in the most effective technique to prevent amphibians from entering a contaminated wetland. The hypothesized techniques (e.g., fence, moat, vegetation removal) form the treatments and are randomly applied to the single wetland with a response variable of number of amphibians crossing the treatment into the wetland. Inference is only possible to the single experimental unit being subjected to the treatments.

Observational or **descriptive studies** are very common in wetland science. Essentially, attributes of variables of interest are measured in multiple experimental units over space and time to describe what is observed. Commonly used to develop monitoring strategies or support conclusions from retrospective studies, observation studies are designed to describe the systems of interest and, through the use of retroduction, suggest causal relationships for the measured variables. In an observational study, the system is not manipulated so that variables of interest are not isolated such as in true experiments. The primary drawback to observation studies is that any one of many potential causes could have resulted in the measured observation. Therefore, it is imperative to have considerable understanding of the wetland type being studied to develop a plausible explanation for the observed patterns. Evidence supporting strong conclusions from observational studies is usually lacking, which prompted Romesburg (1981) to urge researchers to use results from observational studies as hypotheses subjected to more rigorous experiments to test the tentative conclusions.

Treatments in an experiment can be categorized into three types. **Manipulative treatments** refer to those studies where all experimental units have the same probability of being randomly assigned to any of the treatments in the study. An example of manipulative treatments would be the random assignment of soil moisture manipulations (e.g., flood, dry, moist) among all potential wetlands under study to measure the production of certain plant communities. Compared with other treatment types, studies with manipulative treatments have the greatest scientific rigor or reliability of results. **Organismic treatments** are defined when experimental units are a treatment, usually categorical, by definition. That is, it is impossible to randomly assign treatments and it is assumed that experimental units represented by these treatments are representative of a random sample from the target population. Examples of such categories include sex and age of animals, plant species or type, soil series, and wetland type. The third type of treatment, which is frequently important for studies with a temporal factor, is the **repeated measure** of experimental units throughout the study, where time is considered a

treatment. Because successive measurements are relative to and correlated with the initial measurement, distinctive statistical analyses are necessary to account for the correlation among successive measurements of experimental units due to violation of the assumption for most analyses of independence among experimental units.

The use of randomization and replication in a study design determines the target population and extent of inference from conclusions. These tools also maintain study integrity and scientific objectivity (Morrison et al. 2001). Study designs can be distinguished by the rules used to govern randomization and replication. In addition, the application and extent of randomization and replication are critical factors in judging the reliability of conclusions from studies. **Randomization**, according to Fisher (1935), is at the heart of experimental design. Randomization refers to both (1) random selection of representative study units for sampling and (2) random assignment of treatments to experimental units in an experiment. Most traditional experimental designs are based solely on the rules for randomization (Kirk 1982). The extent of inference is directly related to the degree of random sampling, which ensures that the study units are representative of the target population. Failure to randomly assign treatments to experimental units increases the potential impacts of nuisance variables leading to spurious results and potential bias. Underlying statistical theory demonstrates that randomization ensures that estimates of treatment effects and experimental error are unbiased estimates of their respective population parameters. Random assignment of treatments to experimental units is necessary to satisfy the assumption that experimental errors are independent by minimizing the effects of correlation between experimental units on statistical results. Cox (1980: 313) summarized the importance of randomization to studies, in that "randomization provides a physical basis for the view that the experimental outcome in a given study is simply one of a set of many possible outcomes. The uniqueness of the outcome, its significance, is judged against the reference set of all possible outcomes under an assumption about treatment effects, as such effects are negligible. For the logic of this view to prevail, all outcomes must be equally likely, and this is achieved only by randomization." In wetland studies, true randomization within a target population can be difficult. The reasons for this complicatedness are numerous, but include denial of access to study sites, environmental conditions (e.g., drying of wetland when studying aquatic invertebrates), equipment placement requirements (e.g., insufficient water depth to measure water quality, flooding potential of deployed equipment), lack of defined boundaries identifying experimental units, sampling logistics (e.g., travel, time to collect samples), and presence, or suspected presence, of organisms of interest (e.g., certain amphibian species, habitats used by certain avian species).

Replication is the necessary practice of using more than one experimental unit for each treatment. Replication is required to minimize the uncertainty of concluding that differences among treatments are due to treatment effects rather than inherent differences among experimental units or due to random chance. Replication is required to measure the variation among and within treatments to make conclusions regarding treatment effects, which is the basis for drawing inference about treatment effects using traditional, univariate statistical techniques such as

analysis of variance. If only one experimental unit is assigned per treatment, then no statistical inference beyond the experimental units sampled is possible because experimental error cannot be estimated. Much too frequently in wetland science, a study is conducted once with weak evidence for conclusions that results in uncertain knowledge; repeating the study would either strengthen evidence for conclusions or show that the initial conclusions are not supported allowing for the development of alternative hypotheses.

Another form of replication is the practice of repeating a study to strengthen conclusions. Results based on studying a few wetlands in a limited area or constrained environmental conditions (e.g., only wet or dry years) could be confirmed by a similar study or succession of studies conducted over larger temporal and spatial scales to determine if conclusions hold under more general conditions. For example, Luo et al. (1997) concluded that unsustainable accumulation of upland sediment was filling playa wetlands and represented the greatest impact to this unique wetland system. Their results were based on data from 40 playas (20 cropland watersheds and 20 grassland watersheds) in a limited spatial distribution. However, since the initial study, a number of subsequent studies have confirmed that the original conclusion is valid at larger spatial scales and under a variety of environmental conditions with steadily increasing evidence of negative impacts of sediment accumulation on playa wetlands (e.g., Tsai et al. 2007, 2010; Johnson et al. 2011, 2012; Smith et al. 2011; Burgess and Skagen 2012; O'Connell et al. 2012).

The concept of pseudoreplication is frequently confused with true replication (Hurlbert 1984). Replication is based on the number of experimental units whereas pseudoreplication usually refers to multiple measurements from a single experimental unit that are treated as independent experimental units during analyses. For example, if one was interested in biomass production between grazed and ungrazed wetlands (i.e., grazed/ungrazed are treatments) but only applied each treatment to a single wetland of each treatment and clipped and weighed aboveground biomass in 20 plots/wetland, then analyses using plots as experimental units would be pseudoreplicated. Furthermore, because the treatments were applied to the entire wetland, the plots are samples and not individual experimental units randomly assigned to a grazing treatment. Because there is only one experimental unit per treatment, estimation of experimental error is impossible because the variation among samples within each experimental unit would be considered sampling error (i.e., variation among samples of a given experimental unit; see Sources of Error below). Pseudoreplication can easily be avoided when it is understood what the unit is to which the randomization rule applies when assigning treatments to experimental units. The results from studies that include multiple samples from single experimental units should not be considered invalid because it is often difficult or impossible to replicate certain experimental units (e.g., oil spills in a coastal marsh) in applied ecological research (Wester 1992). However, it is critical to realize that inference of results can be strictly extended only to the experimental unit(s) sampled. In most ecological fields, use of pseudoreplication is considered a fatal flaw for studies, but many times this approach is appropriate in wetland studies as there may exist

only one experimental unit or environmental condition to be studied (i.e., the Everglades, Great Salt Lake marshes), interest is only in the experimental units being studied (see BACI study below), or the scale of the study is so large that replication is impractical. As an extreme example, it is not possible to use replication to test hypotheses related to global climate change because Earth cannot be replicated.

1.8 Impact Studies

In wetland ecology and management, biologists are frequently interested in the effects of impacts to wetlands. Impacts can be natural or human-made, planned or unplanned, but cause a change in the system state of the wetland. Impacts can be natural disasters such as hurricanes, extensive prolonged drought, or designed for management to improve ecological conditions (e.g., removal of invasive species, restoration of historical hydrology). An **impact assessment study** includes a design common to wetland studies known as a BACI (before-after/control-impact) design (Green 1979). Principally, these types of study designs are the result of some sort of natural or anthropogenic disturbance. The majority of wetland studies include some sort of measurement or modeling of the effects of disturbance on the abiotic and biotic components of the ecosystem. In wetland ecosystems, disturbance is common and, for many wetland types essential to ecological function, differing primarily in degree of disturbance (i.e., short-term flood, multi-year drought, hurricane effects that last decades). Furthermore, included in definitions of wetland, both ecological and legal, are references to disturbance that must occur prior to the system being declared a wetland. For example, coastal marshes are affected daily by the predictable disturbance of tides that raise and lower water depths and adjust salinity levels in the marshes. In many inland, geographically isolated wetlands such as prairie potholes and playa wetlands, fluctuations between wet and dry states are fundamental to the function of these systems.

When conducting an impact study, the type of disturbance will greatly influence the development of a study design. There are three primary categories of disturbance – pulse, press, and those affecting temporal variation (Bender et al. 1984; Underwood 1994). A **pulse disturbance** is not sustained beyond initial disturbance, but effects persist beyond cessation of the disturbance (e.g., fire, hurricane). A **press disturbance** persists beyond the initial event (e.g., flood, drought, invasive species). A **temporal variance disturbance** results in increasing or decreasing amplitudes (i.e., variance) around a constant mean on some sort of meaningful temporal scale. Documenting a temporal variance disturbance is difficult and requires long-term investigation or system monitoring. For example, some wetlands require precipitation runoff events to flood; however, future climate change may increase variation of precipitation between years or years represented by extreme precipitation events can change over time while average annual precipitation remains relatively constant. Therefore, species adapted to the historical

precipitation patterns (e.g., timing, intensity, and amount) may not persist as they are replaced by species better adapted to the changing precipitation patterns. From a wetland perspective, this is, as yet, an understudied potential consequence of global climate change. Recognition of a human-defined temporal scale relative to disturbance makes it difficult to fully understand the role of natural disturbance in wetlands beyond a few decades.

Because BACI studies are considered pseudoreplicated due to lack of true replication, the inferential scope of these studies is limited to those wetlands studied and not to a larger target population. Such designs are very common in wetland studies to assess effects of proposed anthropogenic activities, especially when mitigation is involved. The basic approach to a BACI-type study design is the collection of a sample prior to the disturbance and another taken after the disturbance at both the disturbed site and representative "undisturbed" control site. A measurable effect due to the disturbance would be represented as a statistical difference in the average value of the dependent variable between the control and disturbed sites prior to and after the disturbance. A wetland that is proposed to be impacted by some activity (e.g., dredging, filling, change in hydrology) has not been chosen randomly, but the impacts of the activity on wetland function must be known to mitigate any negative effects of the action. A BACI study can be used to quantify these impacts. Following identification of the impact site, a particular wetland that is a geographic neighbor and similar enough to the impact site such that both wetlands would be subjected to the same nuisance variables is subjectively identified as the control site (i.e., reference site not experiencing the impact of interest). None of the experimental units were randomly chosen nor were treatments randomly assigned; the goal of this study design is to make inference for only the impacted wetland by measuring the effect size of the impact. Data are collected simultaneously in both wetlands in the same manner prior to the impact and after the impact. Statistical analyses and resulting inference are on the comparison of the magnitude of the differences in recorded data between impact and control sites prior to and after the impact (Stewart-Oaten et al. 1986). Underwood (1991, 1994) details a variety of approaches for statistical analyses of impact studies.

Assessment of impacts not defined prior to impact occurring can be studied using an impact assessment approach, but the lack of pre-impact data results in weaker inference. Unpredicted environmental events such as hurricanes, wildfire, floods, drought, and wind blowdowns are frequently studied for impacts to wetlands without any pre-impact impact data. Skalski and Robson (1992) described an approach to these types of studies defined as an accident assessment study. They suggested using a control site and creating a time series of measurements for the control and impacted site. Inference regarding impact would be based on comparisons of trajectories of the data as the impacted site recovers. In situations where a reference area is not available, an impact gradient study may be appropriate (Skalski and Robson 1992). This approach assumes that impacts related to a disturbance are greatest at the core of the disturbance and declines as one moves away from the core. Therefore, sampling is conducted on the spatial linear distance

radiating from the core location of disturbance. The resulting data are used to model the decline of impact effect with linear distance.

Hannaford and Resh (1999) used a BACI study design to estimate the impact of all-terrain vehicles (ATV) on vegetation in a San Francisco Bay wetland. They found that ATV use caused immediate impact to vegetation but limited use allows for recovery within a year without continuing traffic. Zimmer et al. (2001) used a BACI approach to assess the ecological response to colonization and extinction of minnows in a prairie pothole in Minnesota. The impacted wetland was paired with a fishless site and comparisons were made when both were fishless, following introduction of fish into impact wetland, and after eradication of fish in impacted wetland. They found that introduction of fish into a prairie pothole resulted in increased turbidity, total phosphorus and chlorophyll *a* in water, and decreased abundance of aquatic insects. Removal of fish reversed these effects. Suren et al. (2011) followed a BACI protocol to evaluate the effect of hydrologic restoration of drains within a wetland in New Zealand. Results indicated that restoration of drains was beneficial as invertebrate communities were similar to natural wetlands and cover of exotic pasture grasses declined. In addition, connectivity was improved for recolonization of native wetland plant and aquatic invertebrate communities.

1.9 Sampling

A consistent and common criticism of scientific studies is the scale to which study results are applied beyond the target population (i.e., inference). It is rare to measure every member of a target population (i.e., a census), which is why experimental design and statistical analyses are crucial for study design. Therefore, a subset of potential experimental units from the target population is usually selected to measure the variables of interest, which is termed **sampling**. The selected sampling design, as detailed later, is a contributing limiting factor of the extent of inference from a study. In order for results to be extended to the target population, the sampled experimental units must be representative of the target population (i.e., random selection and replication). Ultimately, statistical analyses of data collected from an appropriate study design enable scientists to make inferences about a target population from its sample.

Representative sampling of a target population allows for the description of spatial and temporal patterns in nature and, through testing competing hypotheses, linking an ecological process to the observed pattern. Ultimately, proper study design should elucidate the linkages between described patterns and the ecological processes that created the pattern. Unfortunately, few studies go beyond description of a pattern with a conclusion based on retroductive speculation on the processes that created the pattern. However, all investigators must realize that wetland data are created by two classes of processes. The first is the ecological process that generated the true pattern. The second is the process inherent in the sampling effort that resulted in the data of interest. The assumption is that the sampling process

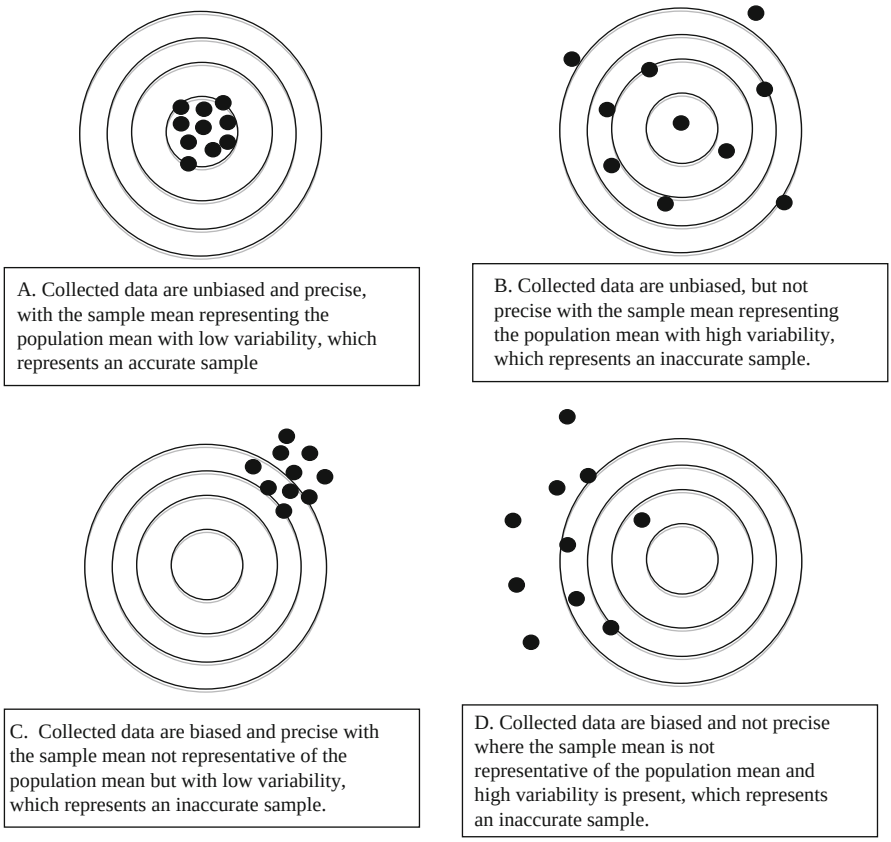

Fig. 1.2 A graphical representation of the concepts of bias, precision, and accuracy where the *center ring* represents the true mean of the target population and *black dots* represents data generated by sampling with the intent to estimate the true mean

does not mask the ecological process and data collected through sampling characterizes the ecological process of interest albeit usually in a much more simplified manner.

Basic to any study design is the goal of sampling randomly-selected experimental units to measure variables of interest. This ensures unbiased inference about some set of population parameters based on a statistic (e.g., mean, variance, standard error) that describes some attribute of interest. The purpose of sampling is to estimate the variable of interest and describe its variation in space and time. When not all members of a target population can be measured (i.e., a census), a sampling design is used to estimate the population value of the variables; statistical methods are used to describe the data and make comparisons regarding tendencies of the data. If one is able to do a census by measuring all subjects of interest (i.e., entire target population), then statistical tests are not necessary. Sampling can be used to both select experimental units for study and control of nuisance variables through a

prescribed strategy. Numerous textbooks are available to assist in designing sampling strategies beyond what is described in this chapter (e.g., Cochran 1977; Scheaffer et al. 1979; Thompson 1992). Points, plots, transects, and marking captured animals are among the techniques used to sample experimental units.

The goal of sampling is to achieve an unbiased (closeness of observed values to true value) and precise (close proximity of repeated measurements of the quantity) estimate of a population parameter value (Fig. 1.2). Ideally, sample measurements for an estimator should have a narrow range of variation (i.e., precision) centered on the population value (i.e., unbiased), which represents an accurate estimate. For example, an objective of sampling must be to produce a sample mean $\approx$ population parameter with low variance around the sample mean. Groups of sample measurements that are centered on the population value but yet have a wide range are considered unbiased, but the presence of a high variance will decrease the reliability of detecting treatment effects. Biased samples generate a mean or other statistic that is not representative of the population parameter, but can have a narrow (precise) or wide range of values. A sample that is both unbiased and imprecise yields little information relative to the target population. Unfortunately, it is rarely possible to determine if one has an unbiased and precise sample because rarely are population means and variances known. Finally, wetland systems are exceptionally complex such that strict adherence to sampling schemes may be difficult, even for laboratory studies. However, it is critical that protocols associated with sampling designs be followed as explicitly as possible. Inference from samples to a target population is conditional on the protocol for selection of study sites and subsequent sampling. Thus, information from any sampling design is subject to interpretation based on the context in which the samples were collected.

Sampling protocols can be categorized as (1) haphazard sampling, (2) judgment sampling, (3) search sampling, and (4) probability sampling (Gilbert 1987). There are many variations of the sampling process within these categories (Gilbert 1987; Gilbert and Simpson 1992) that are beyond the scope of this chapter. The sampling designs described below are not meant to be inclusive of all possible approaches, but rather a description of those that would be commonly used in wetland studies. However, as a caveat, a minimal goal for reliable inference of results is some form of probability sampling where all potential experimental units have the same probability of being selected as a sample. This strategy produces unbiased estimates of the population mean, variance, and other attributes. Frequently, the phrase **sampling frame** is used to describe a list of all members of a target population (i.e., elements) that potentially can be sampled (Jessen 1978). In field studies, the sampling frame is usually spatially (study area) or temporally (study period) defined. In laboratory and human dimension studies, the sampling frame is usually defined by a list of all potential elements that could be selected for study. Finally, it is highly recommended that any proposed sampling design be reviewed by a statistician or quantitative biologist to ensure that all possible contingencies have been addressed and the proposed sampling strategy will allow for the desired inference of results.

Haphazard sampling, also known as **convenience sampling**, is frequently justified due to cost, time, and logistics, or, on occasion, historical merit. This sampling approach greatly limits the number of experimental units within a target population available to be sampled because the strategy employs a protocol that limits sampling only to a limited number of potential experimental units, which can have substantial influence on subsequent inference. Results from haphazard sampling must be placed in the context that the data were recorded and are valid only if the target population is homogeneously distributed (Gilbert 1987). Examples of haphazard sampling would be to only sample wetlands on public land or adjacent to field stations, conduct roadside surveys of wetlands, rely on volunteer reporting of flora and fauna outside a defined study, or use an inconsistent temporal sampling schedule. Similar to pseudoreplication, inference from haphazard studies is limited to the sampled experimental units. For example, if one only sampled wetlands with public access, then reliable inference can only be made to similar wetlands with public access. However, at times, investigators may be interested only in the wetland represented by haphazard sampling and thus, inference can be considered valid if the remainder of the study design is appropriate. Haphazard sampling can be used for initial assessments of an area or hypothesis development (Morrison et al. 2001).

Because of the sheer number and small size of available wetlands, Babbitt (2005) used haphazard sampling across a microhabitat gradient to relate wetland size and hydroperiod on the occurrence of amphibians rather than random sampling. She justified the efficiency of the sampling approach by noting that no new species were found in subsequent sampling efforts. In wetland ecology and management, frequently one is interested in the effects of impacts to wetlands. Hornung and Rice (2003) haphazardly selected grazed wetland treatment locations to evaluate the relationship between the presence of Odonata species and wetland quality in Alberta, Canada. They also used haphazard methods to sample invertebrates. Unfortunately, one conclusion from the study was that the haphazard sampling was insufficient to detect a trend for aquatic macroinvertebrate abundance, diversity, and composition along a gradient of grazing intensity. Due to wind conditions, Pierce et al. (2001) haphazardly sampled fixed sampling stations to document the littoral fish community in Spirit Lake, Iowa. Their results indicate a native species decline of 25 % during a 70 year period, which was attributed to a decline in littoral vegetation and other habitat changes. In wetland science, haphazard sampling is relatively common primarily due to access restrictions precluding random selection of sampling units. It is incumbent upon the researchers to declare the context of the sampling effort, which will appropriately restrict inference of results.

Judgment sampling is based on the presumption that prior experience allows for representative selection of a study area or target population (Gilbert 1987). Deming (1990) stated that judgment sampling was a type of nonrandom sampling based on the opinion of an expert. This approach can be considered subjective and representativeness of results relative to the target population difficult to assess (i.e., uncertainty regarding what population is being sampled). As with haphazard sampling, judgment sampling can be used to assess an area, generate questions

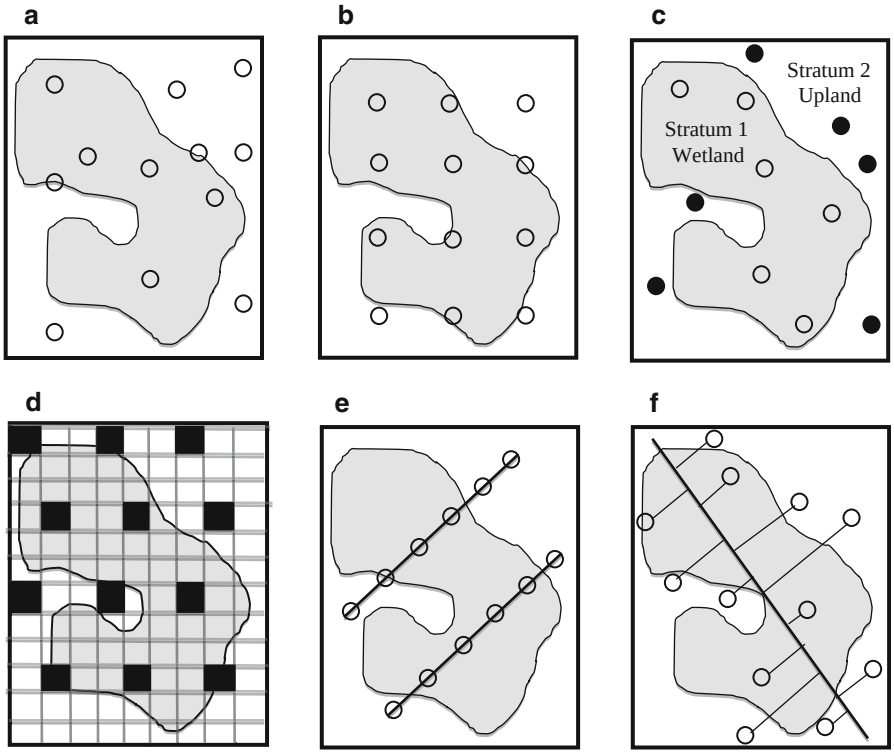

Fig. 1.3 Examples of sampling design for wetlands including (**a**) simple random design, (**b**) systematic sampling, (**c**) stratified random sampling, and (**d**) systematic use of a grid to select sample area. Examples of sampling method are (**e**) plots along transects and (**f**) line transect

that lead to hypotheses, and generate data to be used in a modeling study (Morrison et al. 2001). The most common type of judgment sampling is restricting data collection only to those wetlands that are known to contain the variable of interest (e.g., certain amphibian species, distinctive watershed conditions). Once again, inference is strictly limited to the experimental units sampled, with extension to other subjects of the target population to be considered tentative at best.

Dobbie et al. (2008) stated that professional judgment and opinion were critical in designing monitoring programs of aquatic systems. Cohen et al. (2005) used scientific judgment to rank relative impairment of wetlands and then sampled wetlands based on categories of ecological condition. Hopfensperger et al. (2006) made a case for the use of profession judgment for situations where no prior information is available when evaluating the feasibility of restoring a wetland.

Search sampling is based on historical information. Frequently, this information is available from results of long-term inventory and monitoring programs. Typically, this sampling involves using *a priori* knowledge to select areas to sample. This type of sampling differs from judgment sampling in that sampling

locations are selected based on the known occurrence of variables of interest (e.g., certain plant species, known nesting locations of birds) rather than an informed opinion that the variables of interest would be found at a certain location.

The strongest inference comes from data collected using a form of probability sampling. **Probability sampling** is when all elements within a defined population have an equal probability of being selected to be sampled and that probability is known. There are a number of probabilistic sampling schemes including simple random sampling, stratified random sampling, and systematic sampling (see below, Fig. 1.3). By selecting experimental units at random, statistical properties are unbiased and represent the target population. Furthermore, such samples allow for evaluation of the magnitude of treatment effect size. These sampling strategies can range from rather straight forward to increasingly complex depending on restrictions (e.g., subsets of experimental units and nuisance variables that may need to be addressed). In most wetland studies, elements or experimental units are selected without replacement as each element appears only once in a sample (Levy and Lemeshow 1991). Sampling with replacement is when elements are returned to the target population following measurement and have the potential to be sampled again. Sampling without replacement increases precision of the results (Caughley 1977). There are several types of probability sampling strategies.

One type of probability sampling is **simple random sampling**, which occurs when each element of a sampling frame or target population has an equal probability of being selected. The elements or experimental units are not subdivided or stratified in any manner. Random selection of each element is independent of all other elements. Morrison et al. (2001) outlined five basic steps to achieve a simple random sample. First, the investigator assumes that the target population consists of a finite number of elements (i.e., experimental units) available to be selected. All selected elements can be located, accessed, and the variable(s) of interest can be measured without error. The elements must occur throughout the sampling frame and cannot overlap in any manner. Elements do not need to be identical, but as differences among elements increase in magnitude or subsets occur, then a more complex design may be necessary to avoid biasing a sample with overrepresentation by certain element types. All elements are normally sampled (i.e., consist with all other elements) without replacement.

Use of simple random sampling can be problematic if the target population is comprised of groups or subsets of similar elements. In wetland studies, this occurs when elements are clumped and patchy, such that a relatively small sample size (typical for field studies) may result in an overrepresentation of certain groups or elements with distinctive characteristics that can skew results to properties of subgroups rather than the entire target population. Dividing the elements of a target population into independent subsets or groups (i.e., strata) and then applying a random sampling approach within each stratum can increase the likelihood that results are representative of the target population in addition to increasing knowledge for elements of distinct strata that could be missing using a simple random sample.

Stratified random sampling can be used to increase sampling efficiency and statistical estimation. The key for successful stratified sampling is that the basis for stratification is correlated with the measured dependent variable. For example, if an investigator is interested in the effect of watershed condition on water quality of a wetland then sampling should be stratified using identified watershed conditions (e.g., grassland, cultivation, forested) to ensure that each condition is properly represented in the sample. By stratifying, one ensures that a single watershed condition does not dominate the sample and consequently the final results.

Drawbacks to stratified sampling are (1) spatial and temporal scale of relevant stratification variables can be difficult to determine; (2) increased complications for analyses when homogeneous strata do not exist; and (3) sampling costs are increased. Samples can be distributed among strata either by proportion of strata size or an optimal allocation process. An example of this type of sampling would be to stratify an area of coastal marsh by a salinity gradient (i.e., fresh, intermediate, brackish, saline marsh), estimate the proportion of each strata (e.g., fresh = 0.10, intermediate = 0.30, brackish = 0.50, and saline = 0.10), then determine sample size within each strata by dividing the total number of samples to be taken proportionally among the strata (e.g., if 250 total samples are needed to detect a difference between treatment levels, then 25 would be taken in fresh and saline marsh; 125 in brackish marsh; and 75 in intermediate marsh).

Strata can be defined within the study area (e.g., wetland and upland), study period (e.g., seasons), and target population (e.g., small and large wetlands). Strata cannot overlap, and elements cannot be available for selection in greater than one stratum. For stratification to be useful, elements (experimental units) should be more homogeneous within strata than among strata. If this is the case, by stratifying, sampling standard error of the overall population mean should be reduced to the standard error estimated by simple random sampling. Further, estimates of dependent variables for each strata allows for comparisons among strata, which are frequently of interest. However, it is critical to delineate strata based on knowledge that the identified strata influence variables of interest. For example, one would not test effects of herbicide treatments using strata of wetland size, but rather stratification based on wetland hydrology, soil type, or vegetation would be appropriate.

Systematic sampling represents an interesting approach that is rarely used in wetland studies, but has a role in a variety of settings. Such a sampling approach is possible when a population can be ranked in ascending or descending order of some characteristic (e.g., wetland area, watershed area, salinity gradient). Here, one would rank the population of interest relative to the characteristic and then sample based on some rule (e.g., every 10th ranked object). In addition, systematic sampling is often done on a spatial scale whereby a systematic grid of points or units is established and those to be sampled are chosen by randomly selecting a starting point and then establishing a rule to sample the remaining points or units in reference to the starting point. For example, in a large coastal marsh where one is interested in the distribution of a contaminant, use of an appropriately sized grid overlaid on a map of the marsh provides unique sampling units. Upon randomly

choosing the initial grid cell to sample, the investigator can then systematically assign the remaining cells to be sampled. The usual assumption for systematic sampling is that the study area is relatively homogeneous and thus, the variable of interest is uniformly distributed across the study area. Occupancy modeling (MacKenzie et al. 2006) frequently utilizes this sampling approach. Advantages to systematic sampling include being easier to establish sampling units than random sampling, and it may be more representative (i.e., more precise) because of the uniform coverage of the entire population (Scheaffer et al. 1990; Morrison et al. 2001).

1.10 Errors to Consider in Study Design

A thread linking all aspects of study design is the minimization of errors that impact results, conclusions, and inference of a study. Because sampling is at the core of any study design and the primary goal of any study is to produce reliable data, one must be aware of the potential biases associated with sampling and strive to eliminate or minimize sources of bias or error. Failure to do so confounds subsequent data analyses and results, obscuring the true inference and, frequently, contributes to incorrect conclusions. There are several types of errors that one should be cognizant of throughout the study design process. Such errors can be categorized as theoretical, statistical, mechanical or procedural. While investigators need to be aware of how each type of error affects their study, the best defense against errors disproportionally affecting one's study is strict adherence to a sound design, sampling protocols, and data collection.

An example of theoretical error is in the interpretation of statistical results. Statistical results should be used to support a conclusion or inference based on totality of evidence from a study, rather than an investigator responding exclusively to each statistical result. However, errors associated with statistical results can be found in the inherent uncertainty of statistical tests and expressed in probabilistic terms. In classical null hypothesis testing, the possibility of conclusion errors should be considered in the study design. There are two predominate theoretical decision conclusion errors that can occur in a study. A Type I Error occurs when the null hypothesis is rejected when it is true. The probability of a Type I Error occurring is α, which is set by the investigator prior to conducting statistical tests of the data (conventionally $\alpha = 0.05$) and commonly referred to as the significance level of a statistical test (i.e., the probability level at which a test results in a significant difference between treatments or levels of a treatment). A Type II Error is more serious than a Type I Error and is defined as the probability (β) of failing to reject the null hypothesis when it is indeed false. Determination of β depends on the defined α-level and the sampling distribution of the estimated variable. More importantly, one can derive the value of $1 - \beta$, which is defined as power of the test and defined as the probability of correctly rejecting a false null hypothesis. Power should only be calculated prior to conducting a study when computing the required sample size; it

should not be used following a study to evaluate confidence when failing to reject a null hypothesis (i.e., retrospective power; Gerard et al. 1998).

Statistically, although referred to as error, variation within the target population is important to correctly estimate as it is the foundation of many statistical techniques used for testing differences among levels of dependent variables. Estimation of **experimental error** is the inherent variation among experimental units treated alike or variation not explained by treatments or other variables. Accurate estimation of experimental error is critical for testing treatment effects on response variables. Experimental error differs from **sampling error**, which is the variation among samples (or observations) of a given experimental unit. Sampling error can be due to natural variability among units under study and can result from chance or sampling bias in selecting subjects for sampling (Cochran 1977). Any time that more than one sample or observation is recorded per experimental unit (e.g., multiple plots or water samples/wetland), accounting for sampling error needs to be considered as the study design is developed. An example of experimental error would be variation of above-ground biomass among wetlands; this could be the result of a single sample collected in each wetland or the variation among wetlands of the average of multiple samples taken within a wetland. Sampling error would be the variation among samples within a single experimental unit; that is, the variation of multiple samples of biomass collected within a wetland designated as an experimental unit.

Mechanically, during the course of data collection for a study, a number of errors are possible. Cochran (1977) outlined these and other sources of error in ecological studies for which investigators must be prepared and vigilant. Proper methodology is the primary protection from a study suffering from investigator bias, personal values, and preconceived results. However, if the observations or measurements are made incorrectly or with the inappropriate equipment, then **measurement error** is a likely outcome. For example, species can be misidentified, counts incomplete, flow meters improperly calibrated, and measurements taken at the improper scale (e.g., meters recorded instead of millimeters) are among the countless potential other sources of measurement error. Each observer tasked with data collection must be trained, occasionally assessed, and dedicated to consistent effort to reduce effects of measurement error on final results.

Another source of mechanical error is **missing data** due either to the failure to record the proper measurements or loss of recorded data (e.g., nonfunctional equipment, weather, electronic storage failure, loss of paper copies). Missing data can cause serious issues with subsequent data analysis unless accounted for by an appropriate analysis. Investigators should take steps to avoid missing data by securing data, checking equipment functionality, and ensuring that procedures are understood by all. At times, individuals fail to record an appropriately measured zero in the data, choosing to leave the data cell blank or empty creates the impression of missing data when, in reality, the results may be biased due to lack of a zero. For example, when inventorying plant species in multiple wetlands, one must be careful to ensure that when a species is not detected in a wetland that a zero or absent is recorded and properly transcribed rather than leaving the results for the

species/wetland combination as a blank entry. It is important to realize that missing data and data containing zeros represent vastly different representations of the data.

Observer bias is a mechanical error and constant factor to consider in studies and represent variation among observers. Such bias can be represented in differences in skill of ocular or aural estimate of a variable (e.g., number of birds in flock, percent vertical cover of vegetation, soil moisture relative to field capacity, species of calling amphibians), ability in using a technique (i.e., proficiency with an instrument, ability to distinguish the appropriate scale of measurement) to measure a variable, and human error in recording and transcribing data. If one can measure the magnitude and direction of inter-observer variation, then the data can be adjusted for the bias (Morrison et al. 2001). However, it is quite rare to be able to adjust for observer bias. Therefore, it is important that all observers are trained and tested relative to the data being collected prior to sampling. In most instances, it would be appropriate to consider minimizing the number of observers that record noninstrumented data to reduce observer bias (e.g., same person should conduct bird counts, listen for amphibian calls, estimate percent cover of vegetation types). However, even with limited observers, one must be able to determine if a systematic bias resulting from observer bias where a variable is consistently under- or overestimated due to the selection of sampling points or unit of data measurement (Thompson et al. 1998).

A procedural type of error is what Cochran (1977) termed **gross error** where mistakes are made in transcribing, entering, typing, and editing data and results from analyses. Therefore, all study designs must have a well-defined, unambiguous observation/measurement methodology prior to collection of data. In addition, a protocol for data management is necessary prior to initiating any study. For example, it should be mandated that all paper data sheets be copied at the end of each data collection period and copies placed in safe locations. All electronic data should be backed up in at least two locations and paper copies of electronic data should be printed and stored in a safe location. Loss of complete records of historical data, while rare, can occur due to natural disaster (e.g., hurricane), human error (e.g., inadvertently discarded), loss of electronic data (e.g., hard drive failure), misfiling, or mislabeling. Finally, all data sets and preliminary data analyses results should be reviewed and copy-edited prior to conducting final analyses to ensure observations were accurately transcribed. This is critical if one is using voice-recognition software to enter data. All data should be linked to specific observers. In addition, it is preferable for data review to occur shortly after collection or transcription so that (1) technicians collecting the data remain available to answer any questions, (2) illegible handwriting can be deciphered using unsullied memories, and (3) there is an increased likelihood for recollection of data should issues be identified.

Procedural errors occur when design and sampling protocols are not correctly followed for recording measurements, transcribing and storing data, and conducting data analyses. Development of a structured Quality Assurance and Quality Control (QA/QC) program prior to initiating a project will minimize this bias. A QA/QC program is the foundation for risk management in a study. In addition, any ethical

questions that may arise during a study should be alleviated with an approved QA/QC program. Quality Assurance is a set of activities designed to ensure that the development and/or maintenance process is adequate to ensure a system will meet its objectives. EPA (2001: Appendix B-3) defined Quality Assurance as "an integrated system of management activities involving planning, implementation, documentation, assessment, reporting, and quality improvement to ensure that a process, item, or service is of the type and quality needed and expected by the client." Quality Control is the application of procedures to minimize errors during data collection and analysis. Furthermore, EPA (2001: Appendix B-3) defined Quality Control as "the overall system of technical activities that measures the attributes and performance of a process, item, or service against defined standards to verify that they meet the stated requirements established by the customer; operational techniques and activities that are used to fulfill requirements for quality."

Quality Assurance requires development of a study plan that includes the objectives, design, and implementation of the study with a stated protocol for data recording, storage, analysis, and reporting. The study plan should be reviewed by peers and a biometrician prior to initiation of data collection. The subsequent study plan becomes a dynamic document that should be updated to account for any changes throughout the duration of the study. Examples of Quality Control are stated calibration and maintenance of equipment, training requirements of personnel, methodology procedures and protocols, and use of any data generated during quality control procedures.

Many agencies, private industry, and other organizations require a detailed QA/QC prior to funding an approved study. However, even if a QA/QC plan is not a formal requirement, adherence to and occasional review of an informal QA/QC protocol preserves data integrity. There are countless examples of acceptable QA/QC plans for the U.S. Federal Government (e.g., EPA 1998, 2001, 2008; U.S. Fish and Wildlife Service (http://www.fws.gov/aah/PDF/QI-FWS%20AAHP%20QA%20Program.pdf)), state agencies (e.g., Minnesota and Connecticut http://files.dnr.state.mn.us/eco/wetlands/nwi_comprehensive_project_plan_021012.pdf, http://www.ct.gov/dep/cwp/view.asp?a=2715&q=324958&depNav_GID=1626), and private industry (e.g., Integrated Ocean Drilling Program http://www.iodp.org/qaqc-taskforce/). The U.S. Environmental Protection Agency (EPA) has developed a Wetlands Quality Assurance Project Plan Guidance (QAPP) to assist wetland grant recipients documenting the procedural and data requirements for projects involving environmental measurements (http://www.epa.gov/region9/qa/pdfs/WetlandsQAPPGuidance.pdf).

The EPA Wetlands QAPP guidance is comprehensive and provides a starting point for any wetland project. The guidance includes nine sections with a variety of subjects within each section for consideration prior to embarking on wetland-related studies. Many of these sections can be useful to most investigators of wetland ecology, management, and conservation. The *Project Description* section includes background information and a justification for the study and includes the following items: (1) Project Purpose and Problem Definition; (2) Project Area

Description; (3) Responsible Agency and Participating Organizations; (4) Project Organization Roles and Responsibilities; (5) Permits for Collection of Environmental Measures; and (6) History, Previous Studies, and Regulatory Involvement. The *Project Data Quality Objectives* section ensures that data quality and data management are sufficient to achieve the objectives of the study. The section *Field Study Design/Measurement Protocols* details how data are to be collected (i.e., variables measured) for a suite of abiotic and biotic features. Included in the section *Field Preparation and Documentation* are details related to data management such as (1) Field Preparation; (2) Field Notes (e.g., logbooks, data sheets and forms, and photographs); (3) Documentation of Sample Collections; (4) Labeling of Sample Collections; and (5) Field Variances. The section *Quality Control for Samples Collected for Off-Site Analysis* details handling of samples to prevent contamination and confirmation of lab analyses (i.e., collection of field samples and transport to a laboratory for analyses). Details related to field samples are provided in the section *Field Sample Collection Protocols for Off-Site Analyses*, which can be the most important section for wetland studies. Details related to laboratories are found in the sections *Laboratory Analyses and Section* and *Sample Shipment of Off-Site Laboratory*.

Quality Control is practiced by any entity producing a product. Industry has quality control guidelines and practices to ensure products are functional and within a margin of acceptable variation. That is, identification of defects in products after development but before release. Quality Control is a system of routine technical activities that measures and controls the quality of the inventory as it is being developed. Most Quality Control systems are designed to: (1) provide routine and consistent checks to ensure data integrity, correctness, and completeness; (2) identify and address errors and omissions; and (3) document and archive inventory material and record all QC activities (Penman et al. 2006).

In scientific investigations, Quality Control is project- and method-specific such that development of a Quality Control plan is difficult to generalize. Examples of items to include in a Quality Control plan are (1) equipment monitoring and recalibration, (2) periodic checks for data errors and transcription accuracy, (3) ensuring software and hardware are working correctly, (4) checking integrity of stored data, (5) retraining of technicians and anyone handling or analyzing samples, and (6) confirming that safety protocols are being followed. Therefore, one must identify all fundamental components of a study design and produce a Quality Control plan that addresses each and maintains the highest possible standard of data integrity and accuracy while maintaining a safe environment.

As a final check of the accuracy of the data prior to analyses, one should calculate descriptive statistics (e.g., mean, range, minimum value, maximum value, and variance) or conduct outlier analysis (Barnett and Lewis 1994) to identify extreme values that are inconsistent with the other data and likely to be a result of an error in transcribing data. However, one must have a prepared approach to statistical analyses prior to checking the data to ensure that perceived patterns in the descriptive analyses do not influence subsequent analyses, which can produce spurious conclusions. There is a simple web-based application for the Grubbs' test

for outliers (http://www.graphpad.com/quickcalcs/Grubbs1.cfm) that is available to identify extreme values. If the value came from a contaminated sample, then it is appropriate to recollect the sample if possible or discard the initial observation. Outliers cause considerable problems with most statistical analyses that are based on a particular sampling distribution (e.g., normal) and associated assumptions (e.g., constant variance). If the extreme values were actually recorded and represent a legitimate data point, then the investigator must decide how to handle the value by either removal from the data set, consider data transformation to meet statistical assumptions, or conduct the analyses with techniques robust to outliers (i.e., nonparametric and multivariate methods, or use of generalized linear models linked to a non-normal sampling distribution).

1.11 Sample Size and Effect Size

Although usually an afterthought during study formulation, one must consider the magnitude of a treatment difference or effect size that is biologically meaningful in addition to statistically significant results. **Biological significance** is defined by the investigator but based on a firm understanding of the system being studied and associated literature relative to the system. There is no replacement for sound, extensive biological knowledge of the system that generated the data. If an investigator or reviewer of proposed study design lacks this knowledge, discussions with experienced biologists/ecologists regarding the system and interpretation of results is just as important as use of proper statistical techniques. Not all statistically significant results have biological meaning and, at times, biologically significant differences may not be found to be statistically different. Frequently, the latter is attributed to lack of sample size as an explanation, a situation that would be avoided with proper design prior to collecting the first sample in a study.

Central to a scientific study is the ability to detect a biologically meaningful effect and measure the size of the effect. The primary controlling element for detecting an effect of interest is sample size, where the general rule is "more is better." Increasing sample size decreases the overall variability of the data around a mean for a given treatment, which increases the power of statistical tests (i.e., the probability of finding a difference due to treatment when one truly exists). Therefore, one of the most important aspects of study design is the determination of the appropriate sample size necessary to detect a specified effect. Under classical hypothesis testing (i.e., true experiments), the required sample size to realize a level of power to detect a treatment effect of desired magnitude should be estimated. Calculation of the appropriate sample size primarily depends on the underlying distribution (i.e., variation) of the sample values for the dependent variable, significance level (i.e., α), and minimum effect size to be detected. The concept of effect size is part of study design considerations prior to sampling and after analyzing the collected data. In the effort to determine appropriate sample size prior to conducting a study, investigators need to determine the minimum effect

size of scientific interest. This determination is not a statistical decision but one that must be made by the investigator. Such a decision is important in the context of other constraints to sampling effort and should be made with a realistic expectation that one would expect to find at the conclusion of the study.

Appropriate sample size can be computed explicitly via formulas or through simulations. Use of previously collected data for the variables of interest in the system being investigated can be used to explicitly calculate the necessary sample size. Use of data from preliminary (i.e., pilot study – a preliminary period of reduced data collection using the proposed study design) studies and literature values can be used to estimate necessary sample size. Tragically, many published and most unpublished studies with nonstatistically significant findings contain statements apologizing for such findings and blaming it on the lack of a sufficient sample size. Investigators should strive to avoid such situations to the extent possible because concluding that results from a wetland study are essentially meaningless due to insufficient sample size adds little to scientific process and squanders precious resources and time.

Assessment of the sample size and statistical power to measure an effect is critical to study design. Both of these aspects require an acceptable measure of precision. Therefore, one needs to measure or estimate the level of variation associated with each dependent variable to evaluate the ability of the proposed study design to produce meaningful results. One can accomplish this either through use of values in the literature or conducting a pilot study. There are numerous formulae and approaches for sample size determination and determination of power; many of which are available as calculators on a variety of websites or in statistical software packages. There are a number of on-line and software sample size calculators (see http://www.epibiostat.ucsf.edu/biostat/sampsize.html?iframe=true&width=100%&height=100% for a comprehensive list of available programs). The investigator needs to apply the formula appropriate for their particular study design. There are a minimum of three categories of data that need to be determined or estimated for most sample size formulae – effect size (i.e., the biological effect that one desires to detect, usually represented as probability), a measure of variation related to the dependent variable, and alpha level.

The initial step in determining a necessary sample size is to use the appropriate sample size equation. There are equations for nearly every use of sampling scheme to estimate a population parameter or detect a difference. In addition, there are variations for many equations depending on whether the estimate of variation of the dependent variable is from a pilot study, literature, or known population value (very rare in wetland field studies). Examples of situations where sample size calculations are available include (1) estimation of a population mean, (2) estimation of a population proportion, (3) testing of hypotheses concerning a population mean, (4) testing of hypotheses concerning a population proportion, (5) testing mean differences between two or more populations, (6) testing difference in proportions between two or more populations, (7) testing main and interactive effects in traditional experimental designs, and (8) conducting human dimension survey studies.

Estimation of **effect size** following data collection and statistical analyses is a relatively recent addition to reporting of results from wetland and other natural resource studies, but estimation of treatment effect provides additional evidence and weight for conclusions developed during the study. It is a relatively simple concept that should not be made any more complicated than necessary regarding the magnitude of any found effect. Any significant statistical test reported in the literature should also include the magnitude of the effect to assess biological significance of the results. For example, one can achieve a statistically significant difference between means of a treatment and control population but, depending on the variation within each population, a biologically significant effect may not be an appropriate conclusion. Such an occurrence is more likely in a laboratory setting, but can also be found in field studies. Effect size can be as simple as reporting the percent mean change due to an effect (i.e., increases or decreases by X % due to the application of the treatment). In addition, a number of indices have been developed to quantify the strength of the difference between groups (e.g., levels of independent variables). The most common effect size index is Cohen's *d* (Cohen 1988) or standardized mean difference whereby calculated effect size index values are categorized as 0.20 = small, 0.50 = medium, 0.80 = large. These indices can be calculated for a wide variety of study designs (see for example: http://www.bwgriffin.com/gsu/courses/edur9131/content/Effect_Sizes_pdf5.pdf; http://www.campbellcollaboration.org/resources/effect_size_input.php)

1.12 Other Logistical Considerations of Wetland Study

Wetlands are complex and diverse ecosystems; therefore, it is quite difficult to generalize a logistical approach that can be applied to all wetlands under all study situations. However, there are a number of common information needs to access and become familiar with prior to conducting a wetland study. It is imperative to become well-versed in the system being proposed for study beyond the immediate question being addressed. All biotic and abiotic elements of a wetland ecosystem are potential variables in a wetland study no matter whether one is investigating water quality, hydric soils, plant associations, invertebrates, or animal communities because of the ecological linkages among all elements in the ecosystem. To fully document the effects found in any study, one must consider the totality of effects on all elements of the wetland, which can only be accomplished via a thorough ecological understanding of the system being studied – including the potential ecological states of the system under the environmental variation potentially affecting the wetland that may differ from the state measured during the study.

To define the study population of wetlands and the potential scope of inference relative to research results, one needs to have knowledge of the spatial scale of occurrence of the wetlands of interest. There are a number of sources of wetland occurrence, but quality of locations and associated information varies greatly. Nearly all available mapped locations of wetlands are provided as electronic data

files that can be used and manipulated using software associated with Geographic Information Systems (GIS). Historically, most wetlands (e.g., prairie potholes, coastal marsh) have been identified and mapped by the U.S. Department of Interior, Fish and Wildlife Service, National Wetland Inventory (NWI) (http://www.fws.gov/wetlands/). The data from NWI are available electronically (http://www.fws.gov/wetlands/Data/index.html) and used to produce periodic status and trends reports of wetlands in the United States (e.g., Dahl 2011). Other potential sources of wetland occurrence include individual joint ventures associated with the North American Waterfowl Management Plan that focus on conservation of wetlands for migratory birds, state-specific land cover data bases, U.S. Geological Survey topographic maps (http://nationalmap.gov/ustopo/index.html), state highway departments, and U.S. Department Agriculture, Natural Resources Conservation Service (primarily at state and county levels) wetland determination and soils mapping data. Most states and some nongovernmental organizations have layers of GIS data available on regional location of wetlands; however, at times it requires some searching to find the storage locations of these data.

Because of the variation among wetland types and, to some extent, within a wetland type, it is important to fully describe the wetland(s) under study. The two primary wetland classification/description approaches are the Cowardin et al. (1979) and hydrogeomorphic methods (Brinson 1993; Chap. 2 of Vol. 3). The Cowardin method was developed to serve as a "classification, to be used in a new inventory of wetlands and deepwater habitats of the United States, is intended to describe ecological taxa, arrange them in a system useful to resource managers, furnish units for mapping, and provide uniformity of concepts and terms. Wetlands are defined by plants (hydrophytes), soils (hydric soils), and frequency of flooding. Ecologically related areas of deep water, traditionally not considered wetlands, are included in the classification as deepwater habitats" (Cowardin et al. 1979: 1). This classification approach is used by NWI and all users of these data need to be familiar with the Cowardin et al. system. The hierarchical approach uses System, Subsystem, Class, Dominance Types, and Modifiers, and understanding these is important to maximize the use of NWI data and describe the study wetland using a common language. The hydrogeomorphic classification approach emphasizes hydrologic and geomorphic (i.e., abiotic) controls for wetlands using the three components of (1) geomorphic setting, (2) water source and its transport, and (3) hydrodynamics (Brinson 1993). The geomorphic setting refers to the topographic location of the wetland within the surrounding landscape. The types of water sources are precipitation, surface/near surface flow, and groundwater discharge. Hydrodynamics is the direction of flow and strength of water movement within the wetland. A variety of descriptive terms are available to classify each wetland using this approach. There are many other classification approaches that have been developed, and local wetlands experts should be consulted to determine what approach might work best for the wetlands being studied.

Hydrology is the dominant force driving ecological mechanisms and patterns within wetlands. Abiotic factors represent indices to the hydrology of the wetland and biotic elements represent the response to wetland hydrology. All ecological

functions are ultimately influenced by wetland hydrology; therefore, it is critical for investigators to fully describe the relative source and fates of water for the study wetland type. It is not necessary to provide a detailed water budget but a general depiction of water dynamics assists in understanding the context of wetland studies. In a similar fashion, one must describe as completely as possible the watershed or drainage area associated with study wetlands in terms of size, soil types, land use, anthropogenic features, and any proposed future changes in these characteristics should it be pertinent to the study.

Once a study question and associated hypotheses have been developed, an investigator must define several characteristics of the study. The temporal period for sampling must be appropriate for the question. For example, one would not attempt to test habitat selection for breeding birds during a nonbreeding season. Therefore, it is crucial to understand the life cycle of any species of interest and responses of each species to changing wetland conditions to avoid sampling during unsuitable periods or environmental conditions. Investigators must carefully list potential dependent and independent variables that will provide the most parsimonious information relative to the proposed hypotheses and study objectives. Typically, but not always, dependent variables are defined by the hypotheses and objectives; however, potential independent variables are not as obvious and require a great deal of thought prior to finalizing a study design. Concurrently, confounding and covariate variables must be identified and addressed to avoid any unwanted influence by nuisance variables on the study results.

Finally, in any study design, the project budget must be known with all of the associated restrictions and time sensitive requirements. No study is possible without funding and continuous accounting of project expenses is necessary to avoid situations that would jeopardize the study. It is usually a benefit to keep all investigators and observers informed regarding the budget status to assist in future planning for efforts related to the study. It may be prudent early in the study to develop a number of contingency study plans in the event of unexpected conditions such as loss of funding, natural disasters, destruction of equipment, or greater than anticipated costs. This would allow for the salvage of some information should the study go awry rather than being a complete loss.

1.13 Summary and Additional Considerations on the Application of Study Design

Designing a research project requires a thorough understanding of the wetland system being studied, the question or issue of interest, and those variables that must be measured to address the biological question of interest and test competing hypotheses. Following establishment of the study question (i.e., objective[s]) and associated hypotheses, investigators should then develop the methodology of the study. It is the methods of a study design that attempts to remove any investigator

bias relative to the investigation. Clear, concise, and definite methodology must be produced to not only guide one's study but allow future investigators to replicate and reproduce the original study. During this step, the investigator determines the study population, limits to inference, available resources (e.g., funding, personnel), sampling approach, and initiates an evaluation of the literature relative to the system to be studied. The succeeding step is to state the independent and dependent variables of interest. Each must be expected to have a measureable response or linkage to the treatments, disturbance, impacts, developed hypotheses, or other elements of interest. Again, careful consideration must be made to not attempt to measure all possible variables but only those that are meaningful and unrelated (i.e., not correlated and thus redundant). Pertinent variables can be considered based on literature, prior experience, and results from a pilot study. In addition, one should identify pertinent covariates at this stage so that the influence of typically nuisance or potentially confounding variables can be minimized through appropriate design.

Usually there are a number of potential methods or approaches available for recording data relative to a specific variable. It is most appropriate to choose a measurement method guided by the questions and objectives of the study. Other aspects to consider include techniques used in comparable studies to which collected data will eventually be compared, availability and cost of equipment, type of data being recorded, precision of the measurements, and identification of any identified biases relative to proposed methods. Development of clear, structured, and reliable data recording forms cannot be overstated. Such forms are typically the foundation for data recording, storage, and transfer. A considerable amount of data and information loss occurs with the use of poor data forms. Basic to all data forms are: (1) information to be collected, (2) data collection strategy, (3) order of data recording, and (4) structure of data recording (Levy and Lemeshow 1991). All data must be recorded in a meaningful and legible format that minimizes the probability of recording error. Order of recording data is important for efficiency of data collection and subsequent transfer to electronic format (i.e., data bases). Order of data recording simplifies data collection and minimizes observer effects. The recording structure includes a condensed explanation for sampling protocol that can be referenced in the field; use of "check" boxes or other approaches to minimize mistakes in recording data; and development of variable "keys" to define any shorthand notation or acronyms that can be used on the data sheet. Observers must be trained to consistently complete data forms. Furthermore, during the publication process, a common reason to reject manuscripts is due to poor or inappropriate methodology; therefore, careful consideration of variables to be measured and how to measure and record the variables is necessary for a successful investigation.

Use of pilot studies or some sort of preliminary data collection is recommended, especially for studies where the investigator has little or no experience. As previously mentioned, a pilot study can be used to estimate the variation within data recorded for dependent variables. In addition, a pilot study should be used to develop a suitable sampling protocol allowing all observers to be trained and become familiar with methodology and recording data. Observer bias can be recognized and accounted for by using the results of pilot study usually which

leads to increased training sessions for data collection and recording. Identification of potential nuisance variables may be a result of a pilot study. Finally, data from a pilot study should be subjected to the proposed statistical analyses to identify potential issues such as deviation from statistical assumptions, probability distribution of independent variables, and identification of correlated independent variables.

A primary approach to testing data quality is the resampling of a subset of each data set and comparing results. Other proposed actions to ensure data quality include redundant measurements by two or more observers or analysis of duplicate laboratory samples. Recorded data should be proofed shortly after being measured (e.g., completion of sampling period) by an independent observer to eliminate recording and transcription errors. Researchers should assign unique study responsibilities to each observer so that mistakes or errors can be linked to unique individuals and thus, provides an opportunity to correct any incorrect data.

During data collection and at the conclusion of the study (for long-term studies or monitoring efforts at frequent intervals), a final proofing of collected data is essential. In addition, numerous copies of the data forms and electronic data bases need to be made and stored in secured areas for future reference. Data collection points, plots, or units need to be uniquely identified and locations recorded to assist in interpretation of results, recollection of lost or erroneous data, and for ease of relocation for future investigators or return for a comparison study by current investigators. All equipment must be removed from the field, maintained/serviced, and again tested for accuracy. During the data analysis and hypothesis testing stage, a number of additional steps are necessary. Investigators should use statistics to describe the data (e.g., means, measures of variation, missing data, distribution form, and range of values) prior to primary statistical analyses. Graphical representation of the data prior to analyses is appropriate as long as the subsequent planned analyses are not altered due to perceived patterns observed in the data. Researchers should also realistically assess the sample size relative to planned analyses. For example, some multivariate and modeling approaches require relatively large sample sizes that may not be present. All statistical tests have underlying assumptions (e.g., normality, constant variance) and, although most approaches are relatively robust to at least minor violations of assumptions, it is desirable to test the assumptions for each analysis. It is recommended to use a statistical test that is appropriate for the data rather than alter the data through transformation or some other technique just to use a particular statistical test.

Interpretation and eventual publication of the study results represent the conclusion of a study design. Interpretation of results includes not only describing the data and subsequent analyses but also discussing the relevance and context of the results relative to previously published information (i.e., literature). Care must be taken not to inappropriately extend the inference of the results beyond the study population. Here, one must guard against letting the data and analyses determine the study conclusions rather than using the data and results from analyses to support a conclusion based on the accumulation of evidence. Fixation on statistical results is a poor substitute for critical thinking of the results in an ecological context. Reliable conclusions must be supported by data and be capable of withstanding

future study results using similar methodology to address a common question. No study is complete until publication of results in an accessible source – preferably in a peer-reviewed scientific journal. Conducting research without publication hinders the scientific process and can be considered an inefficient use of resources requiring future investigators to unknowingly recreate a study that delays the scientific process.

As a final point, there is more than one way to conduct a study, test a hypothesis, measure variables, and generate results. Therefore, when judging the merits of results from a study, investigators should independently assess the hypothesis, methodology, study design, statistical approach, and conclusions reached based on results without regard to how they would have conducted the study. One must consider the evidence in its entirety, not just those bits and pieces that support a preconceived conclusion. At all times, the scientific responsibility is to advance our understanding of natural systems, including wetlands, based on the accumulation of evidence from all reliable sources.

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Student Exercises

Classroom Exercise

In wetland studies, there are usually a number of acceptable study designs to generate knowledge regarding an observed ecological pattern or process, effects of management or anthropogenic impacts, or approximation to a desirable condition or state. The key is use of a defensible study design that allows an investigator to make reliable conclusions and inference from the results of data collection and statistical analysis. Use of critical thought through the study design process prior to data collection will ensure dependable results that can be used to advance understanding of the wetland system being studied and hypotheses being tested.

Many wetland systems are actively managed for certain ecological responses through application of specific environmental conditions; for example, water-level manipulation. These ecological responses are typically production of food resources (e.g., seeds, tubers, invertebrates) for wetland-dependent wildlife. Development of management prescriptions to maximize food production typically requires a set of manipulative experiments to test wetland response to a variety of different environmental conditions. However, measurements of food resources in wetlands can occur without manipulated experiments by relating (e.g., correlated) resource production to observed environmental conditions. Such an approach does provide some evidence of influential variables relative to production of food resources, but lacks rigor to produce a complete understanding of causal relationships. Therefore, it is crucial for investigators to properly design studies of appropriate rigor to generate knowledge of sufficient scientific quality to meet the study objectives.

When managing wetlands for wildlife-forage resources, characteristic environmental conditions that are frequently tested include frequency and timing of wetland drawdowns (dewater to expose soils and sediments) and flooding that affects soil moisture and temperature; oxygen content in soil and water (i.e., aerobic vs. anaerobic conditions); and nutrient availability (e.g., nitrogen, phosphorus). Typically, investigators collect and measure invertebrate and plant response to (1) determine species composition in response to treatments and (2) estimate available biomass of forage resources. In addition, relative composition, distribution, and variation among studied wetlands of source populations (i.e., seed and egg banks) for food resources are characteristically considered influential on results but

not a primary interest in a study. Finally, the wildlife species of interest are enumerated in some manner to evaluate the response to available food resources. Much of this volume is devoted to descriptions and recommendations for collecting ecological field and laboratory data for wetlands. The purpose of this exercise is to develop a hypothetical field study of wetlands including development of experimental treatments, objectives, and testable hypotheses.

A public land manager has developed 16, 10-ha wetland units on the floodplain of major river in the southwestern United States. Each unit has been laser-leveled to (1) allow ease in flooding and draining each unit using water-control structures and (2) create a relatively uniform elevation across each unit. Each unit can be manipulated independently, but up to four adjacent units can be manipulated simultaneously. The goal of the land manager is to maximize annual production of natural foods for migratory birds, which use the units for migration and wintering.

The four treatments of interest that coincided with availability of water for flooding include a (1) control, (2) early growing-season drawdown, (3) late growing-season drawdown, and (4) early growing-season drawdown with a late growing-season flood to achieve soil field capacity. All wetland units can be flooded at any time during the migratory and wintering period.

Working in small groups, design a study to test the effect of treatments on forage production and wildlife use of the wetland units. Methodology to measure variables does not necessarily need to be included. In your study design include a description or response to the following questions or statements:

1. List 2–4 detailed study objectives
2. Provide at least two testable research hypotheses or predictions
3. Define and describe a study control
4. Provide a minimum of three dependent variables and three independent variables and the units of measurements for each
5. Describe a strategy for allocation of treatments among wetland units
6. Define the sample frame, study population, and extent of inference from the generated results.
7. Describe a potential sampling strategy for each objective
8. Include a statement on data management and storage

Chapter 2
Wetland Bathymetry and Mapping

Marc Los Huertos and Douglas Smith

Abstract Bathymetry is the measurement of underwater topography. In wetlands, development of bathymetric maps can have many applications, including determining water storage capacity and hydroperiod (depth and timing of flooding) of a wetland, assisting with wetland design and restoration and land use planning, and facilitating legal boundary determination. This chapter provides practical steps for mapping and modeling wadeable wetland bathymetry. By characterizing the bathymetry of wetlands, investigators can better understand key hydrologic, geomorphologic, and ecological processes of wetlands. Using standard survey equipment, investigators can plan and implement a relatively simple survey of wetlands. These data can be used to model and quantitatively analyze the surface area, volume, and bottom topography (bathymetry) of wetlands using standard geographic information system software.

2.1 Introduction

Bathymetry is the measurement of underwater topography. The word is a combination of two Greek words: *Bathus*, which means "deep", and *metron* or measure. Bathymetric maps may be charts with various depths printed at specific locations,

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contour lines of equal depth (depth contours or isobaths), or digital elevation models showing bathymetry in shaded relief. Historically, bathymetric maps were used for navigation (i.e., to prevent ships from running aground), but as field biology and environmental sciences have developed, bathymetric mapping has been applied to address a range of hydrologic and ecological questions in wetlands. Wetland bathymetric maps have many applications, including determining water storage capacity and hydroperiod (depth and timing of flooding), assisting with wetland design and restoration and land use planning, and facilitating legal boundary determination.

Hydrologic conditions in wetlands were typically monitored by determining wetland water level at a fixed point near the deepest part of a wetland. However, water level alone tells us very little about the distribution or evolution of hydrologic conditions in a wetland, and how these conditions influence physical, chemical, and biological characteristics. The usefulness of long-term data sets of wetland water levels would greatly increase if the data described not only the depth of water at a point in the wetland, but also the amount of total wetland areas that was inundated at a specific time (Haag et al. 2010). A survey of wetland bathymetry and the surrounding topography can help us understand how water moves through the landscape, and more specifically, how water influences the hydrologic budget of the wetland. The water budget of a wetland depends on the input and output of water where the storage capacity of the wetland and bathymetry determines storage. In addition, wetland bathymetry will influence residence time, flood retention, sediment trapping (Gallardo 2003; Takekawa et al. 2010), and regional surface and ground water interactions (Poole et al. 2006). Bathymetry plays a key role in plant and animal community dynamics (van der Valk 1981; Ripley et al. 2004) and wetland biogeochemistry (Faulkner and Patrick 1992). For example, the depth of the water and hydroperiod can control the presence-absence of taxa (van der Valk 1981; Bliss and Zedler 1997) and their interactions (Pechmann et al. 1989; Corti et al. 1997; Karraker and Gibbs 2009). In particular, the depth of the water may control vegetation dynamics, such as the establishment and growth of various emergent or floating plant species (van der Valk 1981; Keeley and Sandquist 1992). In summary, with adequate bathymetric maps, we can develop a description of the dynamic changes in wetland conditions instead of a simple snapshot (Takekawa et al. 2010). Moreover, we can translate periodic and widely distributed water-level measurements into a regional view of wetland hydrologic status (Lee et al. 2009).

This chapter introduces several survey strategies and methods for measuring wetland bathymetry, and discusses their attributes and limitations. An overview of the use of geographic information system (GIS) is also provided to assist students with an understanding of how to analyze typical bathymetric measurements (Fig. 2.1). Finally, we include an exercise at the end of the chapter that uses a pre-existing survey data set to provide students with experience in using GIS to analyze bathymetry measurements of a wetland.

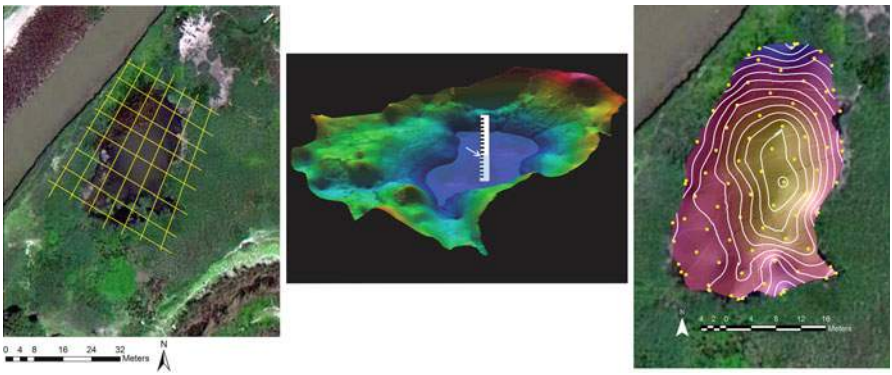

Fig. 2.1 Physical surveys of wetlands and wetland ponds can be used to develop digital models that have the advantage of being visually information-rich and rigorously quantifiable. Popular geographic information system (GIS) software was used to visualize the wetland bathymetry

2.2 Planning for Measuring Wetland Bathymetry

There are three criteria that must be considered when selecting an appropriate method to measure the bottom depths of wetlands: (1) desired accuracy, (2) wetland type, and (3) available resources (e.g., field gear and technology). The level of accuracy of the bathymetry measurements will determine the types of resources needed (e.g., field gear, instruments, and software) as well as the amount of time invested in collecting the measurements. The goals of the survey and mapping project determine the relative accuracy needed to complete the bathymetric analysis. For example, if the goal of the bathymetric analysis is to determine the water storage capacity of a wetland, the relative accuracy for collecting these measurements would be considered low. In contrast, the level of accuracy for collecting the measurements to determine sedimentation rates into a wetland would be considered high. Other goals such as determining the water budget and hydroperiod of the wetland would require medium accuracy.

In general, the type of equipment and time required to collect bathymetry measurements will be limited by whether the wetland is wadeable or non-wadeable. Wadeable wetlands are shallow enough to safely traverse, while non-wadeable wetlands are too deep for wading or may contain a substrate (e.g., muck soil) that is too difficult for walking. In the case of wadeable wetlands, the bottom topography can be measured using a meter tape, hip chain, rotating laser, total station, handheld global positioning system (GPS), or survey-grade GPS. Large wadeable wetlands ($\sim$ 0.5 ha) are usually treated similarly to non-wadeable wetlands, and the tools used are determined based on labor and efficiency.

Non-wadeable wetlands are typically large in aerial extent and the tools used to measure their bathymetry include boats that deploy a lead line or a sonar system for depth coupled with either an optical survey or a GPS system for positioning

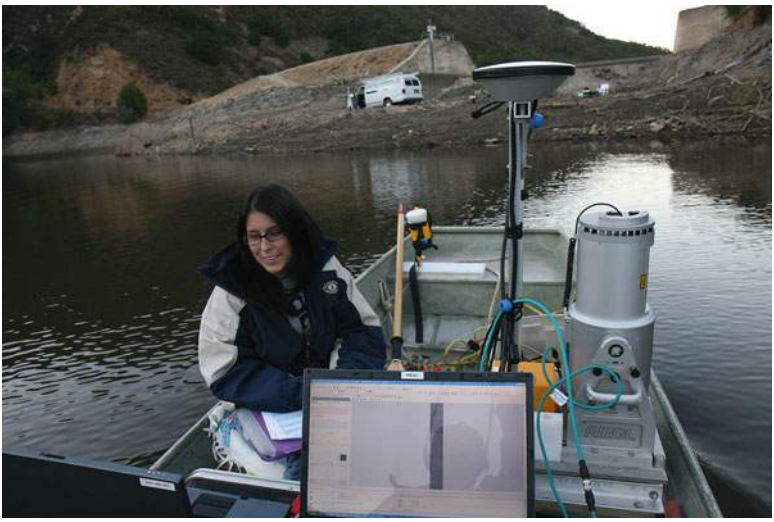

Fig. 2.2 Graduate student operates a vessel-based terrestrial LiDAR unit as she creates a precise digital elevation model of the Los Padres Reservoir in the Carmel River watershed (Published with kind permission of © Rikk Kvitec 2014. All Rights Reserved)

(Fig. 2.2). Of course, some wetlands have special constraints. For example, quaking bogs are challenging because a portion of the water column is inaccessible from the surface, and might require the use of sonar or SCUBA.

Each survey involves collecting the position (X, Y) and elevation (Z) of a number of points on the landscape. The surveying equipment available to collect these data includes measuring tapes, meter sticks, lead lines, stadia rods, survey levels, laser levels, handheld GPS, total stations, survey grade GPS, and both ground-based and aerial LiDAR. A description of each type of equipment is provided in Table 2.1. The instrument selected will be determined by the factors listed above. We present the survey techniques that are applicable to most wadeable wetlands in order from simplest and least expensive to complex and most expensive. We also provide a description of LiDAR technology, which is most suitable for broad wetland environments such as estuarine tidal flats and large scale hummocky environments without vegetation.

Once the data are collected, there are several software packages available for creating maps from the raw data as well as conducting geomorphic analyses. Two of the common professionally used packages are ESRI ArcGIS and Fledermaus. Both are relatively easy to use. Fledermaus has more flexibility for rendering digital hillshade models, which can export models that can be viewed and rotated in a free viewer.

Defining the range of questions to be addressed in a survey of wetlands may help determine the type of methods to be used. Although the relative accuracy is important, other parameters should be defined. For example, will the field survey work be repeated over time? If so, then it may be important to set up permanent markers or benchmarks so the same transects can be used at a later date. Will the

Table 2.1 Description of commonly used survey instruments for collecting bathymetry data

Survey instruments	Description	Data collected	Notes
Meter stick	Layout orthogonal grid and measure the water depth at grid nodes with a meter stick	Depth, using ambient water surface as the datum. Depth precision of 1 cm is typical	Horizontal positions obtained with 50 or 100 m tapes and cross section stakes. Technique is limited by ambient water depth, so surveying at high water will capture more data. If water is flowing, the water surface (datum) will not be constant through space
Survey level	Lay out orthogonal grid for measurement. Horizontal telescope pointed at a calibrated leveling rod. "Autolevels" are able to fine-level themselves after initial coarse leveling	Elevation precision of 1 cm is typical	Horizontal positions obtained with 50 or 100 m tapes and cross section stakes. Requires two surveyors
Rotating laser level	Same as above, but rotating laser level emits a horizontal plane of laser light that strikes a receiver on a calibrated leveling rod	Same as above	Same as above. Can be operated by one person
Handheld GPS	Wide range of models available. Trimble Explorer is a common model	Coarse horizontal position and very poor vertical positions. Post-processing typically achieves 2 m horizontal precision	Used for horizontal mapping. Should not be used for vertical positions. Data are georeferenced
Total station	Spherical 3D points are obtained through laser-based distance measurements and ultra-precise horizontal and vertical angle measurements. Laser is shot at a pole-mounted prism	3D coordinates with 1 cm precision. With experience and research-grade equipment, sub-centimeter precision is possible	Total stations can obtain a precise 3D fix in a few seconds, so hundreds of positions can be shot in a day
Survey grade GPS	3D points are obtained by a tripod-mounted GPS antenna, with reference to satellites and either a local base station or regional set of base stations	3D position with precision limited by local conditions. One centimeter precision is possible under ideal conditions	Surveys are commonly achieved with a "rover" antenna that corrects positional errors by radio communication with a "base" antenna. This provides "precise" positions. "Accurate" georeferencing is achieved by correcting the base station data using fixed GPS stations in the region
Ground-based LiDAR	Tripod-mounted, vessel-mounted, or vehicle-mounted LiDAR scanner	Millions of 3D points are collected in a LiDAR scan. The precision is limited by vegetation, water, and GPS positioning (if it is a mobile system)	Excellent for expansive wetlands, such as large tidal marsh environments. Very complex data collection and post-processing system. Very expensive to purchase and operate

Table 2.2 Various products that might result from a bathymetric analysis

Mapping dimensions	Products
Two dimensions (X and Y)	Wetland perimeter and area (derived from ground surveys, aerial imagery, topographic maps, vegetation, soils)
	Total area
	Reference elevation and elevation datum (benchmark/ground control)
Three dimensions (X, Y, and Z)	Wetland bottom elevation (Z) at various X, Y locations
	Wetland water level (stage) and water depth
	Outflow elevation and potential surface connections with other wetlands (outflow/inflow)
	Wetland drainage basin boundary

wetlands map need to be placed in a larger geographical context with real world coordinates (e.g., longitude and latitude)? If this is the case, then GPS technology will play a role in the field, and you will use GIS tools and map projections to ensure the data are accurately georeferenced to real world coordinates. If the questions about the wetlands include biophysical features (e.g., vegetation patterns, geologic features, or evidence of animal activity) associated with the bathymetry, the survey work might need to map those features too. Successful characterization of the bathymetry will be guided by clearly defining the products that will be needed for conducting the bathymetric analysis (Table 2.2). The products desired will dictate which mapping dimensions will be needed and what type of analyses will be conducted. Finally, the available resources and budget will ultimately constrain what can be accomplished. In general, a well-defined question will result in efficient use of field and analysis time.

The goals of the wetland survey will dictate the boundaries, number of survey points, and needed resolution. The boundaries of interest may be defined legally (i.e., a jurisdictional wetland) or may include a larger context (i.e., the watershed contribution). In either case, the mapped wetland should include enough area outside the wetted area to avoid interpolation inaccuracies near the defined boundary of interest. "Resolution" is a broad term, generally indicating the smallest physical feature visible in the data set. Resolution is a function of survey point spacing, with higher resolution achieved by closer spacing of elevation data. Your choice of resolution will depend in part on the sources of variation in the wetland itself and how much of that variation needs to be captured, the number of sampling points you can afford to survey, and the precision of individual survey shots. For example, Haag et al. (2005) found that bathymetric data containing a high density of data points provided the most useful stage-area and stage volume-relations characterizing isolated marsh and cypress wetlands in Florida. Moreover, bathymetric maps generated from a low density of data points underestimated by 50–100 % the wetland area and volume over certain ranges of stages compared to maps generated by a high density of data points. This emphasizes the importance of collecting data from an appropriate number of data points when determining wetland bathymetry. From a pragmatic perspective, the size of the wetland feature,

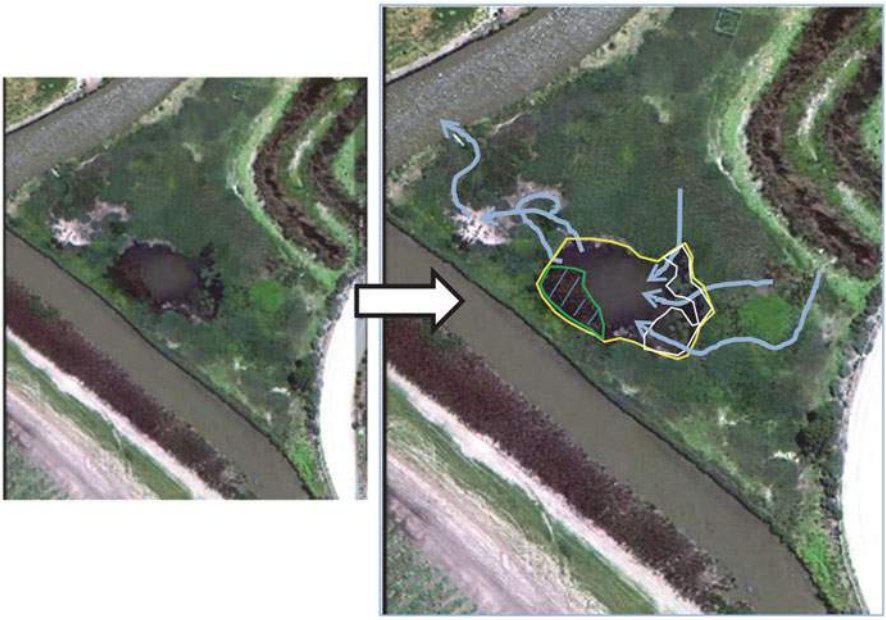

Fig. 2.3 An annotated aerial photograph in the project file indicates that Molera Wetland is a riverine wetland fed by a high water table and upland sources. *Outlines* show the areal extent of vegetative ecosystems present in May 2011, when the photo was taken. Surveys can delimit the true flow pathways and other details

number of sampling points, and resolution will all influence the resources needed to complete the work and ultimately, the quality of the results.

Planning, and obtaining the appropriate resources will increase mapping success. Prior to conducting a survey, we suggest you gather contextual data sets such as U.S. Geological Survey (USGS) topographic maps (http://topomaps.usgs.gov/), historical and recent aerial photographs, and regional digital elevation models. Many of these are available through public databases often overseen by state agencies in the U.S. This overview analysis can help determine the general surface flow patterns that fill and drain the wetland. Review of historic aerial photos can highlight temporal changes, such as gradual infilling of wetlands (sedimentation or land use changes), ecological shifts (such as vegetation changes), and seasonality. The broad view can help constrain the environmental questions and hypotheses, and will serve to plan the survey collection. If resources are limited for initial data collection, a quick tour through Google Earth (http://www.google.com/earth/index.html) can be a useful starting point.

During the planning and preparation process, the following parameters can be explored using the contextual data sets described above or in Google Earth. The general center of the wetland can be described in latitude and longitude or in some other coordinate system. The general setting can be described in terms of access, land use, probable disturbances, position in the watershed, vegetative types, and topography (see Fig. 2.3 as an example). The approximate elevation of the wetland

can be derived from digital elevation models, topographic maps, or Google Earth. The general perimeter and area can be estimated, which will help in planning the number of survey points and the time required for surveying. While this broad overview can also provide initial insight on the types of equipment and gear that will be required for conducting a survey, a pre-survey site visit is essential.

In most cases, it is essential to contact property owners or public lands managers for permission to access a wetland. State or county permits may be required if biological sampling is part of the plan. Advanced planning ensures that the field work will not be interrupted or postponed, potentially leading to missed opportunities related to seasonal water levels.

Finally, before beginning field work, being aware of the appropriate safety measures is important. Safety measures may include bringing a first aid kit, adequate communications devices (walky-talkies, cell phones, or satellite phone), and personal protection gear (bug spray, sun screen, hardhat, boots, personal floatation devices, and safety vests). The necessary equipment will vary with local conditions (weather, proximity to infrastructure, etc.). In general, you want to be as prepared as possible to reduce the risks of accidents or injury.

2.3 Wetland Survey Techniques

2.3.1 Recording Field Data

There are many established techniques for bathymetric and topographic mapping. They all have one thing in common: their data are only as valuable as the quality of the comments and notes that describe the methods and features being surveyed. Without clear field notes, survey data are just numbers with no context. Standard survey notes should be adhered to if the goal is long-term monitoring. Professional-quality notes should be unambiguous, and understood by anyone who tries to re-survey the site. Because some long-term monitoring projects can span generations of students and professors, consistency and reproducibility are key features of data collection. Experience shows that a weatherproof notebook (such as “Rite in the Rain”) and a #3H mechanical pencil lead will create a long-term archive of survey data and field notes. Electronic data records are in common use, but storage media change through the years, so maintaining a hard copy of original survey data and notes in the office is essential.

Information recorded in standard field notes should include date and time, site description (field sketches of key elements such as bench mark locations are valuable), purpose of the survey, type and serial numbers of survey instruments, names and roles of team members, general weather conditions, and wetland characteristics. The notes that record a specific survey should fully describe the benchmarks and datums that were used. Specific survey shots are recorded in a series of data columns. Depending upon the survey technique, these columns might

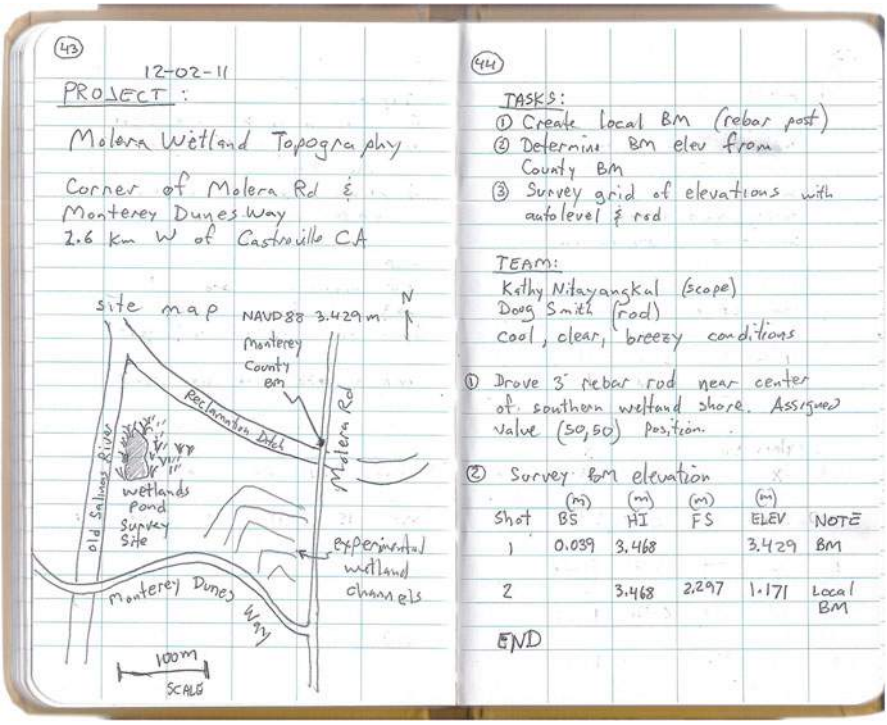

Fig. 2.4 Field book example. Note the descriptive details and map to help interpret how the survey was completed and where the benchmark (BM) is located

include, shot number (a number 1 through n), X coordinate (easting), Y coordinate (northing), Z coordinate (elevation), and a comment column where notes about each shot can be recorded. See Figs. 2.4 and 2.5 for field book examples.

The units of measurement used in each column should be explicitly noted in the column heading, and the units should not change within a column. The numerical values in the columns should reflect the precision of the measurement. For elevations, we commonly read to the millimeter, so an elevation entry might be 3.235 or 4.210, using the zero as a placeholder to show that we are still reading to the millimeter. Harrelson et al. (1994) is an excellent reference for standard environmental survey notes and abbreviations.

2.3.2 Surveying in the Field

Bathymetric survey design is driven by the goals and the available survey tools. Based on the available equipment (Table 2.1) and desired products (Table 2.2), the appropriate survey methods can be selected. For example, if the site is large, but

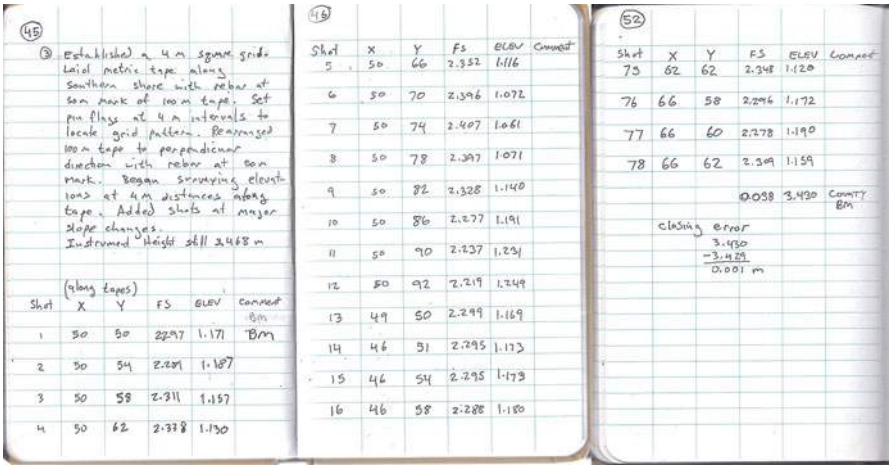

Fig. 2.5 Three pages from the field book showing clear notations for each shot taken during a survey

unvegetated, then LiDAR might be appropriate. On the other hand, if there is a dense of ground cover and the site is small, then an autolevel might suffice. Although automated survey equipment may appear simple, we have found that data processing can be time consuming, so there is a trade-off when the technology exceeds the products needed from the project. The following survey techniques can be implemented using a combination of tools and in conjunction with one another.

The goal of surveying is to collect horizontal coordinates (X, Y), and vertical ordinates (Z). These three-dimensional survey positions may be arbitrarily located in space (e.g., relative to a local benchmark, center of the wetland, or other feature), or they can be georeferenced, which means each point can be positioned on the globe relative to other features. If the data are georeferenced, the horizontal coordinates are often latitude and longitude, or northing and easting in a projected coordinate system. In the georeferenced data set, Z is elevation above sea level, referenced to a standard vertical datum. In some software packages, Z is considered "depth" below a datum such as sea level.

In all mapping projects, you must first establish a horizontal and vertical datum. The datum serves as the reference point from which all measurements will be referenced. Datums can be local (arbitrary) and based on points set in the field, such as rebar stakes, or they can be referenced to published locations. For example, the National Geodetic Service (NGS) has survey data available on the internet (http://www.ngs.noaa.gov/cgi-bin/datasheet.prl). The NGS data system provides a report on each benchmark.

A field survey establishes horizontal and/or vertical locations in relation to a starting point, which is called a benchmark. The selection of permanent horizontal and vertical reference frames is not critical if the wetland is to be surveyed and

Fig. 2.6 National Geodetic Survey Benchmark (*left*) (http://www.ngs.noaa.gov/) and local county benchmark (*right*) used for the Molera Wetland, California, USA

analyzed once. If the object of surveying is to map physical change through time, it is most advantageous to establish at least one long-term, stable benchmark near the survey site. The USGS usually uses brass monuments set in rock, a concrete pylon, or a pipe driven into the ground (Fig. 2.6). Ideally, there will be an NGS or USGS benchmark near your study area and we recommend that you use it. If not, you will need to establish a new local benchmark. This local benchmark can be a wide range of objects such as a chisel mark in exposed bedrock, nails and tin washers driven into a road, concrete pads used for street signs, fencing, or other public infrastructure. You must be sure to select or construct a benchmark that is vertically and horizontally stable, preferably for many years. All bathymetric surveys will follow these basic steps for data collection: (1) establish vertical datum, (2) establish horizontal datum, (3) record wetland water stage, and (4) measure the relative position of wetland features in the X, Y, and Z space. Developing the vertical and horizontal datums allow every survey shot to be referenced to a stationary and stable point—thus, allowing reproducibility and the capacity to measure precision.

The vertical benchmark is the starting point of any topographic and bathymetric survey. We will assume that the ultimate goal of your survey is to develop a "topographic" wetland model, which has elevation values rising in the uplands. "Bathymetric" surveying is analogous, but values rise as you descend into the wetland. Regardless of technique, all surveys will start from a benchmark. There are published benchmarks and local benchmarks that can be used (Fig. 2.6). Local (arbitrary) vertical datums are assigned elevations that are not based on published benchmarks. Assumed elevations can be based on water surfaces, elevations of fixed structures (such as outfalls or crossings), pool points, or staff plate elevations. We recommend you visit the NGS website (http://www.ngs.noaa.gov/) to access published benchmarks. The elevation datum on published benchmarks will either be referenced to the National Geodetic Vertical Datum (NGVD) 1929 or North American Vertical Datum (NAVD) 1988, which are merely vertical scales with a 0 m mark that corresponds to an estimate of mean sea level. You must recognize

that even those scales are somewhat arbitrary, given the dynamic nature of sea level at all time scales. Nevertheless, the vertical datum places the wetland site in a vertical framework so that vertical positions can be compared to one another through time, and in the case of an NGS benchmark, referenced to sea level. Using a published benchmark to determine wetland elevations on an established datum is useful for relating the bathymetric data to other data sources such as Federal Emergency Management Agency flood data, tidal ranges, USGS stream gage records, and USGS topographic information.

Local benchmarks can be established in the field for a specific wetland, and referenced to an arbitrary datum. It is common practice to drive a 1 m long, 1.5 cm diameter rebar vertically into the ground within 1.5 cm of the ground surface for use as a local benchmark. Other local benchmarks can be established with spikes, chiseled "x" on concrete or boulders, and nails in pavement. If a high order of vertical accuracy is desired, the survey should use at least two benchmarks on a common datum and check elevations between the benchmarks regularly. Using more than one benchmark is good practice when establishing control for long-term bathymetric monitoring because it allows recovery of the benchmark even if one is lost. A rebar benchmark can be found (recovered) in future surveys even if it is buried by sediment in intervening years. Carefully sketched maps, GPS locations, a shovel, and a metal detector are standard tools for locating benchmarks.

A "staff plate" (sometimes referred to as a staff gauge) is an acceptable supplemental vertical benchmark (Fig. 2.7). A staff plate can be installed using a graduated meter stick extending vertically from the bottom of the wetland that allows for determination of the water surface elevation. The staff plate can be mounted to a piece of lumber or a metal stake that is driven into the substrate for stability. The 0 m mark on the staff plate is another arbitrary vertical reference for surveys and recording data. The water surface elevation can be converted to NAVD 88, or other external references if the 0 m mark (or any other mark) is related to the external reference by surveying to a nearby-published benchmark or via GPS survey.

If a local benchmark will not be referenced to NAVD 88, you can assign a convenient starting elevation to the benchmark. Standard practice is to select a round elevation value (e.g., 10 m), with the constraint being that it is high enough to keep all the elevations in your survey positive. This practice reduces the common math errors stemming from the use of negative numbers. Otherwise, it is strictly a matter of convenience. Any other arbitrary value will do, as long as you record it in your survey book as reference for future surveyors.

The next step will be to establish the horizontal datum and axial framework. The horizontal position of each elevation point in the survey must be recorded. The position can be considered a point in a Cartesian coordinate system (X, Y). The coordinate system requires defining the physical position of at least one reference point, and the direction of one of the axes (X-axis or Y-axis). The direction of the other axis is taken to be orthogonal from the first. If you are working in latitude and longitude (or UTM coordinates), the Y-axis is defined as the direction to true north.

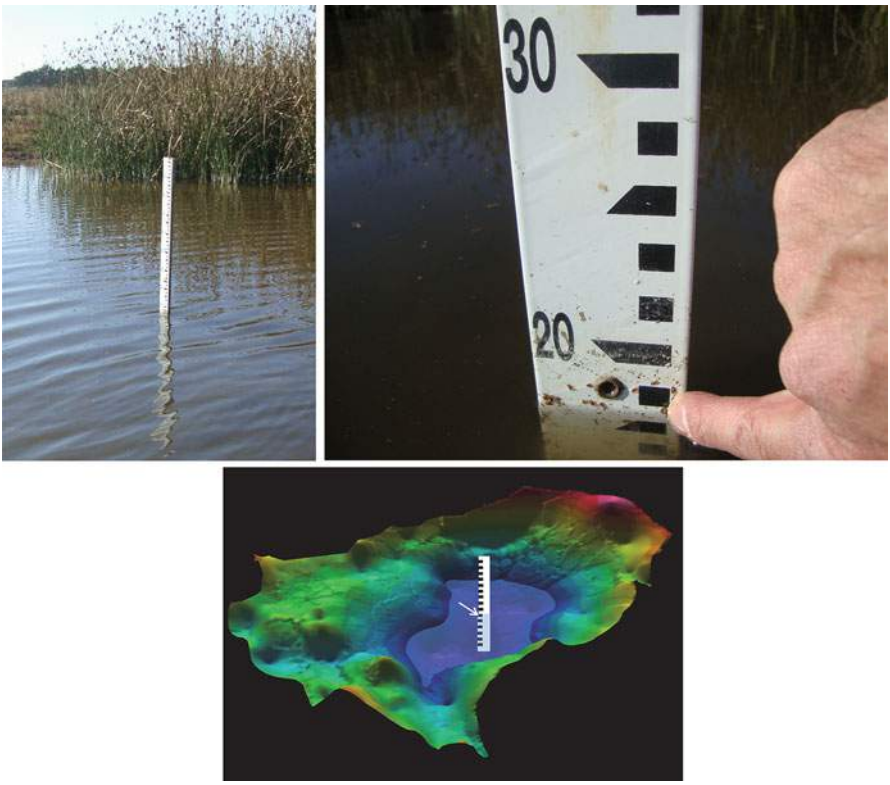

Fig. 2.7 Staff plates indicate the water level elevation with respect to the 0 m mark on the staff plate. The color blocks are 0.01 m tall, and each decimeter is numbered. By interpolating to the millimeter, the water level in the figure is 0.165 m. The staff plate can also be used to relate models with field measurements

Local (arbitrary) horizontal datums are not georeferenced to a standard map projection or coordinate system. Local horizontal datums are assigned arbitrary values for horizontal position relative to a local benchmark to which you have assigned a convenient position such as (0,0), (50,50), or (100,100). As with local elevation datums, it is standard practice to make the coordinates sufficiently large so that the positions will be positive values. Local horizontal datums can be based on fixed structures, such as described for vertical datum, or other objects such as rebar, bridge corners, culverts, or trees. A local horizontal datum allows all measured points to be placed in horizontal space relative to one another, but not necessarily referenced to other datasets that may be available. The other requirement for local horizontal datum to work is to define the direction of one of the axes. It is most convenient to define one of the axes to be the long dimension of the wetland. The axial directions can be recorded as compass bearings from a precise compass, such as a pocket transit.

Georeferencing places the survey shots into a geographic framework, such as latitude and longitude. Georeferencing can be achieved by surveying from a published benchmark, or by placing a GPS antenna above the local benchmark. Published benchmark datasheets will list horizontal coordinates for NGS benchmarks. These published coordinates are either scaled from a USGS topographic map or referenced to State Plane Coordinates based on a horizontal datum such as North American Datum (NAD) of 1983 (NAD83) or of 1927 (NAD27). Although many features are still referenced to NAD27, NAD83 is the official North American datum. It is important to note which datum is used if you plan to make maps that are spatially referenced. The horizontal datum places the wetland site in X, Y space and serves as the initial point from which all measured features will be referenced. Similar to the vertical datum, using a published horizontal datum allows the bathymetric data to be related to other sets of spatial data available.

Once the datum have been established, measurement of wetland features that define the wetland topography can be initiated. The process of measuring wetland feature locations will vary based on the technique employed. The basic bathymetric survey establishes the horizontal and vertical location of points throughout the basin relative to the benchmark. To measure the wetland features, one can establish points along a number of transects traversing the wetland or establish a grid of points. In general, if the wetland is a simple depression, a few transects or a simple grid might be enough to capture the bathymetric variation. But, if the wetland is geometrically complex and large, it might require more numerous measurements. For example, for such a wetland, you may need to establish survey points at a closer spacing where topographic variation is high compared to other parts of the wetland. Determining the density or number of points comes with surveying experience. One strategy is to shoot survey points at major breaks in the slope, but never farther apart than some predetermined value (e.g., 1 or 2 m). Additionally, more points should be used to define key hydrologic features such as the wetland boundary and outlets. If your subsequent analysis shows that more detail is needed, an additional field day can be used to fill in the missing information.

Depending on your objective, you may also be interested in water storage of the wetland. As such, it will also be important to record the wetland water stage while conducting the bathymetry survey because it will be the basis for determining a volume-stage relationship. Stage can be measured by either reading a staff plate (Fig. 2.7) if one is installed or surveying the elevation of the water with respect to your vertical benchmark, which can easily be accomplished at the water's edge.

2.3.2.1 Surveys Using a Taped Grid

The simplest technology for obtaining bathymetric data involves the use of a grid of points determined by long metric tapes placed at set intervals (e.g., every meter) or an equivalent method using a grid pattern (Fig. 2.8). Nodes (points) are created where the tapes cross. The nodes are sampled for elevation data. If the tape positions are referenced with rebar or by some other means, the site can be

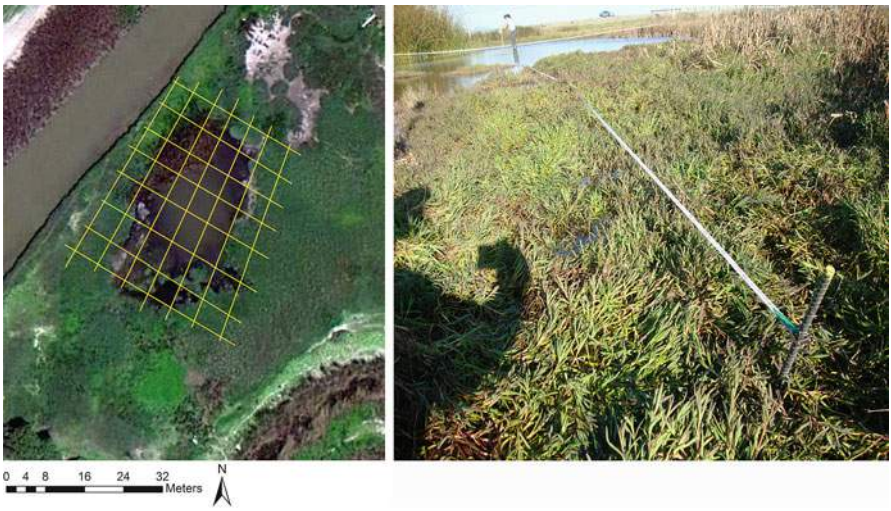

Fig. 2.8 Grid layout and photograph of baseline tape with one crossing tape

resurveyed at precisely the same X, Y points through time. If the elevations are not resurveyed at the same place, elevation differences might be related to space rather than time. The following description provides a means of locating your survey points for time series of bathymetric change (e.g., monitoring siltation levels). You can place two pieces of rebar in the ground, and stretch a long metric tape between them. One of the rebar pieces should be held as the horizontal benchmark. That point will be considered the horizontal origin for the survey and assigned X, Y, and Z coordinates. The tape stretched between rebar stakes forms the baseline, and another tape (or tapes) can be set at right angles to the baseline tape along certain horizontal offsets. For example, if the crossing tapes are set at 4 m intervals along the baseline tape, and the elevations are recorded at every 4 m along the crossing tapes, the result is a 4 m square grid of points that can be reconstructed reliably at future times (Fig. 2.8). The choice of spacing will be based upon the time limits and precision requirements of the particular project. An analysis later in this chapter illustrates the benefits of closer spacing. Also, this grid pattern can be established with just two tapes. One is the baseline tape that does not move during the survey, and the other is the "crossing" tape that can be moved along the baseline tape for each subsequent transect. If many persons are involved, you should bring more crossing tapes which will speed up the process considerably.

Once the sampling points are established, the Z value of the wetland bottom can be obtained by determining depth from a vertical reference point, such as the water surface (Fig. 2.9) or by using an autolevel or rotating laser level which will be discussed in the next sections (Table 2.1). The easiest way to measure the Z value for each point in the grid is a depth measurement using a meter stick or survey rod. In this case, the water surface is the vertical datum from which the measurement is

Fig. 2.9 An example of a researcher recording a water depth measurement at a grid node near the staff plate. Note the falling field book. By using waterproof books, this is not a problem

made, but comparisons with future surveys will be problematic unless the water surface elevation present during the survey is somehow linked to the local benchmark using a staff plate, autolevel, rotating laser level, or more sophisticated gear. Features, such as grade breaks, water surface, and vegetation changes, can be recorded along each transect line. In addition to the grid point measurements, taped locations of the shoreline (0 depth) should be recorded. As with any survey, if the grid spacing is too coarse to capture major breaks in slope or details of interest, more Z values can be collected later, along with their corresponding X, Y positions. This technique used alone will only yield topographic information below the current water level; however, the other survey instruments we describe do not have that limitation.

2.3.2.2 Using an Autolevel

When the wetland vegetation is lower than eye-level, autolevel scopes can be used to survey land-surface elevations. While autolevel scopes are suitable for obtaining precise elevations, they are very poor for measuring horizontal positions, and we

Fig. 2.10 Autolevel surveys are a series of "shots" in which a rod reading is recorded. The colored blocks are 0.01 m tall, and every 0.01 m has a number. This shot indicates that the ground elevation is 3.215 m below the optical center or instrument height (HI) of the scope

recommend using a tape grid for positioning. Once a horizontal grid is established (see previous section), the scope and tripod can be set up as close to the wetland as possible, but with a clear view of all points to be surveyed. The tripod might need to be set up multiple times in different locations if the wetland is large, or if the line of sight is limited. Moving the scope requires the use of a "turning point" in the survey to keep the autolevel scope in the original vertical reference frame. In some wetlands, the canopy may be so dense that the use of a scope is impossible. As a substitute, a compass can be used to obtain direction, and a measuring tape can be used to determine the horizontal distance from a known location.

The autolevel is convenient not only because it can measure ground surface elevations beyond the water surface, but also because it can be used under any wadeable condition. The autolevel is an optical telescope with crosshairs that is mounted on a tripod and provides a level view no matter where it is pointing (Fig. 2.10). The scope person views a leveling rod held vertically by the rod person, and records the elevation value indicated by the intersection of the horizontal, center crosshair, and an elevation value marked on the rod (Fig. 2.10). Each reading of the rod is called a survey "shot." Shots are simply measurements between the ground where the rod is placed and the optical center of the scope, indicated by the horizontal cross hair (Fig. 2.10). The standard notes for an auto level survey are shown in Fig. 2.5.

The basic autolevel set up includes the following steps. First, a surveyor must find a location where the instrument has a clear view of the wetland to be surveyed as well as any benchmarks that will be used to vertically control the survey. Next, the tripod feet must be firmly set so that the tripod top height is at a comfortable viewing elevation and the mounting bracket is approximately level. Then, the

autolevel is mounted to the bracket, without over-tightening the mounting screw. Precise instrument leveling is accomplished with reference to a bubble level as you adjust three leveling wheels on the autolevel base. When the instrument is leveled in this way, yet more precise leveling occurs within the instrument, which is the basis for the instrument name. Once the instrument is level, the horizontal cross hairs in the scope are focused for an individual's eyesight. The cross hairs delineate a horizontal plane as the instrument is rotated about a vertical axis. In other words, everything that the horizontal cross hair hits is at the same elevation. This elevation is called "instrument height," and the value is denoted "HI" in survey notes. The value of the instrument height is determined by the first shot of the survey when the rod is placed on a benchmark (BM) of known, or arbitrarily assigned, elevation. The shot used to determine the HI is called a "backshot" (BS). The instrument height is determined by summing the BM elevation and the BS reading. The HI value is assumed to remain constant, unless the instrument is moved during the survey.

Once the HI is determined, you can survey the ground elevations of any rod positions where the rod can be seen by the scope person. The shots used to determine unknown ground elevations are called "foreshots" (FS). The ground elevation for each FS is determined by subtracting the FS from HI. All that remains is to place the rod on the ground at the grid points determined by the tape grid described above, record the FS, and calculate the elevation. Then, each X and Y position will have a corresponding elevation Z. As with the water depth method, and any other method, additional shots should be taken at many places along the current shoreline and at any points required to capture the details dictated by your survey goals. The last shot of the survey is the "closing" shot. It is a FS taken with the rod on the BM on which you opened the survey. The resulting ground elevation (HI-FS) should match the real (or assigned) BM elevation. The mathematical difference between the elevation from the closing and the real elevation is the "closing error." Closing error is a measure of the precision of the survey. The source of survey error is typically due to physical changes in the tripod or tripod feet positions that change the HI or scope leveling. Given your calculated closing error, you must decide if the precision is acceptable. Based on our experience, autolevel surveys that last 1–2 h typically may have a closing error of less than 1 cm. Errors greater than 1.5 cm are uncommon. Resurveying may be necessary if higher precision is required. If low precision is acceptable, then a larger closing error is allowable.

A standard autolevel practice is to perform a "two-peg" test of instrument calibration prior to a survey or sporadically throughout the survey season. The two peg test involves firmly driving two pegs (A and B) in approximately level ground separated by approximately 30 m. The tripod and scope should be set in the middle of the two pegs and a rod reading should be recorded on pegs A and B. In your notes, the rod readings should be recorded as "a" and "b," respectively. The next step is to move the tripod and scope as close as possible to one of the pegs, but not so close that the rod cannot be read. From this location, another set of readings will be recorded with the rods back on pegs A and B, but record the shots as "c" and "d" in your notes. The difference between "a" and "b" is a measure of the difference in elevations of the tops of pegs A and B. An independent estimate of the difference

Fig. 2.11 An example of rotating laser equipment, which can be used by a single person

is the difference between c and d. Therefore, absolute value of (a-b) should equal absolute value (c-d). If they vary by more than a few mm, consider instrument calibration prior to surveying.

2.3.2.3 Rotating Level Laser

A direct substitute for an autolevel is a rotating laser level (Fig. 2.11). A rotating laser level performs a similar function, but emits a laser (usually red) that can be used to measure the vertical distance from a level plane created by the spinning laser beam. The same notes and data are taken (e.g., HI, FS, and BS), and horizontal control is still provided by the tape grid, but a rotating laser replaces the autolevel on the tripod. Some rotating level lasers have self-leveling servo motors, while others must be leveled by hand using an integrated bubble level and three leveling screws. Once leveled, the rotating laser emits a laser beam from a lens that is spinning about a vertical axis. The beam describes a horizontal plane that represents HI. There is a laser sensor attached to the survey rod that emits a beeping sound when the beam hits the sensor. The rod end is placed on the BM or wetland surface, and the rod is telescoped up or down until the sensor cuts the laser. Then, the rod reading gives the distance from HI to the ground surface as before.

2.3.2.4 Pool-Point Radial Survey Method

Modern, high-precision survey instruments can collect many three dimensional points with little effort. These types of surveys (e.g., pool-point radial survey, total station, real-time, kinetic GPS (RTK-GPS), and ground-based LiDAR), do not require an external grid for determining locations. Instead, they are able to generate a network of X, Y, and Z locations in a digital format which often leads to data sets being composed of thousands or millions of X, Y, and Z point locations that require specialized software for processing beyond the scope of this chapter.

The pool-point radial survey method relies on mapping the maximum wetted perimeter and radial transects with X, Y, and Z values. In simple wetlands, you can take relatively few measurements to obtain accurate bathymetry estimates with two relatively quick site visits. During the first visit, you insert a stake at the pool-point or deepest part of the wetland and then measure ground surface elevation and location from the pool-point to the perimeter along 3–5 radial lines. More radial lines may be required if the wetland is topographically complex. It is important to obtain an adequate number of elevation readings above the maximum height of the wetland so there are no interpolation errors near the high water mark. Later, when the water is at its highest, you survey the perimeter of the high water edge and the stake at the pool-point, which will provide the elevation of the pool relative to the maximum pool depth. This survey can be conducted with a meter tape and stadia rod, lead line, total station, rotating laser, handheld GPS, or survey-grade GPS. The simplest form of pool-point radial survey design is measuring along the long and short axis of the wetland.

If using a meter tape and stadia rod or lead line, the depth of the water at point locations are recorded along the meter tape. One limitation of using water depth as the measurement for establishing elevation is that you are limited to the area of wetland that is inundated. If the survey is conducted at the highest stage of inundation, you can maximize the bathymetric coverage. A total station or GPS that collects horizontal and vertical position can be used in either wet or dry conditions and collect a complete data set regardless of water stage. Using a handheld laser is quite rapid and efficient, as all the dry and wet measurements can be recorded in less than 20 min for a 50 m diameter wetland (Wilcox and Los Huertos 2005).

2.3.2.5 Total Station

Total station equipment ranges in functionality from basic point and shoot (i.e., aim the station at a prism pole and record the X, Y, and Z values) to fully robotic scanners (Fig. 2.12). The basic principle is that the instrument sends out a laser pulse in a known direction and calculates the position of the ground by analyzing the laser signal that is reflected back to the instrument. Some total stations require a prism on a rod to create the reflection, while others can receive the laser reflected from the

Fig. 2.12 An example of total station equipment, which can be used to rapidly capture precise X, Y, and Z points

ground or vegetation. In general, the instrument calculates the horizontal and vertical angles of the aimed laser beam and then calculates the distance to the target by analyzing the reflected light. It converts the resulting spherical coordinates into X, Y, Z coordinates for export to a spreadsheet. Although high precision positions can be used to analyze wetland geometry, the geospatial data are not referenced to any external vertical or geographic reference frame unless the survey is intentionally linked into those frameworks by incorporating published benchmarks or local benchmarks that have been referenced by GPS work. We also note that the use of reflectorless total stations is not practical in wetlands where the laser would be reflected from dense vegetation or water rather than the ground.

2.3.2.6 Real Time Kinematic GPS

The RTK GPS is a positioning system that uses two GPS receivers: a base station and rover (Fig. 2.13). The base station can be positioned over a local BM to determine its georeferenced position, or over a published BM with a known georeferenced position. If the base station is set over a local BM, it should be left

Fig. 2.13 An example of real-time, kinetic GPS (RTK GPS) equipment. The base (*right*) and rover (*left*) can communicate with each other up to several kilometers with radio signal booster. RTK GPS is efficient and requires only one operator

there for up to several hours to record GPS signals. This long record will average out most of the error associated with instantaneous GPS positions. The base station position can be further refined by differentially correcting the data to long-term GPS stations in the region. Additionally, NOAA maintains a free web service (OPUS) for correcting base station data (http://www.ngs.noaa.gov/OPUS/). The differentially corrected positions commonly have less than 1 cm of error in horizontal position and less than 2 cm error in vertical position.

The rover is a GPS antenna that is mounted on a hand-held staff of known length. The rover staff can instantly record a position throughout the wetland. The point position is automatically "corrected" in real time by radio communications with the base station. The same satellite errors affect the base station and rover, but the base station knows its position. Thus, it also calculates the time-specific errors and can correct the rover positions. At a rate of one point per 15 s (including moving from one position to another), it is possible to collect several hundred precise survey points in a survey session. Wilcox and Los Huertos (2005) describe a simple and rapid method for bathymetric mapping using a total station and GPS.

The great value of RTK GPS is that each point is independently georeferenced, and will plot precisely on existing regional map data. The limitations of this technology include potentially poor satellite positions, and the inability to operate when a tree canopy or valley walls block satellite reception.

Fig. 2.14 An example of mobile terrestrial LiDAR positioned by RTK GPS and an inertial motion sensor that was used to map large tidal wetlands in central California

2.3.2.7 Light Ranging and Detection Technology

LiDAR technology offers the ability to survey extensive wetlands with great efficiency and precision. LiDAR is an optical system that sends out several thousand laser pulses each second and records various aspects of the reflected light. In topographic surveys, the primary variables are the direction and distance the laser beam traveled before it was reflected back to the sensor. Those variables are used to calculate an X, Y, Z position to the reflecting surface (e.g., plant, ground, building, etc.). LiDAR has the advantage of shooting thousands of X, Y, and Z points per second from a plane or terrestrial platform, but it has a disadvantage of not being able to easily penetrate water or very dense wetland vegetation. There is currently some experimentation to achieve better water penetration using various light wavelengths. Aerial LiDAR data sets are available from public internet sources including the National Center for Airborne Laser Mapping (http://www.ncalm.cive.uh.edu/). Currently, terrestrial LiDAR scanners are not widely available. California State University Monterey Bay has engineered a mobile LiDAR system that can be attached to an all-terrain vehicle (Fig. 2.14) to precisely digitize soil erosion rates. The early results are very promising, but the instrumentation is beyond the budgets of most practitioners.

2.4 Modeling and Visualization of the Bathymetric Surface

Once the field survey work has been completed, the next step is to create a bathymetric surface. The data are entered into a spreadsheet (e.g., Excel) and processed to create a digital representation of the surface. It is important to note that the depths between the sampling points are interpolated using one of a variety of methods. The points are used to create surface models either as a raster or a triangulated irregular network (TIN) surface model. A raster is a grid of evenly spaced elevation values (points) created by interpolation from the survey data. A TIN is created by making a network of irregular, nonoverlapping triangles between the survey data points. By using relatively affordable software such as ArcGIS with extensions (Spatial Analyst and 3D Analyst), the tabulated data (X, Y and Z) can be calculated to develop a bathymetric surface for display and further analysis. Typical analyses can determine water elevation (stage), and the wetted perimeter, area, and volume as a function stage.

GIS software is now used in all walks of academic and professional environmental science. In the following discussion, we assume that the reader has used GIS software such as ESRI ArcMap. Each new version of ArcMap provides slightly different ways of achieving the desired results we want, so some steps we describe below may become outdated with newer software versions. However, we are confident that the general principles will apply far into the future.

The general steps toward wetland visualization and geometric analysis are:

1. Enter or import the X, Y, and Z data into a spreadsheet;
2. Save the file in a format readable by ArcMap;
3. Import that file to ArcMap;
4. Produce an ArcMap point file;
5. Create a digital elevation model (DEM) by interpolating the data into an elevation raster, or by making a TIN; and
6. Use the DEM to visualize and analyze the wetland structure.

We provide a step-by-step process below using a wetland example.

2.4.1 Molera Wetland GIS Analysis Example

2.4.1.1 Data Preparation and Import

Molera Wetland, which is located along the central California Coast, was selected to provide an example of surveying and data analysis. The example also serves as an exercise at the end of the chapter. To follow our example, use the data available via this weblink: https://sites.google.com/a/csumb.edu/marc-los-huertos/home/molera-wetland-bathymetry. Download the rtk_gps_wetlands.xlsx file in a folder called Molera.

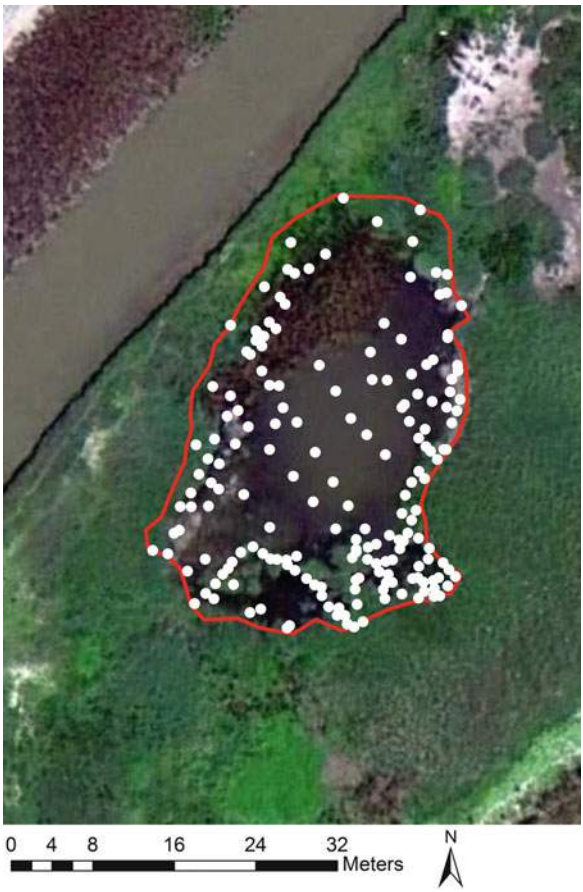

Fig. 2.15 ArcGIS map of Molera Wetland with the survey points

Approximately 270 positions were shot using RTK GPS during one afternoon at Molera Wetland (Fig. 2.15). The base station was placed on a known benchmark. The survey focused on the pond edges because they were more complex than the central part of the wetland pond. In ArcMap, we drew a polygon around the pond perimeter and used the 188 points inside the polygon in the following geometric analysis. Next we entered the data into a spreadsheet with columns labeled "Easting", "Northing", and "Elev". In general, those columns can also represent any X, Y positioning system that was used in the survey, including an arbitrary local survey framework.

In some cases, one must make vertical or horizontal adjustments (e.g., adjust the Z value to relate to stage or NAVD88 for example). We used the GPS system to provide output in WGS 84 UTM, Zone 10 North meters as the horizontal reference and NAVD88 meters as the vertical reference, so no further adjustments were required.

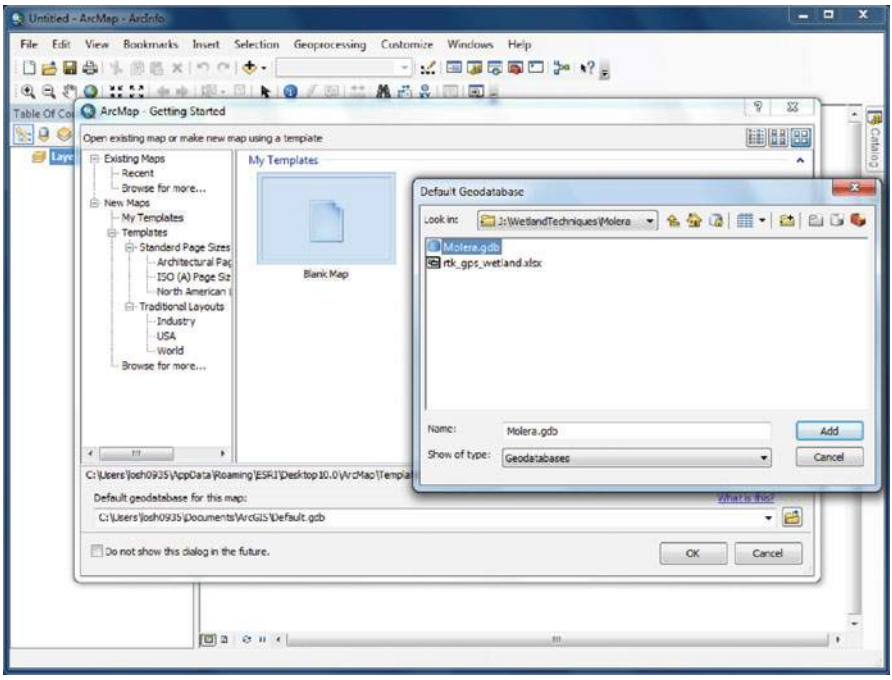

Fig. 2.16 Creating a new geodatabase is the first step when you start ArcMap v10. This screen shows a new directory where we created a new file by selecting the new geodatabase icon (a small grey round cylinder, often used as a database symbol). We then add this file and select OK in the "Getting Started" window

To follow this example, you will need to use ESRI's ArcMap. In the new version of ESRI's ArcGIS, ArcMap v.10 uses a default geodatabase that we will redefine. To begin, you should click on the small folder icon near the bottom of the "Getting Started" window and navigate to the Molera folder using the "Connect to Folder" icon. Once the Molera folder is selected, you should click on the "New File Database" icon and rename it as "Molera" (Fig. 2.16) and click okay. Once this file (geodatabase) is created, you can open the excel file and save the datasheet as a comma delimited ("comma separated values" that is abbreviated as csv) file, which can be done in any spreadsheet software and then add the csv file to the map, using File > Add Data > Add XY Data. You then will select the csv file and assign the X direction as the "Easting" column, Y direction as "Northing" column and Z as the "Elev". You must be sure to assign the appropriate projected coordinate system (Projected Coordinate System > UTM > WGS 84 > Northern Hemisphere > WGS 1984 UTM Zone 10N.prj). ArcMap will give an error (warning) because there is no object-ID field, but you can ignore the warning and proceed by clicking okay. After clicking okay, you should be able to see points displayed on the map. The map should then be saved in the newly created directory with an appropriate name (e.g., Molera). The next step will be to create a shape file, which is a specific file structure

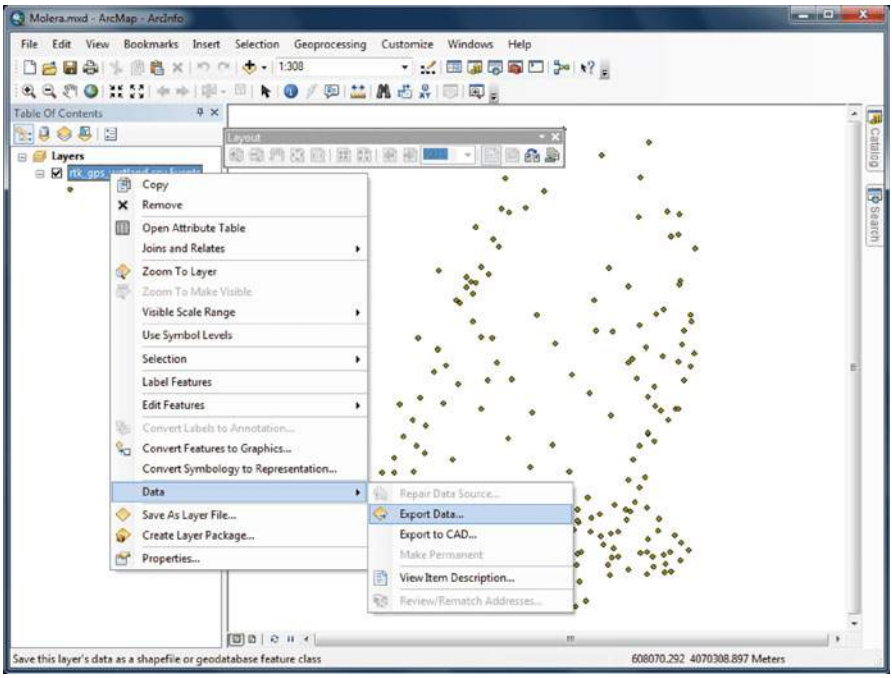

Fig. 2.17 A screenshot of the process of preparing to export the csv file into a shapefile

for maps in ESRI software. You can begin this process by right clicking on the name of the imported file "Events" in the "Table of Contents" in the left panel and choose the Data > Export Data (Fig. 2.17) as a shape file. After you have exported the data as a shape file, you will click on the folder icon and select "shapefile" in the bottom dialog box as the file format. You must be sure that you are saving the file in the Molera directory and as the correct named file; we used "GPSPoints" as the shapefile name. After completing this step, you can remove the original .csv file (by right clicking on it and selecting remove) to clean up the ArcGIS "Table of Contents" and save your Map Document.

Now, you are ready to plot the survey points, which will also allow you to make sure there are no data entry errors (e.g., outlier points located far away from the cluster of survey points). If you observe outlier points, you should check the data entry to make sure the coordinates did not include a typo. By right clicking the database file > plot xy, the data will be prepared for display on the screen. The resulting "point file" will be used in the following analyses. You can also bring up an aerial image, which is available on the website. You should see the points match the extent of the aerial photo. If you do not, then the projection may be incorrectly defined. The exported map should appear as in Fig. 2.15.

2.4.1.2 Creating a Digital Elevation Model

For creating a DEM of the wetland, we recommend using either the "natural neighbor" method or the "kriging" method. Both methods are commonly used for creating a DEM, but there are many choices for grid interpolation methods. Given that we are working with less than thousands of data points, you can create a TIN or you can create a surface by interpolating points to make a raster. We describe kriging in this example. We created a synthetic wetland from our data and sub-sampled it using a variety of sampling grid spacing to synthesize the accuracy achieved by different levels of effort in surveying (Fig. 2.18). We then created both TINs and krig DEMs to illustrate how accurately each one represented the original synthetic wetland in terms of volume, which was analyzed at a variety of depths.

Figure 2.18 shows the rate at which accuracy improves as more survey shots are taken, which results in tighter survey grids being used. However, you should also be aware that there is much less accuracy at lower water stage compared to higher stage. From our experience, we found that kriging improved accuracy between 2 and 10 % in the 5 m grid survey and by 11–30 % in the 10 m grid survey. The results indicate that the advantage of kriging increases markedly when the survey has fewer shots to control elevations in the Molera Wetland.

We now describe the procedure for kriging. Using the "ArcToolbox" (icon with a red tool box), select Spatial Analyst > Interpolation > Kriging with a hammer symbol. Next, you should select the point features created and select the Z value as "Elev". Finally, you must define the results into a DEM folder within your project directory. We used the default options for this kriging. ArcGIS creates a default output cell size, but we rounded the value to 0.1 m. Changes to the "Maximum Distance" for kriging "search radius" might improve the output depending on the bathymetric variability, but we left the value blank (Fig. 2.19). The resulting DEM will appear in the project.

Finally, we created a polygon shape file as a mask that can be used to trim the kriged surface so that we do not extrapolate elevations beyond the GPS collected points. To do this, we opened the ESRI ArcCatalog program and navigated to our project via the Folder Connections where you can right click the folder > new > shapefile and select polygon as file type. We call it "KrigMask," and assign the project's coordinate system. You now add the newly created shapefile to the map in ArcMap and open the editor menu where you select the correct shape file to edit and then go to construction tool and select polygon. After selecting polygon, you can create a polygon on the boundary of the wetland points, and use the aerial photo to help define where to click each point. To end polygon construction, you must double click. You can now save your edits, and then click "stop editing" to complete the process. To further refine the DEM, you should select Spatial Analyst Tools > Extraction > Extract by Mask and create a new trimmed DEM (e.g., Krig_Trim) using the DEM as a raster and the mask shape file (Fig. 2.20).

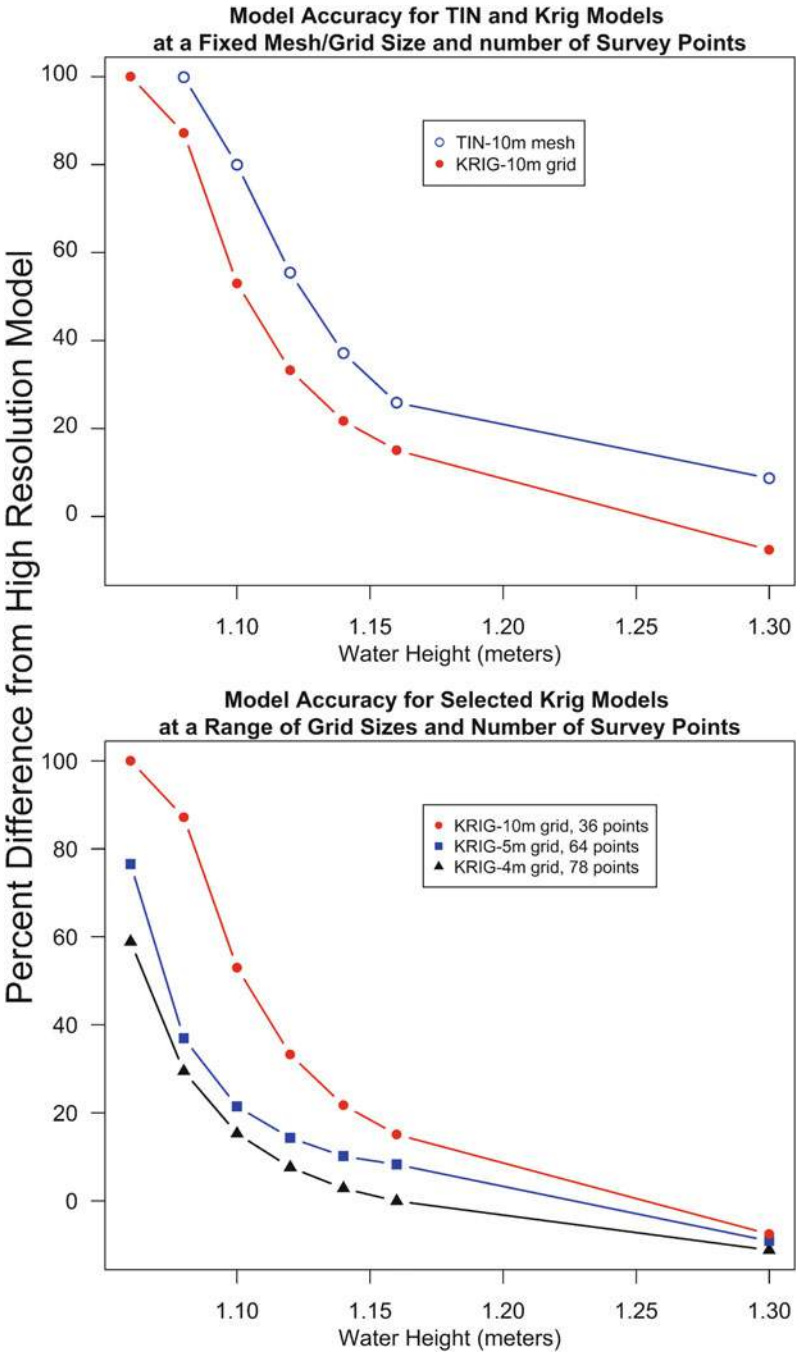

Fig. 2.18 A comparison of the accuracy of krig versus TIN models for calculating wetland volumes at various grid sizes and number of survey points shot

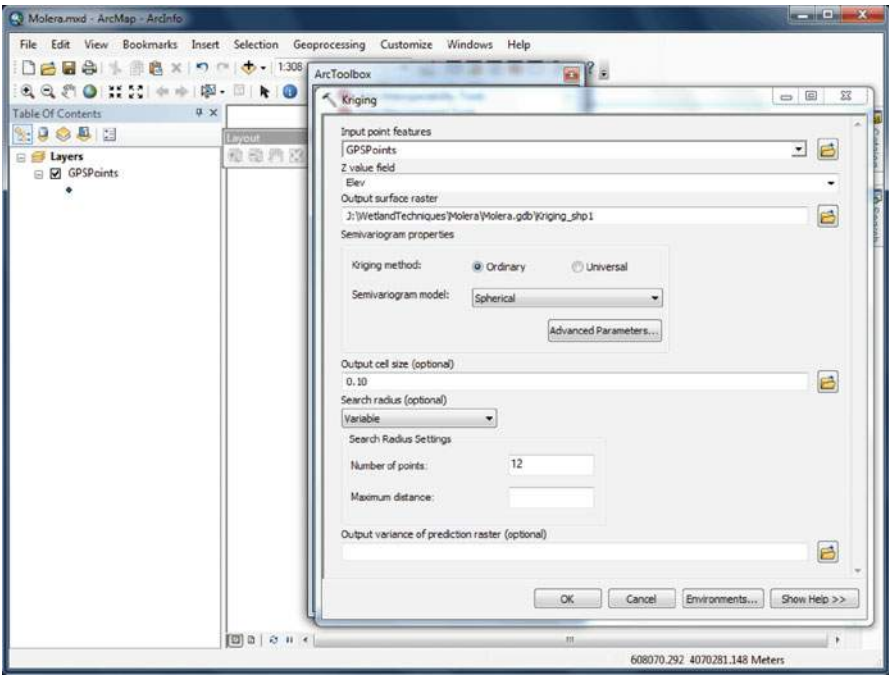

Fig. 2.19 An example of krig dialog box

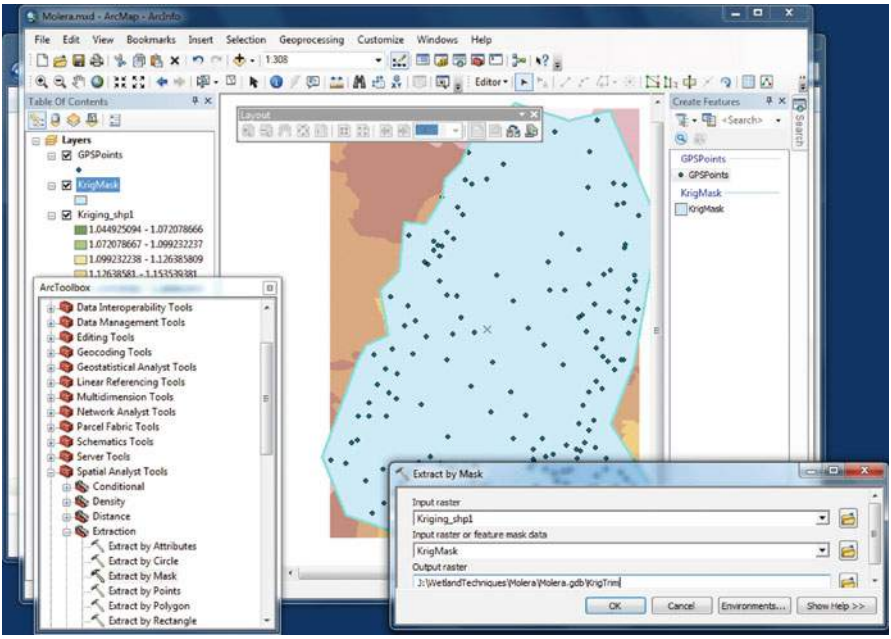

Fig. 2.20 A screenshot of the process of preparing to trim the map

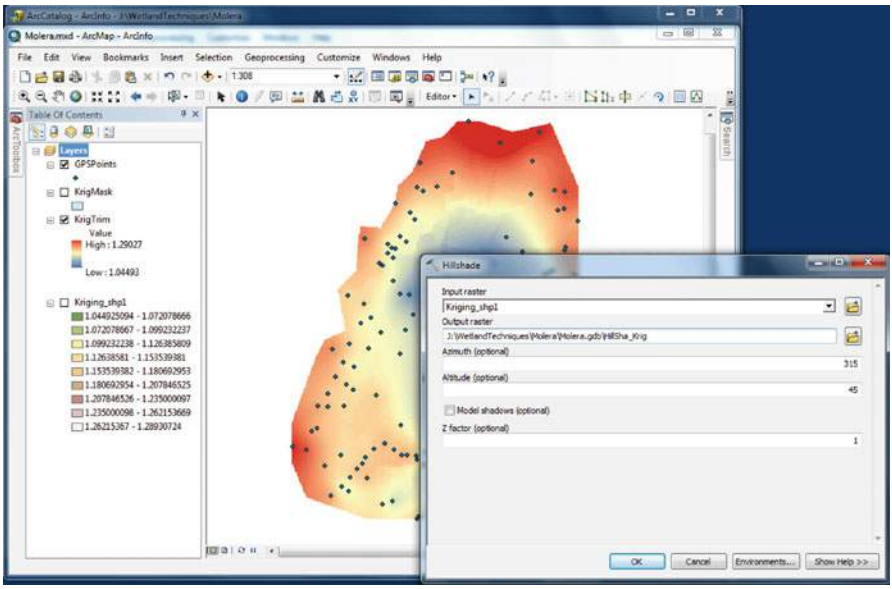

Fig. 2.21 An example of the hillshade dialog box

2.4.1.3 Visualizing Wetland Geometry

The DEM can be displayed so each range of depth can be associated with a color. This operation will color the DEM by elevation. You have many choices, and the selection will be dictated by the information you are seeking. Two standard coloring methods are 1) "stretched" color ramp that gives a continuous gradation of color from high to low elevations or 2) "classified" which gives more information and more control. For stretched color ramp, you will use the following sequence. First, you should right-click the DEM Filename > Properties > Symbology > stretched, and select the color ramp that you think is appropriate. In addition, you can create a shaded image called a hillshade to show topographic variation. From ArcToolBox, you should select 3D Analyst > Raster Surface > Hillshade and make sure the unmasked DEM is the selected file in the dialog box. You will then extract the file using the mask polygon boundary shapefile as before. You will need to control the sun angle and azimuth for illuminating the digital surface (Fig. 2.21). In this example, we used the default values. We suggest you experiment with this option to determine how it influences the results. This process may take some time depending on the speed of the computer, so you should be patient. In this example, the display defaults to a categorical color scheme because the surface is fairly flat, thus, not very useful. To enhance the hillshade, you can manually change the hillshade symbology by setting the high value to 255 and low value to zero. This will create a reasonable hillshade grayscale. To do this, you will right click the hillshade on the left side of the screen and select properties. In the symbology tab,

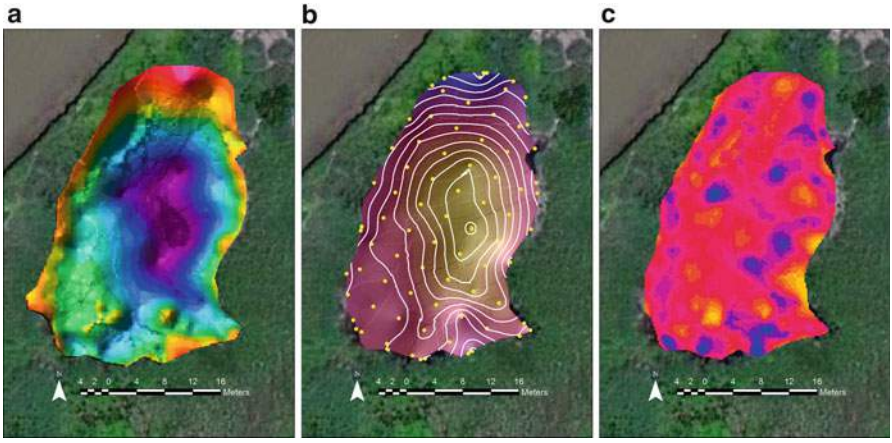

Fig. 2.22 Examples of differences in wetland analyses using kriging. (**a**) Colored hillshade of the synthetic wetland pond created from kriging our 189 GPS shots. (**b**) Colored and contoured hillshade created from kriging the superimposed 4 m grid shots (*yellow*) that were used to subsample the wetland surface in **a**. (**c**) Map of the surface differences between the original model (**a**) and the subsampled model (**b**), with differences mapped as color intensities. *Yellow* indicates areas where the 4 m grid model underestimated the depth, and *blue areas* indicate an overestimate. The average difference was not significant at 95 % confidence level. The maximum local differences were ±0.04 m, indicating a very good model comparison

you will select stretched, set the stretch type to minimum-maximum, check the edit high/low levels box and type in 255 for the high value and zero for the low value.

The DEM can be displayed so each range of depth can be represented. Much more technical information about your wetland can also be displayed by draping (layering) data which may include contour lines, vegetation layers, or elevation coloration on top of a hillshade. The basic technique is to display both the data layer (e.g., colored DEM) and the hillshade in the same map view. For example, Fig. 2.22a illustrates that effect in a Fledermaus project. We can also improve the visual effect by making the layer semi-transparent (Fig. 2.22b), which can be accomplished by right-clicking on the Hillshade > Properties > Display > Transparency and then using trial and error on the transparency level to obtain the desired effect (e.g., in our example, we used 35 %). You may add contours using 3D Analyst > Raster Surface > Contour. We used a 0.2 interval and a base contour of 1.05 (approximately the deepest point). To make the lines more visible, we changed the color to white and simplified the map by turning off other layers (Fig. 2.22b). Figure 2.23b illustrates that effect in an ArcMap project. Further detail can be added as artwork by exporting a map, and using an art program such as Adobe Illustrator. Figure 2.22c demonstrates an analysis of the surfaces mapped in the original and subsampled models.

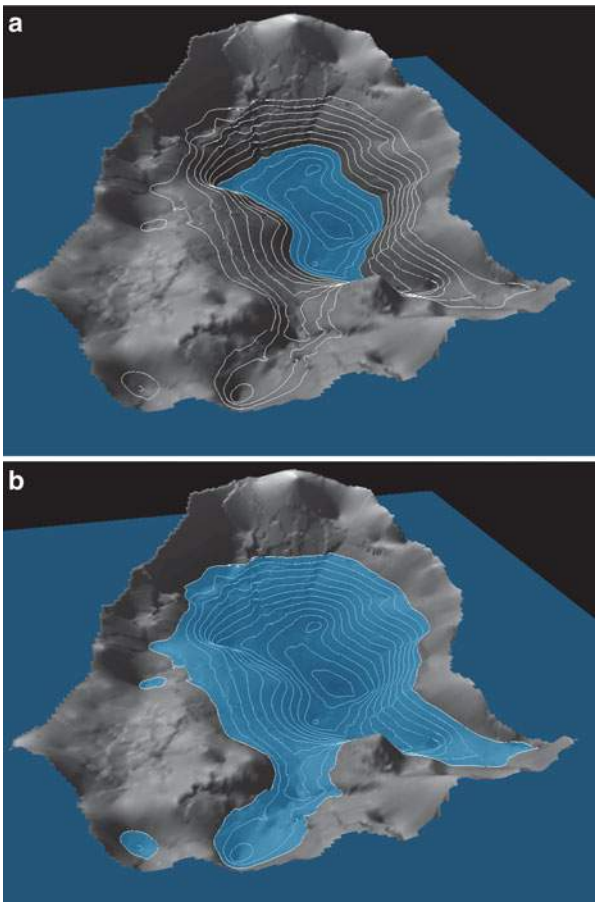

Fig. 2.23 A digital model of Molera Wetland projected using Fledermaus software. The contour interval is 0.02 m and begins at 1.05 m stage. (**a**) Contour lines help to visually define the location of the point of zero volume (pzv). A transparent *blue plane* inserted at an elevation of 1.09 m helps visualize the complexity and general shape of the wetland at that stage. (**b**) A transparent *blue plan*e inserted at a stage of 1.16 m indicates that this stage is very near the point of incipient flooding (pif) for this particular depression. If the water surface were higher, it would flood to adjacent landscape elements

2.4.1.4 Quantifying Wetland Geometry

There are many elements that can be measured in a digital model of wetland topography and several ways to calculate their values. Some of the elements that researchers may need to know include what is the deepest point in the wetland, what is the point where water may overflow to the next basin, and what is the water volume and surface area of the wetland under a variety of different water levels. The deepest point in a wetland is referred to as the point of zero volume (pzv). The pzv will be the last refuge for fully aquatic organisms as the wetland dries. It is the lowest elevation value in the DEM (Fig. 2.22b). The point of incipient flooding (pif) is the elevation where water spills from one wetland depression to another, or to the adjacent terrace. It is important to remember that the pif is not considered the highest elevation in the digital model because water will "spill over" at a "saddle", but is a low point between basins. This is clearly illustrated in the Fledermaus project of Molera Wetland (Fig. 2.23a,b).

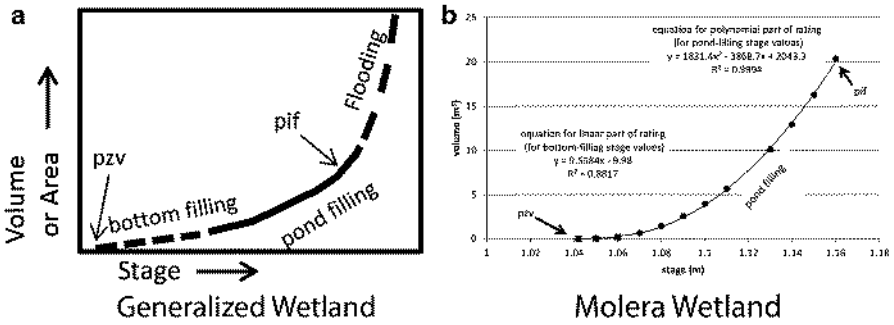

Fig. 2.24 A representation of the relationship between a wetland's water stage and a wetland's volume. (**a**) Wetland ponds can be geometrically complex, so more than one equation is needed to adequately model the volume to stage relation. The simplest wetlands will have a minimum of three volume zones (bottom filling or point of zero volume [pzv], pond filling, and flooding or point of incipient flooding [pif]) as they are analogous to a bathtub with a wide, relatively flat bottom, steep sides, and overflow. (**b**) The bottom of Molera Wetland is relatively flat, and the contour lines are far apart (refer to Fig. 2.23a,b for a representation of its bathymetry), so an increase in stage which inundates the lowest contours results in a very small increase in wetland volume. Nearer to the perimeter, the wetland has relatively steep sides, and the contours are closer together, so that an increase in stage results in a much larger increase in wetland volume

Table 2.3 The relationship of water stage to wetland area and volume in our Molera Wetland example

Water stage (m)	Area (m^2)	Volume (m^3)
1.05	3.54	0.007
1.07	61.5	0.58
1.09	120	2.3
1.11	183	5.3
1.13	256	9.7
1.15	360	15.8
1.17	536	24.7
1.19	748	37.7

Wetland water volume and surface area can be determined at a range of water levels. To calculate water volume and surface area, you will select 3D Analyst Tools > Functional Surface > Surface Volume > to input your trimmed DEM filename, indicate the water elevation for which you want the calculations, and then indicate that the analysis is "below" the plane. You should be sure to create an output file. This output text file stores the analysis results. We placed our file in a folder called "stage_vol directory." If wetland volume or surface area is calculated for a range of stages, a visual graph and mathematical "rating" equation can be created that relates stage to volume or area (Fig. 2.24). Rating equations can be linear, power, or polynomials, and different parts of the data set might require different equations (Fig. 2.24). It is important that you use good judgment in limiting extrapolation beyond the data, given that wetland geomorphology can be very complex as demonstrated by our example of how changes in stage can dramatically change wetland area and volume (Table 2.3). For our calculation of

wetland area and volume at different stages, the calculation output was appended to the text file with each new stage we used when we used ArcGIS version 9.3, but in version 10, we needed to create a new output file for each stage, which is a bit tedious.

2.5 Conclusion

Bathymetric data and analysis can refine our understanding of wetland status and the impacts of human activities on wetlands. The mapping and analysis of wetlands requires several distinct steps that include planning, data acquisition (via field surveys or obtaining digital data), importing data into mapping software, visualization, and analysis. Because the precision and accuracy of wetland bathymetry can play an important role in understanding wetland structure and function, the technology used have become increasingly sophisticated. Survey equipment accuracy and efficiency have increased dramatically in the last 20 years, whereas, software and mapping programs have become more powerful, readily available, and user friendly. The ability to understand and effectively use bathymetric mapping and visualization techniques will convey an advantage to anyone who is interested in wetland science and management.

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Student Exercises

Classroom Exercises

Classroom Exercise #1: Evaluating Wetlands in Google Earth

Developing a familiarity with Google Earth is a good starting point for many mapping projects. This exercise guides you through the use of Google Earth to evaluate the Molera Wetland for this and the following exercises.

1. If you do not have Google Earth installed on your computer, follow this link http://www.google.com/earth/index.html to download the program.
2. To find Molera Wetland, paste the following longitude and latitude coordinates into the search window: 36°46′20.93″N, 121°47′18.60″W. The wetland parcel is bounded on the north by the Tembadero Slough and on the west by the historic Salinas River channel. It is bounded on the south and east by roads.
3. Click on the year at the bottom of the screen and you will get a time slider at the top that can be used to choose the dates of various images. In the case of the Molera Wetland and at the time of writing this exercise, there were several images available from 1993 to 2012. The images vary in terms of resolution and number of color bands.
4. Molera Wetland was constructed as a water treatment wetland. To accomplish that goal, it consists of an elongate, sinuous channel in the southern part and an open water wetland in the northern part. Use the time slider to determine when the land use changed from agriculture to wetland. Note how the "wetness" of the open wetland changes through time. Can you determine if the wetness changes are more related to season differences or annual differences?
5. Select an aerial image year that shows the wetland very wet (e.g., May 2011). We will make a rough estimate of open wetland size in that image. Click on the ruler icon at the top of the Google Earth page.

(a) Use the ruler to measure the perimeter of the wetland.
(b) We can estimate the area of the wetland by approximating it as an ellipse. First, measure and record the long and short dimensions of the wetland and then take ½ of those dimensions and multiply them by pi (π) as demonstrated in the following equation:

$$Area \cong \frac{1}{2}(long\ dimension) * \frac{1}{2}(short\ dimension) * \pi$$

Unfortunately, calculating the area this way has limited value because it is based on what you can see and the water depth at that time. You should also select a few other images and calculate how they have changed in different seasons and different years. Can you say anything about the bathymetry of the wetland based on these dimensions?

Classroom Exercise #2: Creating a Bathymetric Surface for Visualization and Analysis

The following exercise uses locally referenced grid of survey points from Molera Wetland. The data for the exercise are available at the following website https://sites.google.com/a/csumb.edu/marc-los-huertos/home/molera-wetland-bathymetry. Download the "exercise_data.csv" file from a folder called Molera. The X, Y coordinates are linked to a piece of rebar we assigned as (50,50) meters, and the elevations are referenced to a local county benchmark elevation in meters. The data were collected using a grid similar to the one shown in Fig. 2.8a, and a subset of the field notes are shown in Fig. 2.5. This data set is coarser than the one used in the chapter, so you can compare the impact of lower resolution on the analysis values. These data have no real-world horizontal coordinates, and ArcGIS will give you a warning message to that effect, which can be ignored for the purpose of this exercise. Use the same steps outlined in the Molera Wetland example in the chapter. The minor differences are noted below in keeping with a survey that has no horizontal georeferencing.

1. Open a new map project in ArcMap. Make the map units meters by clicking the View menu, then Data Frame Properties > General and select "meters" for the units of the map and display.
2. Import the csv file into ArcMap, and create a point shapefile so that the survey points are displayed on the map, and the points have an attribute table.
3. Create a polygon shapefile to be the mask representing the wetland boundary. Draw the boundary using the outer-most points of the survey as a guide.
4. Interpolate the points into a DEM by kriging. For kriging, use the mask to limit the extent of the analysis.
5. Color the DEM to create a map. You have many choices, and the selection will be dictated by the information you are seeking. Two standard coloring methods

are "stretched" color ramp that gives a continuous gradation of color from high to low elevations, or "classified" which gives more information and more control. For stretched color ramp, use the following sequence. Right-click the DEM filename > properties > symbology > stretched, and select the color ramp. For classified coloration use the following sequence. Right-click the DEM filename > properties > symbology > classified and then select a number in the "classes" box to indicate the number of discrete elevation color bands you want and the color ramp. More statistical information, and coloration controls are present if you click "classify."

6. Add contour lines using a base contour of 1.05 m and a contour interval of 0.02 m, or another value of your choice. Color the contours to your liking.
7. You can calculate the perimeter and area of the analysis region by analyzing the perimeter mask shapefile you created. Right click the mask filename > open attribute table > add a field. Select "short integer" and name the field "perimeter." When you click "OK," you will see a new column in the attribute table. Right click the top of the column and select calculate geometry > perimeter. The perimeter value will appear in the column. Try the same steps for determining the area. These are the values that do not correspond to a specific water level, but are values for describing the wetland in general. How did these values compare with the Google Earth measurements you made earlier in this exercise?
8. Quantify Wetland Volume and Surface Area using the following values for wetland water stage: 1.05, 1.07, 1.09, 1.11, 1.13, 1.15, and 1.17 m. Compare the results to those that we created in Table 2.3. Create a graph of the stage and volume relationship and stage and surface area relationship in a spreadsheet. The values you obtained will differ somewhat, because you are using a lower resolution survey than the one presented in the chapter example. In comparing your results with our results in the chapter example (Table 2.3), you can qualitatively evaluate whether a having a large number of points improves accuracy and provides more information.
9. Practice making a hillshade and making it semi-transparent. We find that using a z-factor of four and lowering the sun angle to about 25° improves the visual impact of the hillshade in low relief settings such as this.

Chapter 3
Assessing and Measuring Wetland Hydrology

Donald O. Rosenberry and Masaki Hayashi

Abstract Virtually all ecological processes that occur in wetlands are influenced by the water that flows to, from, and within these wetlands. This chapter provides the "how-to" information for quantifying the various source and loss terms associated with wetland hydrology. The chapter is organized from a water-budget perspective, with sections associated with each of the water-budget components that are common in most wetland settings. Methods for quantifying the water contained within the wetland are presented first, followed by discussion of each separate component. Measurement accuracy and sources of error are discussed for each of the methods presented, and a separate section discusses the cumulative error associated with determining a water budget for a wetland. Exercises and field activities will provide hands-on experience that will facilitate greater understanding of these processes.

3.1 Introduction

The physical, biological, and chemical properties of a wetland all are greatly influenced by water and chemical fluxes, both to and from the wetland, as well as the temporal variability of these fluxes. Therefore, hydrologic processes are central to the character and features of a wetland and to virtually everything that occurs surrounding and within a wetland basin. A question occasionally posed by wetland scientists is whether a wetland "has hydrology." This terminology likely stems from

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a need to determine whether a landscape has characteristics of a wetland setting for regulatory or protection purposes. Hydrology is basically the study of water as it is distributed over, on, and within the earth. All landscapes, and particularly wetlands, have hydrologic properties that are an integration of all the water-related characteristics and processes that occur there. Wetland hydrology encompasses study of the distribution and flow of all water that is added to, lost from, or stored in a wetland.

A wetland is a portion of a landscape that is wet for a period sufficiently long that physical, chemical and biological conditions are indicative of a wet setting. Wetlands occur in a wide range of settings where geological and hydrological processes enhance the accumulation and retention of water (Winter 1988). Water, therefore, is present at or just beneath land surface at a substantial percentage of the time in wetland settings. Given that water is integral to wetland settings, an overarching challenge in determining the type or persistence or quality of a particular wetland setting is to determine the relative contributions of the various components of wetland hydrology (i.e., precipitation or evapotranspiration or surface-water inputs or groundwater inputs or overland flow). A water-budget approach for making this determination is perhaps the best way to categorize and describe the wide range of wetland types that exist in the world (Winter and Woo 1990; Winter 1992) and is the perspective from which this chapter is presented.

3.2 Wetland Hydrology from the Perspective of a Water Budget

Knowledge and understanding of the storage and mass balance of water and chemicals is critical to understanding a wetland ecosystem. This includes quantifying all of the sources, losses, and changes in storage in the wetland. Simply determining the relative magnitude of various hydrologic components can largely determine a wetland type. For example, surface water may be the dominant source and sink of water and solutes for a riparian wetland whereas overland flow and evapotranspiration may dominate in a prairie wetland. One will have greatly different water chemistry and biogeochemical processes than the other, all because of the relative mix of sources and sinks of water and chemicals.

Wetland stage is an integrated response to all source- and sink-terms in a hydrologic budget. It also incorporates temporal variability in the balance of all hydrologic fluxes and is, therefore, strongly linked to wetland hydroperiod and wetland hydrodynamics, both of which are important to most disciplines that encompass wetland science (Euliss et al. 2004). Wetland stage and volume can also provide a direct and often sensitive response when climate change may be affecting the relative magnitude and importance of specific hydrologic components.

For these reasons and more, an accounting of hydrologic components of a wetland water budget should be one of the first items on a wetland-scientist's agenda (LaBaugh 1986). Preliminary estimates of the relative volume associated with each hydrologic component is often a valuable first step. These estimates will allow attention to be

focused on the most important hydrologic components of a particular wetland setting or type. The importance of quantifying individual hydrological components also depends on the issues and questions being asked. For example, at an extensively studied wetland in the prairie-pothole region of North Dakota, groundwater discharge was a small component (3.5 %) of the water budget, small enough that it might be ignored. However, groundwater discharge delivered a large percentage of chemicals to the wetland and was an important contributor to wetland chemistry (LaBaugh et al. 2000).

A wetland water budget can be written as

$$\frac{\Delta V}{\Delta t} + R = P + O_f + S_i + G_i - ET - S_o - G_o \tag{3.1}$$

where $\Delta V/\Delta t$ is the change in volume of surface water in the wetland per time, P is precipitation, O_f is overland flow, S is surface water, G is groundwater, ET is evapotranspiration, and R is the residual, or unaccounted water, in the water budget. Subscripts i and o refer to water flowing into or out of the wetland. This basic equation should be modified to suit specific wetland settings. For example, some wetlands will have dewfall or stem flow that is substantial and quantifiable whereas other wetlands will not have any surface-water inputs or losses. Many wetlands in northern latitudes also have an input term associated with drifting snow (e.g., Hayashi and van der Kamp 2007). Some wetlands will rarely contain surface water, in which case $\Delta V/\Delta t$ can be based on changes in volume of surface water, groundwater, and soil-moisture storage over time. If surface water is not present, hydrologic fluxes are distributed over an area based on criteria other than areal extent of surface water, perhaps the areal extent of wetland vegetation. In this chapter we will restrict discussion primarily to settings where surface water is present.

Equation 3.1 can be rearranged to solve for any of the components provided the others are known. An example is presented later for determining G_i and G_o as the unknown entities of the water-budget equation. ET also can be a difficult value to obtain and is occasionally solved as the unknown of a water-budget equation. However, the uncertainty associated with ET commonly is much smaller than the uncertainty associated with quantifying G_i or G_o. In many wetland settings, errors associated with quantifying groundwater exchange are so large that solving for ET as the residual would be meaningless.

3.2.1 Determining the Accounting Unit

As mentioned earlier, the change in wetland storage, ΔV, integrates all of the input and loss terms of a hydrologic budget. This term can also be approximated as

$$\Delta V \cong \Delta h \left(A + \frac{\Delta A}{2} \right) \tag{3.2}$$

where A is wetland surface area and h is wetland stage. Details for determination of V in the typical case where A changes with depth are provided in Sect. 3.3.2.

Table 3.1 Errors indicated in % for water-budget components of selected studies conducted on lakes, reservoirs, and wetlands (– indicates parameter was not determined; calc indicates value was calculated as the residual)

	P	E or ET	S	G	Of	ΔV
Winter (1981)	5–10	10–15	5–10	13–36	–	–
LaBaugh (1985)	33	10	5–15	–	–	10
Belanger and Kirkner (1994)	10	10	50	50	–	10
LaBaugh et al. (1995, 1997)	5	10	–	50	–	5
Lee and Swancar (1997)	10	16	–	102–106	–	5
Sacks et al. (1998)	5–9	10	30–100	calc	–	5
Choi and Harvey (2000)	8.5	20	10	10	–	15
Harvey et al. (2000)	15	10	10–15	10	–	15
Motz et al. (2001)	5	20	11–15	50	100	5
Rosenberry and Winter (2009)	5	15	5	25	–	10
Median	9	10	10	36		10
Maximum	33	20	100	106		15
Minimum	5	10	5	10		5

When Δh is small and A is much greater than ΔA, this relation often is simplified by assuming that A is constant (i.e., $\Delta A = 0$). A minimum measurable change in wetland stage is, therefore, a logical accounting unit in a wetland water budget. Precipitation and evapotranspiration already are usually expressed in terms of depth applied over the wetland surface per time (commonly mm/day). Other water-budget components more commonly measured in terms of volume per time, such as surface-water or groundwater inputs and losses, can be expressed as Δh by dividing by A. This seemingly simple task can be a substantial problem at many wetlands, as evidenced by the relatively large errors associated with the ΔV term listed in Table 3.1; errors of 10–15 % are common. Since measuring stage is quite simple and can be done very accurately, often with accuracies of ± 3 mm or better, the estimation of surface area is the source of most of this error.

The shoreline must be identified before wetland area can be determined. Unfortunately, an indistinct shoreline as shown in Fig. 3.1 is common. In some cases, an area of dense emergent vegetation forms an abrupt boundary, not at the shoreline but at the edge of the open-water portion of the wetland, that confounds the determination of the actual shoreline. If this border occurs at a water depth of 0.3 or 0.5 m, an example of which is shown in Fig. 3.1, the actual shoreline, where water depth decreases to zero, can be many meters away and obscured by additional dense emergent vegetation.

For wetlands situated in low-gradient settings, the shoreline can move laterally a large distance in response to a small stage change (e.g., Lee et al. 2009). An accurate bathymetry map, and associated stage-area and stage-volume plots, are particularly important for minimizing error when determining ΔV. Generating an accurate stage-area plot is not nearly as onerous as it once was (see Chap. 2 on wetland bathymetry).

Fig. 3.1 Example of a wetland where the shoreline is not easily distinguishable (Photo by Donald Rosenberry)

Methods that provide high-resolution topographic information, such as a map generated by the light detection and ranging (LIDAR) technique, are particularly useful for determining appropriate areas to assign to specific stages. These methods are best employed when stage is lowest. Also, some wetlands that normally have no surface-water outlet can develop one during extremely wet periods. This process is commonly referred to as "fill-and-spill" (van der Kamp and Hayashi 2009; Shook and Pomeroy 2011; Shaw et al. 2012). Modern approaches based on differential geographic information system (GIS) are capable of determining stage- and scale-dependent contributing areas with regard to net overland flow that contributes to a particular wetland basin. In situations where these relatively new tools and procedures are prohibitively expensive or labor intensive, simplifying assumptions based on general knowledge of wetland shape can provide reasonably accurate stage-area and stage-volume relations (Hayashi and van der Kamp 2000).

Some wetland basins become separate entities during dry periods and then coalesce during wet periods (e.g., Winter and Rosenberry 1998). Water budgets need to be determined for each distinct wetland sub-basin, based on separate stage-area and stage-volume relations, until the wetlands coalesce, at which point a new stage-area relation should be used for the now combined wetland.

Thus far, ΔV has been determined based on the surface area of the open-water or standing-water portion of a wetland. This concept is not appropriate for wetlands that do not contain standing water to any measurable depth; for example,

wetlands on hillslopes or wetlands that drain rapidly following rain or flooding events. In those settings, the accounting unit may need to be set based on topography or areal extent of specific types of wetland vegetation. If the wetland surface is considered to be saturated virtually all the time, then one could reasonably assume that ΔV is zero. In this case, any additions of water to the wetland must instantly be balanced by an equal volume of loss terms. The accounting unit also could be the quantification of water stored in the vadose zone or the sum of water contained in the vadose zone and in groundwater beneath the vadose zone. Assumptions may need to be made regarding the level of saturation of the wetland soils to quantify a change in stored volume. If the water table decreases to below the wetland bed, water-volume change could be estimated based on water-level measurements in monitoring wells and assumptions about volumetric storage capacity of the wetland soil. A further complication is associated with the typically small distance of the water table below the land surface or wetland bed. The capillary fringe is a zone of tension saturation that exists above the water table in all settings; the thickness is inversely proportional to the grain size of the soil. In the generally fine-grained sediments found in most wetland settings, the soils may be essentially saturated beneath much to all of the wetland bed. If this is the case, even small rainfall or recharge events can bring the water table directly to land surface and result in surprisingly large amounts of overland flow to the wetland (Gerla 1992).

3.2.2 Determining the Accounting Period

The proper time interval (Δt in 3.1) over which a water budget is determined depends on the questions asked, the duration of the study, or the reasons for quantifying a water budget (Healy et al. 2007). If the question is related to wetland response to climate change, an annual water budget may be all that is necessary. If determining the relative significance of a particular hydrologic component is important, the study may need to extend over several years and quarterly or monthly time steps would be appropriate. If the concern is related to the response of a wetland to individual recharge events or to specific physical, chemical, or biological processes, daily time steps may be the most appropriate. In general, because of technological advances in data collection, scientists are tending to use shorter time steps. Whereas monthly measurements may have been the norm during previous decades, it is more likely that data are collected every minute to every hour and hourly or daily values are then calculated based on those data.

3.3 Water-Budget Hydrology

The volume of water contained in a wetland, V, is an integrated response to all of the hydrological processes that add or remove water. Therefore, if all of the components of a wetland water budget were measured perfectly, the sum of those

processes should equal the volume of water stored in a wetland for any given accounting period. By simply measuring the change in the elevation of the wetland surface, and multiplying that change by the surface area of the wetland, we can obtain a change in wetland volume over an accounting period and relate that change to the hydrologic inputs and losses that occurred over that same accounting period. Relative to the complexities associated with measurement of all of the input and loss terms, measurement of wetland stage should be relatively simple and error free. However, even small measurement error, and poor characterization of wetland bathymetry and geometry, can still result in substantial errors (e.g., Winter 1981).

3.3.1 Stage Measurement

The relative height of the wetland water surface commonly is referred to as wetland stage, herein symbolized as *h*. This is sometimes confused with wetland elevation, which is the height of the wetland water surface relative to a citable datum (reference elevation); for example, North American Vertical Datum of 1988 (NAVD88). This also is not to be confused with wetland water depth, which is the vertical distance from the sediment-water interface (herein, referred to as the wetland bed) to the water surface. Stage typically is determined relative to a local datum, such as a painted mark on a rock outcrop or stable concrete fixture, a pipe or rod driven into the ground, a lag screw placed near the base of a nearby tree, or a benchmark if one is located nearby. Wetland hydrologists commonly make the assumption that the wetland surface is flat and that wetland stage can be measured at any location in a wetland (see Sect. 3.3.3.3 below on how to address seiches for large wetlands). Therefore, measurements typically are made either at a location convenient to the observer or at the deepest point in the wetland if it is expected that the wetland might go dry. In some cases, the wetland bed is artificially deepened at the point of measurement so that the water level can be measured for a short distance below the deepest portion of the wetland during drawdown. A water-table monitoring well installed in the wetland is required to track further water-level drawdowns during prolonged dry periods. Several of the more commonly used methods for measuring stage are described below. Greater detail is provided in a U.S. Geological Survey (USGS) report on methods for making stage measurements (Sauer and Turnipseed 2010).

3.3.1.1 Staff Gage

The simplest and most common method for measuring wetland stage is to visually observe the value where the water surface cuts across a graduated plate placed vertically in the water (Fig. 3.2). Commonly made of fiberglass or enamelled metal, the staff gage is bolted to a stable surface or placed on a pipe or solid rod driven into the wetland bed. Data-collection interval commonly is variable and depends on the

Fig. 3.2 Surveying a wetland staff gage to a local datum. Rod is held on a screw projecting from the plank on which the staff plate is mounted. Water-level is at 26.38

timing and frequency of observer visits to the site. Staff gages are subject to movement because wetland sediments tend to have a relatively large content of organic material and are, therefore, often poorly competent, meaning they are loosely compacted and may readily deform. Pipes or rods to which staff gages are attached should be driven deeper to provide a stable anchor if sediments are soft. If the wetland surface freezes during winter, any change in the elevation of the ice surface over the course of the winter, such as a rising ice surface during snowmelt, can also move the staff gage, either horizontally or vertically. Therefore, the height of the staff gage relative to the local datum needs to be determined at least annually to provide inter-annual continuity of stage data. Staff plates can be stacked vertically if wetland stage varies over a distance greater than the length of a single staff plate. Because this method is so simple and relatively robust, staff-gage values commonly are used as the reference value when automated sensors are used to collect more frequent stage data. Staff plates need to be cleaned regularly to remove chemical or biological accumulations at or near the water surface.

3.3.1.2 Float-Based Gage

A float and counterweight connected to opposite ends of a tape or wire draped over a rotating pulley is another wetland-stage measurement method that has been in use for many decades. The float moves up and down with the wetland water level,

which turns a shaft on which the pulley is mounted. The counterweight maintains tension on the system and keeps the tape or wire taut against the measurement wheel. Earlier versions usually were linked to a mechanical chart recorder, but the rotating shaft now more commonly used is attached to an electrical potentiometer or a device that generates an electrical pulse for a specific degree of shaft rotation (shaft encoder), either of which can easily be interfaced with a digital datalogger. Drag or frictional resistance associated with movement of the float, the float wire, and the rotational resistance of the potentiometer or shaft encoder cause the float to ride higher in the water during a falling water table than during a rising water table (instrument hysteresis). The accuracy of this system is related to a large extent to the diameter of the float. The float displaces a greater volume of water during rising than during falling stage. Because displacement volume is equal to float-immersion depth times the cross-sectional diameter of the float, variation in immersion depth becomes smaller as the float diameter is increased.

3.3.1.3 Bubbler System

A bubbler system, also commonly referred to as a bubble gage, measures water level above an orifice submerged beneath the water surface. A very accurate non-submersible pressure transducer is often used to measure the pressure required to push gas through the orifice; the pressure is proportional to the height of the water column above the orifice. Gas (typically nitrogen or air) supplied by a pressurized cylinder or a small pump is pushed through a flexible hose or pipe to the orifice that is affixed at a stable location beneath the water surface. The tubing or pipe often is buried beneath the wetland bed to prevent disturbance, damage or vandalism. Systems can be designed to either pump gas continuously or to intermittently purge the orifice line and then collect a pressure reading once the gas flow has stabilized. The latter design either uses less gas if a compressed cylinder is the supply or requires less power consumption if a pump supplies the pressurized gas. A bubbler system also allows measurements beneath an ice-covered surface. Data of poor quality may result from siltation of the orifice or if the orifice is placed where surface-water currents are substantial.

3.3.1.4 Capacitance Rod

Capacitance is a measure of the charge that builds up between two plates relative to an applied voltage. Capacitance is directly proportional to the area of the plates and to the dielectric property of the material between the plates, and inversely proportional to the distance between the plates. Since water has a greatly different dielectric property than air, output from a capacitance rod that is partially submerged in water is proportional to the submergence distance. Therefore, capacitance rods should be suspended from a fixed point, such as a stilling well placed in a wetland, so that the water level in the wetland does not go below the bottom or above the top of the rod.

Capacitance rods generally are available in lengths ranging from 0.5 to 2 m and some models contain integrated dataloggers as well as a temperature sensor that provides water temperature output and also corrects for the influence of changing temperature on sensor output. Accuracy generally depends on the length of the rod and is commonly about 1 % of the full scale (e.g., ±20 mm for a 2-m rod). Care must be exercised in determining the vertical placement of the rod so a rising water level does not overtop the rod and possibly damage the data-processing hardware.

3.3.1.5 Submersible Pressure Transducer

A submersible pressure transducer measures the pressure of a column of water above the sensor while it is submerged in the fluid. The most common type is a silicon strain gage, in which electrical resistance across a silicon wafer changes in proportion to the slight deflection (strain) that occurs in response to differential pressure applied across the plane of the wafer (Freeman et al. 2004). Sensors that are vented contain a small-diameter tube that extends from the transducer to the point at which the power and signal wires are connected to a computer or datalogger. The vent allows changes in atmospheric pressure to be transmitted to the sensor so that the output reflects only changes in the height of the water column above the sensor. Non-vented sensors measure the combined pressure of the water column and overlying atmosphere and require use of a separate pressure sensor (barometer) to allow atmospheric pressure changes to be subtracted from output from the non-vented sensor. The advantage of vented sensors is that only one measurement is required, minimizing cost, complexity, and eliminating any measurement error associated with a separate sensor. The problem with a vented sensor is the vent itself. The vent needs to remain completely unobstructed; the vent can become blocked if the cable is inadvertently kinked, for example. If moisture condenses inside the vent so that a water drop extends across the cross section of the vent, changes in atmospheric pressure no longer are completely transmitted to the sensor. Furthermore, corrosion and corruption of sensor output is likely if the water or moisture is transmitted to the sensor housing. Non-vented sensors not only eliminate the vent problem, they also commonly contain an on-board datalogger and do not have any electrical wires extending beyond the sensor housing. This minimizes problems with cable-related leaks. The disadvantage, in addition to the requirement of a separate sensor to measure atmospheric pressure, is that the sensor generally needs to be retrieved to download the data. Measurement error can occur if the sensor is not re-suspended at exactly the same elevation below a control datum.

Output in pressure (P) is converted to stage with the relation

$$\psi = \frac{P}{\rho g} \tag{3.3}$$

where ψ is pressure head (m), P is pressure (Pa), ρ is density of water (kg/m^3), and g is acceleration of gravity (m/s^2). Since pressure is force per area and can be expressed as kg m/s^2/m^2, units for ψ are kg/s^2/m divided by ρg (kg/s^2/m^2), which yields m. As long as the pressure transducer is positioned somewhere within the open column of water in the wetland (i.e., not buried in the sediment beneath the wetland), the pressure head is directly proportional to wetland stage because, within a water column, pressure head is exactly offset by the elevation head (i.e., the height at which the pressure head is being measured). The deeper the transducer is positioned in the wetland, the smaller the elevation head but the larger the pressure head. For example, if the transducer is mounted at a depth 0.1 m above the wetland bed, and the converted output from the transducer is 0.3 m, then wetland depth is equal to elevation head (0.1 m) plus pressure head (0.3 m), which is equal to 0.4 m. If instead the transducer is mounted at 0.25 m above the bed, then the height of the water column above the transducer would be smaller because the transducer would be immersed at a shallower depth in the wetland. In this case, elevation head would be 0.25 m, output from the transducer converted to pressure head would be 0.15 m, and wetland stage would still be 0.4 m.

3.3.1.6 Manual Measurement with a Graduated Rod

In seasonally frozen wetlands, a staff gauge needs to be surveyed annually to account for movement due to frost action or moving ice. Conly et al. (2004) presented a simple and practical method to eliminate this requirement using a water-depth measurement rod. In this method, permanent markers, typically metal stakes, are driven into the wetland sediments at locations where standing water commonly is present. The elevation of the top of the marker stake is surveyed once, which is used to establish the elevation of the wetland bed, which is in turn used to calculate the wetland stage from depth measurements. An observer places a wooden measuring rod, approximately 2 cm in diameter and 1.5 m in length and graduated to 1 mm resolution, on the bed next to each marker stake. A 6-cm diameter metal or plastic disk attached to the base of the rod prevents the rod from being pushed below the sediment-water interface, ensuring greater repeatability of measurements. A notch is cut into the base so the rod can be slid along the side of the metal stake, ensuring consistency of the measurement location among site visits. The observer slides the rod downward until the base touches the bed. Distances from the marker base to the water surface, from the marker top to the water surface, and from the marker top to the wetland bed are recorded, providing redundancy in the water-surface measurement. Conly et al. (2004) reported the accuracy of this method to be on the order of 1–2 cm. While this degree of uncertainty may be too large for daily water budgets, the method provides a useful approach to maintaining inter-annual consistency during long-term monitoring of wetland stage and in determining monthly to seasonal water budgets.

3.3.1.7 Remote Sensors

Several other sensors are capable of measuring wetland stage without coming in contact with the water, some with considerable accuracy. Acoustic sensors transmit an acoustic wave to the water surface and record the time of transmission upon reflection of the acoustic wave back to the sensor. This provides a useful method to monitor the stage of a seasonally frozen wetland when a pressure transducer could suffer damage caused by freezing (Hayashi et al. 2003). Corrections need to be made for air temperature and density to maintain a high level of accuracy. Sensors that transmit and receive a radar pulse operate under the same assumptions. For the radar sensors in particular, the diameter of the water surface over which stage is being determined depends on the transmission beam angle as well as the distance the sensor is mounted above the water surface. Therefore, any object(s) projecting above the water surface that are within the cone of influence can corrupt the measurement. Several acoustic and radar sensors can provide water-level measurements that are within 3 mm of the true value. Laser-based devices also are available, but for water that is particularly clear, the laser beam may penetrate the water rather than reflect off it. Use of a floating reflector positioned inside a stilling well may minimize this problem.

High-resolution satellite images or aerial photographs provide reasonably accurate estimates of inundated wetland areas under ideal conditions. If the relation between wetland area and stage is known (e.g., Eq. 3.4 below), then the wetland stage can be estimated with reasonable accuracy. However, the accuracy of this method depends on the delineation of inundated area, which may be difficult with the presence of emergent vegetation (e.g., Fig. 3.1), and on the accuracy of the stage-area relation.

3.3.2 Converting Stage Change to Volume

Measurement of wetland stage commonly is determined on a short time interval, perhaps every 15 min or once an hour, unless the measurement is made manually. This allows quantification of stage in response to individual precipitation events if precipitation also is determined on a short time interval. However, for the purpose of determining a water budget, change in stage should be determined on the same time interval as the hydrologic component with the longest measurement interval. In most situations, the time-limiting parameter will be evaporation, which rarely is determined on less than a daily interval. Therefore, assuming that all other hydrologic components are determined at least on a daily basis, stage change should be determined based on subtracting wetland stage at midnight from wetland stage during midnight of the subsequent day. In this way, daily change in wetland stage will be integrated over the day, just as is the case for measurement of the rest of the hydrologic components.

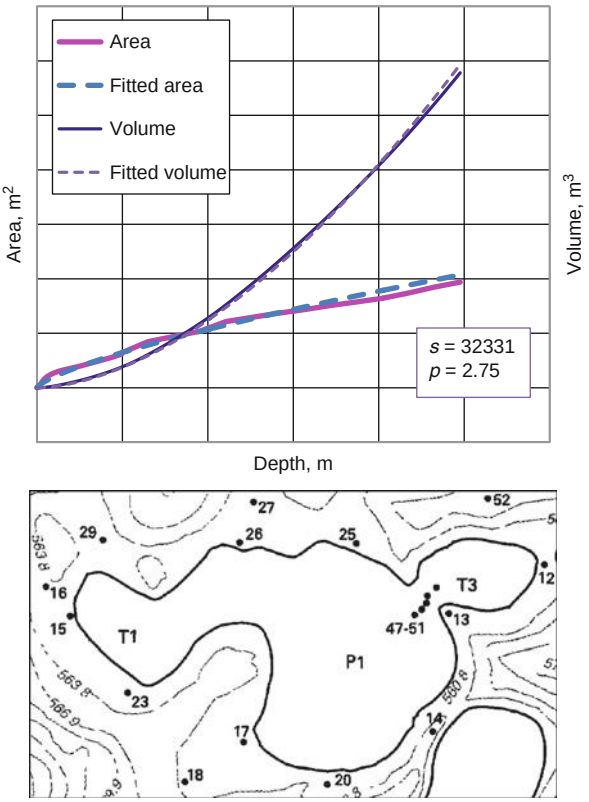

Fig. 3.3 Wetland area and volume related to stage based on a detailed map of wetland bathymetry for an irregularly shaped wetland. D.O. Rosenberry unpublished data for wetland P1, Cottonwood Lake Area, North Dakota

A relation between wetland stage and surface area or volume is needed to determine a volume associated with change in stage. If detailed bathymetry data are available, curves relating wetland area and volume with stage can be generated, from which wetland volume can be determined for any given stage value (e.g., Fig. 3.3). In this case, it is a simple matter of taking the difference between volumes associated with two sequential values of wetland stage to determine change in wetland volume.

Unfortunately, it often is not a simple matter to determine wetland bathymetry. Wetlands commonly are situated in a low-gradient landscape where small changes in stage can result in large changes in surface area. Dense or tall emergent vegetation also can hinder bathymetry determinations based on remote-sensing technology or even on direct observation, as previously noted in Fig. 3.1. It often is necessary to use the brute-force approach and collect high-density measurements of the elevation of the wetland bed at well-determined locations, either with detailed on-site surveying or a combination of surveying and differential global positioning system (GPS). A study of cypress wetlands in Florida, for example, determined location and elevation at 86–145 measurement points/ha in order to generate wetland areas and volumes for every 3 mm increase in wetland stage (Haag et al. 2005).

Lacking such detailed data, it is possible to determine change in volume with reasonable accuracy by making some basic assumptions related to wetland geometry. Assuming a symmetrical wetland basin with the deepest part located at the center of the basin, wetland area can be determined by making an assumption about the change in slope of the wetland basin with distance from the center. Using this approach, Hayashi and van der Kamp (2000) developed the following relation:

$$A = s\left(\frac{H}{H_0}\right)^{\frac{2}{p}} \tag{3.4}$$

where A is wetland surface area, H is wetland depth, H_0 is unit depth (e.g., 1 m), s (m^2) is a scaling factor that is equal to the wetland surface area at H_0, and p is a dimensionless scaling factor that is related to the shape of the wetland basin. For example, if the profile of the wetland bed extending from the center to the perimeter is a straight line, then p is equal to 1. If the wetland is bowl shaped, then p is close to 2. If the wetland has a broad, flat basin that steepens near the wetland edge, then p is somewhere between about 5 and 100, with p increasing to infinity for a rectangular cross section. Wetland volume also can be approximated using the same fitting factors and the equation

$$V = \frac{s}{1 + 2/p}\frac{H^{1+2/p}}{{H_0}^{2/p}} \tag{3.5}$$

(Hayashi and van der Kamp 2000).

An example of comparing fitted and measured values for area and wetland area and volume is shown in Fig. 3.4. Even with an irregularly shaped wetland basin, Eqs. 3.4 and 3.5 approximate values for A and V reasonably well based on measured bathymetry.

3.3.3 Sources of Error

Measurement of wetland stage is conceptually very simple and any given observation has a high likelihood of being very accurate. However, several sources of error can increase as study duration extends to multiple months or years and greatly diminish the accuracy of wetland stage that is very important to a water-budget analysis. The significance of these errors depends on the accuracy requirements. The U.S. Geological Survey, for example, requires an accuracy of ±0.01 ft (3 mm) for water-level measurements over the range typically encountered in most wetland settings (Sauer and Turnipseed 2010).

If daily water budgets are a goal, then measuring stage to a level of precision and accuracy similar to hydrologic fluxes summed over a day would be appropriate.

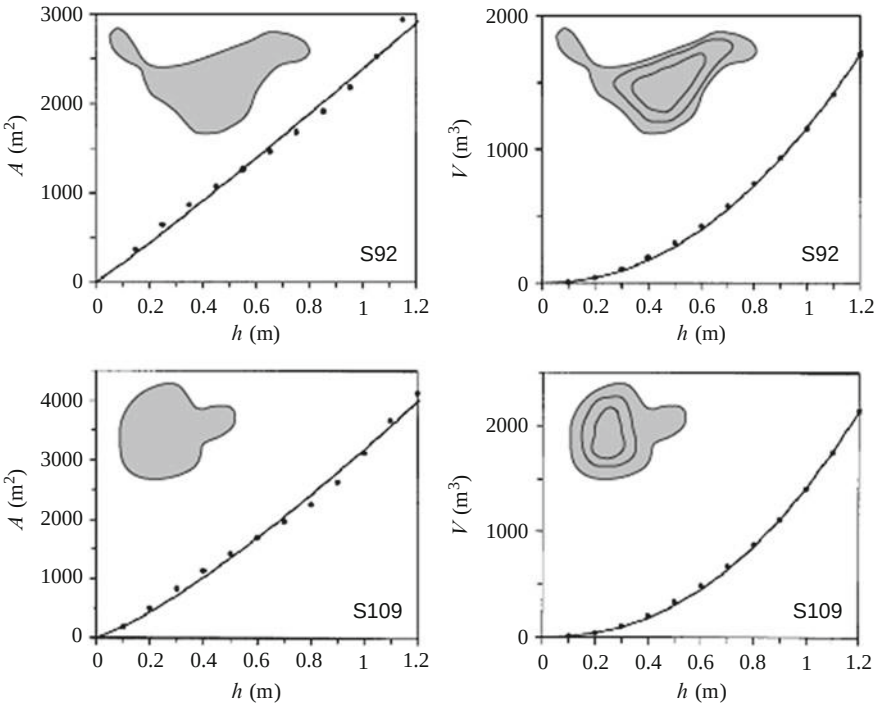

Fig. 3.4 Measured versus modeled areas and volumes of two wetlands in the St. Denis Wildlife Area, Saskatchewan, Canada. Wetland shape and bathymetry shown in *upper left* portion of plots (Modified from Hayashi and van der Kamp (2000)). Published with kind permission of © Elsevier 2000. All Rights Reserved)

Precipitation commonly is measured to the nearest 0.3 mm with an accuracy generally considered to be ±5 to 15 % (Winter 1981). Daily evaporation commonly ranges from 0 to 4 mm and rarely exceeds 6 mm. Accuracy of surface-water measurements depends on the surface-water discharge relative to the wetland surface area. Even if inputs are relatively large and wetland surface area is relatively small, measurement error expressed in terms of wetland stage usually is less than 3 mm. Given these magnitudes of daily hydrologic fluxes common to wetland settings, measuring wetland stage to within 1 mm is not an unreasonable goal, even though it is rarely achieved with current technology.

3.3.3.1 Staff-Gage Errors

Although accuracy of wetland stage to within 1 mm is desirable, it is quite difficult to read a staff gage more accurately than about ±3 mm. Most staff gages are incremented no finer than 3 mm and many display 10-mm increments. Observation errors result from waves that cause the water surface to fluctuate during a gage

reading. Clear water makes it difficult to see specifically where the water cuts across the staff plate. Corrosion or algal growth can obscure the values on the staff plate. A tilted staff plate causes the indicated stage change to be larger than the actual stage change.

As mentioned earlier, a staff gage mounted on a pole or stake driven into the bed can move over time, resulting in a bias in time-series trends. This problem is greatly enhanced in locations where the surface water freezes during winter. Staff gages commonly are pulled upward in the spring when recharge to the wetland causes the floating ice to rise before the ice melts enough to release contact with the staff gage. If wind causes the ice to move horizontally during the melting process, the staff gage can be tilted or even sheared from the pole or stake. In these cases, staff gages need to be resurveyed to a stable datum annually.

3.3.3.2 Local Datum Errors

Elevation of the local datum to which the staff gage is related also can change over time. Lag screws at the base of large trees are surprisingly stable, but a strong wind can cause a tree to lean or fall to the ground. Therefore, many installations make use of multiple datums and annual surveys are made from the staff gage to each one. Some studies have used nearby monitoring wells as stable reference marks for maintaining inter-annual elevation control. Wells associated with wetland studies commonly are shallow because the water table is close to land surface. Shallow wells are not well anchored, leaving them susceptible to frost expansion; some have been observed to move 10 cm or more in a single winter. Therefore, use of a well as a stable reference point is not recommended in environments where soil frost occurs (Rosenberry et al. 2008).

3.3.3.3 Automated Sensor Errors

Automated sensors also are subject to error. Each sensor has specifications that indicate sensor resolution and accuracy. In addition to stated sensor limitations, problems can develop that are specific to particular types of sensors.

Systems that include a float are subject to hysteresis, as described earlier. Errors are proportional to the square of the float and pulley diameters and generally will be less than 3 mm if the float diameter is greater than about 60 mm and the pulley diameter is 0.1 m. Floats also can ride deeper in the water or even sink due to a leak. Debris accumulation on the float can cause the float to ride deeper in the water, creating a bias. A potentiometer connected to a float can fail (basically, wear out) over specific depth increments if the float remains at essentially the same height but waves cause the float and potentiometer to move back and forth (“paint”) for extended periods. During exceptionally high or low water levels, the counterweight may exceed its length of travel, resulting in faulty measurements.

Bubble gages can drift or fail if sediment covers the bubble orifice. The system will fail if the pressurized-gas source is exhausted or if the battery supply is insufficient to power the pump. Output from submerged pressure transducers can drift due to cable or wire stretch, slippage of the point from which the cable or wire is suspended, or simply due to sensor electronic drift (formerly, a common problem). Non-contact sensors that make use of radar or sonar are sensitive to air temperature and sensor height above the water surface. Floating debris, wind-generated waves, rain or snow, and other falling debris (e.g., pollen) can further degrade data quality.

Errors can result from sensor location as well. If the sensor is not located in the deepest part of the wetland basin, the sensor might indicate that the wetland has gone dry while there is still standing water in a deeper portion of the wetland. For larger wetlands, strong wind can generate a seiche, or oscillation of water stage associated with piling of water on the downwind side of the wetland. The effects of seiche can be minimized by averaging multiple measurements over a period longer than the characteristic period of oscillation, which is roughly proportional to the length of the wetland, and inversely proportional to the square root of the depth of water (Wilson 1972).

3.4 Precipitation

Precipitation is the main driver of most wetland water budgets through direct application to a wetland surface and indirect inputs via surface runoff and groundwater discharge (Winter and Woo 1990). Accurate measurements or estimates of precipitation are essential. Compared to other water-budget components, precipitation measurement at a given location (i.e., a point measurement) using a properly designed precipitation gage is relatively straightforward and accurate. However, there are a host of issues that can introduce error in these simple point measurements, such as poor installation or maintenance of instruments. Scaling up from point data, sparse data, or off-site data to precipitation distributed over a watershed also increases the uncertainty of a representative value. In the following sections, we will present methods for making point measurements, indicate potential sources of errors, and then present methods for scaling from point measurements to determining precipitation on a watershed scale.

3.4.1 General Consideration for Point Measurements

To obtain representative values of precipitation over an area of interest, the choice of measurement site, the type and exposure of the instrument, the prevention of evaporation loss, and the reduction of wind effects and splashing all are important considerations (WMO 1994:91). Ideally, a precipitation gage should be located

within an open space in a fairly uniform enclosure of trees, shrubs, or fences, so that wind effects (see below) are minimized. None of the surrounding objects should extend into the volume of an imaginary inverted cone with the apex positioned directly above the sensor and the sides extending from the sensor orifice at a 45° angle (Dingman 2002:112). The gage orifice should be horizontal even on a sloping ground surface. The gage height should be as low as possible to minimize wind effects, which increase with height, but high enough to prevent splash of rain drops from the ground. If the gage is used to measure snowfall, the orifice should be located above the maximum snow height. An orifice height of 0.3 m is used in many countries in areas that receive little snow. A standard of 1 m is suggested for most areas that accumulate larger amounts of snow during winter (WMO 1994:92).

Orifice size needs to be sufficiently large to minimize edge effects and should be known to the nearest 0.5 % for an accurate conversion of volume of water collected (m^3) to equivalent depth of precipitation (mm). An orifice area of 200–500 cm^2 is common (WMO 1994:94). The collection cylinder should be deep enough and the slope of the funnel steep enough to prevent rain from splashing out of the gage. For storage-type gages (see below), a smaller-diameter restrictor should be positioned between the orifice and the collection cylinder, and the cylinder should be covered with a highly reflecting material to minimize loss of water by evaporation. Adding a layer of non-volatile immiscible oil floating on the collected water also minimizes evaporation. Low viscosity, non-detergent motor oils are recommended for this purpose; transformer and silicone oils have been found to be unsuitable (WMO 1994:95).

3.4.2 Type of Precipitation Gages

Precipitation gages can be classified into non-recording and recording types. Non-recording gages generally consist of an open receptacle with vertical sides or a funnel, and a reservoir that stores the collected water. Precipitation is determined by weighing or measuring the volume of water collected in the reservoir, or by measuring the depth of water using a calibrated measuring stick or scale. Care must be taken to minimize observation errors for both graduated-cylinder and weighing-device measurements (see WMO 1994:96–100 for detailed discussion). If measurements are made infrequently, evaporation loss can cause substantial negative bias in the data, or overflow of the collector may occur as a result of unusually heavy storm events. Despite these potential sources of errors, carefully operated non-recording gages present a useful alternative to recording gages because they are simple, accurate, and relatively inexpensive. They are particularly useful for applications that require a large number of points to capture spatial variability of rainfall at a low temporal resolution (e.g., weekly or monthly).

The most commonly used recording devices are the weighing gage and the tipping-bucket gage. Weighing gages measure the weight of the storage reservoir and its contents using electronic sensors, such as a load cell or strain gage, or a

spring. Since these gages store water in a reservoir, similar to non-recording gauges, they also are susceptible to evaporation loss or overflow resulting from infrequent site visits. Weighing gages record cumulative precipitation data at a specified frequency (e.g., hourly). Estimating the amount of rainfall or snowfall during specific time intervals may not be straightforward due to instrument noise. In particular, electronic sensors are sensitive to temperature; even after temperature compensation routines are applied, the data may contain substantial temperature-related error. Therefore, it is best to use weighing gages for recording precipitation over a relatively long interval (e.g., weekly, in which case temperature-related effects can be integrated over a longer time), and tipping-bucket gages for studies requiring greater temporal resolution.

Tipping-bucket rain gages introduce the water received in a funnel to one of a pair of identical vessels (buckets) balanced on a fulcrum. When one bucket is filled, it tips and sends an electronic pulse to a recording device, and the other bucket is brought into position for filling (Dingman 2002:105). These instruments are useful for collecting rainfall data at a high temporal resolution, but they also have some disadvantages. For example, during high-intensity rainfall the sensor measures less rainfall than actually occurs. The bucket does not tip instantly and during the first half of its motion, rain is being fed into the compartment already filled with the designed amount of rainfall (WMO 1994:103). This delay results in systematic negative bias in measured rainfall for high intensity events. Events with less than a minimum amount of precipitation required to tip the bucket also are not recorded (these are called "trace" events). Similarly, a small amount of water collected at the end of an event commonly is left in the bucket and subsequently evaporates, resulting in underestimation of precipitation. The sensitive balance of buckets requires periodic calibration of the amount of precipitation per each tip. Without such calibration, the data may contain a substantial degree of positive or negative bias.

Optical devices are less commonly used, but represent promising new technology (Nitu and Wong 2010). When water passes through an optical scintillation gage it alters the frequency of an infrared beam, which can be analyzed to deduce the time, amount, and intensity of precipitation. In the second type, the sensor measures the extinction caused by precipitation droplets falling through a thin sheet of light, from which precipitation type (rain or snow), amount, and intensity are deduced. The third type measures the forward optical scattering by the particles, from which the precipitation type, amount, and intensity are estimated. In comparison to conventional gages, these optical devices tend to have a larger degree of measurement uncertainty, but they provide useful alternatives when tipping-bucket or weighing gages cannot be used, for example, on a ship or a floating platform affected by wave motion (Nystuen et al. 1996).

Recording precipitation gages are commonly used with internal or external data-logging devices that record the data at a fixed interval (e.g., every 30 min) or record the time stamp of individual tips of a tipping bucket. Many data-logging devices can also accommodate other environmental sensors, such as water-level or water-quality sensors, and transmit the data via telephone, satellite, or wireless communication network.

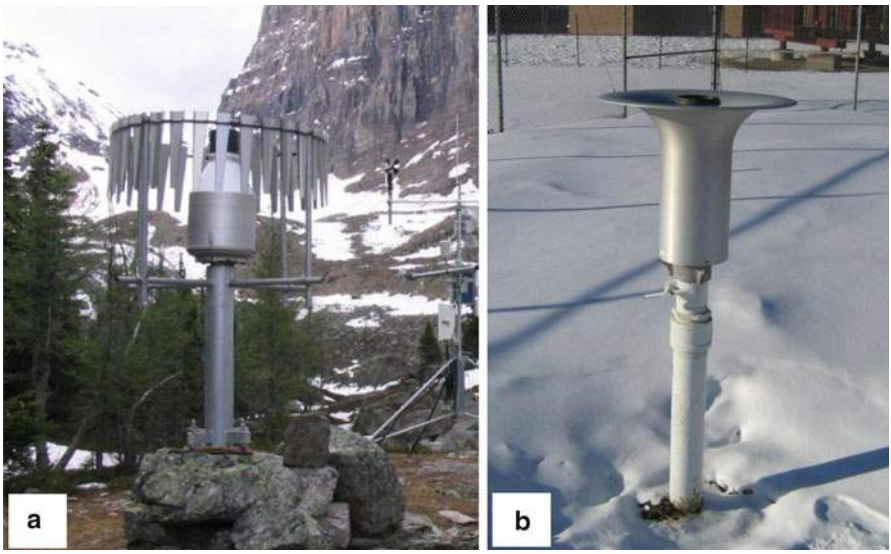

Fig. 3.5 Examples of precipitation gages equipped with a windshield. (**a**) a weighing gage equipped with an Alter shield. (**b**) a non-recording gage equipped with a Nypher shield

3.4.3 Effects of Wind

Precipitation gages that project above the ground surface generate wind eddies that tend to reduce the catch of the smaller raindrops and snowflakes (Dingman 2002:109). This can be a major source of error in precipitation measurements. The amount of measured precipitation relative to "true" precipitation is referred to as gage-catch deficiency. The degree of deficiency generally increases with wind speed and can be on the order of 20 % (Yang et al. 1998). Several types of wind shield are commonly installed around a precipitation gage to reduce wind eddies above the gage orifice (Fig. 3.5). Even with a wind shield, catch efficiency is significantly less than one, especially for snowfall. Empirical correction formulas can compensate for the negative bias caused by wind. Coefficients are usually determined by fitting a regression curve to data that relate precipitation to wind speed (Fig. 3.6). Unfortunately, these curves are fitted to what often are noisy data sets (e.g., Goodison et al. 1998:36–37). Therefore, protection from wind should be a high priority in selecting a site for measuring precipitation, particularly for measurements of snowfall.

3.4.4 Snowfall Measurement

Measurements of snowfall using a precipitation gage have additional challenges even if wind effects are minimized. If a weighing gage is used, the reservoir needs

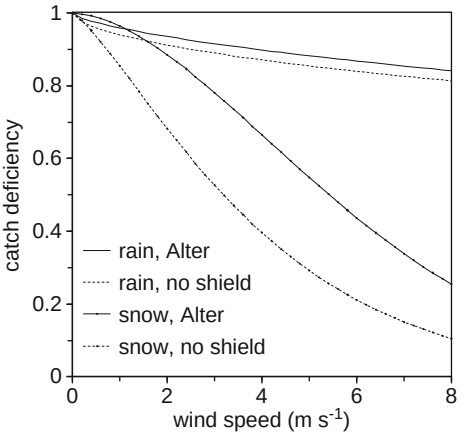

Fig. 3.6 Gage-catch deficiency for rain and snow as a function of wind speed at the orifice height. Data are for the U.S. National Weather Service standard 8-in. gage with and without Alter shields (equations compiled by Dingman 2002:111–112)

to have a sufficient amount of antifreeze solution to melt incoming precipitation, even at very low temperatures. If a heated tipping bucket gage is used, the effects of evaporation due to heating need to be considered. During high-intensity snowfall, snow may pile up at the gage orifice and subsequently blow off, causing negative bias in measured precipitation. If a gage is located in an exposed area, drifting snow may fall into the gage, causing positive bias.

Blowing snow causes redistribution of accumulated snow and substantial sublimation loss, resulting in large local-scale variability of snow density and accumulation. Depending on the surface condition of a wetland and the surrounding upland, the wetland may accumulate higher or lower amounts of snow than recorded by precipitation gages. For example, if a wetland has a smooth-ice surface and the surrounding upland is covered by tall grasses, much of the snow that falls on the wetland may drift to the wetland edge or the upland, where it is trapped by grasses. On the other hand, if a wetland is situated in a relatively deep basin and has extensive emergent vegetation, snow may drift from the upland and accumulate in the wetland. For these reasons, it often is better to quantify and distinguish between snow accumulation on the wetland, and snow accumulation on the surrounding terrain, rather than trying to relate snowfall measured with a precipitation gage to distribution across a landscape of interest.

A snow survey conducted at peak snowpack accumulation, just before the snowpack starts to melt, can integrate, both spatially and temporally, all of the processes that apply and redistribute snow over a landscape, including drifting, loss due to sublimation, and mid-winter melt events. A snow survey should be conducted along an established line, called a snow course, that encompasses the local-scale heterogeneity of landforms and vegetation across the area of interest. At a fixed interval (typically 10–50 m) along the snow course, a snow tube is used to measure the depth of snow and collect a sample for determining snow density by weight. Depending on the scale of survey, the process can be abbreviated by measuring depth at all of the sites and measuring density at a subset of those sites. In this case, the depth-density relation needs to be established to estimate

density for those points with only depth measurements. The amount of snow water equivalent, commonly abbreviated as SWE, is calculated from the product of depth and density. Details on snow course measurements are described in WMO (1994:117–122).

In many studies, the amount of precipitation is recorded without differentiating whether it is rain or snow. Local air-temperature data can be used to separate rainfall from snowfall. The transition from snow to rain normally occurs at temperatures between 1 and 3 °C (e.g., Dingman 2002:108).

3.4.5 Calibration and Maintenance

Precipitation gages need to be recalibrated at specific intervals to operate at their stated accuracy. Detailed calibration procedures are specific to the instrument, and are usually found in the manufacturer's operation manual. Calibration is simple for manual gages; it involves accurate measurement of the gage orifice to confirm that orifice dimensions are as stated in the manufacturer's specifications. If not, a custom multiplier can be applied to the data. For weighing gages, known weights of water are added to the gage to validate the gage reading. For tipping-bucket gages, a known volume of water is slowly poured through the gage (e.g., 5–10 tips per minute) and the total number of tips is used to determine the amount of precipitation (mm) per tip, which can be different from the manufacturer's specification by as much as 10 % (Marsalek 1981).

Precipitation gages need to be periodically cleaned (at least once every field season) to prevent clogging of the funnel opening, accumulation of dust in the tipping buckets, and to minimize friction of all moving parts. Gages should be checked to ensure they remain level. Any potential obstruction, such as fallen woody debris or vegetation growing close to the orifice, should be removed. Recording instrument calibrations and servicing in a maintenance record is useful. Such information should ideally be kept in a metadata file accessible in the same area as the data. This advice applies to collection of all automated data related to wetland hydrology and eliminates confusion during subsequent data processing and analysis, or when someone other than the operator uses the data.

3.4.6 Spatial Variability and Interpolation

Precipitation can have large spatial variability due to variation in elevation, topography, prevailing wind direction, and distance from moisture sources or from urban areas. For example, the amount of precipitation may systematically decrease on the lee side of mountain ranges. Urban "heat islands" may have complex effects on the local distribution of precipitation. Convective storms have a relatively small area of influence, often resulting in a large difference in rainfall between sites located

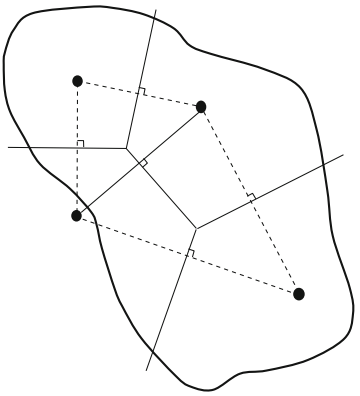

Fig. 3.7 Construction of Thiessen polygons by: Step-1, drawing the dashed lines connecting adjacent gages, Step-2, drawing solid lines perpendicularly bisecting the dashed lines, and Step-3, dividing the area into four polygons defined by the solid lines

within and beyond the narrow storm track. For these reasons, precipitation gages need to be located as close as possible to the wetlands being studied. Determining precipitation over a large wetland or drainage basin of wetlands may require multiple gages to capture spatial variability. If precipitation data from nearby weather stations are used, instead of an on-site gage, errors resulting from spatial variability should be considered carefully.

To estimate areal precipitation using data from multiple stations, or to estimate local precipitation using data from distant weather stations, data need to be interpolated spatially using one of several surface-fitting method (Dingman 2002:118–130). The weighted-average method estimates precipitation (P_{av}) by the sum of the product of individual station data (p_i) multiplied by a station-specific weighting factor (w_i):

$$P_{av} = \sum_{i=1}^{N} w_i p_i \quad \text{and} \quad \sum_{i=1}^{N} w_i = 1 \tag{3.6}$$

where N is the number of stations. The simplest averaging scheme is the arithmetic mean that uses $w_i = 1/N$ for all stations. Another commonly used scheme is the Thiessen polygon method, in which the area of interest is divided into N polygons as shown in Fig. 3.7, and w_i is given by the area of each polygon divided by the total area. Other surface-fitting methods are more convenient for constructing precipitation maps from a large or relatively dense network of gages. In these methods, the area is divided into a large number of grid cells and precipitation for each grid cell is computed from station data using a weighted average scheme similar to Eq. 3.6. The most commonly used weighting schemes include the inverse distance method, where w_i is inversely proportional to the distance between the grid cell and the station, and the kriging method, which assigns w_i based on geostatistical correlation among station data (e.g., Kitanidis 1997). These methods are available in popular software packages such as ArcGIS (Environmental Systems Research Institute Inc.) and Surfer (Golden Software Inc.). The software packages can be used to estimate total precipitation over the area, or point values of precipitation for a location that does not have a local precipitation gauge.

3.4.7 Missing Data and Other Issues for Using Archived Weather Data

It is not uncommon to have gaps in precipitation records due to malfunctioning equipment, temporary closure of a station, and various other factors. Missing data are often estimated using data from other precipitation gages, preferably located close to the site. This estimation can be made using the weighted average method (Eq. 3.6), or using the normal-ratio method (Dingman 2002:115) that estimates missing values (p_m) from long-term average precipitation at the station with missing record (P_0), the long-term average precipitation at each station (P_i), and the data for the missing time interval (p_i) at other stations

$$p_m = \frac{1}{N} \sum_{i=1}^{N} \frac{P_0}{P_i} p_i \tag{3.7}$$

where N is the number of nearby stations with valid data. The long-term average may be annual or monthly depending on the site climatic condition and the nature of analysis (e.g., weekly or monthly water budget).

Archived precipitation data from other sources should to be examined for consistency, particularly if the user does not have first-hand knowledge of the station history and conditions. Changes in measurement method, gage location, or the surrounding environment can induce artificial offsets or trends in the data (Dingman 2002:117). It is important to review the station history, if available, and also use a double-mass curve technique to identify any suspicious data. A double-mass curve is a graph showing cumulative monthly or annual precipitation from a reference station on the horizontal axis and cumulative precipitation from the station of interest on the vertical axis (Searcy and Hardison 1960). The slope of a double-mass curve should be constant if there has been no change in the station of interest. If there is a statistically significant change in slope, the data from the station of interest can be multiplied by a correction factor to compensate for the change. The reference station should have consistent data and be located reasonably close to the station of interest.

In addition to checking the consistency, attention should be paid to gage calibration and correction procedures. For example, some weather stations operated by governmental and municipal agencies may not apply corrections for gage-catch deficiency, resulting in a negative bias in measured precipitation, snowfall in particular. In recent years, gridded precipitation data have become available from government agencies such as the U.S. National Oceanic and Atmospheric Administration (NOAA). These data are generated over a large region (e.g., North America) typically by interpolating observation data and/or refining numerical weather model outputs. While these data sets offer convenient means to estimate precipitation for a given region, the data are not intended as a surrogate for local precipitation measurements. Therefore, gridded precipitation data should be validated using observational data from local stations.

3.4.8 Summary

Precipitation measurement provides critical input data to water-budget analysis. With proper installation and operation of precipitation gages, and sufficient attention to calibration and site maintenance, it is feasible to achieve an accuracy of 5–15 % in point precipitation measurement. Spatial variability adds another degree of uncertainty for larger areas of interest, but the uncertainty can be reduced by using multiple gages and appropriate spatial interpolation techniques.

3.5 Evapotranspiration

Evapotranspiration (ET) is the combined flux from surface water to the atmosphere resulting from evaporation and transpiration from plants. Evaporation is the process of converting liquid water to water vapor, along with the transport of that vapor from the water surface to the atmosphere. Transpiration is a similar process, but one that occurs in plants. Liquid water is pulled through roots from the soil and transported to the plant leaves. The vaporization process occurs within plant leaves and the release of water vapor to the atmosphere occurs via small openings on the leaf surface called stomates. We usually cannot distinguish between evaporation and transpiration so hydrologists generally combine the measurement of these processes (e.g., Shoemaker et al. 2011). Most of the time, this is sufficient from a water-budget perspective. Unless loss of wetland water to groundwater is unusually large, ET usually is the largest water-budget loss term for wetlands that do not have a surface-water outlet and is, therefore, an important term to quantify as accurately as possible.

3.5.1 Commonly Used Methods

Vaporization of water is an energy-intensive process; 4.2 J are required to raise 1 g of water 1 °C whereas approximately 2,500 J are needed to vaporize 1 g of water. Both evaporation and transpiration are dependent on the amount of energy available to drive the process. They also depend on the relative availability of liquid water as well as on the ability of the atmosphere to remove the water vapor once it is formed, thereby allowing for the formation of additional water vapor at the wetland surface. Since wetlands by definition are settings that generally (although not always) have an ample water supply, it is usually assumed that ET occurs at a rate that is unlimited from a water-supply perspective. Therefore, ET is assumed to occur at the maximum potential ET rate based on available energy. This same assumption cannot be made regarding the ability of the atmosphere to remove the recently vaporized water, however. The vapor-removal component of ET depends on the

degree of instability of the lower atmosphere. If the atmosphere is stable, in which case, temperature and density both decrease with increasing elevation, there is very little vertical movement that could otherwise aid the vapor-removal process. Therefore, most methods for quantifying ET also measure the vapor-pressure gradient, wind speed, or both, just above the surface supplying water for ET.

3.5.2 Direct Measurements

Of the options available for quantifying ET, only two can be considered capable of directly measuring the process. The evaporation pan is very simple, but requires a coefficient that can be difficult to determine. The eddy-covariance method often is considered a direct method because it has a sound theoretical foundation and requires no assumptions regarding atmospheric stability or the wind-speed velocity profile. It also requires extensive instrumentation and data processing.

3.5.2.1 Evaporation Pan

The evaporation pan is perhaps the most direct method for quantifying ET. It consists of a cylinder nearly filled with water with the top open to the atmosphere. Evaporation is determined by totalling the water added to the system, minus water removals following rainfall, required to maintain the water at a constant level. Pans of a variety of shapes, sizes, and depths have been used over many decades. Since the 1950s and 1960s, many national networks (e.g., Jovanovic et al. 2008) have adopted the class A pan configuration of 1.21-m diameter and 0.25-m height with a stilling well positioned inside the tank to facilitate accurate measurement of the water level (Fig. 3.8). To ensure uniformity, the standard installation is located in an open area at least 20 m by 20 m with close-cropped grass on a wooden platform 15 cm above ground (Allen et al. 1998). Some sites also measure wind speed just above the rim of the pan as well as air and water temperature to make additional empirical corrections for those environmental variables (e.g., Harbeck et al. 1958).

The amount of water lost from an evaporation pan almost always is larger than water lost from a nearby lake or wetland because of enhanced wind flow across the pan surface and radiational heating of the sides of the pan. A pan coefficient of 0.7 is generally applied to convert measured ET to actual ET on an annual basis (Dingman 2002), although values have been reported ranging from 0.64 for the Salton Sea, California, to 0.81 for Lake Okeechobee, Florida (Linsley et al. 1982). Numerous studies have determined site-specific monthly coefficients that vary over a much larger range to achieve greater accuracy (Kohler 1954; Abtew 2001; Masoner et al. 2008). Several studies have employed a floating pan situated in the water body of interest. For small wetlands, this method works well because waves do not become large enough to overtop the floating pan and corrupt the data. Data from a floating pan situated in a small wetland near Norman, Oklahoma, were within 3 % of

Fig. 3.8 Two standard class-A evaporation pans in use near a lake in northern Minnesota. One is serviced daily and the other weekly (Photo by Donald Rosenberry)

evaporation-chamber measurements and were considered to represent actual ET with no correction factor (Masoner and Stannard 2010). Comparisons with the empirical Priestley-Taylor method (discussed in the Combination methods section) indicated that the Priestley-Taylor method over-estimated ET during the day because the air over the hot, dry landscape surrounding the wetland did not represent atmospheric conditions directly over the evaporating water in the wetland. The study also indicated that the floating-pan measurements over-estimated ET during mid to late afternoon and under-estimated ET during nighttime to early morning. This was attributed to the shallower depth of the water inside of the pan being more sensitive to diurnal air-temperature changes than the deeper water column adjacent to the floating pan. However, the errors were largely offset so the floating pan provided daily ET values with little bias.

3.5.2.2 Eddy-Covariance Method

The process of evaporation can be viewed as vapor-rich rotating eddies of various sizes that rise because they are less dense than other volumes of drier air that descend to occupy the volume that the moist, rising air just vacated. The process is 3-dimensional and also occurs on horizontal axes, but for the purpose of determining evaporation from a wetland surface we are most concerned with the vertical axis. Sensors measure the vertical velocity of these air packets, as well as their "concentrations" (either temperature or absolute humidity) to obtain the vertical velocity of the upward or downward flux of these properties. Vertical flux is then

represented as a covariance between measurements of vertical velocity and concentration of either humidity (vapor flux) or temperature (sensible heat flux).

Mathematically, the process can be presented for latent heat flux ($\lambda\rho_w E$) and sensible heat flux (H) as:

$$\lambda\rho_w E = \lambda\overline{w'\rho'_v} \tag{3.8}$$

$$H = \rho_a c_p \overline{w'T'} \tag{3.9}$$

where overbars indicate an average (typically a 30-min average), primes indicate departures from the mean, λ is the latent heat of vaporization (J kg^{-1}), ρ_w is the density of water (kg m^{-3}), E is evapotranspiration flux (m s^{-1}), w is the vertical wind speed (m s^{-1}), ρ_v is the absolute humidity (also called vapor density) (kg m^{-3}), ρ_a is the density of air (kg m^{-3}), c_p is the specific heat capacity of air (J kg^{-1} °C^{-1}), and T is the air temperature (°C). Measurements are typically made 10–20 times a second. For both latent and sensible heat fluxes, units are in J m^{-2} s^{-1} or W m^{-2}. E rather than ET is used in Eq. 3.8 to be consistent with other literature that describes the evaporation process. The process is identical in wetland settings, although the source for some of the water is via the stomates of leaves. In this chapter, ET refers to the process of evapotranspiration and E refers to evapotranspiration flux in units of distance per time.

The above description is suitable for flat, open areas with uniform vegetative cover over long distances upwind of the sensors. The process is a bit more complex on sloping land surfaces or where air streams converge or diverge upwind of the sensors, in which case covariances are determined on three axes and coordinate rotations to the data may need to be performed before E is determined (Wilczak et al. 2001). A krypton hygrometer or infra-red gas analyzer, and sonic anemometer, are typically used to measure humidity and wind speed, respectively. Temperature is provided either as a by-product of the sonic anemometer or from a separate sensor. Instrumentation has improved rapidly in this field; newer sensors are much more robust and are now capable of being deployed during rain events. This method is sensitive to misalignment; sensors need to be deployed and maintained on a stable platform and at a constant orientation (Wilczak et al. 2001). Height of deployment is strongly related to the upwind area that the measurement represents. The roughness of the upwind area also greatly affects the sensor signal.

3.5.3 Estimation Methods

Numerous empirical methods have been developed to estimate evaporation, ranging from methods that require only measurement of air temperature to methods that have a sound physical basis and require numerous parameters. Methods can be grouped into those that quantify available energy to determine evaporation as the residual,

those that quantify evaporation based on the aerodynamics of the near-surface atmosphere, and those that combine energy and aerodynamic approaches.

3.5.3.1 Energy-Balance Method

Energy removed in the evaporation process is usually offset by resupply of energy from the wetland and the atmosphere. Solving for evaporation by accounting for all of the other energy terms can be expressed as:

$$Q_n - \lambda\rho_w E - H = Q_x - Q_v \tag{3.10}$$

where Q_n is net radiation, Q_x is increase in energy stored in the wetland water column, and Q_v is the net amount of energy advected to the wetland from the sum of streamflow to and from the wetland, groundwater flow to and from the wetland, and rainfall. Atmospheric terms are on the left side and water and sediment terms are on the right side of the equation. Because of the errors associated with determining Q_x and Q_v, the accounting period for this method historically has been 5 days or longer but newer instrumentation has led some to determine evaporation using this method on a daily basis.

Unfortunately, neither $\lambda\rho_w E$ or H can be directly measured, requiring the use of the Bowen ratio (B):

$$B = H/\lambda\rho_w E \tag{3.11}$$

The Bowen ratio can be determined by measuring differences in temperature and vapor pressure in the atmosphere directly above the evaporating surface:

$$B = \gamma\left(\frac{T_s - T_a}{e_s - e_a}\right) \tag{3.12}$$

where γ is the psychrometric constant, T_s is the temperature at the water surface, T_a is the air temperature, e_s is the saturation vapor pressure at the temperature of the water surface, and e_a is the atmospheric vapor pressure. T_a and e_a are measured at the same height above the water surface, commonly 2 m. The psychrometric constant is not really a constant but is a function of specific heat capacity and atmospheric pressure. It is equal to

$$\gamma = \frac{c_p P}{0.622\lambda} \tag{3.13}$$

where c_p and λ are as described above, P is atmospheric pressure, and 0.622 is the ratio of the molecular weights of water vapor and air (Perez et al. 1999).

Substituting B into Eq. 3.10, and assuming that Q_v is negligibly small (a reasonable assumption for most wetland settings), we obtain

$$E = \frac{Q_n - Q_x}{\lambda \rho_w (1 + B)} \tag{3.14}$$

This expression is commonly termed the short form of the energy-budget evaporation equation. For a version that includes individual terms for net radiation, as well as Q_v and heat transfer to and from wetland sediments (Q_b), see Parkhurst et al. (1998) or Winter et al. (2003).

Determination of Q_x can be challenging for some wetlands where water depth is sufficiently large that the wetland column is thermally stratified. In that case, several temperature sensors are required at different water depths to represent the change in heat stored for each depth increment, or horizontal slice, of wetland water. The change in heat can be expressed as

$$Q_x = \sum_{i=1}^{n} \rho_{wi} c_{wi} h_i \frac{\Delta T_{wi}}{\Delta t} \tag{3.15}$$

where h_i and T_{wi} are the thickness and average temperature, respectively, of each horizontal slice of wetland water, ρ_w and c_w are defined as before but now apply to the water in each specific depth increment of the wetland, and Δt is the time between two successive temperature measurements. Q_x is the sum of the change in heat for all depth increments. The same procedure can be used in the soil where standing water is not present if temperature sensors are installed at several depths beneath the wetland bed, with the deepest sensor at a depth where temperature does not change considerably over periods less than weeks to perhaps a month. In this case, the term for changes in heat stored in the sediment commonly is referred to as G rather than Q_x.

In drier environments, where soil at the surface is not at or near saturation, the assumption cannot be made that vapor pressure at the surface is at saturation. In these cases, temperature and vapor pressure need to be measured at two heights. Because differences often are very small, and instrument bias could lead to substantial error, it is common to use a device that alternates the positions of the upper and lower sensor so that bias can be subtracted.

Several other empirical formulations are used to determine ET based on estimation of available energy. Most require measurement of either solar or net radiation, or air temperature, or both. Three that use solar radiation and air temperature are Jensen-Haise, Makkink, and Stephens-Stewart (McGuinness and Bordne 1972). Most radiation-temperature models can perform fairly well in the environment for which they were developed but do not transfer well when applied in other climates (e.g., Rosenberry et al. 2004, 2007). Several others require measurement only of air temperature and day length (e.g., Blaney-Criddle and Hamon methods) or air temperature only (e.g., Papadakis and Thornthwaite methods). The Thornthwaite method, in particular, has been widely used because of its simplicity.

However, it has been shown based on comparisons in several settings that these rather simplistic formulations provide ET values with much greater uncertainty than methods that are more physically based (Winter et al. 1995; Dalton et al. 2004; Drexler et al. 2004; Rosenberry et al. 2004, 2007; Elsawwaf et al. 2010).

3.5.3.2 Energy Balance Method with Large Aperture Scintillometer (LAS)

Sensible heat flux from the ground causes variations in the refractive index of the atmosphere, referred to as scintillation. These variations can be detected by a scintillometer using a beam of light emitted by the transmitter and detected by a receiver, from which sensible heat flux is estimated (Hill et al. 1992). A large aperture scintillometer (LAS) measures scintillation over a horizontal line at a scale of several hundred meters. Using sensible heat flux estimated by the LAS method with measurements of net radiation and other components of the energy balance, it is possible to estimate latent heat flux and evapotranspiration from Eq. 3.10. Brunsell et al. (2008) applied the LAS method at a tall grass prairie site in Kansas and obtained ET values that were comparable to measurements by the eddy-covariance method. This method is not as widely used as the eddy-covariance method, but offers a promising alternative for larger-scale measurements of ET.

3.5.3.3 Aerodynamic Methods

Numerous equations have been developed to determine ET based on the removal of water vapor (mass transfer) from the evaporating surface. Many are referred to as Dalton-type equations, named after the English chemist John Dalton who first formulated such an equation in 1802 (Dingman 2002), and are of the form

$$E = N \cdot f(u)(e_s - e_a) \tag{3.16}$$

where N is a locally-determined coefficient (not required for some formulations), $f(u)$ is a function of wind speed, and $e_s - e_a$ is the vapor-pressure difference presented in Eq. 3.12. N, if present, generally is determined by regression of the product of the wind function and the vapor-pressure difference (the mass-transfer product) with another means of determining evaporation.

Rasmussen et al. (1995) presented a comparison of seven versions of this basic equation applied to a number of lakes in Minnesota. Singh and Xu (1997) presented results of 13 mass-transfer equations applied to data from 4 climate stations in Ontario, Canada. Results indicated a general insensitivity to wind speed. In addition, methods with parameters determined for one location did not transfer well when applied to other locations.

A range of methods for quantifying ET, including the mass-transfer method with locally determined values for N, were compared with the energy-budget approach at a

small wetland in North Dakota and at a small lake in New Hampshire. Mass-transfer results were within 20 % of energy-budget results 45 and 57 % of the time at the North Dakota and New Hampshire sites, respectively, and were among the poorest methods in comparison to energy-budget results (Rosenberry et al. 2004, 2007).

3.5.3.4 Combination Methods

Combination methods determine ET based on measuring both available energy and aerodynamic efficiency. Probably the best known is the Penman method (1948), which often is written as

$$E = \frac{\Delta}{\Delta + \gamma} \frac{Q_n - Q_x}{\lambda \rho_w} + \frac{\gamma}{\Delta + \gamma} E_a \tag{3.17}$$

where Δ is the slope of the saturation vapor pressure versus temperature curve at the mean air temperature, γ is the psychrometric constant described in the *Energy-balance methods* section, and E_a is described as the drying power of the air and is basically a mass-transfer product, as described in the *Aerodynamic methods* section. The first and second terms on the right side of the equation are the radiation and aerodynamic terms, respectively; hence, a combination method. Many mass-transfer products have been associated with the Penman method. A form of the equation that includes an often-used mass-transfer product in place of E_a, along with a multiplier to convert to units of mm/day, is (Rosenberry et al. 2007):

$$E = \frac{\Delta}{\Delta + \gamma} \left(\frac{Q_n - Q_x}{\lambda \rho_w} \right) \times 86.4 + \frac{\gamma}{\Delta + \gamma} (0.26(0.5 + 0.54 u_2)(e_{sa} - e_a)) \tag{3.18}$$

where u_2 (m s^{-1}) is wind speed measured at 2 m above the water surface, e_{sa} (hPa) is the saturation vapor pressure at the air temperature, and e_a (hPa) is the measured vapor pressure at 2 m height. One of the benefits of using the Penman equation is temperature and vapor pressure only need to be measured at one height. Penman originally formulated this method to use T_a as the temperature at which to obtain Δ and e_{sa} because T_s was considered difficult to measure.

The Penman method requires a lot of data: net radiation, temperature of the water body at multiple depths, air temperature, vapor pressure of the air, and wind speed. Numerous simplifying assumptions have been made to reduce the data requirements. The Priestley-Taylor (1972) method is likely the best of these alternate approaches. It assumes that the aerodynamic portion of the equation is 26 % of the energy term and replaces the aerodynamic term with the coefficient 1.26 applied to the energy term. Therefore, only T_a, Q_n, and Q_x need to be determined:

$$E = 1.26 \frac{\Delta}{\Delta + \gamma} \frac{Q_n - Q_x}{\lambda \rho_w} \tag{3.19}$$

Despite its simplicity, the Priestley-Taylor method applied to data from a wetland in the prairie-pothole region of North Dakota produced better data that the Penman method when compared to results from the energy-budget method (Rosenberry et al. 2004).

Another type of combination method that may work well for small wetlands, or wetlands that are moisture limited, is the complementary relationship method. First proposed in 1963, the complementary-relationship method makes the assumption that actual evapotranspiration (*AET*) is reduced relative to evapotranspiration in a wet environment (ET_{wet}) to the same extent that potential evapotranspiration (*PET*) is larger than ET_{wet}; the drier the environment, the greater the difference between *AET* and *PET*. Following this logic,

$$AET + PET = 2ET_{wet} \tag{3.20}$$

By calculating ET_{wet} based on available energy, and measuring *PET*, one can then determine *AET* with

$$AET = 2ET_{wet} - PET \tag{3.21}$$

Any number of methods can be used to determine ET_{wet} and *PET* (e.g., Morton 1983a). One approach is to determine ET_{wet} using the Priestley-Taylor method and determine *PET* using the Penman method (Brutsaert and Stricker 1979). As one might expect, the Brutsaert-Stricker method applied over a wetland in North Dakota (Rosenberry et al. 2004) and over a small lake in New Hampshire (Rosenberry et al. 2007) gave results very close to either the Penman or the Priestley-Taylor methods alone because the environment was wet; ET was occurring at the potential rate. However, wetlands do not always have an ample water supply. During times when wetlands are relatively dry, the complementary relationship method likely would produce better results than other methods designed to indicate ET at the potential rate. The method also can be used to determine the extent to which warm, dry air may increase ET along the upwind edge of wetlands, a particular concern for wetlands that are small or situated in arid environments (Morton 1983b). This also is a concern for other ET methods that rely on the assumption that vapor and temperature gradients are adjusted to the wetland surface at the point of measurement. However, at a wetland in North Dakota, the open-water portion of which varied in size from 1.5 to 3 ha, insufficient fetch was found to cause errors in ET estimates of less than 2 % (Stannard et al. 2004).

3.5.4 Measurement Parameters for Estimating ET

Quantification of net radiation is required for most of the ET methods. This involves measurement of downward shortwave and longwave radiation from the atmosphere, upward reflected shortwave and longwave radiation, and upward longwave

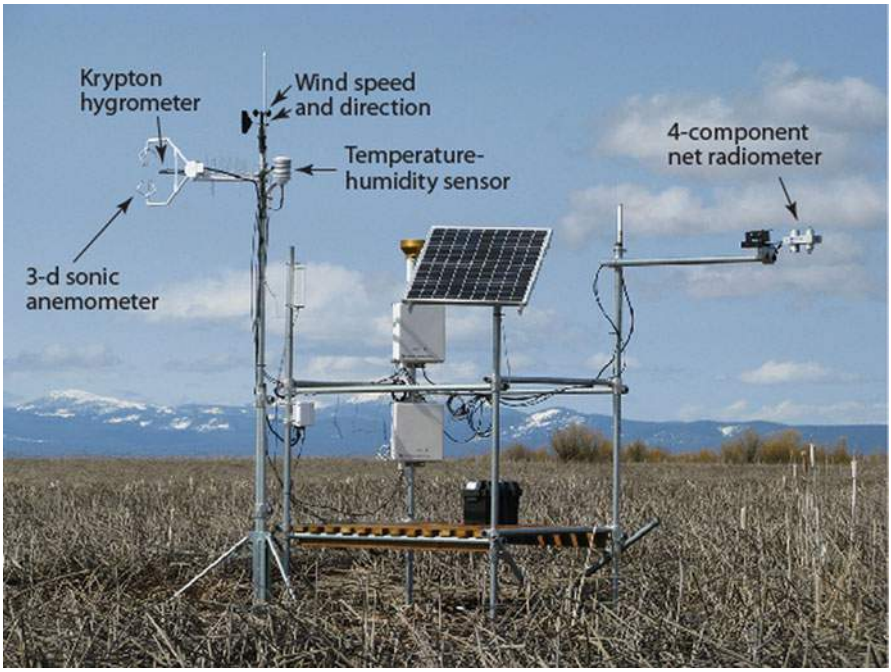

Fig. 3.9 Upward and downward facing 4-component net radiometer deployed over a large wetland in southern Oregon. Other sensors also are labeled (Photo printed with kind permission of © David Stannard, U.S. Geological Survey 2014. All Rights Reserved)

radiation emitted from the land or water surface proportional to the temperature of the surface. Several broad-spectrum radiometers with sensors facing both upward and downward can provide a single net-radiation value. Another option is to deploy four sensors, two facing upward to separately measure shortwave and longwave radiation, and two facing downward to separately measure shortwave and longwave radiation (Fig. 3.9). Although deploying four sensors provides greater accuracy, it comes at a substantially larger cost. If only two upward-facing sensors are deployed, the upward radiation vectors can be calculated with reasonable accuracy (Sturrock et al. 1992; Parkhurst et al. 1998; Winter et al. 2003). Sensors need to be deployed on a stable platform and maintained in a level, horizontal orientation to minimize bias.

All evaporation methods require measurement of air temperature and most also require measurement of humidity at the same location. For gradient-based methods, sensors usually are deployed that provide both temperature and relative humidity. With both parameters, humidity output can be converted to vapor pressure, vapor density, or whatever form of humidity is needed for a specific ET method. Several ET methods require measurement of the water-surface or land-surface temperature and make the assumption that vapor pressure is at saturation based on the surface temperature. Methods that quantify change in heat stored in the water body (Q_x) require a temperature sensor for each horizontal slice of water contained in the

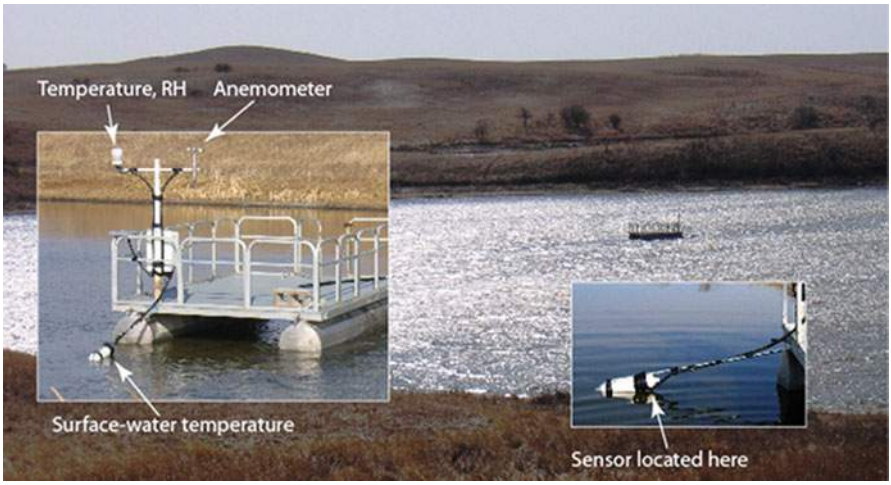

Fig. 3.10 ET raft deployed on Cottonwood Lake Area Wetland P1. *Inset on left* shows raft being put into place just after ice out. Surface-water-temperature sensor is located near the shallow end of the float so it is resting just below the water surface. The white float minimizes heating from solar loading (Photos by Donald Rosenberry)

water body. For settings where substantial horizontal variation in temperature exists, such as in a wetland that has several embayments of differing total depth, profiles of temperature sensors may need to be deployed at several locations to accurately represent change in heat stored in the entire wetland. For other wetlands, a single deployment of sensors at different depths may be sufficient. Rosenberry et al. (1993) provide additional information regarding the errors associated with making measurements at a single location relative to multiple locations. Temperature usually is measured with a thermistor, which has a non-linear, inverse electrical resistance relative to temperature. Polynomial functions are applied to the resistor output to provide temperature that commonly is within ±0.1 to 0.5 °C of true temperature. Thermocouples are inexpensive and stable temperature-measurement devices based on the principle that an electrical current is generated when a junction is made between two dissimilar metals. The current is proportional to the temperature difference between two electrical junctions at different locations in the same circuit. Output is linear and stable, but small. Temperature difference usually is related to a separate temperature sensor located very near one of the bi-metal junctions. If wires are well shielded with insulation so that temperature gradients are small (or at least remain constant) in the vicinity of the reference thermometer, modern dataloggers are well capable of resolving the small current changes that occur in response to temperature changes.

Wind speed often increases approximately logarithmically with height above a surface. Measurement of wind speed needs to be specified with regard to height above the water surface. Measurement height is somewhat arbitrary, but 2 m is common. Wind speed, air temperature, and humidity should all be measured at the same height (Fig. 3.10). All anemometers have a threshold below which the sensor

indicates zero wind speed. A sensor should be selected with the lowest threshold that is affordable. Most sensors have cups or propellers that spin in response to the wind and generate either a pulse per revolution or direct current (DC) volts proportional to the rotational velocity. Ultrasonic anemometers operate based on resolving the difference between times of transmission of ultrasonic pulses sent in both directions between two transducers. Combinations of sensors are commonly oriented along two or three axes to determine horizontal or 3-dimensional wind speed, respectively.

Either dataloggers or field computers can be used to measure and store data from analog and digital sensors that provide data to feed evaporation methods. Sensor scan rate usually is determined based on the suite of sensors connected to a particular datalogger as well as requirements specific to evaporation methods or other products generated from the installation.

The most representative data are collected over the wetland in a location designed to maximize fetch in all directions, or at least in the directions of the prevailing winds. Sensors commonly are deployed from a floating raft or a platform fixed to the bed, depending primarily on the wetland depth. Sensors that need to be deployed at a constant height above the water surface, such as temperature-humidity sensors and anemometers, need to be adjusted as the water level of the wetland rises and falls. The temperature probe at the surface should be installed so it rests as close to the water surface as practically possible. A float is commonly used to keep sensors at the proper height above the water surface (Fig. 3.10). Data from land-based sensors often are used if a raft or platform is not available. In some cases, use of land-based sensors results in little additional error (Rosenberry et al. 1993).

3.5.5 Measurement Errors

ET is difficult to estimate or measure in part because of the numerous sources of error associated with the large number of sensors required. Error varies substantially depending on the chosen measurement method.

The evaporation pan has sources of error associated with the actual physical measurement as well as conceptually. A sensitive depth gage is very important when adding or removing water in response to evaporation or rainfall. For a class-A evaporation pan, evaporation of only 1 mm of water equates to 1.15 L of water that needs to be added back to the pan to maintain a constant water level. Pans also eventually develop leaks. If the pan is buried to minimize the effect of sidewall heating, it would be difficult to diagnose a leak. Similarly, a floating pan is subject to waves splashing over the side of the pan during windy periods. Numerous modifications also have been made to minimize the effect of animals drinking from the pan water. If a wire mesh is placed over the pan, a correction may need to be made to account for the shading effect from the wire mesh. From a conceptual perspective, a pan deployed in an arid environment, and elevated relative to the land

surface around it, is subject to what commonly is termed the oasis effect; the pan represents a surface that is evaporating at the maximum potential rate. However, if the surface around the pan has a limited moisture source, and the rate of ET is substantially smaller, then the cooling effect associated with evaporation is reduced. The air mass that passes over the pan is both warmer and drier than what it would be if the surface surrounding the pan was losing water at the same rate as the pan. The warmer, drier air enhances evaporation from the pan, just as described for the complementary-relationship method.

Sources of error, in addition to simple measurement error associated with each sensor, often are the result of prolonged sensor deployment. Some sensors, such as humidity sensors and radiometers, drift and require regular recalibration. Bearings in anemometers wear out and require replacement. Thermistors deployed in water experience algal growth that delay the response time of the sensor. If the wetland water level changes and the sensor heights are not adjusted, then bias can occur in temperature and vapor-pressure differences. Loss of data due to power interruptions, sensor failure, or breaks in the sensor wires also occur. An error that often is overlooked is extrapolation of measured evaporation rates over the evaporating surface. As wetland stage changes, along with the corresponding change in shoreline location and wetland surface area, substantial error can occur by applying measured evaporation rates to an incorrect wetland surface area (see Sect. 3.2.1).

3.5.6 Cost Effectiveness

All scientists would like to quantify fluxes and processes as accurately as possible, but the benefit associated with greater accuracy has to be balanced with the cost of instrumentation and methodology. The eddy-covariance method may yield ET rates with the smallest uncertainty, but sensors are expensive, data processing is lengthy, and installation costs can be large. At the opposite extreme, temperature is one of the least expensive parameters to measure, making methods such as Thornthwaite particularly attractive if the scientist is willing to sacrifice accuracy in the interest of economy. To a large extent, the choice of method depends on the importance of accurate quantification of ET. If ET is a large component of a wetland water budget, then a substantial investment in time and money is warranted. If the wetland is dominated by surface-water flow and ET is a relatively small component, then perhaps a method based on temperature, or temperature and radiation, is sufficient. At a small lake in New Hampshire where all components of the water budget were characterized as accurately as possible, the Priestley-Taylor method was deemed the best compromise between accuracy and cost. It produced data nearly as good as the energy-budget method but did not require measurement of surface-water temperature or quantification of advected energy sources and sinks (Rosenberry et al. 2007). Another study of a reservoir in northern Florida came to a similar conclusion regarding use of the Priestley-Taylor method despite the water budget being dominated by surface-water inputs and losses (Dalton et al. 2004).

3.6 Surface Water Inflow and Outflow

Surface water flow is commonly the largest component of a water budget for wetlands in humid regions, making it an important component to measure accurately. Accurate measurements are possible if flow is confined to well-defined channels. Measurement uncertainty can be very large, however, in wetland settings with exceptionally low topographic gradients where flow commonly occurs through a network of poorly constrained channels. Furthermore, the distinction between surface-water flow and diffuse overland flow can be difficult to determine. In this section, we limit discussion to settings where a well-defined channel exists and briefly discuss low-gradient settings in the section on flow estimation using indirect methods.

Surface-water flow in terms of volume per time (m^3/s) is often referred to as discharge in the stream hydrology literature. A more precise definition is the volume rate of flow through a stream cross-section at a right angle to the flow direction. Since it is impractical to manually measure stream discharge with 15-min to daily temporal resolution over long periods of time, it usually is calculated from stage measured at a stream-gaging station installed in a channel, or stage measured in an artificial control structure (i.e., flume or weir) using a pre-established stage-discharge relation. Often referred to as a stage-discharge rating curve, this empirical function is determined with direct measurements of stream discharge over a range of stages or by calibration of theoretical formula. In the following section, we will briefly describe establishment of a stream-gaging station, measurement of stream discharge and stage, development of a rating curve, and the associated potential problems and sources of error.

3.6.1 Stream Gaging Station

The accuracy of stream discharge measurement is strongly dependent on the accuracy of the rating curve, which is dependent on the degree of flow control. Therefore, whenever possible, it is best to establish gaging stations in a stream section with good natural flow control (see below) or in a specifically designed control structure such as a weir or flume. Flow control can be defined as a feature some distance downstream of a gaging station that controls the stage-discharge relation upstream at least as far as the location of the gaging station.

Ideally, a gaging station should be located in a place where: (1) the channel is straight about 100 m upstream and downstream, (2) the total flow is confined to one channel at all stages, (3) the streambed is not subject to scour and fill and is free of aquatic vegetation growth, (4) banks are permanent, free of brush, and high enough to contain floods, (5) the downstream flow control is unchanging, (6) a pool is present upstream from the control, (7) the site is far enough upstream from the confluence with another stream that flow in the tributary does not affect flow at the gaging station, and (8) a suitable stream section for making manual discharge

measurements is available near the gaging station (Rantz 1982:5). It is usually impossible to meet all the above conditions, and judgement must be exercised to select an adequate location despite some shortcomings. For measuring outflow from a wetland, conditions 6 and 7 can easily be satisfied by using the wetland itself as a pool and measuring stage in the wetland near the outlet. However, controlling the areal extent and density of vegetation in the outlet channel is often a challenge in wetland settings.

Some of the conditions above can be improved by modifying a channel; for example, reinforcing the bank or removing flow obstacles, without significantly altering the habitat characteristics for aquatic life. The most accurate data are obtained by installing an artificial control structure, but installation may require an environmental impact assessment if the required channel modifications are extensive.

3.6.2 Characteristics of Flow Control

Some stream reaches are relatively straight for a long distance with constant slope, channel geometry, and bed roughness. This situation, where the control on the upstream stage-discharge relation is the channel itself, is called channel control. Other flow-control settings include a place where the stream flows across bedrock, a reach where the channel narrows, or the point beyond which the downstream stream reach steepens (i.e., the upstream end of a riffle). Some artificial structures that are not designed specifically to be a flow control, such as a bridge or a culvert under a road, may serve as a good flow control. Two attributes of a satisfactory flow control are stability and sensitivity (Rantz 1982:11). If a control is stable, then the stage-discharge rating curve does not require frequent adjustment. For example, a constriction provided by rock-ledge outcrop is not affected by flood events (stable), but upstream boulders and gravel may move during floods (unstable). Regarding sensitivity, a control section ideally should have a relatively narrow width at low discharge condition so that a small change in discharge is reflected by a significant change in stage.

If natural conditions do not provide adequate stability and sensitivity, artificial control should be considered. In natural streams having a wide range of discharge conditions, it is common to use broad-crested weirs (Fig. 3.11a) that conform to the general shape and height of the streambed (Rantz 1982:12). It is generally impractical to build the control high enough to avoid submergence at high discharges. Therefore, broad-crested weirs are effective for low to medium discharge only. In canals and drains where the range of discharge is limited, thin-plate weirs (Fig. 3.11b) or flumes (Fig. 3.11c) may be used to cover the complete range of discharge. Thin-plate weirs are suitable for channels in which the flow has relatively low sediment load and the banks are high enough to accommodate the increase in stage (backwater) upstream of the weir. Flumes are largely self-cleaning and can be used in channels with high sediment load, and do not cause significant backwater. However, flumes are generally more costly to build than weirs (Rantz 1982:13). Other types of control structures include weirs with moving gates installed in canals, commonly referred to as “head gates”.

Fig. 3.11 Examples of artificial flow control structures: (**a**) a broad-crested weir in a stream in northern Manitoba, Canada, (**b**) a thin-plate weir in a stream in Banff, Canada, (**c**) a Parshall flume. The streamflow direction is from the left to the right in all photographs (Photos a and b by Masaki Hayashi; photo c by Donald Rosenberry)

Ideally, artificial controls should have structural stability, their crest should be as high as practical to eliminate the effects of variable downstream conditions, and the stage should be sensitive to discharge. As a general rule, a weir is more advantageous than a flume because it is less expensive, can be designed to have greater sensitivity, and its rating curve can be extrapolated beyond the normal operation range without serious errors (Rantz 1982:18). Flumes are more advantageous in streams carrying heavy sediment load or when backwater created by a weir is undesirable. Specific design and characteristics of different types of weirs and flumes can be found in Rantz (1982:294–326).

3.6.3 Stage Measurement

The first step in measuring stream stage is to establish a permanent datum (reference elevation) and a staff gage (Fig. 3.2). The zero for a stream-stage datum should be below the elevation of zero flow on a natural control, and usually at the elevation of zero flow in an artificial control such as a weir. Changing the datum during a

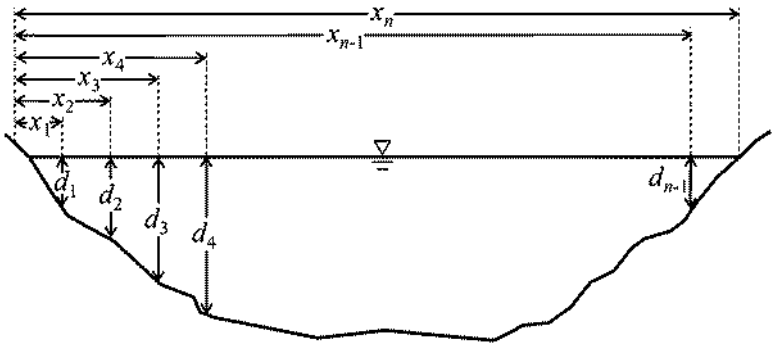

Fig. 3.12 Schematic diagram showing a channel cross section divided into rectangular subsections for the velocity-area method of discharge measurement (Adapted from Rantz 1982:81. Published with kind permission of the U.S. Geological Survey. Figure is public domain in the USA. All Rights Reserved)

monitoring period should be avoided, but if change is unavoidable, the relation between the new and old datum needs to be clearly established and recorded.

Details regarding installation, operation, and maintenance of stage-measuring devices are described in the Sect. 3.3.1 on wetland stage. For stage measurements in streams, a water-level sensor can be protected from flowing debris by installing it in a stilling well, which also dampens waves generated by wind or turbulence and provides a more stable reading. A stilling well is a vertical pipe or culvert that is hydraulically connected to the stream water level. It can be installed in a stream bank and connected to the stream by a subsurface horizontal pipe, or installed directly in a channel. It must be secured to a stable anchor so that its elevation does not change over time. The bottom of a stilling well should be at least 0.3 m below the minimum stage and its top should be above the peak flood level. Details regarding construction of stilling wells and their shelters is described by Rantz (1982:41–52) and Sauer and Turnipseed (2010:6–11).

3.6.4 Discharge Measurement

3.6.4.1 Velocity-Area Method

Among several methods for discharge measurement, the most commonly used is the velocity-area method, which involves direct measurement of flow velocity using a current meter at successive locations along a channel cross section (Fig. 3.12) and summation of measured values to calculate the total discharge. The ideal stream cross section is located within a straight reach having streamlines parallel to each other; the streambed is relatively uniform and free of numerous boulders and extensive aquatic vegetation; flow is relatively uniform and free of

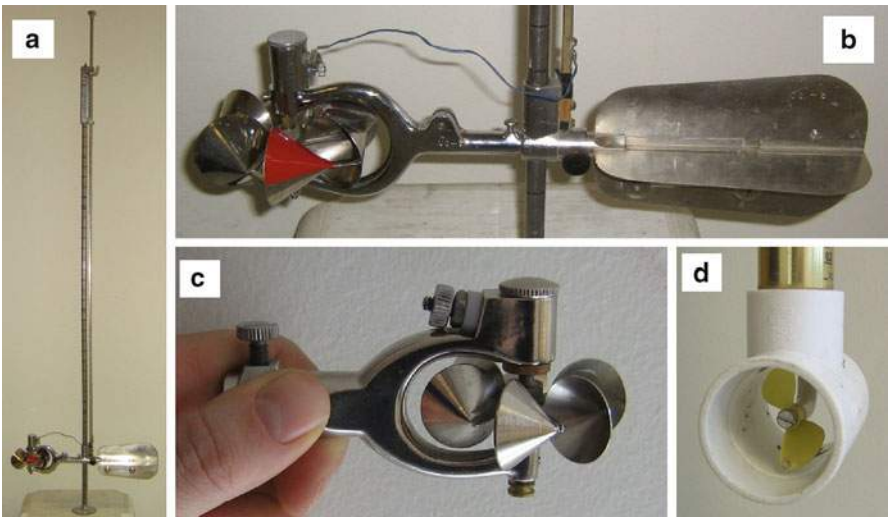

Fig. 3.13 Example of flow-velocity measuring devices that are commonly used in North America: (**a**) top-setting wading rod with a meter attached, (**b**) vertical-axis current meter (Price AA), (**c**) vertical-axis current meter (Price Pigmy), (**d**) horizontal-axis current meter (Global Water FP101)

eddies, slack water, and excessive turbulence; velocity and depth are within the range for which measuring devices give accurate results; and the observer can safely carry out measurements (Rantz 1982:139; Dingman 2002:609). If necessary, the condition can be improved by removing obstructions in, above, and below the channel section without affecting discharge.

Measurements can be made by wading, from a boat, or from bridges. After the selection of a suitable section, the first step is to extend an incremented line across the channel above the water surface and measure the total channel width. A tape measure, a rope marked at constant intervals, a marked tag line for boat measurements, or marking constant intervals on a bridge can be used for this purpose. Except for bridge-based measurements, the line is placed at right angles to the direction of flow. The next step is to determine the measurement interval, or the width of individual rectangular subsections in Fig. 3.12. Rantz (1982:140) recommend 25–30 subsections with no single subsection contributing more than 5 % of total discharge, meaning that smaller widths need to be assigned for subsections in deeper and faster portion of the channel.

Depth of water and average velocity is determined at the middle of each subsection. When flow velocity and depth allow measurements to be made while wading the stream, a flowmeter commonly is mounted to a top-setting wading rod specifically designed to indicate water depth and easily placed at the proper depth for each velocity measurement (Fig. 3.13a). The observer should stand in a position that least affects the velocity being measured; for example, by standing downstream of the tag line and facing either bank, or in the case of a sufficiently

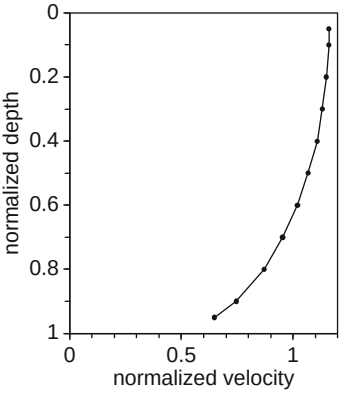

Fig. 3.14 Typical velocity profile plotting dimensionless velocity normalized to the profile-mean value against dimensionless depth normalized to the total depth of profile. At a normalized depth of 0.6, normalized velocity is approximately equal to 1 (Adapted from Rantz 1982:133. Published with kind permission of the U.S. Geological Survey. Figure is public domain in the USA. All Rights Reserved)

narrow channel, standing on an elevated plank crossing the channel (Rantz 1982:146). A cable with a heavy depth-sounding weight at the bottom is used to measure depth and suspend the current meter at the proper depth when measurements are made from a boat or bridge or cableway (Rantz 1982:101–102).

Flow velocity in a channel also varies vertically. Within a rectangular subsection, the velocity is very small at the streambed and increases upwards. Figure 3.14 shows the standard velocity profile used by the U.S. Geological Survey (Rantz 1982:133) based on intensive investigation of open channel hydraulics. In this profile, the velocity measured at a normalized depth of 0.6 (six tenths of the distance from the water surface to the bed) is almost identical to the profile-mean velocity, and also the arithmetic mean of two velocity values measured at normalized depths of 0.2 and 0.8 is almost identical to the profile-mean velocity. Based on these observations, it is common practice to represent the mean velocity of a subsection by the arithmetic mean of two measurements at 0.2 and 0.8 depths (2-point method), or a single velocity measurement using a current meter placed at 0.6 depth (six-tenth method). The 2-point method generally gives more reliable results than the six-tenth method, but in shallow streams making a measurement at 0.2 depth may not be possible, depending on the size and specification of the current meter, in which case the six-tenth method is preferred (Rantz 1982:134–135). An alternative approach is to use a current meter that integrates velocity measurements and calculates a profile-mean velocity as the meter is continuously moved up and down the entire profile at a uniform rate for a sufficiently long period (integration method).

Several types of current meters are commonly used to make streamflow measurements. Mechanical current meters have long been the sensor of choice, but electromagnetic current meters and acoustic Doppler velocimeters (ADV; Winter 1981)

are rapidly gaining popularity; ADV's may already be the most widely used type of current meter in North America. These sensors are preferable because they output direct velocity values, and in the case of the ADV, they have the ability to measure flow velocities below the limit of mechanical meters (see Turnipseed and Sauer (2010:44–58) for an overview of mechanical and non-mechanical current meters). Mechanical current meters are classified into vertical-axis (Fig. 3.13b, c) and horizontal-axis meters (Fig. 3.13d) depending on the alignment of moving cups or a propeller with respect to the flow direction. In North America, the standard equipment used by U.S. Geological Survey and the Water Survey of Canada are Price[1] AA and Price pigmy vertical-axis current meters. These meters require an operator to place it at a fixed depth, count the number of rotations of the cups, and convert the count to velocity using a calibration table or rating function. This operation, formerly conducted manually, is automated in most meters that are currently available on the market. Horizontal-axis meters also require a count of the rotations of a propeller and conversion to velocity. They are suitable for continuous profiling of flow velocity, and are often equipped with electronics designed for the integration method of velocity measurement (see above).

Regardless of the meter type, it is important to ensure that the meter is correctly functioning and well calibrated. The meter should be visually checked before and after its use, and cleaned if necessary. It should be periodically checked during measurements, when it is out of the water, to ensure that it spins freely. It is not uncommon for debris or aquatic weeds to become trapped between moving parts causing underestimation of velocity. Rantz (1982:93–94) describes the maintenance and care of mechanical meters in detail. A meter should be periodically calibrated to ensure that its rating function (i.e., relation between the speed of rotation and flow velocity) has not changed beyond a specified tolerance. The rating function is normally established by towing the meter at a constant velocity through a long water-filled trough (Turnipseed and Sauer 2010:53). While it is desirable to have current meters calibrated in a dedicated facility operated by experienced staff, it is possible to carry out similar calibrations using a more accessible facility. For example, if access to a swimming pool can be obtained for a short period of time then a flow meter can be attached to a cart and towed at a reasonably constant velocity under completely calm pool conditions (Fig. 3.15).

Once the depth (d_i, m) and mean velocity (u_i, m s^{-1}) of each individual subsection are measured accurately, total discharge (Q, m^3 s^{-1}) is calculated by the summation of discharge in all subsections:

$$Q = \sum_{i=1}^{n} u_i A_i = \sum_{i=1}^{n} u_i d_i \frac{x_{i+1} - x_{i-1}}{2} \tag{3.22}$$

[1] Any use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

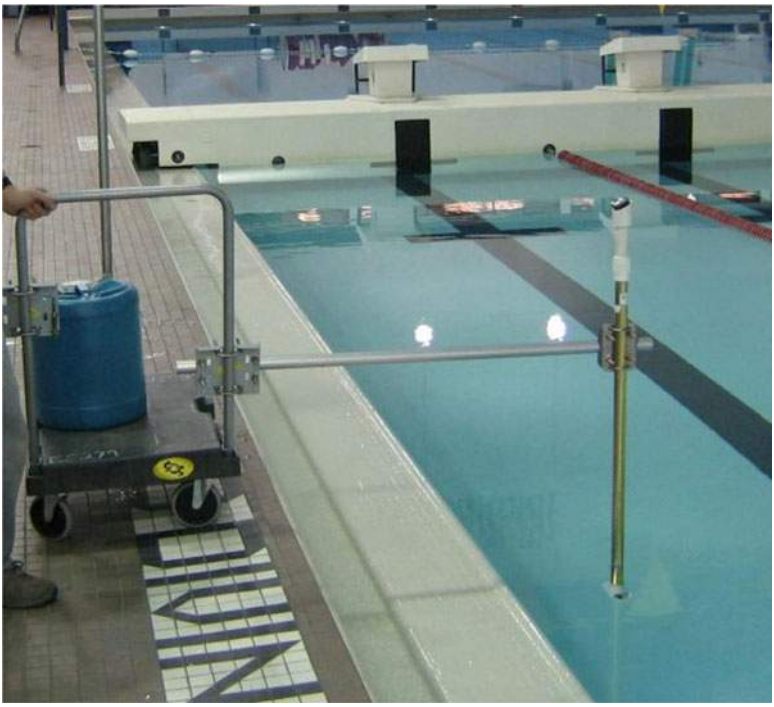

Fig. 3.15 Calibration of a horizontal-axis current meter in a swimming pool. The meter is attached to a cart that is pushed at a constant velocity (Photo by Masaki Hayashi)

where A_i (m^2) is the cross sectional area of each rectangular subsection, and x_i (m) is the distance along the line of measurements. This method of computation is called the midsection method, and is known to provide more accurate values of Q than many other methods (Dingman 2002:611).

In Eq. 3.22, total discharge is calculated using a number of data points measured over the length of time required to quantify flow across the entire cross section. In stream sections undergoing rapid flow transience (for example, during a storm event or water extraction/release upstream), discharge may change substantially during the measurement period. If these conditions are suspected, stream stage should be monitored during the measurement period to assess the potential magnitude of error caused by the transience.

3.6.4.2 Tracer-Dilution Methods

Tracer-dilution methods are useful alternatives to the velocity-area method where it is difficult or impossible to use a current meter due to high velocities, turbulence, debris, rough channels, shallow water, or other physical reasons; or where the cross-sectional area cannot be accurately measured (Kilpatrick and Cobb 1985). However,

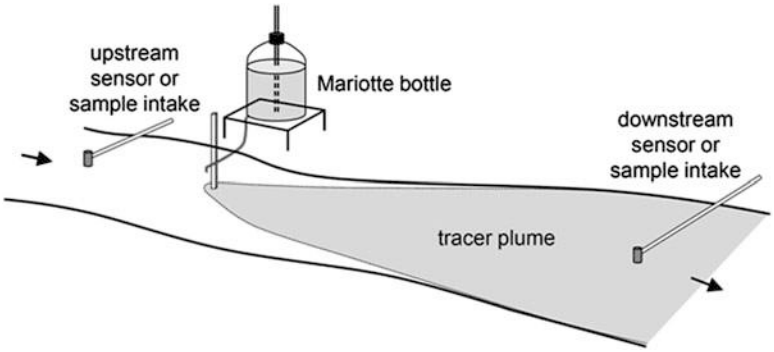

Fig. 3.16 Typical set up for the constant-rate injection method of tracer-dilution gaging. The tracer solution is released in the center of a stream section using a Marriote bottle system. The background value of tracer concentration is monitored at an upstream point, and the fully mixed value of tracer concentration is monitored at a downstream point. The set up is the same for the sudden injection method with the exception that the Mariotte bottle is not used because the tracer is instantaneously poured into the stream (From Dingman (2002) modified with kind permission of © S. Lawrence Dingman 2002. All Rights Reserved. The figure was created based on an illustration originally appearing in Gregory and Walling (1973:134))

tracer-dilution methods are generally more difficult to use than current-meter methods, and under most conditions the results are less reliable. Therefore, these methods should not be used when conditions are favorable for a current-meter measurement (Rantz 1982:212).

In a typical tracer-dilution measurement, a tracer solution is injected into the stream and the stream discharge is estimated from measurements of the tracer-solution concentration, tracer-solution injection rate, and tracer concentrations at a sampling cross section downstream from the injection site (Fig. 3.16). Two methods are commonly used; the constant-rate injection and the sudden injection. For both methods, it is assumed that the tracer is completely mixed at the downstream measurement point. In the constant-rate injection (CRI) method, a tracer of known concentration C_1 (kg/m) is injected at a constant rate q (m^3/s) in the center of a stream channel using a constant-flow device such as Mariotte bottle (e.g., Moore 2004). If the injection is continued for a sufficiently long period, monitoring of concentration at a downstream sampling cross section will show a plateau of constant concentration C_2 (kg/m). Discharge (Q) is estimated from the dilution ratio by:

$$Q = q(C_1 - C_2)/(C_2 - C_b) \tag{3.23}$$

where C_b (kg/m) is the background tracer concentration in the stream.

In the sudden-injection or slug-injection (SI) method, a slug of tracer solution is instantaneously applied to a stream, and concentration is monitored at a downstream sampling cross section to generate a concentration-time curve (Fig. 3.17). The total mass of the injected tracer must equal the total mass of tracer going

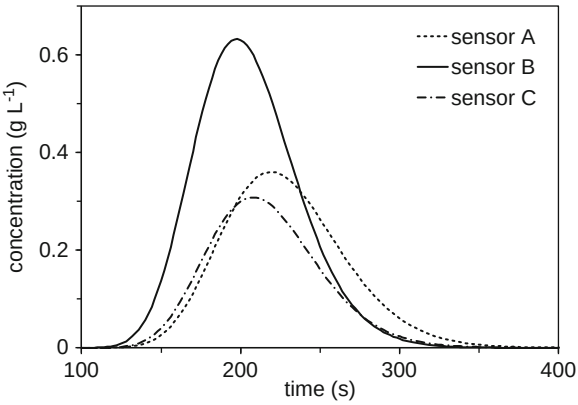

Fig. 3.17 Hypothetical concentration-time curves at a downstream sampling section during a tracer-dilution gaging by the sudden-injection method. *Three curves* represent concentrations recorded by three sensors placed at different locations in the same cross section. The areas under the three curves should be identical if the tracer is completely mixed. In this example, sensor B is recording a much larger mass of tracer passing the section, indicating incomplete mixing

through the sampling section, which is given by the integration of concentration under the curve, assuming that discharge does not vary during the monitoring period. Therefore, discharge is estimated from the mass balance by:

$$Q = C_1 V_1 \Big/ \int_0^\infty [C(t) - C_b]dt \tag{3.24}$$

where V_1 (m^3) is the volume of tracer solution introduced to the stream and $C(t)$ (kg m^{-3}) is the time-varying concentration at the sampling cross section. In practice, the integral in Eq. 3.24 is approximated by:

$$\sum_{i=1}^{n} (C_i - C_b)\Delta t \tag{3.25}$$

where C_i are the concentrations measured at a discrete time interval Δt (s) until C_i becomes indistinguishable from C_b at the nth sample.

Both tracer-injection methods assume that complete vertical and lateral mixing of the tracer with stream water has occurred at the sampling cross section. Vertical mixing usually occurs very rapidly, but a substantial distance is required for lateral mixing. Therefore, it is important to establish the sampling location a sufficient distance downstream of the injection point to ensure complete mixing. It also is important to sample at the downstream location long enough to establish a concentration plateau for the CRI method or to capture the entire concentration-time curve for the SI method. Depending on the site condition and access, it may be difficult to achieve complete mixing, as illustrated in Fig. 3.17, in which case a large degree of

Fig. 3.18 (**a**) Example of a portable V-notch weir (Photo by Masaki Hayashi), (**b**) portable 2.54-cm Baski flume (Photo printed with kind permission of © Kirk Miller, U.S. Geological Survey 2012. All Rights Reserved)

error can be introduced in estimated Q. It is strongly recommended that the methods be tested in the field according to procedures described by Rantz (1982:216–220). Other sources of error include the loss of tracer solution between the injection and sampling points to groundwater or hyporheic exchange, photochemical and other reactions, sorption to streambed materials, and interference of concentration monitoring devices by turbidity and other sensor-calibration issues. Errors due to reaction and sorption, and to some extent sensor issues, can be minimized by the choice of tracer. The ideal tracer has very low natural concentration in the stream, is chemically and biologically conservative (no reaction, absorption, release or uptake), is readily detectable at low concentration, and is harmless to the observer and aquatic life. Sodium chloride and fluorescent dye are commonly used as tracers.

3.6.4.3 Other Methods of Discharge Measurement

For narrow streams with small discharge, it is often possible to divert the entire flow to a container having a known volume, and measure the time it takes to fill the container to calculate discharge (the bucket-stopwatch method). Examples of sites presenting the opportunity for this method are a V-notch weir or a cross-section of natural channel where a temporary earthen dam can be built over a small-diameter pipe (Rantz 1982:263). If a temporary structure is built for volumetric measurement, the stage behind the structure (e.g., a dam) should be allowed to stabilize before the measurement. The measurements should be repeated several times to obtain consistent results. Where a portable V-notch weir or Parshall flume can be installed in the stream (Fig. 3.18), discharge is estimated from the stage measurements in these devices using a laboratory-calibrated formula (see next section).

In situations where no current meter or tracer-dilution equipment are available, or the condition does not permit the use of other methods, estimates of discharge can be obtained using surface floats. Any distinguishable article can be used as a

float, such as wooden disks, bottles partially filled with water, or even oranges. Two cross sections are selected along a reach of straight channel, so that the time the float takes to pass from one cross section to the other can be measured accurately. Distance from the upstream to the downstream cross sections needs to be measured and floats should be applied far enough upstream of the upper cross section that they are travelling at the same speed as the surface current when they pass the upper cross section. For best estimates of discharge, a number of floats are distributed uniformly across the stream width, and the position of each with respect to distance from the bank is noted. The stream-channel width can be segmented just as with the velocity-area method described above, depth for each channel section can be measured, and discharge calculated as the sum of each velocity-area product. This method will over-estimate discharge because the surface velocity is faster than the depth-integrated velocity. Therefore, a coefficient of 0.85 is commonly used to convert the surface velocity to mean velocity (Rantz 1982:262).

3.6.5 Stage-Discharge Rating Curve

Rating curves for discharge gaging stations are normally determined empirically with periodic measurements of stage and discharge made over the full range of stage at a particular station. Thereafter, only periodic measurements (commonly a minimum of 10 per year) are needed if it has been demonstrated that the rating curve does not vary with time (Rantz 1982:285). However, shifts in rating curves are common at gaging stations with natural control due to changing channel conditions, including scour and fill, vegetation growth, boulder movements, and ice formation. Rating curves are created by plotting the stage-discharge data and fitting a suitable mathematical function to the entire data set, or fitting a set of functions to separate segments of the data. Power functions are commonly used to fit the data:

$$Q = a(h - h_0)^m \tag{3.26}$$

where Q ($m^3\ s^{-1}$) is discharge, a ($m^{3\text{-}m}\ s^{-1}$) is a scaling constant, h (m) is stream stage (i.e., water level), h_0 (m) is the stage at zero flow, and m is a dimensionless constant. This function generates a straight line when Q and $h - h_0$ are plotted on logarithmic axes. For gaging stations with no control or natural control, a and m are determined by minimizing the difference between the measured discharge and the predicted discharge using Eq. 3.26.

For gaging stations with an artificial control structure, a formula developed in the laboratory is provided with the device. For example, a rectangular thin-plate weir has a rating curve in the form:

$$Q = Cb(h - h_0)^{3/2} \tag{3.27}$$

where b (m) is the width of the weir through which water is flowing, and C ($m^{1/2}/s$) is a coefficient dependent on the geometry of the weir (Rantz 1982:296–297). A V-notch weir has a rating curve in the form of:

$$Q = C \tan(\theta/2)(h - h_0)^{5/2} \tag{3.28}$$

where θ is the angle of notch (θ = 90° for the standard V-notch weir), and C ($m^{1/2}/s$) is a weir coefficient, which has a value of 1.38 in the ideal condition where the weir plate is perfectly made and vertical and water in the pool upstream of the weir has negligible velocity approaching the weir (Rantz 1982:304). The formulas for several types of broad-crested weirs and flumes, including the Parshall flume, are described by Rantz (1982:306–326).

Stage-discharge rating formulas for artificial controls were developed based on laboratory and modeling studies conducted under ideal conditions. To achieve maximum accuracy in the field, data from these instruments should be compared with separate measurements of discharge conducted at the weir or flume installation (Rantz 1982:295) and correction coefficients applied as necessary. The velocity-area method (Fig. 3.11a) or volumetric method (Fig. 3.11b) is most often used for this purpose.

The stage-discharge relation is sensitive to changing conditions of the flow control and the channel reach in the vicinity of the gaging station. Therefore, changes in the channel (e.g., boulder and gravel movement, streambed scouring or filling, bank erosion, vegetation growth) may cause a shift in the rating curve, which can only be detected through periodic stage-discharge measurements. When an observer measures discharge at an existing gaging station, it is a good practice to calculate discharge before leaving the site and plot the new measurement on the rating chart. If the new point deviates noticeably from the established rating curve, a second discharge measurement is carried out to confirm a shift (Rantz 1982:346). If the second measurement confirms a shift, a note should be made to pay particular attention to the site conditions during the next visit. If several consecutive measurements show a consistent shift, the rating curve will need to be adjusted to account for the new channel conditions.

3.6.6 Flow Estimation by Indirect Methods

In low-gradient wetland settings, surface-water flow can be very slow and occur over a broad area without a well-defined channel, often covered with extensive emergent vegetation, making it impossible to measure surface-water flow using any of the methods above. In this case, surface-water flow may have to be treated as the residual of a water-budget equation. Surface-water flow can still be estimated, although often with a large degree of uncertainty, using one of several hydraulic equations with estimated parameters. If the depth of water is sufficiently large

compared to the height of submergent vegetation, and if the flow is turbulent, a standard equation for open-channel flow, such as Manning's equation, can be used to estimate average flow velocity (v, m/s):

$$v = y^{2/3}\ S^{1/2}/n \tag{3.29}$$

where y (m) is depth of water, S is slope of the water surface (unitless), and n ($\mathrm{s\ m^{-1/3}}$) is a roughness coefficient whose values can be found in the literature (e.g., Kadlec and Wallace 2009:40).

The use of open channel equations such as Eq. 3.29 is inappropriate for those commonly encountered situations in wetlands where flow is laminar, water depth is shallow, stream slope is very small and nearly impossible to measure, and flow resistance exerted by vegetation is strongly related to water depth. In these settings, it is more appropriate to use an empirical equation having the form

$$v = \alpha y^{\beta-1} S^{\gamma} \tag{3.30}$$

where α ($\mathrm{m^{2-\beta}\ s}$) is an empirical coefficient representing hydraulic resistance, and β and γ are additional empirical coefficients (Kadlec and Wallace 2009:35). A disadvantage of Eq. 3.30 is that all empirical coefficients are site-specific (and possibly season-specific) and have to be determined locally. Kadlec and Wallace (2009:39) listed ranges of α from 70 to 2,300 $\mathrm{m^{2-\beta}/s}$, β from 1.4 to 3.0, and γ from 0.7 to 1 from a survey of the existing literature and recommended the following values be used when no data are available: $\alpha = 120$ and 580 m/s for densely and sparsely vegetated wetlands, respectively, $\beta = 3$, and $\gamma = 1$.

Indirect estimates of discharge using equations such as 3.29 and 3.30 have a very large degree of uncertainty unless the equation is calibrated using site-specific field data. Therefore, they should be used as a last resort to obtain an order-of-magnitude estimate of flow.

3.6.7 Errors and Challenges

Using well-calibrated instruments for velocity and depth measurement in a carefully chosen location with appropriate flow control, it is possible to measure stream discharge with accuracy of 3–6 % using the velocity-area method (Turnipseed and Sauer 2010:80). Similar accuracy can be achieved for the volumetric measurement using a well-calibrated vessel. However, flow often has to be measured in non-ideal locations caused by unsatisfactory flow control, local flow lines not crossing the measurement section at a right angle, obstruction of flow by boulders and vegetation, vertical and lateral leakage of surface water to fluvial sediments, loss of water to side channels, or several other causes. It is advisable to conduct duplicate or triplicate velocity-area discharge measurements on different sections located within the same stream reach to evaluate the uncertainty in flow measurement. Compared

to the velocity-area method and volumetric measurements, tracer-dilution methods tend to have a larger degree of uncertainty due to lack of complete mixing, loss of tracer to groundwater, sensor calibration issues, and insufficient time for complete capture of tracer. Other methods likely have greater degrees of uncertainty.

Errors associated with the stage-discharge rating curve are expected to be reasonably small for well-maintained and calibrated artificial control structures such as weirs and flumes. However, rating curves for naturally-controlled gaging stations can be greatly affected by short- and long-term changes in channel conditions. It is not uncommon to have root-mean-squared (RMS) errors of rating curves exceeding 20 %, particularly for those locations that are susceptible to changes in density and extent of vegetation or shifts in boulders or gravel bars. Therefore, as stated earlier, it is important to visit stations frequently to conduct maintenance, make manual discharge measurements, detect any shifts in the rating curve, and make appropriate adjustments. Errors in the stage measurement (see Wetland stage section) also affect discharge data, and should be kept at a minimum.

The formation of channel ice in cold regions subject to freezing temperatures can have a major effect on the stage-discharge relation by causing backwater due to increased flow resistance. The backwater effect is dependent on the quantity and nature of the ice, as well as the amount of discharge, which necessitates frequent discharge measurements, particularly during freeze-up and thaw when the flow is highly variable. Rantz (1982) describes procedures for making discharge measurements in ice-covered streams and adjustments to stage-discharge rating curves to account for the backwater effect (Rantz 1982:360–376). If it is not feasible to maintain ice-free condition or to measure discharge frequently, the record may be regarded as "seasonal", with no data available during the ice-covered period.

3.6.8 Summary

Surface-water flow can be a major component of a wetland water budget, and the uncertainty of surface-flow measurements can dominate the cumulative uncertainty of a water-budget calculation. Measurement accuracy of 10 % or better can be achieved at well-maintained gaging stations with appropriate flow control provided the stage-discharge rating curve is frequently checked and adjusted. However, stream channels in many wetland settings are ill-defined and conditions can be highly variable over time. If it is necessary to measure discharge in an undesirable location, the observer should strive to obtain estimates of measurement error and uncertainty, which can be reflected in the water-budget calculation.

3.7 Diffuse Overland Flow

During snowmelt or storm events, surface runoff generated within the drainage area may directly reach the wetland by flowing over the land surface without entering stream channels. This process is called diffuse overland flow. Since it is practically

impossible to measure diffuse overland flow over a large area, this component is frequently neglected or treated as a residual in the water-budget equation, especially when there is evidence indicating that the magnitude of diffuse overland flow is much smaller than stream inflow or groundwater inflow. However, diffuse overland flow is a major water-budget component, at least temporarily, in some wetlands without channelized stream inputs. Examples include many prairie wetlands in the Northern Prairies region of North America (Winter 1989) and ephemeral forest pools in the New England region of the United States (Brooks 2009).

Diffuse overland flow generally moves toward a wetland and is nearly always considered as an input term in wetland water budgets, but this is not always the case. In low-gradient settings where surface-water outflow is very slow and occurs over a broad area with an ill-defined channel, loss of wetland water could be considered either as slow surface-water flow or diffuse overland flow that is moving away from the wetland. Although some have separated diffuse overland flow into separate input and loss terms (LaBaugh 1986), here we will consider any slow-moving flow occurring over a broad area that is leaving a wetland basin to be surface-water outflow, as quantified with Eqs. 3.29 and 3.30.

Since the amount of diffuse overland flow input entering a wetland is proportional to the length of wetland perimeter, the effect of diffuse overland flow is particularly pronounced in relatively small (e.g., $<10^4$ m^2) wetlands that have large perimeter-to-area ratios. A shallow water table in areas adjacent to a wetland can rise to the surface with a relatively small amount of infiltration during storm events (Gerla 1992), which precludes further infiltration and generates runoff (Dunne and Leopold 1978:268). In cold regions that have seasonally or permanently frozen soil, reduced infiltrability of frozen soil causes a large amount of snowmelt runoff in the surrounding uplands, which can be the dominant mode of water input to wetlands (e.g., Winter and Rosenberry 1995; Hayashi et al. 1998). Using the same logic, low-permeability soils also will retard infiltration, resulting in a greater percentage of precipitation reaching the wetland as diffuse overland flow. Although difficult to quantify over a large area, there are a number of methods for measuring flow volume on a local scale or for individual storm events, or for obtaining order-of-magnitude estimates of flow volume using simple models.

3.7.1 Measurement of Diffuse Overland Flow

Flow traps are commonly used to measure diffuse overland flow over a small area. Figure 3.19 shows a very simple flow trap consisting of an isolated area and a pit to collect water. In this example, the pit is emptied after each storm event to determine the overland flow volume for individual events. A pit also can be equipped with a V-notch weir and water-level recorder for continuous monitoring of overland flow. If the area contributing diffuse overland flow to a wetland is delineated with reasonable accuracy, then the data obtained using small flow traps may be extrapolated to a larger area to estimate the total water input to the wetland.

Fig. 3.19 A simple overland flow trap consisting of the contributing area delineated by concrete walls and a pit to collect water (Photo by Masaki Hayashi)

If diffuse overland flow occurs in an area of relatively uniform vegetation and slope, and depth of surface water is measured with reasonable accuracy, approximate flow rates can be estimated from the empirical Eq. 3.30 described in the section on streamflow measurements. This method has a large degree of uncertainty due to uncertainties and errors associated with empirical coefficients, uncertainty in delineation of the area contributing to overland flow, and uncertainty in estimating the water depth of the overland flow.

The methods described above can be applied to areas with uniform characteristics; results from each of those areas are then summed to generate O_f for the entire wetland. Methods described below all provide a single, integrated value for the entire wetland.

If a wetland does not have inflow and outflow streams, and the groundwater flow rate is much slower than the overland flow rate, the amount of diffuse overland flow during a short-lasting storm event can be estimated from the water-balance equation. Omitting stream and groundwater flow terms and assuming negligible evaporation during the event, Eq. 3.1 can be written as

$$O_f = \Delta V / \Delta t - P \tag{3.31}$$

Integrating (3.31) for the entire event, the total overland flow volume (O_{ftot}) is given by

$$O_{ftot} = V_{fin} - V_{ini} - P_{tot} \tag{3.32}$$

where V_{ini} and V_{fin} are the initial and final volumes of water contained in the wetland, respectively, and P_{tot} is the total volume of precipitation applied at

the wetland surface. Although this method requires specific conditions and is not applicable to all water-budget calculations or all wetland settings, data obtained using Eq. 3.32 may be used to calibrate or validate models used for estimation of diffuse overland flow (see below).

3.7.2 Estimation Using Simple Rainfall-Runoff Models

If the measurement of diffuse overland flow is impossible, it can be estimated from other variables using a hydrological model. One of the simplest models is the "rational method", which estimates the volume of overland flow (O_{fc}, m^3) generated from a contributing area (A_c, m^2) as a fixed ratio of precipitation (Mitsch and Gosselink 2007:128):

$$O_{fc} = R_c p A_c \tag{3.33}$$

where R_c is a dimensionless "rational coefficient" taking values between 0 and 1, and p (m) is the amount of precipitation. Equations similar to Eq. 3.33 are commonly used by engineering hydrologists for estimating storm runoff generation in urban areas. However, their applicability to wetlands is limited because R_c is dependent on many factors including soil, vegetation, and the depth to the water table. It is usually difficult to represent the variable conditions in a wetland catchment, both in time and space, with a single parameter.

More sophisticated models consider soil type, vegetation, land use, and numerous other factors, and treat soil moisture (and water-table) conditions as time-dependent variables. A relatively simple example of such models is the "curve number" (CN) method developed by the U.S. Soil Conservation Service in the 1950s and 1960s to estimate storm runoff from agricultural lands. Since then, the CN method has become a standard tool for hydrologists and has been used in numerous computer-based hydrological models such as the Soil and Water Assessment Tool (SWAT) (http://swatmodel.tamu.edu/) and Hydrologic Engineering Center (HEC) (http://www.hec.usace.army.mil/) models. This method computes O_{fc}/A_c in Eq. 3.33 as a non-linear function of precipitation. The non-linear dependence of runoff on precipitation is represented by a CN coefficient, which is determined by soil texture and drainage condition, land use, and the amount of rain during a period prior to the storm event (i.e., antecedent moisture). Step-by-step instructions of the CN method and examples of applications are found in introductory hydrology textbooks such as Dunne and Leopold (1978:291–298), and can be easily implemented in a computer algorithm.

Regardless of model sophistication, estimates made by these methods commonly have a large degree of uncertainty resulting from violations of model assumptions and uncertainty in model parameters and input variables. Therefore, estimated overland flow should be verified with measured data whenever possible using the methods described above.

3.8 Groundwater Inflow and Outflow

Determining exchanges between wetland water and groundwater can be a surprisingly complex task for wetland hydrologists. Low-permeability, organic-rich soils are often situated beneath and adjacent to wetlands; they reduce rates of exchange and can greatly increase residence time of water in pore spaces, enhancing geochemical processes. Flow across the sediment-water interface is variable both in space and time on multiple scales. Directions of flow between groundwater and surface water can reverse seasonally or in response to individual precipitation events or evapotranspiration (e.g., Doss 1993; Rosenberry and Winter 1997).

A variety of tools and methods are available to quantify exchange between groundwater and surface water, the selection of which should include strong consideration of the appropriateness of the scale of the measurement method relative to the scale of the goals of the study. On the larger end of the scale spectrum, suitable methods include watershed-scale rainfall-runoff modeling, groundwater-flow modeling, quantifying changes in streamflow along stream-reach segments (commonly called a seepage run), and making use of aerial imagery to locate areas of focused groundwater discharge. On a local scale, appropriate for smaller wetlands or specific shoreline segments or riparian reaches, methods include measurement of hydraulic properties using piezometers and water-level monitoring wells, use of seepage meters to quantify flow across an isolated portion of submerged bed sediment, and measurements of temperature to determine quantitatively or qualitatively distribution and rates of groundwater discharge to specific portions of wetland beds. Several of the most-commonly utilized methods are described below.

3.8.1 Darcy Flux Method

Use of the Darcy equation to determine flow through porous media is one of the core concepts of hydrogeology and is commonly employed to estimate exchanges between surface water and groundwater in wetland settings. The Darcy equation (e.g., Freeze and Cherry 1979) can be expressed as

$$Q = -KA\frac{h_1 - h_2}{l} \tag{3.34}$$

where Q is the volume of water that flows across the bed of the wetland to enter or leave the wetland, K (hydraulic conductivity, m/s) is a proportionality constant that represents the ease with which water can flow through porous media, A is the area of the sediment-water interface through which water flows to enter or leave a wetland, $h_1 - h_2$, or Δh, is the difference between hydraulic head measured at a nearby monitoring well and the wetland surface, and l is the distance from the monitoring

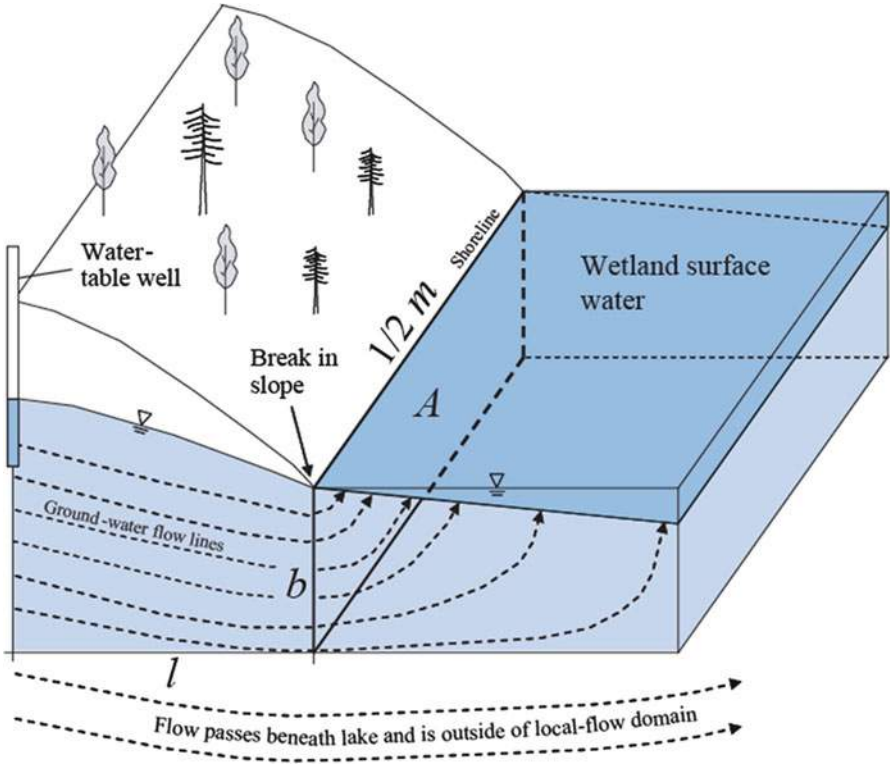

Fig. 3.20 Components required to determine exchange between groundwater and surface water using the Darcy method (Modified from Rosenberry et al. (2008). Published with kind permission of the U.S. Geological Survey. Figure is public domain in the USA. All Rights Reserved)

well to the shoreline of the wetland (Fig. 3.20). The minus sign is included to indicate that the flow (Q) occurs in response to a decrease in hydraulic head along a groundwater flowpath. The ratio $\Delta h/l$ is also called the hydraulic gradient and is commonly indicated with lower-case i, leading to the shorthand version of the Darcy equation:

$$Q = KiA \tag{3.35}$$

Note the absence of the negative sign that appears in Eq. 3.34. In Eq. 3.35, i is formulated so that the difference between heads is positive. The hydraulic gradient is usually the easiest term to quantify, requiring only measurements of the water level in a nearby monitoring well, wetland stage, and the horizontal distance between the well and the shoreline. The elevation of the top of the monitoring well relative to the water surface of the wetland also is needed to relate the water level in the monitoring well to wetland stage.

A as described above is the wetted portion of the wetland bed across which water flows to enter or leave the wetland. By definition, flow across *A* is perpendicular to the plane of *A*, the orientation of which ranges from nearly vertical to horizontal at the wetland bed. Therefore, *A* in Fig. 3.20 is instead oriented vertically and located at the shoreline of the wetland, which simplifies determination of *i* by allowing it to be made on a horizontal axis. Flow across this vertical plane is assumed to be equal to flow across the sediment-water interface of the wetland.

A is the product of *m*, the shoreline reach associated with the monitoring well, and *b*, the thickness of the portion of the aquifer that exchanges with the wetland (Fig. 3.20). Both *m* and *b* are conceptually simple but often difficult to determine. The distance, *m*, of a shoreline segment associated with the gradient between a monitoring well and the wetland is based on how far one can reasonably extrapolate the hydraulic gradient along the wetland shoreline. Because the monitoring well for each shoreline segment is usually located at the center of the segment, *m*, and therefore *A*, as shown in Fig. 3.20 are both half of what they should be because the figure does not show the half of the shoreline segment that would extend out of the page. If the shoreline reach is straight and hydrogeological conditions are expected to be uniform, the shoreline segment corresponding to a monitoring well could be quite long and the extent relatively easy to determine. If the shoreline is curvilinear (commonly the case in wetland settings) and hydraulic gradients are expected to vary substantially along the shoreline reach, then additional monitoring wells should be installed and shoreline segments should be correspondingly shorter. The other component needed to determine *A*, and one that often is the most difficult to estimate, is *b*, the thickness of the vertical plane through which water has to flow to enter (or leave) the wetland. Water passing through any portion of the aquifer deeper than *b* will not exchange with the wetland but will instead pass beneath the wetland. This is shown by the two flowlines that extend beyond the wetland in Fig. 3.20. Unless the subsurface geology is known to constrain exchange with the wetland, or unless the wetland depth extends to the base of the aquifer, *b* has to be estimated. One common approach is to arrive at a reasonable estimate for *b* through the use of a simplified groundwater flow model.

The terms of the Darcy-flux method are extrapolated along an entire shoreline segment, the extent of which is based on what is determined to be reasonable. The longer the segment, the weaker the assumption that *K*, *A*, and $\Delta h/l$ indeed are uniform along the segment. Therefore, the wetland perimeter is divided into several segments, each of which is associated with a specific monitoring well located near the wetland. This commonly is referred to as the segmented-Darcy approach, an example of which is depicted in Fig. 3.21. Once the hydraulic gradient, shoreline-segment length, and estimated *K* and *b* are determined for each shoreline segment, all of the information is available to calculate *Q* for each segment and the entire wetland. The example shown in Fig. 3.22 is of a flow-through wetland, one that both receives groundwater discharge and contributes wetland water to the adjacent groundwater system. This approach allows for estimation of total groundwater discharge, total recharge of wetland water to groundwater, as well as the net (G_i minus G_o) term.

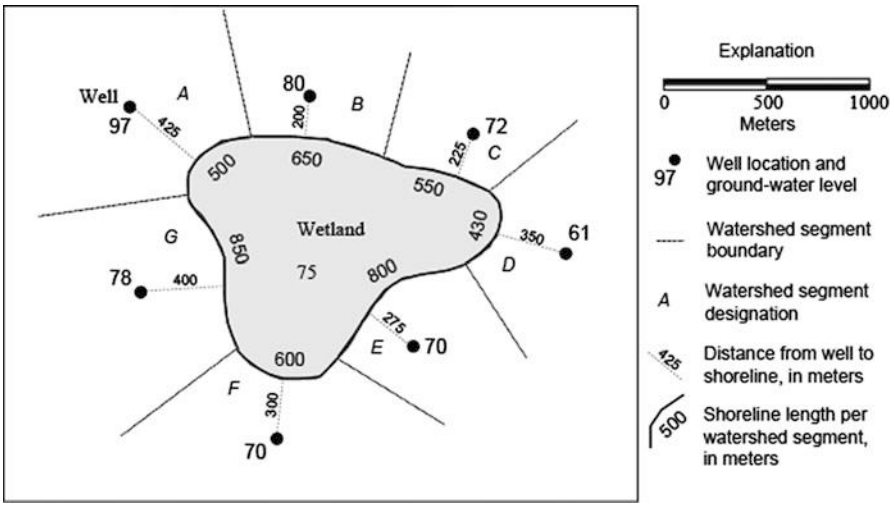

Fig. 3.21 Segmenting of a hypothetical wetland for determination of Darcy fluxes based on locations of seven monitoring wells surrounding the wetland (Modified from Rosenberry et al. (2008). Published with kind permission of the U.S. Geological Survey. Figure is public domain in the USA. All Rights Reserved)

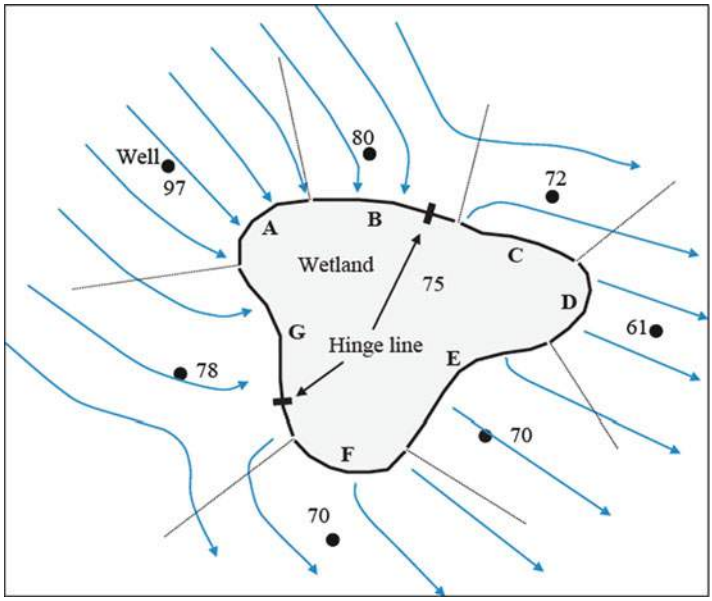

Fig. 3.22 Wetland perimeter divided into segments for determination of Darcy fluxes with groundwater flowlines shown in *blue* (Modified from Rosenberry et al. (2008). Published with kind permission of the U.S. Geological Survey. Figure is public domain in the USA. All Rights Reserved)

3.8.1.1 Location of Monitoring Wells, Assumptions, and Errors

The value of a single monitoring well for determining exchange between groundwater and wetland water is slight. Analysis of sediment and distribution of soil types based on materials removed during the well installation has local value but the lateral extent of these properties is unknown. Difference in hydraulic head between the single well and the wetland can be determined, but little can be known about the actual direction of flow of the ground water with only a single well and wetland stage. Two wells provide additional information about the local-scale geology and hydraulic gradient. If holes augured on opposite sides of a wetland both indicate similar geology, then confidence is increased that geology surrounding the wetland is somewhat uniform. However, information still will be insufficient to characterize hydraulic gradients around the entire wetland perimeter with any certainty.

Unless aquifer-gradient information is known a-priori, the minimum number of wells required to estimate groundwater exchange with a wetland, at least qualitatively, is three. If wells are distributed approximately evenly around a wetland, contour lines of equivalent hydraulic head (equipotential lines) can be drawn based on the head values from the wells and the stage of the wetland. Once equipotential lines are drawn, groundwater flowpaths can be drawn perpendicular to the equipotential lines. Flowpath lines will provide an indication of the direction of groundwater flow. For flow-through wetlands that both receive groundwater discharge and recharge water to groundwater, the locations of hinge lines, defined as those points along a shoreline that separate reaches where groundwater discharges to a wetland from reaches where wetland water flows to groundwater, can be drawn (Fig. 3.22).

This rudimentary analysis forms the beginning of a groundwater flow-net analysis, which is another method for estimating the direction of groundwater flow, described more completely in Rosenberry et al. (2008). Examples are shown in Fig. 3.23 based on a variety of combinations of monitoring wells. For example, heads from any two wells selected from the array of wells shown in Figs. 3.22 and 3.23, with the exception of wells A and D or A and E, will lead to an incorrect interpretation of directions of groundwater flowpaths in the vicinity of the wetland. Data from only wells C and G will lead to the assumption that flow is to the northeast (Fig. 3.23b). Data from wells C and F will lead to the assumption that the wetland is losing water to groundwater at least along the majority of the wetland margin (Fig. 3.23c).

Heads only from wells B, E, and G would result in a correct interpretation of the direction of groundwater flow (Fig. 3.23d). However, without data from wells A or D, the interpretation would be that far less groundwater exchanges with the wetland. Groundwater flow to the wetland would occur only along the shoreline represented by well B. Assuming that segments A, B, and C were assigned to the gradient at well B, that b is 20 m, and that K is 30 m/day, inflow would total 25,500 m^3/day. Outflow, assuming well E is assigned to segments D, E, and F, well G is assigned only to segment G, and b and K remain the same at 20 and 30, respectively, would be −19,964 m^3/day. This would result in an imbalance of over 5,000 m^3/day or 20 % of inflow. If wells were located near the three protruding bays of the wetland (wells A,

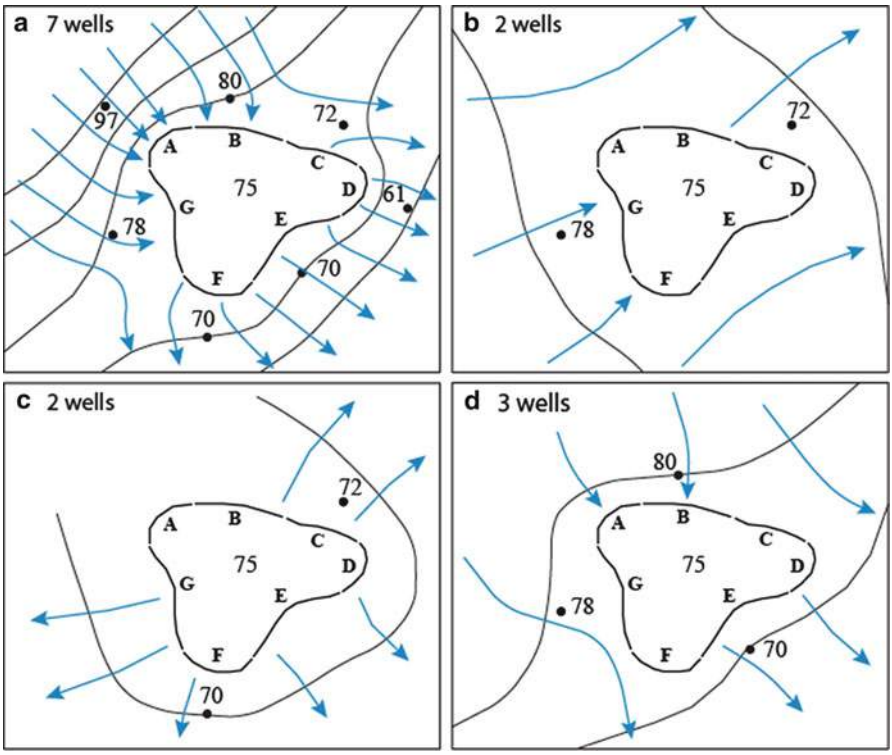

Fig. 3.23 Groundwater equipotential lines and flowlines based on (**a**) seven wells, (**b**) two wells, (**c**) a different combination of two wells, and (**d**) three wells

D, and F), inflow would total just over 62,000 m^3/day and outflow would total just over −43,000 m^3/day. Groundwater discharge to the wetland would be more than double the estimate based on seven monitoring wells and flow of wetland water to groundwater would be only two-thirds of groundwater discharge. It is clear that the location and number of monitoring wells are crucial for determining a reasonably accurate indication of groundwater exchange with a wetland.

The segmented-Darcy approach assumes that groundwater flow vectors are perpendicular to the wetland shoreline. This clearly is not the case for some wetland settings, including the example shown in Fig. 3.22. Groundwater flow is primarily tangential to shoreline segments B and G, where hinge lines are located. Those hinge lines also indicate that, not only is the hydraulic gradient not uniform along the entire segment reach, the gradient is in opposite directions from one end of the segment to the other. Contouring the hydraulic-head data and drawing flowlines provides additional information about groundwater exchange with the wetland. For example, flowlines can provide a much better indication of the locations of hinge lines. Knowing that, shoreline segments can be modified to end in the vicinity of hinge lines and more realistic values of flow can be determined for each shoreline segment.

Errors of interpretation and incorrect assumptions regarding directions of flowlines can be greatly reduced through the use of one of a variety of commonly available flow-modeling techniques. Perhaps the simplest and oldest is the previously mentioned hand-drawn flow-net approach. However, many analytical and numerical computer-based models can provide a quick analysis of likely groundwater flowpaths and estimations of volumes of exchange between groundwater and water in the wetland. The influence of upper and lower bounds of K and b on volumetric exchange also can be determined with numerical simulation.

3.8.1.2 Location and Installation of Monitoring Wells

The interpretation of groundwater-surface-water exchange, whether by segmented-Darcy, flow-net, or analytical or numerical models, depends on data from monitoring wells. Fortunately, installation of wells for the purpose of measuring the elevation of the upper extent of saturated sediments (the water table) is often relatively simple and inexpensive. Shallow water tables and small depths to water, not to mention soft and often nearly saturated sediments in near-shore wetland margins, allow monitoring wells to be installed manually rather than with a drilling rig. Although wells can sometimes be driven to depth with a post driver, sledge hammer, or hydraulic-push rig, it usually is better to auger a test hole, collect, describe, and analyze the sediments removed from the hole, and then install the well in the test hole. This may be difficult in some sediments, either because the sediments are poorly consolidated and slump back into the hole, or because sediments contain a large fraction of cobbles or larger particles, making hand auguring difficult or impossible.

Two types of monitoring wells are used for wetland-hydrology investigations. A water-table monitoring well is designed to indicate the level of the top of the aquifer, where total pressure is equal to atmospheric pressure and below which all the pores in the soil are filled with water; in other words, the water table. Because the water table can fluctuate over a range of several meters in some wetland settings, a water-table well needs to have a well screen that is long enough that it intersects the water table whether the water table happens to be high or low at the time. Note the long well screen for the water-table well shown in Fig. 3.24 that extends above the water table. The other type of monitoring well is often termed a piezometer. A piezometer is designed to represent hydraulic head at a single point in an aquifer. Ideally, such a well would just have an opening at the bottom of the well casing to represent pressure. Because many piezometers also are designed as water-quality sampling points and need to produce some water for sampling, they often have short screened intervals. In such cases, the mid-point of the screened interval is the depth to which pressure head indicated by the piezometer is generally associated. Two piezometers are indicated in Fig. 3.24. One represents a piezometer installed near a wetland and will provide a pressure head to compare with the adjacent water-table well. The other is installed in the sediments beneath the wetland bed and is designed to provide a hydraulic gradient on a vertical axis as

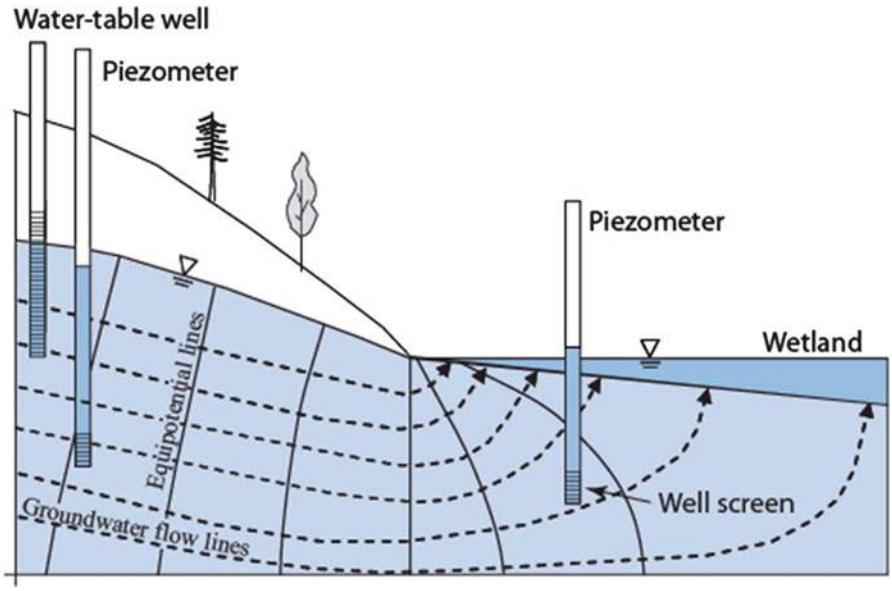

Fig. 3.24 Depiction of the two types of monitoring wells commonly installed in wetland settings. The water level in the piezometer on the left is lower than the water level in the water-table well, indicating that a downward gradient exists in the aquifer at that point. The piezometer to the right positioned in the wetland has a water level higher than the wetland surface, indicating an upward gradient exists in the aquifer beneath the wetland

determined by the pressure difference between the mid-point of the piezometer screen and standing water in the wetland. Actual flow vectors beneath a wetland bed cannot be known with a single piezometer installed in standing water in a wetland. However, because the hydraulic gradient is determined on a vertical axis, the direction of flow in these settings is almost always assumed to be vertical.

Regardless of the preferred installation method or monitoring-well type, well screens should be selected so the width of openings in the screen, commonly called the slot size, is representative of the grain-size distribution of the sediment adjacent to the screen. This ensures that the monitoring well is in good hydraulic connection with the surrounding sediments. The screen length and installation depth should adequately represent the elevation of either the water table or pressure head at a specific depth below land surface. If a water-table well is installed at too great a depth, and the well has a screened interval that is substantially below the water-table elevation, the well will likely function as a piezometer, providing a water level indicative of hydraulic head at some depth beneath the water table rather than the actual water-table elevation. This effect generally is a concern only when vertical hydraulic gradients are large.

Wetland settings, although generally conducive to studies involving monitoring wells, can present unexpected challenges in data interpretation. Evapotranspiration from emergent vegetation and dense riparian vegetation can extract groundwater until the water-table elevation is below the wetland stage. Recharge to groundwater

is more rapid and extensive in these near-shore margins where the unsaturated zone is thinnest, and may result in the water table being higher than elsewhere, effectively forming a hydraulic dam between the wetland and groundwater farther from the wetland. Numerous examples of either a water-table trough or ridge between a monitoring well and the wetland shoreline are reported in the literature (e.g., Rosenberry and Winter 1997). If either a trough or a ridge is present between a wetland and nearby monitoring well, water cannot flow from the well to the wetland or vice versa. It often is prudent to install two or more monitoring wells at different distances from shore to determine if transient water-table ridges or troughs occur, are frequent, or persistent.

Once installed, determining the water level in a monitoring well can be accomplished with several methods ranging from something as simple as lowering a chalked steel tape into the well to immersing a pressure transducer that includes a self-contained datalogger for collecting time-series data. Details for measuring water levels in wells are presented in Cunningham and Schalk (2011).

3.8.1.3 Methods for Determining Hydraulic Conductivity (K)

Of all the factors that control the degree of exchange between groundwater and wetland water, K is the most spatially variable and often the most difficult to determine. A complex history of erosion and deposition of organic and inorganic sediments is commonly encountered in many wetland settings where stage and shoreline location can vary by a large amount over time. Organic-rich sediments, typically with small values of K, can be situated next to wave-washed sand and gravel in these dynamic environments, complicating the determination of K on a scale that is relevant to a wetland water budget. Furthermore, determination of K is itself scale dependent (e.g., Rovey and Cherkauer 1995). Point measurements may represent conditions within a few meters of a monitoring well, but will not be representative of a more transmissive portion of the sediments that may route most of the groundwater to or from a wetland. Most sediment is more permeable to horizontal flow than to vertical flow. In addition, K commonly decreases with sediment depth (Hayashi et al. 1998). Reduction in K with depth also is particularly common in peat. An additional complexity of peat is that it is compressible, which also affects K (Surridge et al. 2005; Hogan et al. 2006). For these many reasons, determinations of K require careful consideration and several avenues of investigation.

A single-well slug test provides a reasonable indication of K at a scale comparable to the size of the well screen. This method involves recording the water level within a well, typically with a submerged pressure transducer, while the water level is suddenly increased or decreased (e.g., Fetter Jr 2001). The rate of recovery of the water level in the well is proportional to the hydraulic conductivity of the sediments that surround the well screen. Analysis of the recovery curve assumes that flow to or from the well is primarily horizontal and requires use of one of several analytical methods such as Bouwer (1989), Bouwer and Rice (1976) or Hvorslev (1951) to calculate K.

Obtaining a single value for K that is representative of an entire wetland is possible where wetland stage can be changed rapidly, either by pumping wetland water elsewhere or by altering wetland stage with a control structure. As an example, water was pumped from a 60-m-diameter wetland in Florida until wetland stage was lowered 0.3 m and the recovery of wetland stage was recorded following the end of the pumping period (Wise et al. 2000). Because the recovery occurred during a time of minimal rainfall, and the rate of recovery was much larger than potential effects of rainfall or evapotranspiration, recovery of wetland stage following pumping was attributed to seepage from groundwater. By measuring the vertical hydraulic gradient, i_v, between wetland stage and several piezometers installed within the wetland basin, carefully measuring wetland bathymetry to obtain a good estimate of A for each increment of wetland stage, and knowing the amount of water pumped from the wetland (Q), Darcy's law can be manipulated to calculate the vertical component of hydraulic conductivity, K_v, of the wetland sediments:

$$K_v = Q/(i_v A) \tag{3.36}$$

Once K_v is known, i_v can be monitored with measurements of piezometers installed in the wetland and G_i or G_o can be determined depending on whether i_v is indicating upward or downward flow potential.

3.8.2 Direct Seepage Measurements

Most devices or methods for quantifying exchange between groundwater and surface water are based on indirect measurements. For example, hydraulic gradient and hydraulic conductivity are determined using the segmented-Darcy approach, but the actual quantity of interest is the flux across the sediment-water interface. A seepage meter is an instrument that directly measures flow across the sediment-water interface between groundwater and surface water. Although several early versions developed in the 1950s and 1960s were unwieldy and quite complex (listed and described in Carr and Winter 1980), the meter generally in use since the mid 1970s, the "half-barrel" seepage meter, is very simple and inexpensive. The device consists of an open-ended seepage cylinder placed on the bed to which an attached plastic bag is used to record the time-averaged rate of flow (Lee 1977). The open-ended cylinder isolates a portion of the bed, commonly 0.25 m^2, and all flow across the bed area covered by the cylinder is routed to (or from, depending on the direction of flow) the attached plastic bag (Fig. 3.25). By recording the volume contained in the bag at the times of emplacement and removal, the volumetric seepage rate is determined:

$$Q = \frac{V_{t1} - V_{t2}}{t_1 - t_2} \tag{3.37}$$

Fig. 3.25 Half-barrel seepage meter with seepage bag located inside a bag shelter for protection from currents and waves (Photo by Donald Rosenberry)

where V_{t1} is the volume contained in the bag at the start of the measurement period, V_{t2} is the volume in the bag at the end of the measurement period, and t_1 and t_2 are the times at the start and end of the measurement period. Dividing that result by the area covered by the seepage cylinder gives seepage flux in length per time:

$$q = \frac{Q}{A} \tag{3.38}$$

Although conceptually very simple, the device is not necessarily simple to use. Inferior data have been collected and published, likely because the simplicity and low cost of the meter have resulted in insufficient understanding of sources of error and attention to measurement commensurate with the cost and complexity of the instrument. However, given sufficient measurement care, the half-barrel seepage meter can provide reliable and repeatable data (Rosenberry et al. 2008).

Several modifications to the basic design are commonly employed to reduce measurement error and improve measurement efficiency. Perhaps the most important is to place the seepage bag inside of a shelter to minimize the influence of currents and waves, as shown in Fig. 3.25. Seepage bags exposed to currents can fill with water due to velocity-head effects not normally considered by groundwater scientists (Sebestyen and Schneider 2001; Rosenberry 2008). Other modifications

include deploying the bag 1 m or more from the seepage cylinder to minimize local disturbance during attachment and removal of the bag, increasing the diameter of bag-connection hardware to improve meter efficiency, and connecting multiple cylinders to a single bag to reduce measurement time and increase the bed area represented by each measurement (Rosenberry 2005). Additional discussion of modifications and sources of error (and how to minimize them) are presented in Rosenberry et al. (2008).

Seepage meters also have been modified for use in streams (Rosenberry 2008). Such a meter would be useful for riparian wetland settings where flow, although usually relatively slow, could still corrupt seepage measurements made with meters not modified for use in flowing water. For the typically slow flow velocities associated with wetland settings, the most important consideration is to place the bag inside of a bag shelter.

As indicated earlier, many of the errors associated with seepage measurements can be attributed to problems associated with the seepage bag. Furthermore, any variability in seepage rate is integrated over the duration of each bag attachment. To address these concerns, the bag can be replaced with alternate means of quantifying flow ranging from chemical-dilution methods to heat-pulse flow technology to mechanical or electromagnetic flowmeters (Rosenberry et al. 2008). Much finer temporal resolution is possible with these designs that allow quantification of processes that would otherwise be impossible with standard designs (Rosenberry and Morin 2004; Rosenberry 2011).

3.8.3 *Determining Groundwater Fluxes as the Residual of a Water Budget*

The wetland water budget presented earlier (Eq. 3.1) can be reordered to solve for net groundwater exchange:

$$G_i - G_o \pm R = \Delta V/\Delta t - P + ET - S_i + S_o - O_f \tag{3.39}$$

where the terms are as described earlier. This is a common approach for determining net groundwater contribution to lakes, wetlands, or stream reaches where groundwater fluxes are difficult to determine with more direct measurements. Hood et al. (2006), for example, used Eq. 3.39 to estimate the contribution of groundwater to an alpine lake. Note that this provides only the net groundwater exchange ($G_i - G_o$). For wetlands where either G_i or G_o dominates, determining the net groundwater contribution may be all that is needed, but for many other wetland settings determining the net term may not be sufficient. For example, by only knowing net groundwater exchange, the water residence time cannot be determined. Fortunately, if we also have a chemical constituent of some sort that is associated with each of the water terms, then both G_i and G_o can be determined.

The water-budget equation, including chemical concentrations associated with each term, can be written as a mass-balance equation:

$$\Delta(C_W V)/\Delta t = C_P P + C_{Si} S_i + C_{Gi} G_i + C_{Of} O_f - C_{ET} ET - C_{So} S_o - C_{Go} G_o \pm \varepsilon \tag{3.40}$$

where C is the concentration of the chemical constituent, ε is the total hydrologic and chemical-measurement error, subscripts are related to the various components of the water budget, and $_W$ refers to surface water in the wetland. In many settings, surface water in the wetland is well mixed; therefore, it is reasonable to assume that the concentration associated with G_o and S_o are the same as that of the wetland surface water, C_W. Under this assumption, Eq. 3.39 is rearranged to isolate G_o and this expression is substituted into Eq. 3.40 to obtain

$$G_i \pm \varepsilon = \frac{V\frac{\Delta C_w}{\Delta t} + (C_W - C_P)P + (C_W - C_{Si})S_i + (C_W - C_{Of})O_f + (C_{ET} - C_W)ET}{C_{Gi} - C_W} \tag{3.41}$$

Note that the residual term R in Eq. 3.39 was omitted in this substitution because all errors are now lumped into ε. Lastly, G_i can be inserted into Eq. 3.39, which can be rearranged to solve for G_o:

$$G_o \pm \varepsilon = P + S_i + G_i + O_f - ET - S_o - \frac{\Delta V}{\Delta t} \tag{3.42}$$

Another important assumption is that the chosen chemical constituent is conservative, meaning that it is not altered by any chemical or biological process. Water solutes are commonly used in this analysis and chloride is often considered conservative in many settings. Stable isotopes of water, usually deuterium (^{2}H) or oxygen-18 (^{18}O), are an excellent choice because they are not a dissolved solute but part of the water molecule. If chloride or another solute is used, the equation is simplified somewhat because the evaporation process distils the water and no solute is lost with the evaporating water; therefore $C_{ET}ET$ is zero. If a stable isotope of water is used, the isotopic value of the evaporating water needs to be determined. This value is rarely available, is relatively difficult to obtain, and often is estimated based on other studies conducted within the area or region (e.g., LaBaugh et al. 1997).

This method is not well suited for wetland water budgets dominated by groundwater discharge. As G_i becomes large, the difference between the two terms in the denominator of Eq. 3.41, $C_{Gi} - C_W$, becomes small, at which point measurement errors can greatly affect the solution. If water isotopes are used, the method is not very robust when the water residence time of the wetland is short or seasonal variation in isotopic composition is large (Krabbenhoft et al. 1994). In such instances, it is better to use a conservative major ion. Errors can be substantial for some of the terms and in some cases the residual term, ε, can approach or exceed

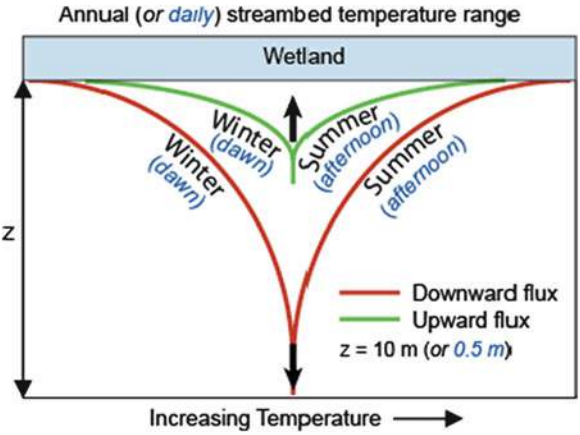

Fig. 3.26 Streambed temperature profiles for downward water flow and upward water flow (Modified from Constantz (2008). Published with kind permission of the U.S. Geological Survey. Figure is public domain in the USA. All Rights Reserved)

either G_i or G_o. A good discussion of errors associated with use of the water budget to determine groundwater exchange terms can be found in LaBaugh and Winter (1984), Krabbenhoft et al. (1990), or Choi and Harvey (2000).

3.8.4 Measurement of Temperature to Quantify Groundwater-Surface-Water Exchange

Diurnal and seasonal temperature changes in wetland water are attenuated with depth beneath the sediment-water interface. Attenuation is controlled by the capacity of the sediments to conduct heat. Direction and rate of groundwater flow modifies the conduction-driven attenuation. Net upward flow reduces, and net downward flow increases, the amplitude of diurnal or seasonal temperature responses with depth beneath the sediment-water interface (Figs. 3.26 and 3.27).

Robert Stallman (Stallman 1965) developed a method that could determine vertical flow of water through sediment based on measurement of temperature. The method required measurement of diurnal (or seasonal) fluctuation of temperature at two depths, the volumetric heat capacity of the water, and estimates of thermal conductivity and volumetric heat capacity of the bulk sediment. If the time series of the diurnal or seasonal temperature data are sinusoidal, this method provides a reasonable indication of the groundwater flow to or from a surface-water body. However, when applied to the typical wetland setting where wetland stage (and, therefore, i) is often highly variable over time, a method is needed that can solve for flow that may be substantially non-uniform.

Fortunately, several numerical models exist that simultaneously solve for heat and fluid flow in porous media. In addition to the parameters listed above, reasonable assumptions regarding boundary conditions, and values for i_v and K_v, are all that is needed to determine the flow of water across the sediment-water interface.

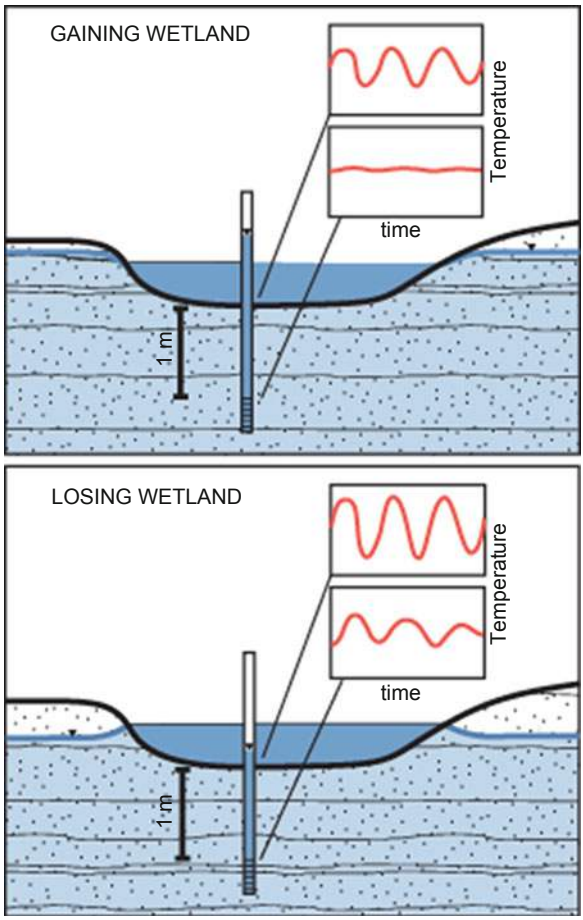

Fig. 3.27 Diurnal temperature response at the wetland bed and at a well screen installed 1 m beneath the wetland for upward and downward groundwater flow beneath the wetland (Modified from Stonestrom and Constantz (2003). Published with kind permission of the U.S. Geological Survey. Figure is public domain in the USA. All Rights Reserved)

The vertical hydraulic gradient can be obtained with the installation of a shallow piezometer in the wetland bed. This installation also allows convenient deployment of a temperature sensor to provide data at depth to compare with temperature at the bed of the wetland. Although K_v can be determined in-situ, K_v often is heterogeneous and is scale dependent. Therefore, K_v usually serves as the model calibration factor that is adjusted until the simulated time-series temperature data generated by the model match the measured time-series temperature data. Once K_v is calibrated so the modeled and measured temperatures are in good agreement, the model produces q, the specific groundwater flux across the sediment-water interface.

Temperature sensors are among the most accurate, robust, and inexpensive devices commonly used in the earth sciences, making this method particularly attractive. Thermal conductivity (K_T), the heat-flow equivalent of hydraulic conductivity (K), is a property that varies over a much narrower range than hydraulic conductivity. It can be reasonably estimated based on the type of sediment present

Table 3.2 Typical values of parameters required to determine q using numerical models that solve for heat and fluid flow through porous media (Modified from Stonestrom and Constantz 2003)

Material	Porosity	Density (10^6 g/m^3)	Volumetric heat capacity (10^6 J/m^3/ °C)	Thermal conductivity (W/m/°C)	Thermal diffusivity (10^{-6} m^2/s)
Liquid water	1.0	1.0	4.2	0.6	0.1
Ice		0.9	1.9	2.2	1.2
Quartz		2.7	1.9	8.4	4.3
Soil minerals	0.2–0.4	2.7	1.9	2.9	1.5
Clay minerals	0.4–0.7	2.7	2.0	2.9	1.5
Soil organic matter	0.4–0.9	1.3	2.5	0.25	0.1
Organic estuary[a]	0.8		2.3	0.9	0.2

[a]Values from Land and Paull (2001)

beneath the wetland. Other properties, such as porosity, diffusivity, and heat capacity of the sediments, affect the solution to a lesser extent. These values commonly are estimated based on the type of sediment present at the site of interest (Table 3.2).

Any numerical model that simultaneously solves for fluid flow and heat flow can be used. Two commonly used models are SUTRA and VS2DH (Stonestrom and Constantz 2003). If a substantial horizontal component of flow is suspected, such as near the perimeter of many wetlands, additional wells can be installed near the wetland, temperature measured at several depths, and a 2-d version of the model can be created and calibrated to solve for flow along the wetland bed (Fig. 3.28). Although both Figs. 3.27 and 3.28 indicate that temperature is measured at several depths beneath the wetland bed, a single temperature measurement at depth will suffice. With only one temperature measurement in the sediments, the assumption is that K of the sediments is uniform. Additional temperature measurements provide additional data regarding the variability and distribution of K within the sediments. An example of adjusting K to obtain a good fit of modeled output to measured time-series data is presented by Stonestrom and Constantz (2003:88).

Vertical flow velocity also can be calculated by comparing the ratio of amplitudes or the phase shift of time-series data from temperature measured at different depths (Hatch et al. 2006). The selection of amplitude ratio or phase shift depends on the vertical velocity; very fast seepage rates are better resolved with the phase-shift solution whereas the slower seepage rates common in wetland settings are better determined with the amplitude-ratio method. This procedure has the added benefit of calculating changes in seepage over time and does not require measurement of hydraulic-head gradients. Although the degree and type of filtering of the data can be rather complex, advancement and use of this technique has been rapid and several variations of the original concept have been presented (Keery et al. 2007; Swanson and Cardenas 2010; Vogt et al. 2010). An automated data-processing routine has recently been developed to make the method faster and more user-friendly (Gordon et al. 2012).

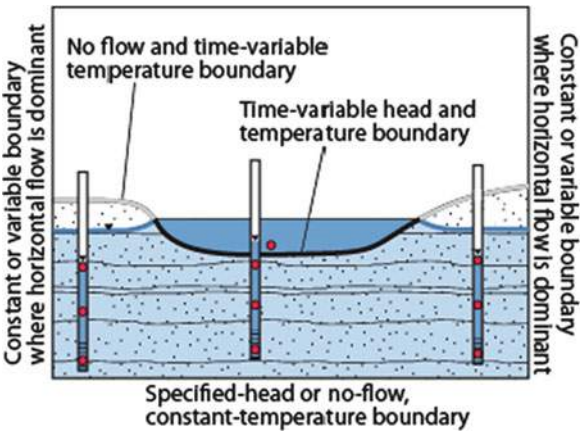

Fig. 3.28 Boundaries associated with a coupled water- and heat-flow model to simulate groundwater exchange with a wetland in two dimensions. *Red dots* are locations where temperature is measured. Head is measured at the wetland and at the screened interval of two monitoring wells (Modified from Stonestrom and Constantz (2003). Published with kind permission of the U.S. Geological Survey. Figure is public domain in the USA. All Rights Reserved)

Measuring and mapping the temperature of the submerged wetland bed also can be used to determine rates and distribution of groundwater discharge, but only for wetlands where groundwater discharge is prominent and pervasive (Schmidt et al. 2007). This method uses the Turcotte and Schubert (1982) solution for steady-state 1-dimensional advection-diffusion heat flow and relates temperature measured at about 20-cm depth in the bed sediment to an assumed constant temperature at greater depth in the sediment. The method requires that the surface-water temperature has small diurnal variability prior to and during the mapping of the temperature of the wetland bed, a condition best met during winter or during prolonged cloudy periods. Although the method was developed for use in streams, it should provide acceptable results for many wetlands that receive groundwater discharge; bed-sediment temperatures should be measured during periods when diurnal fluctuations are minimal. The method should work particularly well for wetlands that are ice covered during winter.

Mapping the bed temperature has become much easier with the growing use of what is now commonly called the distributed temperature system (DTS) (Selker et al. 2006; Fleckenstein et al. 2010). This system uses a device that sends a laser pulse down a length of fiber-optic cable that can be up to several km long. The light signal is reflected back to the sensor from every point along the cable. By timing the return, and resolving the frequency distribution of the light scattering, temperature can be determined to about 0.1 °C resolution and averaged over cable increments of 0.5–1 m. Temperature mapping of a sediment bed can be done as frequently as every minute to several minutes, allowing a qualitative determination of temporal as well as spatial variability of groundwater discharge (e.g., Henderson et al. 2009).

3.9 Subsurface Storage Above the Water Table

The storage term (ΔV) in the water balance equation commonly represents the amount of surface water in a wetland. For those wetlands with seasonal or ephemeral surface water, subsurface water storage also is an important hydrological consideration. Moisture content of the exposed soil influences the transport of oxygen and other gases, thereby affecting redox condition and biogeochemical processes. Moisture conditions affect the viability of soil fauna and the growth of plants adapted to high moisture environments.

After surface water in the wetland dries up, water loss from the wetland soil continues, mainly due to transpiration by wetland vegetation, which causes the water table to drop and the soil to become unsaturated. The volume of sediment between land surface and the water table is called the vadose zone. The soil remains nearly saturated immediately above the water table due to surface tension that holds water in the soil pores. As soil dries and the water table continues to decline, it becomes increasingly difficult for plant roots to extract water from the soil. As a result, the rate of transpiration decreases, the rate of water-table decline decreases, and the water table eventually reaches a relatively stable position. This condition persists until something changes; most often the change is a subsequent recharge event that adds water to the unsaturated sediments. The amount of water required to saturate the soil completely and bring the water table to land surface is called the moisture deficit. A wetland with a small moisture deficit can recover from a dry condition relatively quickly when wet meteorological conditions return. Therefore, subsurface moisture storage is an indicator of the resilience of a wetland to fluctuations in water inputs.

Subsurface moisture storage is determined by the depth to the water table and soil water content in the vadose zone. Methods for determining the position of the water table are described in the section on groundwater flow. Here we describe methods for measuring soil-water content and then introduce the concept of specific yield, S_y, that relates subsurface storage to water-table depth.

3.9.1 Thermo-Gravimetric Method for Measuring Soil Water Content

This method starts with collecting a sample of undisturbed soil in a metal cylinder of precisely known volume (e.g., 100 cm^3) using a soil corer, or inserting the cylinder into the side wall of a soil pit. Care must be taken to fill the cylinder completely while at the same time avoiding soil compaction. The top and bottom of the sample are leveled using a metal scraper so that the soil volume is equal to the volume of the cylinder, and sealed with plastic caps and electrical tape to prevent evaporation. Samples are transferred to the laboratory and weighed using a balance to determine the pre-drying weight. Samples are placed in an oven with

a temperature set at 105 °C for 24–48 h to evaporate all liquid water without volatilizing organic components of the soil. The sample weight may also be measured periodically during drying until it does not change any longer. Samples are cooled in a sealed container with desiccant to prevent absorption of atmospheric vapor during cooling, and weighed again. Volumetric soil water content, θ_v (cm^3/cm^3), is given by

$$\theta_v = [(\text{original weight} - \text{dry weight})/\text{density of water}]/\text{sample volume} \quad (3.43)$$

The same sample may be used to determine other soil parameters such as dry bulk density or porosity (see Chaps. 4 and 8 on soil sampling).

The water content determined with this method is commonly used as the reference to test or calibrate other methods. However, it should be noted that the thermo-gravimetric method does not necessarily yield exact results (Topp and Ferré 2002) because the measured value may be affected by the drying temperature and time, vapor absorption during cooling, and most importantly, errors in measurement of sample volume and weight. A major disadvantage of this method is that it requires the removal of the sample and is not suitable for continuous, in situ monitoring of soil water. Therefore, instrumental methods are commonly used for continuous monitoring.

3.9.2 Time Domain Reflectometry

Of the various types of instruments available for continuous monitoring of soil moisture, time domain reflectometry (TDR) and the capacitance method are the most widely used (Ferré and Topp 2002). These methods both make use of the fact that the velocity of electromagnetic (EM) waves is equal to the speed of light ($c = 3.0 \times 10^8$ m s^{-1}) in a vacuum but is lower in other media. EM-wave velocity is determined by a property called dielectric permittivity. The dielectric permittivity of water relative to a vacuum is much greater (≈80) than that of air (≈1) or soil solids (≈3–8). Therefore, volumetric water content can be estimated from measurements of soil dielectric permittivity.

In the TDR method, a very sharp voltage pulse from the signal source travels through the soil along a wave guide, typically consisting of parallel stainless steel rods (Fig. 3.29), and is reflected back to the source. The reflected signal is recorded as a time series of voltage values, commonly called the wave form (Fig. 3.30), in which the time to the negative peak (t_1) indicates the two-way travel time of the EM wave between the source and the top of the wave guide, and the time to a rapid rise (t_2) indicates the two-way travel time between the source and the bottom of the wave guide. If the length of wave guide is L (m), then the apparent velocity of the EM wave (v_{EM}) in the soil is

$$v_{EM} = 2L/(t_2 - t_1) \quad (3.44)$$

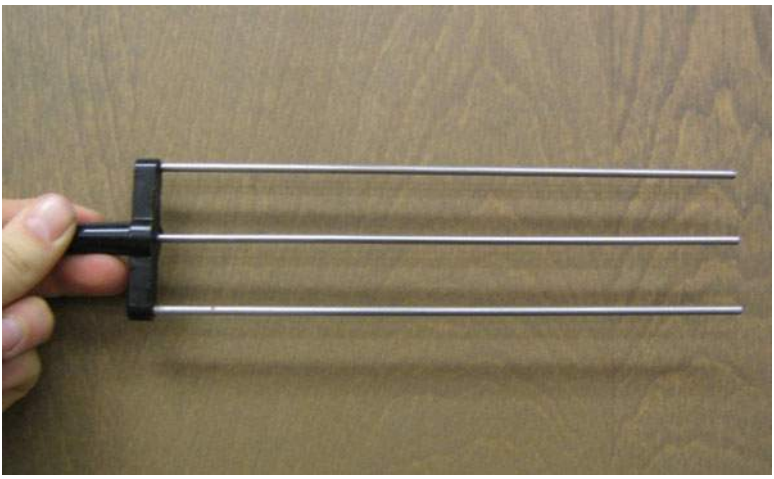

Fig. 3.29 Example of a parallel-rod wave guide for time domain reflectometry (TDR)

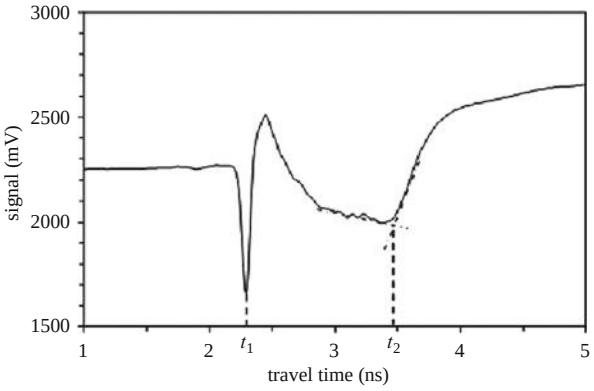

Fig. 3.30 Example of waveform data obtained by a TDR device with a 0.2-m parallel-rod wave guide (see Fig. 3.29) installed in a mineral soil. Times t_1 and t_2 indicate the travel time of signals reflected from the top and bottom of the wave guide

For low-salinity soils, relative dielectric permittivity (ε_r) is given by

$$\varepsilon_r = (c/v_{EM})^2 \tag{3.45}$$

Equation 3.45 only gives approximate values for soils with high electrical conductivity (Ferré and Topp 2002). Volumetric water content is estimated from a calibration curve relating θ_v and ε_r. For a large variety of agricultural mineral soils, a "universal" formula of Topp et al. (1980) has been found to yield reasonably accurate values of θ_v:

$$\theta_v = -0.053 + 0.0292\varepsilon_r - 0.00055{\varepsilon_r}^2 + 4.3 \times 10^{-6}{\varepsilon_r}^3 \tag{3.46}$$

However, significant deviation has been noted in organic soils. Therefore, it may be necessary to develop soil-specific calibration curves for application of the TDR method in organic soils typically found in wetlands (see next section for calibration methods).

TDR wave guides (or probes) can be installed vertically from the surface or horizontally in a soil pit, which should be refilled very carefully to prevent preferential infiltration affecting the measurements. The measurement is sensitive to soil disturbance and gaps between the probe and soil matrix. Therefore, probes must be inserted straight into the soil, with minimum wobble, to minimize disturbance. A TDR probe measures an average θ_v over the entire probe length in a cylindrical region within a diameter of approximately 1.5 times the rod separation (Ferré and Topp 2002). If the depth to the water table is shallow (e.g., <0.3 m), vertical probes may be used to cover the entire vadose zone. Where the water table is deeper, horizontal probes need to be installed at multiple depths to measure a profile of water content from the surface to the water table.

Probes may be connected to a portable field device for manual recording of wave forms and determination of ε_r, or connected to a digital datalogger. Most of commercially available TDR devices determine θ_v using internal algorithms and output the value of θ_v, which is a convenient feature for long-term monitoring. However, it is prudent to store the raw waveform data and periodically check the accuracy of automatically determined θ_v. Detailed discussion on the TDR method and useful guidance for its application are found in Ferré and Topp (2002).

3.9.3 Capacitance Method

This method also utilizes the large contrast in dielectric permittivity between water and other soil components, but it is based on the principle that frequency of oscillation of a circuit consisting of an electrode-soil capacitor is a function of dielectric permittivity (Starr and Paltineanu 2002). Since the functional relationship is dependent on electrode configuration and soil type, soil-specific calibration is required to calculate θ_v from the frequency measured with a capacitance probe. Compared to the TDR method, which yields reasonably accurate results using the universal formula, the disadvantage of the capacitance method is the necessity of soil-specific calibration. On the other hand, once well calibrated, the capacitance method offers a much more robust and convenient tool for continuous monitoring of soil water content than the TDR method. Most commercially available capacitance probes are designed to work with standard dataloggers and can be used as part of the standard collection of sensors that make up hydrological monitoring stations. This is another advantage over TDR, which typically requires an expensive control unit in addition to a datalogger.

Depending on the monitoring objectives and probe length, capacitance probes may be installed vertically from the surface, or horizontally at multiple depths in a soil pit. Similar to TDR wave guides, measurement with capacitance probes is

sensitive to soil disturbance and air gaps between electrodes and the soil matrix. Therefore, probes need to be inserted carefully to minimize sensor error. Detailed discussion on the capacitance method and practical procedures are found in Starr and Paltineanu (2002).

If a large block of undisturbed soil representative of field conditions can be removed intact, capacitance probes may be calibrated in the laboratory. Briefly, the sample is placed in a sealed container, a probe is placed in the sample, and the sample is brought to saturation. The total weight is measured and the output of the probe is recorded. The sample is then drained and dried in several stages with each stage given enough time to establish uniform water content in the container, and total weight and probe output are recorded. At the end of drying, subsamples are collected from the container to determine the dry bulk density of the soil, from which the weight of the soil is converted to volumetric water content. Starr and Paltineanu (2002) describe detailed procedures for laboratory calibration using disturbed and repacked soil, which is suitable for agricultural mineral soils but may not be applicable to organic soils. If it is not feasible to conduct laboratory calibrations, soil samples can be collected from the probe depth at the time of installation, and θ_v determined with the thermo-gravimetric method is then compared to the initial probe data for a single-point calibration.

3.9.4 Specific Yield

Fluctuations of the water table represent changes in subsurface storage. Since it is easy to measure the water-table elevation in monitoring wells, attempts have been made to estimate changes in subsurface storage (ΔS_{sub}) from changes in water-table elevation (Δh_{WT}) using a concept called specific yield (S_y), also known as drainable porosity. When ΔS_{sub} is expressed as depth of water, S_y is the ratio of ΔS_{sub} to Δh_{WT}, or more precisely, it is the volume of water released from or taken into storage per unit cross sectional area following a unit change in water-table elevation (Freeze and Cherry 1979:61).

Despite being conceptually simple, S_y is somewhat complex because soils can retain a sizable and variable amount of water above the water table that is related to the size of void spaces in the soil matrix. The relation between water content and the magnitude of tension force holding water in pores is called the soil water characteristic (SWC) curve. Under static conditions in the absence of vertical flow, the magnitude of tension force is proportional to distance above the water table. Therefore, SWC is commonly shown as a vertical profile of θ_v, which represents the theoretical distribution of water content after complete gravitational drainage of the vadose zone following complete saturation (Fig. 3.31). Suppose that a certain amount of groundwater is extracted, causing the water table to drop. This extraction induces drainage of water from the vadose zone until the new static condition is reached (Fig. 3.31). The amount of extracted water should be equal to the difference between the pre- and post-extraction profiles (Fig. 3.31). In other words,

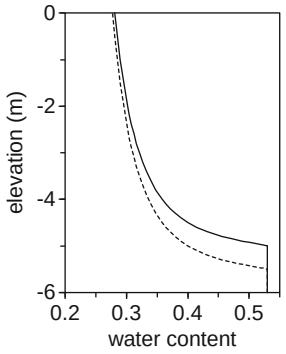

Fig. 3.31 Example of soil water characteristic (SWC) curve of agricultural mineral soil having a clay-loam texture. *Solid line* shows the soil water-content profile for the water table located 5 m below the ground surface, and *dashed line* shows the profile for the water table lowered to 5.5 m below the surface

$$\Delta S_{sub} = \int_{z1}^{0} \Delta\theta_v(z)dz \tag{3.47}$$

where $\Delta\theta_v(z)$ is change in volumetric water content as a function of elevation (z), and z_1 is a reference point below the water table. Figure 3.31 shows an example for a relatively deep water table. For a shallow water table, for example 2 m below the surface, the value of ΔS_{sub} for the same amount of Δh_{WT} is much smaller, indicating the strong dependence of ΔS_{sub} and S_y on the water-table position. In addition, depending on soil hydraulic conductivity, complete drainage of soil water following the water-table drop may take a long time, meaning that S_y is dependent on the time scale of measurement (Healy and Cook 2002).

Despite the variable nature of S_y, a constant value is used in most practical applications. Healy and Cook (2002) reviewed several field and laboratory methods for estimating S_y. If the SWC is available from laboratory analysis of soil samples, S_y can be calculated directly from the comparison of theoretical water-content profiles resulting from a given drop in the water table (e.g., Fig. 3.31). If the water table is sufficiently deep, for example deeper than 3 m for the soil shown in Fig. 3.31, the calculated S_y is little affected by the assumed position of the water table.

While this method is theoretically simple, determination of SWC is time consuming and labor intensive. The column-drainage approach offers an alternative method for laboratory measurement of S_y. In this method, a column is filled with undisturbed or repacked soil taken from a field site and then saturated with water from the bottom. Water is then allowed to drain from the bottom of the column. The top of the column is open to the atmosphere, while avoiding evaporation, and the bottom is placed in a very shallow (e.g., <2–3 mm) water reservoir to maintain a water-table condition. Dividing the amount of drainage (ΔS_{sub}) by the length of the column (Δh_{WT}) gives S_y. Care should be taken to drain the soil column completely, which may take several hours or days, depending on the soil type. The column should be sufficiently long to reduce the influence of column length on S_y. For example, the soil used in constructing Fig. 3.31 would require a column longer than 3 m, although such a long column is not practical.

Complete drainage of soil water, which is assumed in laboratory methods, may take up to several weeks. Since such assumptions do not reflect the dynamic water-table conditions in the field, it is preferable to use field methods to estimate S_y. Several methods have been proposed based on conducting an aquifer pumping test and interpreting the data. However, pumping tests require installation of a network of observation wells and the interpretation is strongly influenced by model assumptions, such as the boundary conditions or time scale of soil-water drainage (Healy and Cook 2002).

An alternative to aquifer pumping tests is the water-balance approach, where ΔS_{sub} over a given time period is estimated from careful measurements of other water balance components:

$$\Delta S_{sub} = P + O_f - E + S_i - S_o + G_i - G_o \tag{3.48}$$

Since it is usually impossible to measure all components in Eq. 3.48, periods are chosen so that some of the hydrologic components are negligible and can be omitted. For example, if a wetland does not have surface water input and output, overland flow is negligible, and the magnitude of net groundwater input ($G_i - G_o$) is expected to be much smaller than that of net atmospheric input ($P - E$), then plotting Δh_{WT} observed in response to rainfall events against the amount of net precipitation ($P - E$) may yield a linear relation between $P - E$ ($\approx \Delta S_{sub}$) and Δh_{WT}. The slope of this linear relation is equal to S_y. This type of approach is commonly used in wetland studies (e.g., Gerla 1992; Rosenberry and Winter 1997).

Considerable uncertainty and discrepancy is noted in values of S_y estimated using different methods. Field-based methods generally give smaller values than laboratory methods, presumably because laboratory methods usually allow a long time for complete drainage of soil samples compared to the time scale of field processes (Healy and Cook 2002). Therefore, investigators must be aware of the time scale of processes under investigation, as well as the assumptions associated with the definition of S_y.

3.10 Use of Conservative Tracers

In many low-gradient wetlands with extensive areas of vegetation it can be difficult to quantify several of the terms in the wetland water budget. In these situations, water chemistry may provide a separate or perhaps better estimate of some of the water-budget terms. In Sect. 3.8.3 we discussed the procedure of writing two equations, one a water-budget equation (Eq. 3.39) and the other a mass-balance equation for a conservative tracer (Eq. 3.40), for the purpose of determining either G_i or G_o. These equations can be rearranged and this procedure can be used to solve for any of the water-budget components, not just G_i or G_o. Here we discuss further the characteristics of a conservative tracer, assumptions associated with this method, and procedures for proper sample collection.

3.10.1 Evaluation of Water and Mass Balance Equations

An ideal tracer for the mass-balance method is chemically and biologically conservative (no reaction, absorption, release or uptake), is readily measureable, and is harmless to the observer and aquatic life. Chloride and bromide are commonly used as tracers (e.g., Choi and Harvey 2000; Parsons et al. 2004). For relatively shallow and small wetland ponds, water is often assumed well mixed so that tracer concentration is uniform within the pond. When this assumption is justified based on field data, for example by the analysis of samples from multiple points and depths in a pond, Eq. 3.40 is simplified to

$$\Delta(C_W V)/\Delta t = C_P P + C_{Si} S_i - C_W(S_o + G_o) + C_{Gi} G_i + C_{of} O_f \tag{3.49}$$

(note that $C_W V$ is equal to mass and that $C_E E$ has been neglected as it is assumed that no tracer mass is lost through evaporation or transpiration). If the assumption is not justified, C_{So} and C_{Go} need to be measured for each stratified layer, or for each isolated embayment of the pond, and mass needs to be determined based on the concentration and volume of each separate entity of the pond (e.g., Choi and Harvey 2000). Further simplification to Eq. 3.49 can be made in some settings. For example, if chloride is used in a region that receives very small amounts of chloride in precipitation relative to other components, then C_P is assumed negligible. Tracers also can be applied to an entire wetland in some cases. If bromide is used as an artificial tracer in a system that has very low natural bromide concentrations, then C_P, C_{Si}, C_{Gi}, and C_{Of} are all assumed negligible.

Mass balance calculation requires measurements or estimates of all "known" terms in the water balance equation and all concentrations in the mass-balance equation. Specialized monitoring devices, such as ion-specific electrodes, are available for continuous monitoring of tracer concentration, but these devices are prone to instrument drift and calibration issues. Therefore, best results are obtained by collecting water samples and analyzing them in the laboratory. This process can be conducted as part of a routine sampling program for water-quality monitoring (see Chap. 6 on water quality). Using automated water samplers (Fig. 3.32), concentrations can be determined daily or even hourly and the mass balance calculated on a fine temporal resolution. When a longer sampling interval, such as weekly or monthly, is used the water and mass balance equations need to be evaluated using average values. The equations can be written as:

$$(V_2 - V_1)/(t_2 - t_1) = <P> - <E> + <S_i> - <S_o> + <G_i> - <G_o> + <O_f> \tag{3.50}$$

$$(C_{w2}V_2 - C_{w1}V_1)/(t_2 - t_1) = <C_P\ P> + <C_{Si}S_i> - <C_w(S_o + G_o)> + <C_{Gi}G_i> + <C_{Of}O_f> \tag{3.51}$$

where subscripts 2 and 1 indicate sampling dates 2 and 1, respectively, and $< >$ indicates an average value for the time interval. Definition of average is

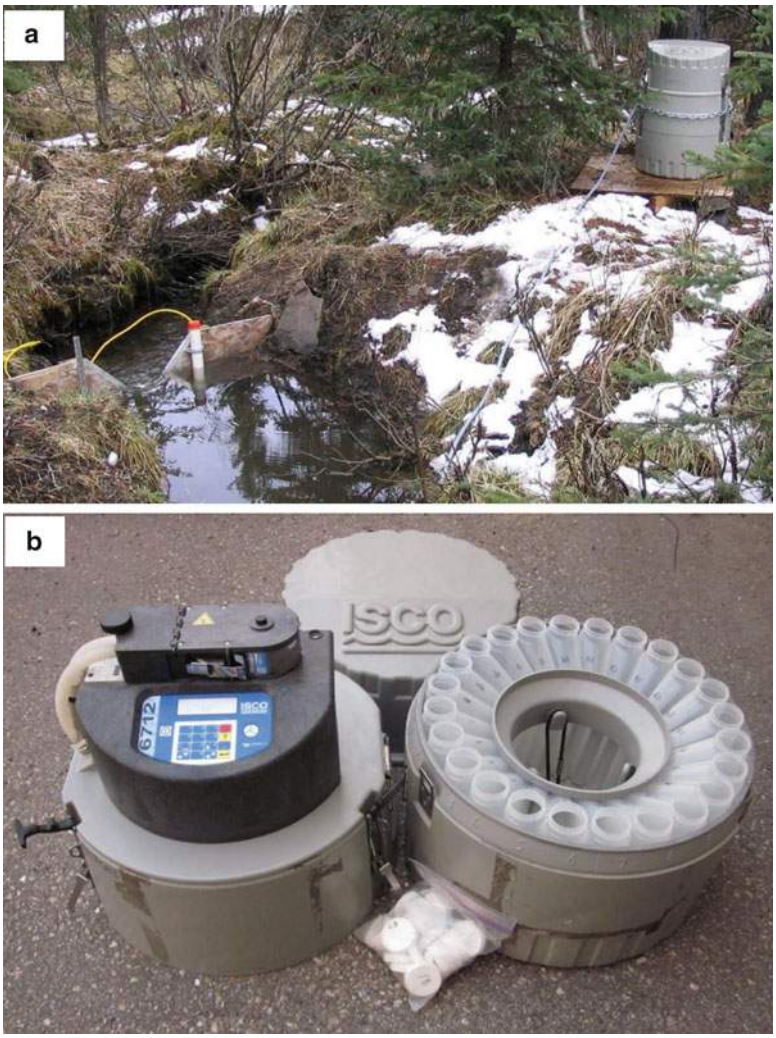

Fig. 3.32 Automated sampler for collecting stream water samples at a prescribed time interval. (**a**) A field sampler with the sample intake in a pool above a V-notch weir. (**b**) The same sampler showing the control unit and 24 sample bottles inside the sampler body (Photos by Masaki Hayashi)

straightforward for some terms; for example, $<P>$ is given by the total amount of precipitation divided by $t_2 - t_1$. However, assumptions may have to be made for averages of other terms. For example, if continuous surface inflow data are available but only two values of concentration are available, a reasonable approximation is to use an arithmetic mean for concentration

$$<C_{Si}S_i> \approx (C_{Si1} + C_{Si2})/2 \times <S_i> \tag{3.52}$$

Fig. 3.33 Application of bromide tracer in a prairie wetland (Parsons et al. 2004). To ensure even distribution of tracer, the tracer solution was slowly released to the wetland from a boat moving around the wetland (Photo printed with kind permission of © David Parsons 2012. All rights reserved)

Once the balance equations are written for measured values, they can be implemented in a spreadsheet or a simple computer program to compute desired components. For example, Choi and Harvey (2000) used chloride to quantify groundwater inflow and outflow in constructed wetlands in Florida, U.S.A. Parsons et al. (2004) used bromide to quantify evaporation and groundwater outflow in a prairie wetland in Saskatchewan, Canada.

3.10.2 Remarks on Water Sample Collection

Successful application of the mass-balance approach depends on how well the simplifying assumptions are satisfied and how well all the concentration terms are represented by measured values. If a single value of C_W is used to represent the whole wetland pond, it is important to verify that the pond is well mixed by periodically sampling and analyzing water from different locations and depths. If an artificial tracer is applied to a pond, it needs to be applied evenly in the entire pond area to ensure a uniform initial concentration (Fig. 3.33). Samples near the water surface can be collected by simply submerging a clean bottle in water. Deeper samples need to be collected using a tube with an intake at the sampling depth connected to a pump or plastic syringe, or using a van Dorn, Kemmerer, or other type of sampler (see Ward and Harr 1990). In water up to approximately 1 m in

depth, a sample can be collected from the entire water column using a cylinder inserted vertically into the wetland water (Swanson 1978). If $C_P P$ is a significant component of tracer input, then a specialized sampling device must be used to collect atmospheric deposition (see Allan (2004) for specification). If a specialized device is not available, it may be better to use published data as a proxy rather than using erroneous data collected via inappropriate methods. Precipitation chemistry data are available from national and international atmospheric deposition monitoring networks, such as Global Atmospheric Watch (http://gaw.empa.ch/gawsis/).

Temporal and spatial variability in C_{Si} and C_{Gi} need to be properly represented by collecting multiple samples over time. If there are multiple inflow streams, samples should be collected at all streams and volume-weighted averages of concentrations should be used in the mass-balance equation. Due to geologic heterogeneity, solute concentrations in groundwater may have large spatial variability even within a relatively small area. Therefore, it is necessary to collect groundwater samples from several locations (and depths) in the areas of anticipated groundwater discharge. Water samples usually are collected directly from monitoring wells and piezometers, or springs; they also can be extracted from sediment core samples (see Adams (1994) for methods). Wells should be bailed or pumped prior to sample collection to ensure that the sample represents the composition of groundwater in the aquifer surrounding the well screen. A common purging protocol is to pump 3–5 times the volume of water in the well to ensure that the stagnant water in the casing has been completely removed prior to sample collection. Another option is to slowly pump water through an intake tube that is placed in the screened interval of the well (low-flow sampling method). The pumping rate needs to be slow enough that virtually no drawdown occurs in the well, in which case nearly all of the water supplied during pumping originates from the aquifer. Detailed procedures for groundwater sampling are found in manuals and handbooks on this subject (e.g., Yeskis and Zavala 2002; Wilde 2006). The number of groundwater samples commonly used is insufficient to determine the precise value of an average tracer concentration, $<C_{Gi}>$. Therefore, a recommended practice is to use one standard deviation from the arithmetic mean of all groundwater samples to represent the uncertainty in the mass-balance calculation.

3.10.3 Use of Multiple Tracers

If more than one conservative tracer is available for water-budget determinations, and their concentrations are not correlated, then Eq. 3.51 can be written for each individual tracer. This increases the number of equations and thus, the number of unknowns that can be determined. For example, using naturally occurring chloride and artificially introduced bromide, a set of three equations can be solved for three unknowns. Alternatively, the water-budget equation is solved separately with each mass-balance equation to provide separate estimates of the same unknowns. If resulting values using separate tracers are greatly different, then possible errors in estimation or measurement of known terms or missing terms in the equations are indicated.

The mass balance of a conservative tracer can be used with a potentially reactive tracer to identify a possible reaction and estimate its rate. The mass-balance equation for a reactive tracer can be written as

$$\Delta(C_w V)/\Delta t = C_P P + C_{Si} S_i - C_w(S_o + G_o) + C_{Gi} G_i + C_{of} O_f + R_{xn} \qquad (3.53)$$

where R_{xn} (kg/m^3/s) is the rate of reaction per volume. If all other terms in Eq. 3.53 are known from solving the water-budget equation and the mass balance equation for a conservative tracer, then R_{xn} can be determined as the residual of Eq. 3.53. For example, Heagle et al. (2007) solved the water-budget equation with the mass balance equations for naturally occurring chloride and sulphate to estimate the rate of sulphate reduction in a prairie wetland in Saskatchewan, Canada.

3.10.4 Final Remarks

The tracer mass-balance approach provides a useful tool for estimating the water-budget components that are difficult to measure directly. Unlike other methods for estimating groundwater flow, the mass-balance method evaluates the flow averaged over the entire wetland, while giving no information about the spatial distribution of groundwater recharge or discharge within the wetland. Therefore, it is beneficial to use this method in combination with other methods that give local values of flow, such as a seepage meter or mini-piezometer (see Sect. 3.8). The mass-balance method provides a constraint on the possible range of total groundwater flow, whereas a local-scale method is useful for delineating areas of focused recharge or discharge, which may have significant influence on the distribution of wetland flora and fauna (e.g., Rosenberry et al. 2000).

3.11 Estimation of Errors

As introduced in Sect. 3.2, a wetland water budget can be written as the change in wetland volume per time (plus residual) equal to the sum of all inputs and losses (Eq. 3.1). If all of the hydrological components are measured as accurately as possible, it is almost certain that the sum of those components will not equal the change in volume in the wetland over an accounting period. R in Eq. 3.1 can be disturbingly large relative to ΔV for some water budgets. Error stems from (1) incorrect measurement of a parameter (instrument error), (2) misapplying point measurements to specific areas or volumes of a wetland (a common but often neglected error), and (3) misinterpreting the hydrologic setting, usually by not

measuring one or more parameters that are important to the water or chemical budget. Positive errors often are offset by negative errors, so R commonly is smaller than the error associated with one or more individual terms. A first-order error analysis takes this into account when determining error as a function of multiple parameters. When errors are additive, first-order analysis (e.g., Taylor 1982) involves calculating the square root of the sum of the squared values of each of the parameters:

$$\delta = \sqrt{\delta_P^2 + \delta_E^2 + \delta_{Si}^2 + \delta_{So}^2 + \delta_{Gi}^2 + \delta_{Go}^2 + \delta_{Of}^2 + \delta_{\Delta V}^2} \tag{3.54}$$

where

δ = error,
P = precipitation,
E = evapotranspiration,
Si = surface-water flow to the wetland,
So = surface-water flow from the wetland,
Gi = ground-water discharge to the wetland,
Go = loss of wetland water to ground water,
Of = overland flow,
ΔV = change in volume of water contained in the wetland (positive for increase in volume).

First-order error analysis assumes that errors are independent and randomly distributed. This clearly is a poor assumption. For example, most of the water-budget parameters are dependent on precipitation. If substantial interdependence is suspected, a more rigorous analysis can be conducted where covariances between terms are considered (e.g., LaBaugh 1985). However, a large percentage of water-budget studies, if they present error estimations at all, simply assume parameters are independent and apply an equation similar to 3.54.

By estimating the error associated with each of the water-budget components of a wetland, cumulative error, δ, can be compared with the residual, R, of Eq. 3.53. If differences are large, it is likely that at least one of the components has been determined incorrectly or that errors associated with one or more of the water-budget components have been poorly estimated.

A common question among wetland scientists is just how large are these errors? Estimates vary substantially depending on the setting, goals of the study, and methods of measurements. Errors reported in a selection of publications that provide error estimates for water-budget components of studies of lakes, wetlands, and reservoirs generally are smallest for precipitation and largest for groundwater. Based on values presented in ten such studies in Table 3.1, median estimates for error associated with P, E, S, G, and ΔV are 9, 10, 10, 36, and 10 %, respectively.

3.12 Chapter Summary

After reading this chapter, the reader may come away with the thought that wetland hydrology is complex but there are many different approaches and tools that can be used, or that quantifying hydrologic fluxes in wetland settings is difficult and fraught with error. Either impression would be correct. Perhaps the most important conclusion is that the pursuit of parallel lines of evidence, using multiple methods for achieving the same goals, will lead to a better understanding of these complex processes and a more accurate assessment of the various hydrologic components that constitute the hydrologic setting of a wetland. Armed with the numerous methods at our disposal, and knowledge of the various sources and magnitudes of error associated with each approach, the wetland hydrologist can feel comfortable in pursuing quantification of the various hydrological components with a judicious selection of methods appropriate to the goals and budget associated with the investigation.

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Student Exercises

Classroom Exercises

Short Exercise 1: Converting Pressure to Water Depth and Stage

Measuring wetland stage and hydraulic head, and determining direction and potential for flow between groundwater and surface water, are among the most basic requirements in wetland hydrology. A sketch of a common monitoring installation appears below (Fig. 3.34). A piezometer designed to indicate hydraulic head beneath the wetland bed is instrumented with a submersible pressure transducer. The sensor is suspended from the surface of the well casing by a metal wire. The distance from the attachment point to the sensor port commonly is described as the hung depth. This particular type of sensor stores the data on a circuit card; the sensor must be retrieved and the data downloaded periodically. Some installations instead have a data cable extending from the sensor to a datalogger that can query and store data from multiple sensors. In some models the cable contains a vent tube that allows changes in atmospheric pressure to be transmitted to the pressure sensor. Venting allows the pressure measurement to be relative to atmospheric pressure. The transducer in this example is not vented to the atmosphere; some would argue this is preferable because there is no associated opportunity for water vapor to reach and damage the sensor electronics. However, without

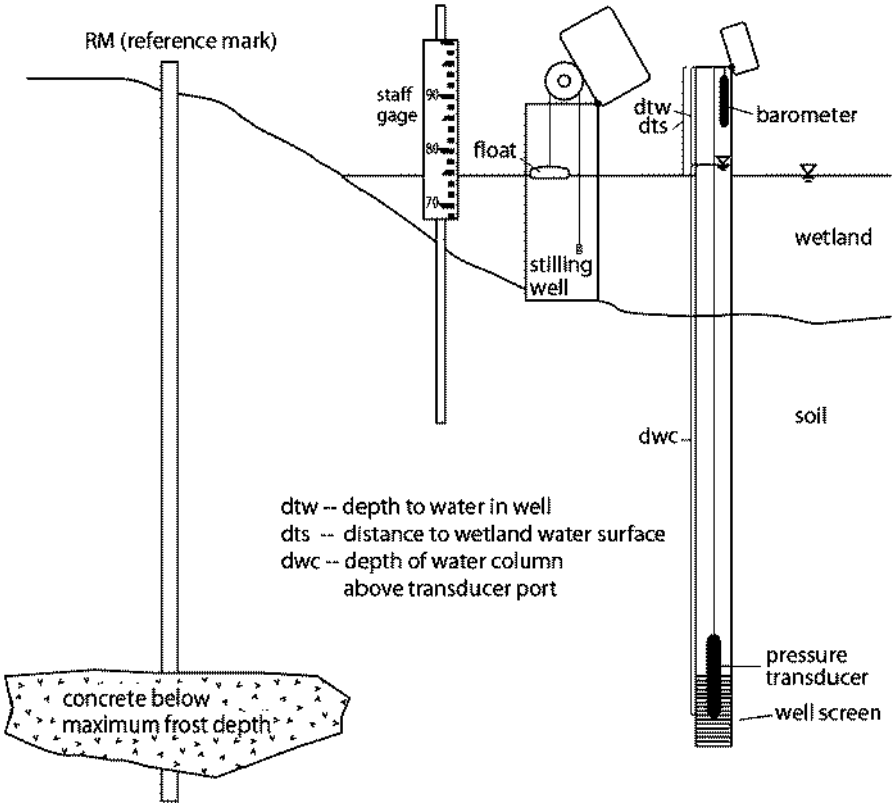


Fig. 3.34 Installations commonly used to determine wetland stage, elevation, and vertical hydraulic-head gradient

venting, the sensor output is the sum of hydrostatic pressure of the water column above the sensor port (the dwc or depth of the water column that we want to know) and atmospheric pressure. Therefore, atmospheric pressure needs to be measured and subtracted from the output of the submerged pressure transducer to obtain the height of the water column above the submerged sensor. A barometer is suspended in the piezometer casing, well above the water level, to provide atmospheric-pressure measurements. If the well is susceptible to occasional flooding, the barometer could instead be located anywhere nearby as atmospheric pressure does not change appreciably over distances of several km.

Output from pressure transducers, as well as many other sensors, commonly is converted to units in which field check measurements are made. In wetland settings, that unit usually is feet or meters of water head. Meters will be used here. To convert output in pressure to head, recall that Pressure $= \rho g h$ where ρ is density of water (kg m^{-3}), g is acceleration due to gravity (m s^{-2}), and h, hydraulic head, is the height to of a column of liquid that would exert a given pressure, in m. Output from pressure transducers commonly is in units of Pascals. Recall that a Pascal is a

Newton per square meter and that a Newton, a unit of force, is determined in terms of mass times acceleration (kg m s^{-2}). Therefore,

$$h = \frac{P_{trans} - P_{bar}}{\rho g} + Offset \tag{3.55}$$

where P_{trans} is the output from the submerged pressure transducer, P_{bar} is the output from the barometer, and *Offset* is a value that equates the sensor output to a local datum or reference elevation.

A stilling well also is displayed in the drawing. Although another submerged pressure transducer could have been used to indicate wetland stage, this stilling well contains a float and counterweight that together rotate a pulley connected to a potentiometer or pulse-counting device. As water level changes, the float moves and the pulley rotates, changing either the electrical resistance if the sensor is a potentiometer, or causing electrical pulses to be sent to a data recording unit if the sensor is a pulse-counting device (often called a shaft encoder). The output of the sensor in the stilling well commonly is set to be equal to the water level indicated by a nearby staff gage.

The staff gage is connected to a metal pipe driven into the wetland bed. This simple device is designed to provide a direct indication of the relative stage of the wetland. The units on the "staff plate" in this example are in meters, but units of feet are perhaps more common in the US. Some wetland sediments are relatively soft, and some wetlands freeze during winter, providing the potential for the staff gage to move over time. To determine whether this occurs or not, we need a stable reference point to which the staff gage can be compared; hence, the reference mark, commonly called an RM. The term RM is used so as to not confuse it with BM (bench mark), which is an official surveying location that is part of a national geodetic survey. This particular RM consists of a pipe that extends into the ground. However, in areas where soil frost is common and can extend a meter or more beneath ground surface, pipes also can move. Therefore, this particular RM was set in a mass of concrete that was installed beneath the deepest expected extent of soil frost.

Our tasks here are to:

1. compare the potentiometer output from the stilling well to the output from the submerged pressure transducer in common units,
2. make separate measurements of water levels inside of the well and of the wetland surface,
3. determine the difference in hydraulic head (Δh) between the wetland and the piezometer, and
4. verify that our sensors are providing the correct output.

Field site data

Staff gage	0.750 m (manually read)
Potentiometer	0.755 m
dts	0.198 m (manually measured)
dtw	0.178 m (manually measured)
Barometer	100.510 kPa
Pressure transducer	110.610 kPa
Pressure-transducer offset	−0.250 m

1. What is the dwc in m of water? Assume fresh water at 20 °C. (therefore, density = 998 kg/m^3)_____
2. What does the pressure transducer indicate for head in the piezometer in m relative to the local datum?_____
3. What do the sensors indicate for Δh?_____
4. What is the manually measured Δh?_____
5. Is the potential for flow upward or downward based on the measured values? _____
6. How does the Δh indicated by the sensors differ from the Δh calculated from the manual measurements?_____
7. What is the gradient assuming the midpoint of the well screen is 0.75 m below the wetland bottom?_____
8. If the top of the staff gage plate is at an elevation of 102.550 m, what is the elevation of the water level inside of the piezometer?_____

Short Exercise 2: Wind Correction of Precipitation Data

Table 3.3 shows daily mean air temperature and wind speed, and daily total precipitation recorded by a weighing precipitation gauge with an Alter wind shield (similar to Fig. 3.5a), at a hydrological research station in Calgary, Alberta, Canada, in 2008. There were two precipitation events, on December 7 and 12.

1. Based on the air temperature, determine the form of precipitation (rain or snow).
2. If the precipitation occurs as snow, then a correction must be made to account for the gage-catch deficiency (see Fig. 3.6). Use the following equation (Dingman 2002:111–112) to compute the catch deficiency factor (CD) from wind speed (u, m s^{-1}) for each day.

$$\mathrm{CD} = 100 \exp\left(-4.61 - 0.036u^{1.75}\right) \tag{3.56}$$

3. Divide the uncorrected precipitation by CD to estimated true (i.e., corrected) precipitation.
4. Calculate the total of two precipitation events for both uncorrected and corrected data. What is the degree (percentage) of underestimate by not correcting the data?
5. Many winter precipitation data sets available on the internet have not been corrected. Discuss the potential problem of using such data for a water-budget analysis.

Table 3.3 Daily mean air temperature and wind speed, and daily total precipitation

Date	Air temp. (°C)	Wind spd. (m s^{-1})	Recorded pcp. (mm)	CD	Corrected pcp. (mm)
Dec. 7	−1.4	1.7	13	___	___
Dec. 12	−5.4	3.7	17	___	___
Total			___		___

Short Exercise 3: Spatial Interpolation of Precipitation Data

Table 3.4 shows monthly total precipitation (mm) at three meteorological stations in Alberta, Canada. Olds Station is located between two other stations, approximately 50 km south of Red Deer and 70 km north of Calgary. The first three columns list the long term average for 1971–2000; the last three columns list the data recorded in 2010. The 2010 data for Olds are missing.

1. Using the normal ratio method (Eq. 3.7), estimate monthly total precipitation in Olds for the three missing months.
2. Actual precipitation data recorded at the Olds station were 77 mm for June, 85 mm for July, and 79 mm for August. Discuss the magnitude of uncertainty associated with this method.

Table 3.4 Long-term average monthly precipitation and 2010 monthly precipitation (mm) at three meteorological stations in Alberta, Canada

	1971–2000 average			2010		
	June	July	Aug.	June	July	Aug.
Red Deer	84	92	70	138	144	62
Calgary	80	68	59	64	66	87
Olds	90	87	65	___	___	___

Data source: Environment Canada National Climate Data and Archive (http://climate.weatheroffice.gc.ca/climateData/canada_e.html)

Short Exercise 4: Calculation of Discharge from Tracer Data

Tracer dilution methods were used to estimate the discharge of two small streams flowing into a wetland. The constant injection method was used in the first stream, where chloride solution having a concentration of 60 g L^{-1} was injected at a rate of 12 L min^{-1}. The tracer concentration in the stream reached a steady value of 100 mg L^{-1} by 150 s after the start of injection (Fig. 3.35). The background chloride concentration in the stream was 1 mg L^{-1}.

1. Using Eq. 3.23, estimate the stream discharge from concentration data.

   The slug injection method was used in the second stream, where 10 L of tracer solution containing 3 kg of chloride mass was instantaneously injected in the stream. The tracer concentration reached a peak about 40 s after the release and declined quickly afterwards (Fig. 3.35). The background chloride concentration in the stream was 2 mg L^{-1}.
   Concentration data are listed in Table 3.5.
2. Using Eq. 3.25 with $\Delta t = 10$ s, estimate the integral in the denominator of Eq. 3.24.
3. Using Eq. 3.24 with $C_1V_1 = 3$ kg, estimate the stream discharge.

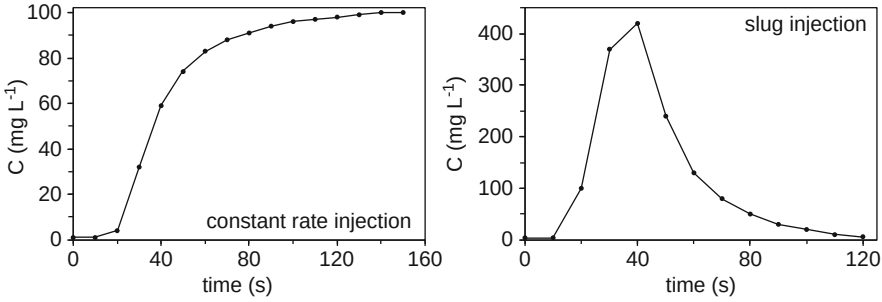


Fig. 3.35 Concentration of chloride tracer in streams. *Left*: constant-rate injection test. *Right*: slug injection test

Table 3.5 Data for slug injection test

t (s)	C (mg L^{-1})	$(C - C_b)\Delta t$ (kg m^{-3} s)	t (s)	C (mg L^{-1})	$(C - C_b)\Delta t$ (kg m^{-3} s)
0	2	_____	70	80	_____
10	2	_____	80	50	_____
20	100	_____	90	30	_____
30	370	_____	100	20	_____
40	420	_____	110	10	_____
50	240	_____	120	4	_____
60	130	_____		Total =	_____

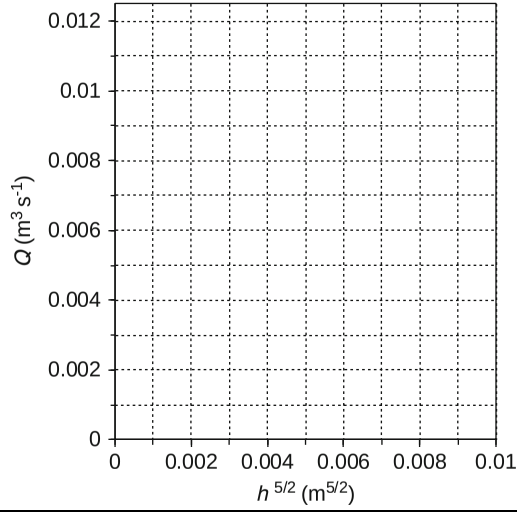
Short Exercise 5: Calibration of Weir Coefficient

V-notch weirs provide stable and reliable flow measurements, particularly when the coefficient C in the weir formula (Eq. 3.28) is determined to reflect site-specific conditions. Table 3.6 lists measurements of water level (h) and discharge (Q) for the V-notch weir shown in Fig. 3.11b. The water level is measured with respect to the base of the weir. Therefore, $h_0 = 0$ in Eq. 3.28.

1. Compute $h^{5/2}$ and convert Q to $m^3\ s^{-1}$.
2. Plot $h^{5/2}$ and Q in the graph and determine the slope of the plot.
3. Determine C in Eq. 3.28. Note that $\theta = 90°$; thus, $\tan(\theta/2) = 1$. Compare this value to the theoretical value for an ideal weir, $C = 1.38$.

Table 3.6 Water level (h) in a 90° V-notch weir and independently measured discharge (Q) in the weir shown in Fig. 3.11

Date	h (m)	Q (L s^{-1})	$h^{5/2}$ ($m^{5/2}$)	Q (m^3 s^{-1})
Jun. 18, 2009	0.055	0.8	_____	_____
Aug. 7, 2009	0.096	3.6	_____	_____
Oct. 30, 2009	0.114	5.5	_____	_____
May 28, 2010	0.144	9.7	_____	_____
Jul. 8, 2010	0.156	12.1	_____	_____
Oct. 5, 2010	0.132	7.5	_____	_____



Short Exercise 6: Determination of Stage-Discharge Rating Curve

Coefficients for the stage-discharge rating curve (Eq. 3.26) of a stream gauging station can be determined from a series of measurements of stage (h) and discharge (Q) encompassing different flow conditions. Table 3.7 lists the measured h and Q in a small stream in Calgary, Alberta, Canada. The stage at zero flow (h_0) is 0.35 m at this gauging station. Equation 3.28 can be written in a logarithmic form

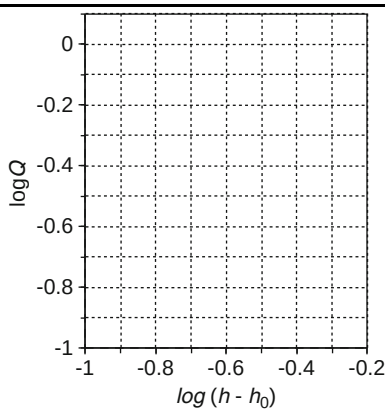
$$\log Q = \log a + m \log(h - h_0) \tag{3.57}$$

When the logarithms of data are used to fit a straight line, the intercept and slope of the line give log$a$ and m, respectively.

1. Compute $\log(h - h_0)$ and log$Q$ for each measurement.
2. Plot $\log(h - h_0)$ and log$Q$ in the graph and fit a straight line.
3. Determine the intercept and the slope of the plot, and compute a and m.

Table 3.7 Water stage (h) and discharge (Q) measured in a small stream near Calgary, Alberta, Canada in 2011

Date	h (m)	Q (m^3 s^{-1})	$\log(h - h_0)$	$\log Q$
June 9	0.65	0.56	_____	_____
June 14	0.59	0.46	_____	_____
June 21	0.88	1.10	_____	_____
June 28	0.59	0.45	_____	_____
July 6	0.50	0.27	_____	_____
July 13	0.53	0.29	_____	_____
July 26	0.52	0.24	_____	_____
Aug. 8	0.47	0.15	_____	_____
Aug. 24	0.44	0.11	_____	_____



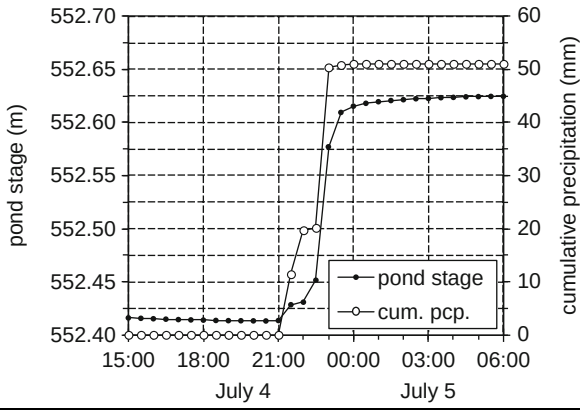
Short Exercise 7: Estimation of Diffuse Overland Flow

The amount of diffuse overland flow can be estimated using a wetland as a natural overland flow trap. If the wetland does not have inflow or outflow streams, and the contribution of groundwater flow is negligible during a short-duration storm, then the water balance equation for the wetland pond is given by Eq. 3.32. Total overland flow during the storm (O_{ftot}) is estimated from measuring the volume of pond water before (V_{ini}) and after (V_{fin}) the storm. The figure embedded in Table 3.8 shows the pond stage and cumulative precipitation in Wetland 109 in the St. Denis National Wildlife Area in Saskatchewan, Canada, on July 4–5, 1996 (see Hayashi et al. 1998 for a site description). The cumulative precipitation (p_{cum}) during the entire storm was 51 mm. The pond stages recorded at 21:00 and 02:00 are listed in Table 3.8. Water depth (H) at the deepest point in the pond is given by subtracting 551.68 m from the pond stage. The area of pond surface (A) and the volume of pond water (V) can be estimated using Eqs. 3.4 and 3.5 with $s = 3{,}180$ m^2 and $p = 1.61$ (Hayashi and van der Kamp 2000). The effective drainage area (A_{eff}) of Wetland 109 is 20,100 m^2.

1. Calculate the initial (21:00) and final (02:00) pond area and volume from the stage data.
2. Calculate the total amount of precipitation (P_{tot}) falling within the pond by multiplying p_{cum} by the pond area (A_{fin}) at 02:00.
3. Using Eq. 3.32, determine O_{ftot}.
4. Runoff-contributing area to the pond is given by $A_{eff} - A_{fin}$. From O_{ftot}, estimate the areal average runoff (mm) in the contributing area.
5. Estimate the runoff coefficient (R_c = runoff/precipitation) for this storm.

Table 3.8 Pond stage in Wetland 109 in the St. Denis National Wildlife Area, Saskatchewan, Canada on July 4, 1996

Time	Stage (m)	H (m)	A (m^2)	V (m^3)
21:00	552.41			
02:00	552.62			



Short Exercise 8: Calculation of Groundwater Flow Using the Segmented-Darcy Method

The segmented-Darcy approach shown in Fig. 3.21 provides values for Q_{In} and Q_{Out} that are based on data from monitoring wells and wetland stage. The figure below (Fig. 3.36) is identical to Fig. 3.21 but heads for three of the wells are changed slightly. Use the data shown in Fig. 3.36, along with the assumptions that K is 30 m/day and b is 20 m, to fill out the data in Table 3.9. Sum the positive values to determine Q_{In} and sum the negative values to determine Q_{Out}. Then answer the following questions.

1. Where is the greatest rate of exchange (Q/A) between groundwater and the wetland? Why?
2. A hinge line is a point along a shoreline that separates a shoreline reach where groundwater discharges to the wetland from a shoreline reach where wetland water flows to the groundwater system. What are the approximate locations of the hingelines?
3. If there is no surface-water exchange with the wetland, and overland flow is negligible, what does this analysis tell you about the other terms of the water budget?

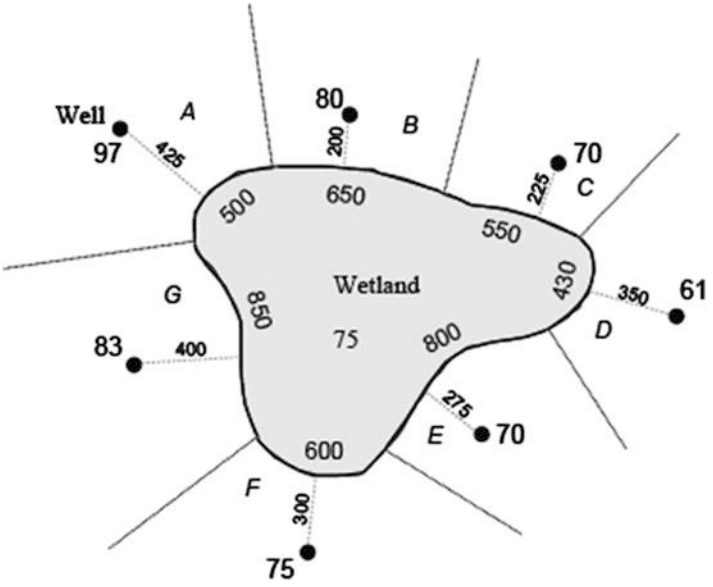


Fig. 3.36 The same wetland setting shown in Fig. 3.21 but with several different head values. Figure legend is shown in Fig. 3.21

Table 3.9 Parameters needed to determine Q_{In} and Q_{Out} using the segmented-Darcy approach

Watershed segment	Horizontal hydraulic conductivity (K), in m/d	Effective thickness of the aquifer (b),in m	Hydraulic head in well minus surface-water stage (h_1–h_2), in m	Distance from the well to the shoreline (L), in m	Length of shoreline segment (m), in m	Water flow (Q), in m^3/d
A						
B						
C						
D						
E						
F						
G						
					ΣQ_{In} =	
					ΣQ_{Out} =	
					In - Out =	
					% imbalance =	

Short Exercise 9: Simple Flow-Net Analysis

We do not need a sophisticated numerical model to give us a good first estimate of groundwater flows to and from wetlands. Reasonable values for exchange between groundwater and a wetland can be calculated with: (1) a map showing the locations of a few monitoring wells and their hydraulic-head values, (2) a value for stage of the wetland, and (3) estimates of hydraulic conductivity. In this brief exercise you will make a flow-net analysis to determine flow between groundwater and a wetland and also compare those values with values that were obtained with the segmented-Darcy approach in short exercise SE 8.

The flow-net analysis is a graphical approach for determining 2-dimensional groundwater flow. The Darcy equation is used to solve for flow through individual "stream tubes" that are drawn based on contour lines drawn from head data. The method assumes steady-state flow is two-dimensional. The flow net can be drawn in plain view, as we did with SE 8, or in cross-sectional view. We will assume that the aquifer is homogeneous and isotropic, although modifications can be made when drawing the flow net if the aquifer is known to be anisotropic. A brief description of how to draw a flow net follows. More detail can be found in Fetter Jr. (2001) and Cedergren (1997).

A flow net consists of equipotential lines (contour lines of equal hydraulic head) that are drawn perpendicular to flow lines that indicate the direction of groundwater flow. The net is bounded by no-flow boundaries or constant-head boundaries. The equipotential lines intersect no-flow boundaries at right angles and the flow lines intersect constant-head boundaries, if present, also at approximately right angles. A simple example is shown in Fig. 3.37. Equipotential head drops consist of the area

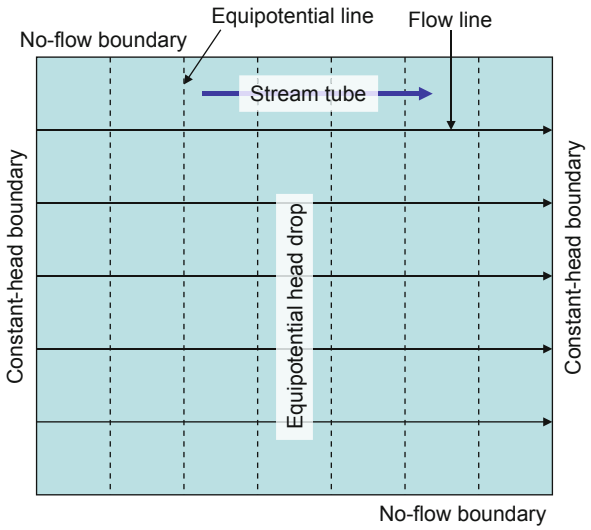


Fig. 3.37 Diagram of a simple rectangular flow net showing boundary conditions, equipotential lines, and stream tubes

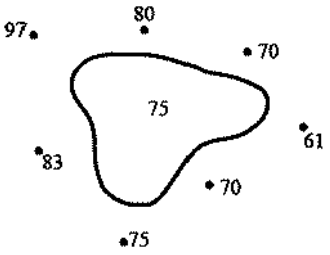


Fig. 3.38 Draw contour lines based on the heads displayed at the monitoring wells and the wetland stage

of the flow net bounded by adjacent equipotential lines and stream tubes consist of the area of the flow net bounded by adjacent flow lines.

The example in Fig. 3.37 contains seven equipotential head drops and six stream tubes. The flow-net equation can be written as

$$Q = \frac{MKbH}{n} \tag{3.58}$$

where M is the number of stream tubes, n is the number of equipotential head drops, K is the assumed hydraulic conductivity, b is the sediment thickness in the third dimension, and H is the total head drop across the flow net. M is commonly presented as m in most texts, but we use upper-case M here to distinguish it from m, the shoreline length presented earlier in Fig. 3.21. Q is in units of volume per time.

Some basic steps to follow are:

1. Determine boundaries and boundary conditions,
2. Draw equipotential lines by contouring head data from wells and wetland stage,
3. Draw flow lines to create approximate squares (you should be able to draw a circle bounded by the equipotential lines and flow lines),
4. Flow lines cross equipotential lines at right angles (assuming we have isotropic conditions) and flow lines also intersect constant-head boundaries at right angles,
5. You can draw half-equipotential lines for areas with smaller gradients.
6. Five to ten flow lines usually are sufficient,
7. Count up stream tubes and equipotential drops to determine M and n,
8. Determine H, and estimate b and K.
9. Calculate Q for flow to and/or from the wetland.

Let's see how well this can work. The same wetland setting in Short Exercise 8 is displayed in Fig. 3.38. This is the same wetland shown in Fig. 3.21 but with head values changed for three of the seven wells. Your task will be to determine the extent to which changes in head will affect the interpretation of flow of groundwater to and from the wetland. Draw contour lines based on the head data and then draw flow lines based on the instructions provided above. After that, you will count up flow tubes and head drops and calculate flow to the wetland and flow from the wetland. Use K and b values from Short Exercise 8. You will then be able to answer the following questions:

1. How does flow to the wetland compare to flow from the wetland? If the values are different, why are they different?
2. How do the values for flow to the wetland and flow from the wetland compare to those you obtained with the segmented-Darcy approach? Which method do you prefer? Which method provides more realistic results? What might be sources of error for both methods?
3. How do the flowlines you have drawn compare with the flowlines shown in Fig. 3.22? What effect do the different head values have on the positioning of the hinge lines?

References

Cedergren HR (1997) Seepage drainage and flow nets, 3rd edn. Wiley, New York

Fetter CW Jr (2001) Applied hydrogeology, 4th edn. Prentice Hall, Upper Saddle River

Short Exercise 10: Measurement of Groundwater Flow Using a Half-Barrel Seepage Meter

Seepage meters were used to quantify rates and distribution of exceptionally fast flow through a lake bed (Rosenberry 2005). In this exercise you will use data from that report to determine groundwater-surface-water exchange and also compare standard flow measurements with those based on connecting multiple seepage cylinders to a single seepage bag.

Mirror Lake is a small, 10-ha lake in the White Mountains of New Hampshire. A dam built in 1900 raised the lake level by about 1.5 m, increasing the lake surface area and inundating what had previously been dry land. Water leaks out of the lake through a portion of the southern shoreline that, because of the stage rise following dam construction, has been covered by water for only about 110 years. More water is lost as seepage to groundwater than from the lake surface-water outlet (Rosenberry et al. 1999). Seepage meters were used to determine where rapid rates of seepage were occurring and to determine the rates of seepage from the lake to groundwater.

Data shown in Table 3.10 were collected from 18 seepage meters that were installed in the area shown in Fig. 3.39. The photo inset shows the locations of some of the seepage cylinders that were installed prior to the installation of seepage bags and associated bag-connection hardware. Most of the measurements were made from standard seepage meters similar to Fig. 3.25. However, two sets of measurements were made from four seepage cylinders that were all connected (ganged) to one seepage bag. Your task is to fill in the missing data in Table 3.10 for meters 3 and 13 and then answer the following questions. To convert from ml/min to cm/day you will assume that 1 ml = 1 cm^3 of water. You will divide your result in cm^3/min by the area covered by the seepage cylinder (2,550 cm^2) and then multiply by the number of minutes in a day to obtain units in cm/day.

1. What are the averages of seepage measurements made at each of meters 3, 4, 5, and 6? Values for 4, 5, and 6 are already provided. What is the range in seepage rates at these 4 m? How does the variability in seepage among these 4 m compare with the ranges of values at each meter based on repeat measurements?
2. Repeat this analysis for meters 13, 17, 18, and 20. How do these seepage rates compare with meters 3 through 6? How does the range in seepage among meters compare with the ranges of measurements at individual meters?
3. Calculate average values for the two sets of ganged measurements (13, 17, 18, 20 and 3, 4, 5, 6). How do these values compare with the sums of seepage rates based on measurements made at individual meters? What can you say about summed versus ganged measurements for areas of slow versus fast seepage?

Table 3.10 Values collected from Mirror Lake, NH, during July 16–18, 2002

Seepage measurements at Mirror Lake, Campton, New Hampshire										
Dates: 7/16/02–7/19/02				Meter area = 2,550 cm^2						
Date	Meter	V1 (ml)	V2 (ml)	T1	T2	ΔV (ml)	Δt (min)	ΔV/Δt (ml/min)	ΔV/Δt (cm/d)	Ave.
7/16/02	3	1,000	675	9:26	9:42					
7/16/02	3	1,000	200	9:46	10:28					
7/16/02	3	1,000	860	13:14	13:21					
7/16/02	3	1,000	720	16:07	16:21					
7/16/02	3	1,000	650	16:29	16:47					
7/16/02	4	1,000	910	9:17	9:30	−90	13	−6.9	−3.9	
7/16/02	4	1,000	780	9:35	10:04	−220	29	−7.6	−4.3	
7/16/02	4	1,000	710	10:24	11:07	−290	43	−6.7	−3.8	
7/16/02	4	1,000	540	11:08	12:12	−460	64	−7.2	−4.1	−4.2
7/16/02	4	1,000	625	12:13	12:57	−375	44	−8.5	−4.8	
7/16/02	4	1,000	910	14:55	15:06	−90	11	−8.2	−4.6	
7/16/02	4	1,000	910	16:07	16:24	−90	14	−6.4	−3.6	
7/16/02	4	1,000	870	16:29	16:47	−130	18	−7.2	−4.1	
7/16/02	5	1,000	970	9:42	10:14	−30	32	−0.9	−0.5	
7/16/02	5	1,000	995	10:19	10:49	−5	30	−0.2	−0.1	
7/16/02	5	1,000	940	10:50	11:28	−60	38	−1.6	−0.9	
7/16/02	5	1,000	905	11:29	12:49	−95	80	−1.2	−0.7	−0.6
7/16/02	5	1,000	890	13:19	14:57	−110	98	−1.1	−0.6	
7/16/02	5	1,000	990	16:04	16:14	−10	10	−1.0	−0.6	
7/16/02	5	1,000	980	16:27	16:41	−20	14	−1.4	−0.8	
7/16/02	6	1,000	560	13:19	14:47	−440	88	−5.0	−2.8	
7/16/02	6	1,000	970	16:04	16:14	−30	10	−3.0	−1.7	−2.3
7/16/02	6	1,000	940	16:27	16:41	−60	14	−4.3	−2.4	
7/17/02	13	1,000	210	17:34	17:37					
7/17/02	13	1,000	465	17:43	17:45					
7/17/02	13	1,000	440	18:30	18:52					
7/18/02	17	1,000	800	9:45	9:51	−200	6	−33.3	−18.8	
7/18/02	17	1,000	830	10:09	10:13	−170	4	−42.5	−24.0	

(continued)

Table 3.10 (continued)

Seepage measurements at Mirror Lake, Campton, New Hampshire										
Dates: 7/16/02–7/19/02				Meter area = 2,550 cm^2						
Date	Meter	V1 (ml)	V2 (ml)	T1	T2	ΔV (ml)	Δt (min)	ΔV/Δt (ml/min)	ΔV/Δt (cm/d)	Ave.
7/18/02	17	1,000	840	10:24	10:29	−160	5	−32.0	−18.1	
7/18/02	17	1,000	930	10:49	10:51	−70	2	−35.0	−19.8	−20.4
7/18/02	17	1,000	940	15:14	15:16	−60	2	−30.0	−16.9	
7/18/02	17	1,000	960	16:31	16:32	−40	1	−40.0	−22.6	
7/18/02	17	1,000	920	16:36	16:38	−80	2	−40.0	−20.6	
7/18/02	18	1,000	280	9:48	9:56	−720	8	−90.0	−50.8	
7/18/02	18	1,000	550	10:09	10:13	−450	4	−112.5	−63.5	
7/18/02	18	1,000	490	10:24	10:29	−510	5	−102.0	−57.6	
7/18/02	18	1,000	770	10:49	10:51	−230	2	−115.0	−64.9	−53.5
7/18/02	18	1,000	510	10:58	11:03	−490	5	−98.0	−55.3	
7/18/02	18	1,000	840	15:19	15:21	−160	2	−80.0	−45.2	
7/18/02	18	1,000	920	16:31	16:32	−80	1	−80.0	−45.2	
7/18/02	18	1,000	840	16:36	16:38	−160	2	−80.0	−45.2	
7/18/02	20	1,000	900	11:05	11:07	−100	2	−50.0	−28.2	
7/18/02	20	1,000	800	11:26	11:30	−200	4	−50.0	−28.2	−28.9
7/18/02	20	1,000	900	15:08	15:10	−100	2	−50.0	−28.2	
7/18/02	20	1,000	945	16:08	16:09	−55	1	−55.0	−31.1	
7/18/02	13,17,18,20	1,000	670	15:27	15:28	−330	1	−330.0	−46.6	
7/18/02	13,17,18,20	1,000	650	15:32	15:33	−350	1	−350.0	−49.4	−47.6
7/18/02	13,17,18,20	1,000	670	15:37	15:38	−330	1	−330.0	−46.6	
7/16/02	13,17,18,20	1,000	660	15:41	15:42	−340	1	−340.0	−48.0	
7/16/02	3,4,5,6	1,000	670	15:17	15:27	−330	10	−33.0	−4.7	
7/16/02	3,4,5,6	1,000	670	15:46	15:56	−330	10	−33.0	−4.7	
7/17/02	3,4,5,6	1,000	680	16:56	17:06	−320	10	−32.0	−4.5	−4.5
7/17/02	3,4,5,6	1,000	680	17:07	17:17	−320	10	−32.0	−4.5	
7/17/02	3,4,5,6	1,000	690	18:14	18:24	−310	10	−31.0	−4.4	

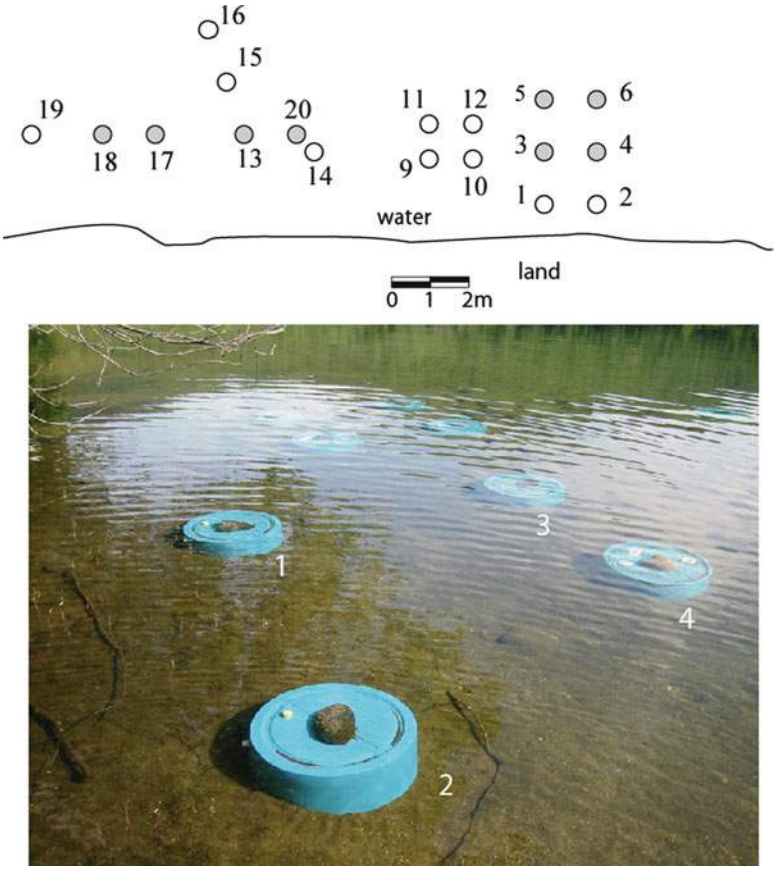


Fig. 3.39 Distribution of seepage meters installed in Mirror Lake, New Hampshire, USA. Seepage cylinders that were ganged for a single, integrated measurement are shown by *shaded circles*. *Numbers* in the photo inset correspond to the numbered seepage meters in the drawing. Note the rocks positioned on top of the seepage cylinders to counteract the buoyancy of the plastic cylinders, and that bag shelters have not yet been attached to the seepage cylinders

References

Rosenberry DO (2005) Integrating seepage heterogeneity with the use of ganged seepage meters. Limnol Oceanogr Methods 3:131–142

Rosenberry DO, Bukaveckas PA, Buso DC, Likens GE, Shapiro AM, Winter TC (1999) Migration of road salt to a small New Hampshire lake. Water Air Soil Pollut 109:179–206

Short Exercise 11: Estimation of Seepage Flux Using Temperature Data

Diurnal oscillation of temperature in wetland-bed sediments can be used to estimate groundwater seepage flux based on mathematical analysis of vertical heat transfer. When the temperature at the sediment-water interface oscillates in a sinusoidal manner with a fixed period (τ), (here we will assume 1 day), and amplitude A_0 (°C), then the temperature T (°C) of the sediment at depth z (m) is given by:

$$T(z,t) = T_m(z) + A_0 \exp(-az)\sin(2\pi t/\tau - bz) \tag{3.59}$$

where $T_m(z)$ is the time-averaged temperature profile representing the effects of a long-term temperature gradient, t is time, and a (m^{-1}) and b (m^{-1}) are constants defined by the thermal properties of the sediment and the magnitude and direction of seepage flux (Stallman 1965, equation 4; Keery et al. 2007, equation 2).

Equation 3.59 indicates that the amplitude of oscillation decreases with depth, and the phase delay of the sinusoidal signal increases with depth. Both amplitude and phase delay are dependent on the thermal properties of the saturated sediment and seepage flux. Suppose that the data recorded at two temperature sensors located at depth z_1 and z_2 ($z_1 < z_2$) have amplitudes of A_1 and A_2, and a phase shift (i.e., time difference of peak temperatures between two depths) of Δt (s). Seepage flux q ($m\ s^{-1}$) is positive for downward seepage in this example, which is the opposite of its definition elsewhere in this chapter. Seepage is defined this way in this exercise to be consistent with the construct used by Keery et al. (2007). Seepage flux is related to temperature amplitude by (Keery et al. 2007):

$$\frac{H^3 D}{4(z_2 - z_1)} q^3 - \frac{5H^2 D^2}{4(z_2 - z_1)^2} q^2 + \frac{2HD^3}{(z_2 - z_1)^3} q + \left(\frac{\pi^2 c^2 \rho^2}{{\lambda_e}^2 \tau^2} - \frac{D^4}{(z_2 - z_1)^4} \right) = 0 \tag{3.60}$$

where c ($J\ kg^{-1}\ °K^{-1}$) and ρ ($kg\ m^{-3}$) are the specific heat capacity and density, respectively, of bulk sediment, λ_e is the effective thermal conductivity of bulk sediment, and c_w ($J\ kg^{-1}\ °K^{-1}$) and ρ_w ($kg\ m^{-3}$) are the specific heat capacity and density, respectively, of water. In addition,

$$H = c_w \rho_w / \lambda_e \quad \text{and} \quad D = \ln(A_1/A_2) \tag{3.61}$$

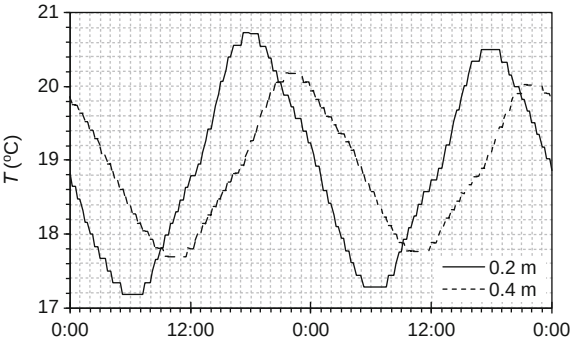
It also follows that the magnitude of q is related to Δt by (Keery et al. 2007):

$$|q| = \sqrt{\frac{c^2 \rho^2 (z_2 - z_1)^2}{\Delta t^2 {c_w}^2 {\rho_w}^2} - \frac{16\pi^2 \Delta t^2 {\lambda_e}^2}{\tau^2 (z_2 - z_1)^2 {c_w}^2 {\rho_w}^2}} \tag{3.62}$$

Therefore, q can be estimated from the analysis of temperature signals using Eqs. 3.60, 3.61 and 3.62.

Table 3.11 Temperature measured in sandy sediments underlying a wetland at depths of 0.2 m and 0.4 m over a period of 2 days

	0.2 m	0.4 m
T_{max} Day 1		
T_{min} Day 1		
T_{max} Day 2		
T_{min} Day 2		
Peak time Day 1		
Peak time Day 2		
Amplitude, 0.2 m =	(°C)	
Amplitude, 0.4 m =	(°C)	
Δt = (h)	(s)	



Accurate estimates of q using this method requires pre-processing the signals using Fourier transform or a dynamic harmonic regression algorithm (Keery et al. 2007; Gordon et al. 2012). In this exercise, a simple graphical technique is used for demonstration purposes.

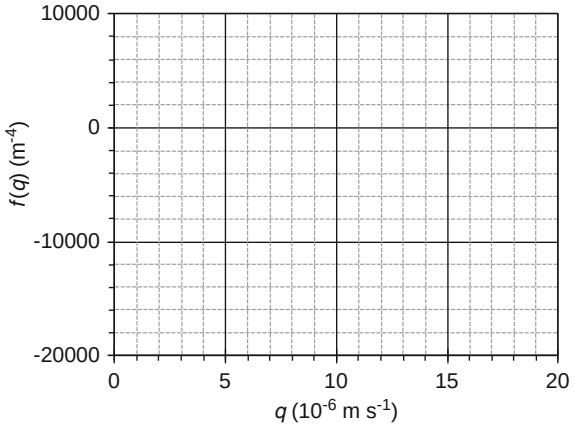
The figure embedded in Table 3.11 shows the temperature data collected in sandy sediments underlying a wetland.

1. Record the maximum and minimum temperature recorded on Day 1 for the 0.2 and 0.4 m sensor depths and enter the values in Table 3.11. Repeat the procedure for Day 2.
2. Record the time of peak temperature on Day 1 at 0.2 and 0.4 m depths and enter the values in the table. Repeat the procedure for Day 2.
3. Estimate the average amplitude of temperature oscillation by calculating $(T_{max} - T_{min})/2$ and taking the average of the 2 days.
4. Estimate the average phase shift Δt by calculating the difference in peak time for each day and taking the average of the 2 days.
5. Calculate D and H in Eq. 3.61 assuming: $c_w = 4{,}160$ J kg^{-1} °K^{-1}, $\rho_w = 1{,}000$ kg m^{-3}, and $\lambda_e = 2.0$ W m^{-1} °K^{-1}.
6. Calculate all constants in Eq. 3.60 assuming $c = 1{,}400$ J kg^{-1} °K^{-1}, $\rho = 2{,}000$ kg m^{-3}. Note that the period of oscillation τ is 86,400 s (24 h).
7. Solve Eq. 3.60 for q. The third-order polynomial equation has three roots, but only one is a real number. Various numerical tools are available; for example, MATLAB[2] software or its freeware equivalents have a line command for solving polynomial equations. The solution also can be obtained graphically by treating the left hand side of Eq. 3.60 as a polynomial function $f(q)$ and

[2] Any use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

plotting $f(q)$ against q on the graph below. Starting with $q = 1 \times 10^{-6}\text{ms}^{-1}$, keep plotting $f(q)$ for increasing values of q until $f(q) = 0$ is reached, which is the solution. A positive value of q indicates downward flow, and a negative value upward flow.

8. Calculate the magnitude of q using Eq. 3.62 and check the consistency of the values calculated from Eqs. 3.60 and 3.62.



References

Gordon RP, Lautz LK, Briggs MA, McKenzie JM (2012) Automated calculation of vertical pore-water flux from field temperature time series using the VFLUX method and computer program. J Hydrol 420–421:142–158

Keery J, Binley A, Crook N, Smith JWN (2007) Temporal and spatial variability of groundwater–surface water fluxes: Development and application of an analytical method using temperature time series. J Hydrol 336:1–16

Stallman RW (1965) Steady one-dimensional fluid flow in a semi-infinite porous medium with sinusoidal surface temperature. J Geophys Res 70:2821–2827

Short Exercise 12: Estimation of Specific Yield

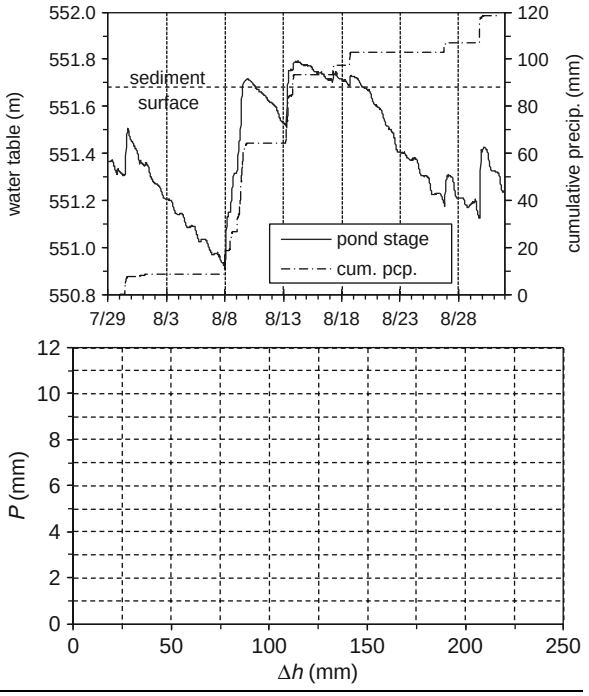
When inflow to and outflow from a wetland containing no surface water are negligible over a short-duration storm, the change in subsurface storage (ΔS_{sub}) is approximately equal to the net vertical input or loss of water from the wetland ($P - E$) (see Eq. 3.48 and the associated paragraph). Assuming that E is much smaller than P during the storm, specific yield can be estimated as the proportionality constant between ΔS_{sub} ($\cong P$) and increases in the water table (Δh) caused by storms:

$$\Delta S_{sub} = S_y \Delta h \tag{3.63}$$

The figure embedded in Table 3.12 below shows the water-table elevation recorded beneath Wetland 109 in the St. Denis National Wildlife Area in Saskatchewan, Canada (see Hayashi et al. 1998 for the site condition), in July-August 1995 when the water table was mostly below the sediment surface (551.68 m). During this period, there were five storms that caused measurable increases in the water table without bringing it to the surface (see Table 3.12 below).

Table 3.12 Total precipitation and water-table increases during storms recorded in July-August 1995 at Wetland 109. The graph shows the water-table elevation and cumulative precipitation

Date	P (mm)	Δh (mm)
July 30	7.8	193
Aug. 7	10.1	188
Aug. 8	7.6	149
Aug. 26	3.8	94
Aug. 29	10.8	240



1. Plot P and Δh in the graph.
2. Draw a straight line that goes through the origin and provides the best fit with all five points.
3. Determine the slope of the straight line and estimate S_y.
4. The sediments in this wetland are rich in clay (20–30 % by weight). Discuss the relation between S_y and the texture (i.e., grain size distribution) of the sediments. Would sandy sediments have higher or lower S_y than the value computed in this exercise?

Reference

Hayashi M, van der Kamp G, Rudolph DL (1998) Water and solute transfer between a prairie wetland and adjacent uplands, 1. Water balance. J Hydrol 207:42–55

Short Exercise 13: Influence of Error on the Water Budget

Whatta Wetland is a hypothetical 1.5-ha wetland situated in a humid environment where annual precipitation is nearly three times larger than evaporation (Table 3.13). The stage of Whatta Wetland is controlled by a small dam that increases the water level about 0.3 m. As such, it has a well-defined outlet channel, which allows accurate measurement of surface-water flow from the wetland using a weir. A weir also is used to measure surface-water flow to the wetland. In fact, great care was taken to measure all input and loss terms of the Whatta water budget. Based on a report from the wetland observer indicating that she has never seen overland flow at this sandy location, we assume that overland flow, if any, is insignificant. Maximum errors associated with individual components of the water budget are estimated to be:

Precipitation	P	±5 %
Evapotranspiration	ET	±15 %
Streamflow into the wetland	S_i	±5 %
Streamflow from the wetland	S_o	±5 %
Groundwater flow to the wetland	G_i	±25 %
Wetland flow to groundwater	G_o	±25 %
Change in lake volume	ΔV	±10 %

We can write our water-budget equation as

$$R \pm \varepsilon = P + O_f + S_i + G_i - ET - S_o - G_o \tag{3.64}$$

where R is the sum of all of the water-budget components (except change in wetland volume) and ε is the cumulative error associated with all of the water-budget terms on the right hand side.

We are interested in determining how R compares with our measured value for ΔV, which will tell us if we have any bias in our water budget or whether there are some unknown or missing terms. Ideally, R will be very close to ΔV. If this is not the case, we want to know if the difference between R and ΔV can be attributed to measurement error or if there really is a missing component or some substantial bias in our estimates of one or more of the water-budget terms.

The uncertainty associated with determination of each term also is presented in Table 3.13. After quick calculation, you can confirm that the sum of all the input and loss terms, R, is more than eight times larger than our measured annual change in wetland volume, ΔV. If we make the worst-case assumption that all errors are at the positive extreme and then sum all of the error terms, the value based on a summation of the positive error terms is so large that it encompasses the measured value for ΔV. Alternately, manipulating the sum to obtain a minimal cumulative error cannot be supported either. Thus, simple sums of the error values do not provide a means of discriminating whether R is a valid measure of the residual.

Table 3.13 Water-budget terms of Whatta Wetland, including percent of input our output terms, maximum percent error, and maximum error in m^3 per year

Water-budget term	Volume (m^3/year)	Percent of input or loss	Percent error	Error (m^3/year)
P	18,200	26 %	5 %	±910
S_i	46,900	68 %	5 %	±2,345
G_i	4,250	6 %	25 %	±1,063
ET	6,540	10 %	15 %	±981
S_o	49,730	79 %	5 %	±2,487
G_o	6,940	11 %	25 %	±1,735
R	6,140			
ΔV	700		10 %	70

If we can justify making two simple assumptions, we can estimate our cumulative error with far less uncertainty. First, we assume our errors are distributed normally. Given that measurements were made approximately biweekly, making our number of measurements around 26, this assumption appears reasonable. Second, we assume that errors in our measurements are independent. Given that precipitation is measured with a rain gage, streamflow with a flow-velocity meter, evaporation with a suite of sensors, and groundwater with a tape measure of some sort, there is small possibility that any of our sources of measurement error are dependent on another. Assuming errors are normally distributed and independent, cumulative error is reduced based on an equation similar to Eq. 3.54, but without the ΔV term:

$$\varepsilon = \sqrt{\varepsilon_P^2 + \varepsilon_{ET}^2 + \varepsilon_{Si}^2 + \varepsilon_{So}^2 + \varepsilon_{Gi}^2 + \varepsilon_{Go}^2} \tag{3.65}$$

Using ε as a measure for the cumulative error, Eq. 3.64 indicates that $\Delta V = R \pm \varepsilon$.

Based on the above information, answer the following questions:

1. How does R compare with ΔV? Are these values reasonably close? If not, suggest a reason for why they are different.
2. What is the additive error associated with determination of R (what is $R \pm \varepsilon$?) What is the error associated with R based on Eq. 3.65? Based on ε determined with Eq. 3.65, are you comfortable with stating that R is different from ΔV?
3. What if our weir failed and we had to use floating oranges all year to make estimates for the S_i term. Recalculate the maximum error for S_i assuming an error of 20 %. How does this affect R, ε, and your assessment of the water budget relative to ΔV?
4. What if the weir was fine but, instead, we had only air temperature data and were forced to estimate evaporation using the Thornthwaite method, which we decided had a maximum error of 50 %. How would increasing the error associated with evaporation from 15 to 50 % affect the determination of R relative to ΔV?

Field Exercises

Field Activity 1: Installation of a Wetland Staff Gage, Water-Table Well, and Piezometer

With a staff gage to indicate wetland stage and measurement of the depth to water in a nearby water-table well, a wetland scientist can determine whether groundwater has the potential to flow to the wetland or whether the wetland is likely to lose water to the adjacent groundwater system. If we know hydraulic conductivity (K) at the well, and make the assumption that K is uniform in the vicinity of the well and the wetland, we can calculate flow (Q) between the wetland and groundwater in an area for which we think data from the well is representative. Lastly, two additional measurements of Q can be made; one utilizes a seepage meter installed in the wetland bed and the other makes use of changes in temperature gradients in the wetland sediments. The temperature method requires installation of sensors at various depths beneath the wetland bed. Since we have to auger a hole or pound a pipe a meter or two into the sediment to install these sensors, it also makes sense to put a well screen at the bottom, in which case we can determine the hydraulic gradient on a vertical plane as well as K based on a single-well test. With that information, and our measurement of Q from the seepage meter, we can use Darcy's law to calculate K of the wetland sediment on a vertical axis. This will give us an idea of anisotropy, the ratio of horizontal to vertical hydraulic conductivity. With this small investment of time and money, we will have learned a great deal about wetland hydrology and hydrogeology at this site.

This first of three exercises near the wetland shoreline will demonstrate the installation of a monitoring well and a staff gage. Detailed instructions and parts lists presented here, and also those presented in the other field exercises, represent the authors' preferences and describe only one of many different ways to achieve these objectives. Students are encouraged to seek other descriptions and opinions for accomplishing these tasks and then develop their own impressions and methods for collecting data in the field.

Wetland Staff Gage

Figure 3.40 shows a wetland staff-gage installation and illustrates some of the problems that can be associated with their use. First, note that there are two staff gages in the photograph. In settings where wetland stage changes substantially, it may be necessary to have multiple staff gages so that when one gage is completely submerged during periods of high water another situated at a higher elevation can be read to indicate wetland stage. Secondly, note the substantial angle from vertical of the staff gage in the distance. This is the result of ice on the wetland surface having moved at some point during the winter, tilting the staff gage. If the ice moves enough, the staff gage can be completely removed from the wetland bed and sometimes transported a considerable distance. The surveyor holding the rod on the

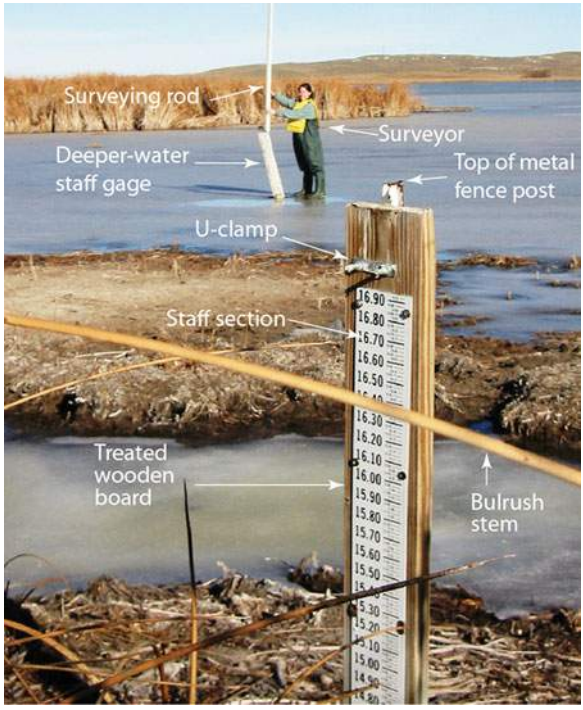


Fig. 3.40 Staff gages installed in a wetland in the Nebraska sandhills with a surveyor standing on the frozen wetland surface and holding a survey rod at the distant gage. Note that ice movement has tilted the staff gage in the distance. Staff-gage movement is an annual occurrence in locations where ice forms on the wetland surface during winter, requiring re-surveys to maintain year-to-year continuity of wetland stage data

staff gage in Fig. 3.40 will also record the angle from vertical of the staff gage so that corrections can be made to any stage measurements obtained while the gage is tilted. Once straightened, the gage will need to be re-surveyed.

Construction of the staff gage in the foreground is typical of many installations. A steel fence post is attached to a piece of lumber that is treated to resist rot (the example in Fig. 3.40 uses U-clamps to attach a wooden board to the post). An incremented staff section, usually made of enameled metal or fiberglass, is screwed to the wood. The fence post can be attached to the wood and then driven into the wetland bed, or if the wetland sediments are very resistant, the fence post can be driven first and then the board complete with face plate is subsequently attached. A length of steel pipe is often substituted for the fence post. Many installations also have a bolt or screw projecting out of the wood next to the face plate so that a survey rod can be placed on the bolt and held in a constant position relative to the values on the face plate while surveying the relative elevation of the staff gage.

Monitoring Well Installation

Two types of monitoring wells, or piezometers, will be installed as part of this field activity, one constructed to indicate the elevation of the water table adjacent to a

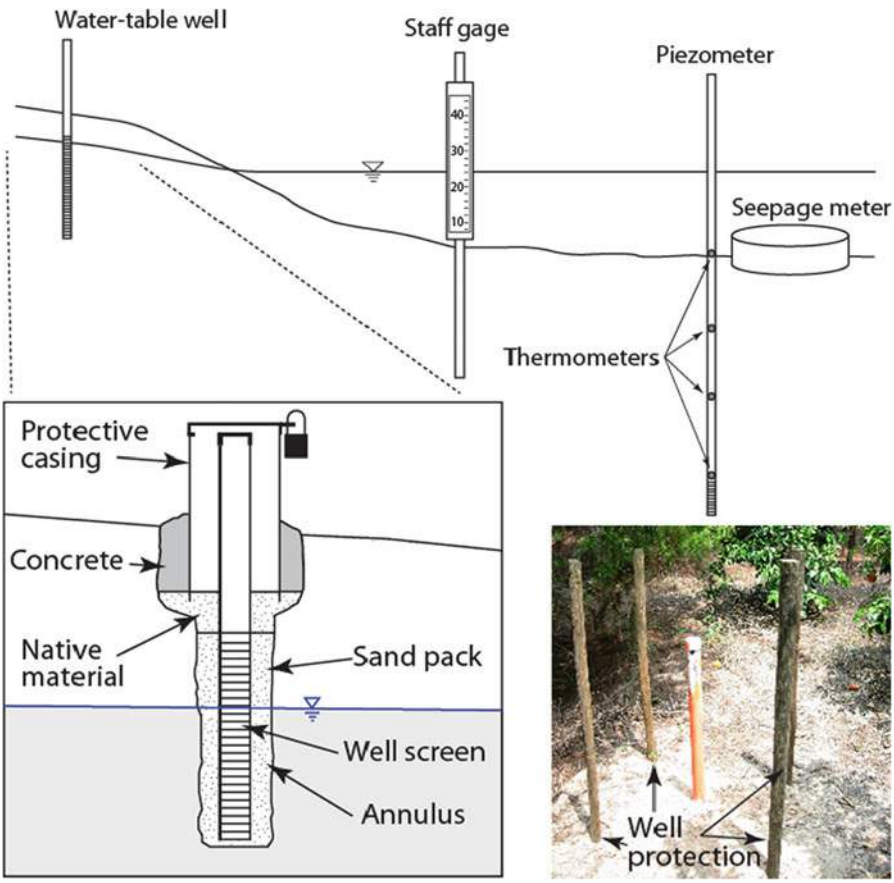


Fig. 3.41 Typical installation to quantify horizontal and vertical hydraulic gradient, seepage rate, and hydraulic conductivity

wetland and the other constructed to indicate hydraulic head at some point beneath the water table (Fig. 3.41). Although both can be considered as piezometers, we will refer to the first as a water-table well.

Water-Table Well Installation

A water-table well is designed to indicate the elevation of the top of the saturated portion of the sediments where pressure head is equal to atmospheric pressure (the water table). Installation of a water-table monitoring well can be simple and inexpensive if the land surface slopes gently away from the wetland edge, in which case the vertical distance from land surface to the water table is usually small. In these shallow, near-shore margins a monitoring well can usually be installed by hand, precluding the need for a large, mechanical drill rig. Such is

the assumption for the following field activity describing the installation of a shallow monitoring well. Items you will need include:

- Polyvinyl chloride (PVC) pipe (a wide range of diameters are available but 5.1-cm diameter is very common)
- PVC well screen (see Fig. 3.42c for examples of commercially made screens. See the section on piezometer installation for making screens from regular pipe)
- Associated couplings and caps and PVC cement
- Bucket auger and associated hardware (8.9-cm (3.5-in.) diameter is common)
- Supply of medium sand (approximately 5-L but amount will vary depending on the diameter of the augered hole relative to the diameter of the monitoring well)
- Shovel
- Tamping rod (handle of the shovel or unused sections of auger rod can suffice)
- Hand saw
- Sledge hammer
- Tape measure or folding rule
- Water-level measurement device (e.g., chalked-steel tape, electric tape)
- Notebook, hand lens, sediment-sample bags

First, select a location for installation of the water-table monitoring well. The well should be located so that it is representative of conditions along a specific reach or area of the wetland. Criteria that are commonly considered when locating a water-table well include topographic gradient, vegetative cover, aspect, geology and soil type. Once the location is selected, use a shovel to remove the vegetation from an approximately 0.25-m^2 area surrounding the intended well site. Note the vegetative cover and organic soil type and thickness.

Install an appropriate auger head on a section of rod (Fig. 3.42a) (closed-head for sand and loosely consolidated sediment, open-head for cohesive sediment) and begin turning the auger in a clockwise direction until the auger bucket is full. Remove the bucket from the hole and shake or push the sediment out of the auger head (Fig. 3.42b), allowing the sediment to fall onto a clean surface, such as a board or tarp. Record the depth of the hole with a tape measure. Describe the sediment in the field notebook. Place a sample from the auger in a sample bag for later lab analysis of percent organic matter and grain-size distribution. Repeat this process until you reach the water table or the intended depth. As you auger deeper, you may need to add one or more rod extensions to the soil-auger assembly. You also may encounter large rocks that inhibit continued augering. Persistence will sometimes get you past a rock or rocky layer, but you also may have to abandon the hole and try again a short distance away.

The water table may not necessarily be obvious if the permeability of the sediment is small enough that water does not readily flow into the auger hole. In some cases, squeezing the sediment with your hand can indicate whether the sediment is saturated or not. If the sample was removed from below the water table, water will be released from the sediment as you squeeze the sample. In settings where the sediment is sandy and poorly cohesive, it is likely that saturated sediment will slump back into the hole as sediment below the water table is removed. The

Fig. 3.42 Hand auger for removing sediment prior to installation of a water-table monitoring well. (**a**) Auger head, rod, and handle with two rod extensions and an additional auger head; (**b**) Augering a hole with the bucket inverted for removal of sediment; (**c**) PVC wound well screen, PVC slotted well screen, and well-screen swab. Note the two different types of fittings at the end of the well screen (standard PVC cap and cone-shaped PVC point). If the slotted screen is inverted and the cap is attached to the opposite end, the non-slotted interval becomes the sump

common solution to this problem is persistence. Keep augering through this sediment with strong downward force on the auger handle. You may need to change to an auger head that has solid sides and a narrower opening between the cutting fins so that loose, wet sand is better retained when the auger is pulled from the hole. The hole below the water table will gradually deepen as you continue to remove sediment and the loose slurry occupying the hole will become less and less dense

as you continue to remove sediment from the hole. Once the desired depth has been reached, commonly about 1–1.5 m below the water table, it is time to assemble and install the well.

Record the total depth of the hole by marking the auger rod at the point where it is even with land surface when the auger is at the bottom of the hole. Remove the auger from the hole and measure the distance from the mark to the bottom of the auger. Add a distance, commonly 0.6–1 m, for the extent of the well casing that will be above the ground. This is often called the "stickup." The sum of these distances will be the total length of the monitoring well. Assemble the well screen by gluing a cap to the bottom of the well screen and a coupling to the top of the screen (Fig. 3.42c). If available, it is desirable to use a well cap that either is cone shaped or that has the same outer diameter as the well screen to reduce resistance when pushing the assembly into the loose sediments below the water table. The well screen should be sized to be long enough that the water table is usually within the screened interval of the well. The slot size (the width of the openings in the screen) should be selected so that most of the sediment cannot pass through the well screen.

Well screens often have an interval at the bottom of the screen that does not have any slots. This is called the sump, or the volume below the screen where fine sediments that pass through the screen can accumulate without blocking the well-screen openings. Be sure to record the presence of a sump and indicate the length of the sump. This information will be important in determining the precise screened interval of the well. The existence of a sump becomes particularly important if the water table is below the bottom of the screened interval. Measurements of depth to water will indicate an erroneous water level equivalent to the elevation of the bottom of the well screen because water will be trapped in the sump. Drilling small holes in the bottom of the sump prior to well installation may allow trapped water to drain from the sump if the well goes dry.

Cut the PVC casing so that the total well length is the distance of the hole depth plus the desired stickup length. If the hole is relatively deep, you may need to attach another PVC coupling and another length of well casing to reach the desired total assembly length. By now, the sediment in the auger hole may have settled and solidified and it may be necessary to remove several additional buckets full of recently slumped sediment from the hole. Keep removing sediment from the hole until the auger has reached the bottom of the hole and the sediment is once again poorly consolidated. At this point it is important to move rather quickly, especially in sediments that readily slump and solidify, such as medium to fine sand. As soon as the last bucket of sediment is pulled out of the hole, immediately shove the completed well casing and screen into the hole and push it down until it stops. You may need to pound lightly on the top of the well casing with the sledge hammer to drive the well to the intended depth. It is prudent to place a board or drive cap on the well casing to prevent damage to the top of the well casing. While pounding lightly, grab the well casing and push downward, essentially vibrating the well downward through the loose sediment. In most cases, you will be able to reach or get very near the desired well depth. Once the well is in place, it is a simple matter of filling the annular space between the edge of the augered hole and the well casing with

sediment that was removed from the hole. Tamp the sediment repeatedly as you fill the hole so the sediment is tightly consolidated. This will prevent any preferential flow of water along the outside of the well casing during recharge events. If unused segments of auger rod are used for this purpose, place duct tape over the end of the rod to prevent damage of the threads.

If the sediment is sufficiently cohesive that the augered hole remains open below the water table, inserting the completed well screen and casing is as simple as placing the assembly into the auger hole. In this case, you will then need to pour sand coarser than the well-screen slot size down the hole so that it surrounds the entire screened interval. This backfill, often called a sand pack, will ensure that the well screen does not become clogged with fine-grained sediment that otherwise would be situated next to the well screen. Once sufficient sand is added to fill the annular space to just above the screened interval, material removed from the auger hole can be added to fill the remainder of the augered hole. As described before, this sediment should be tamped to ensure that the density of the sediment filling the annular space is not less than the undisturbed material. It is common to add soil to create a small mound of soil at the base of the well that will direct rainfall away from the well casing.

Now all that is left is to install a well cap, install well protection, and make several measurements. A well cap can be as simple as a plastic slip cap that stays on the casing via friction and gravity. You might instead wish to glue on an assembly that has a threaded cap or that allows access to the well to be protected with a keyed lock. In either case, make sure that the well cap can easily be removed from the casing for measurements of depth to water. Shallow monitoring wells are not well anchored to the soil because of the smaller contact area with the soil that surrounds the well casing. Some wells can easily be moved, even in an attempt to remove a firmly attached well cap, which may change the vertical positioning of the top of the well and introduce error in determinations of hydraulic gradient. A small hole also may be drilled through the well casing to facilitate equilibration of the pressure inside of the well casing with changes in atmospheric pressure. If air cannot readily enter the well casing, the position of the water table inside of the well may not represent the water table.

In many areas, regulations require some form of protection that will minimize the chance of the well casing being inadvertently broken by a falling tree or branch or a wayward automobile or lawnmower. This may entail placing a steel casing of larger diameter over the top of the well casing and into the ground (Fig. 3.41), or installation of three or four wooden or metal posts positioned so that wayward objects will strike the posts rather than the well casing (Fig. 3.41 photo inset). Lastly, make measurements of the stickup length and the distance to the bottom of the well. Survey to the top of the well casing and determine the spatial coordinates of the well with a global positioning system (GPS) or similar device.

Piezometer Installation

The piezometer will be installed in a location where the wetland bed is beneath the water surface. In this situation, the piezometer will indicate the vertical hydraulic

gradient. In order to ensure that the difference in head between the piezometer screen and the wetland stage will be measurable, the screen needs to be placed a considerable distance below the sediment-water interface, often 2–3 m or more below the sediment-water interface. If the sediments are well consolidated and do not readily slump, it may be possible to use a bucket auger to create a hole in which the well screen and casing are placed, as described previously for installation of a water-table well. If augering is possible, the augered hole should not be larger than the outside diameter of the well to prevent vertical preferential flow of water along the outside of the well casing, which could alter hydraulic head at the well screen. However, in most inundated settings the sediments simply collapse into the augered hole and it is extremely difficult to auger a hole deep enough for a piezometer installation. It is much more common to drive a piezometer to depth with a well pounder or post driver. That is what we will do here. The items you will need include:

- Well screen, cap, couplings, and casing (typically steel to withstand the rigors of pounding)
- Device for driving the well and casing to the desired depth
- Cap to protect the top of the well casing
- well swab (a device to shove water through the well screen)
- bailer or pump for removing or adding water to the well
- Measuring tape

You will want to select a well diameter that is small enough to permit the driving of the well to depth but large enough to allow installation of monitoring equipment inside of the well casing, such as a pressure transducer or temperature sensors. A common diameter for these purposes is 1.9–3.2 cm (0.75–1.25 in.). Commercial well screens are preferred because of the large surface area open to the sediments, although holes or slots can be drilled or cut with hand tools to create simple screens in coarser-grained settings. If the latter option is pursued, the much smaller aggregate surface area of the holes and slots relative to a commercial well screen may result in an unacceptable response time of the well to changes in hydraulic head.

Considerable care is needed to ensure that the well screen is not clogged during installation, especially if a well screen is made by cutting or drilling holes in the well casing. To minimize this possibility, a well swab can be constructed to force water through the screen and to clean out the screened interval of the well during and following the well installation. A well swab can be as simple as a rubber washer or washers attached to the end of a metal rod (Fig. 3.42c) so that the rubber washer rubs against the side of the well casing and screen as it is pushed up and down inside of the well casing. By pushing the rod downward, water inside the well casing is forced through the screen. An upward motion pulls water through the well screen into the well casing. Repeated up and down motion generally is sufficient to remove particles that may be stuck in the screened openings, improving the connection with the aquifer sediments and reducing the time required for the head inside of the well to become representative of the adjacent saturated sediments.

Whether a post driver or well-head driver or sledge hammer is used to advance the well assembly, it should not directly strike the top of the well casing if threads

are present. Doing so could deform the threads and make it impossible to attach a coupling or additional sections of casing that would otherwise allow the well screen to be driven deeper into the sediment. A drive cap or coupling should be screwed onto the threads at the top of the well casing before striking the top of the casing to drive it farther into the sediment. The drive cap or coupling should be tightened occasionally as the casing is driven into the sediment; not doing so also may result in damaged threads. It is prudent to periodically stop driving the well and swab the well to remove sediment that may have clogged the well screen. It may be necessary to pour water into the top of the well casing so the swab pushes and pulls water, and not air, through the well screen. If additional sections of pipe are required, Teflon tape or pipe dope should be used liberally, and the fittings tightened using pipe wrenches, to ensure that no leaks occur at the junctions between pipe segments. Once the well is driven to depth, it should be thoroughly developed by repeatedly swabbing the well and screen, including periodic removal of water and suspended sediment from the well with a pump or bailer, until the water level inside the well casing recovers readily to the static water level. Once this occurs, the well is considered developed and is functioning as a piezometer.

After well installation and development you will want to measure and record:

1. Distance from the top of casing to the well bottom,
2. Distance from top of casing to the wetland bed,
3. Screened interval, sump interval (if present), and
4. Distance from the water surface to the wetland bed.

With these values determined, the distance from the sediment-water interface to the mid-point of the screened interval can be calculated. Commonly referred to as l in the Darcy equation (or sometimes l_v to indicate that the gradient is distributed on a vertical axis), this is the distance that the head difference is divided by to determine the vertical hydraulic gradient. The head difference can easily be determined by measuring the distance from the top of casing to the wetland water surface and subtracting the distance from the top of casing to the water surface inside of the well. For a small-diameter well completed in low-permeability sediments, measurements of depth to water can be corrupted if a portion of the measuring device needs to be immersed in the water to make a measurement. The volume of the sensor device immersed in the water will cause the water level to rise inside of the well. Low-permeability sediments will not permit the water level inside the well to return to static equilibrium in a sufficiently short time, resulting in a false depth-to-water measurement. Care should be taken to prevent this possibility by using a measurement method that does not require immersion of a large sensor relative to the well-casing diameter during a water-level measurement. The cut-off end of a chalked-steel tape is a particularly good device for this purpose because the volume of the steel tape immersed to make a measurement is very small.

Once the piezometer is installed, GPS coordinates and well-top elevations are determined, and measurements are made to determine the hydraulic gradient. Sensors also can be installed to continuously monitor hydraulic head, and temperature at one or more depths, inside of the piezometer (Fig. 3.41).

Field Activity 2: Single-Well Response Test

In Field activity 1, a piezometer was installed either on the margin of or beneath a wetland bed. Figure 3.43a demonstrates a piezometer in a wetland with the screen (slotted portion in the bottom) in direct contact with the sediments, and panel b demonstrates a piezometer completed in a dry margin of a wetland (the water table is below the ground surface). The latter has been installed in an augered hole with a sand pack around the screen and a clay seal above to prevent "short-circuiting" of water through the annular space. A horizontal line beneath an inverted triangle is a commonly used symbol to indicate surface-water level. This symbol is displayed here to indicate the pond water level in (a) and the water table in (b), as well as the undisturbed water levels (also called static head) in the piezometers.

A single-well response test, often referred to as a slug test, is initiated by changing the water level in a water-table well or piezometer very quickly (within a few seconds) and monitoring the recovery of the water level from the initial disturbed value to the static level. A number of methods are available for creating this near-instantaneous water-level change (Butler 1998). The easiest method is to quickly lower a solid cylinder (typically made of metal or high-density plastic) attached to a length of rope into the piezometer. This solid "slug" displaces a known volume of water as it is rapidly lowered into place and the slug remains stationary for the duration of the test. The water level in the well returns to the static level at a rate that is controlled by the hydraulic conductivity of the porous medium around the well screen. After the static level is reached, a second test can be initiated by rapidly removing the cylinder, thereby causing an instantaneous drop of the water level. It is always good practice to conduct two response tests (positive and negative displacement) and check the consistency of results.

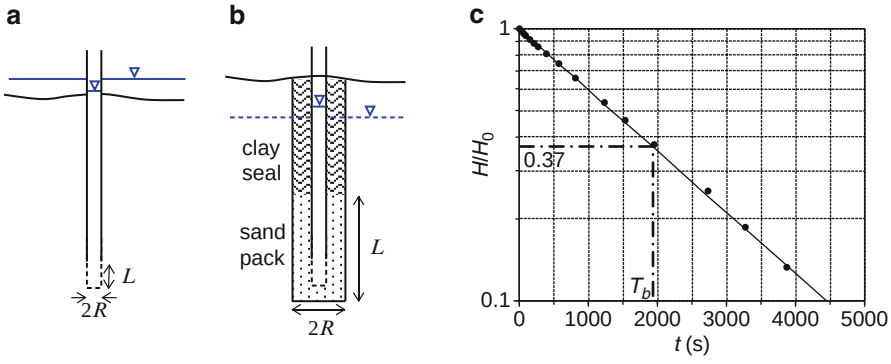


Fig. 3.43 Schematic diagrams of piezometers with screen length *L* and radius *R* without (**a**) and with (**b**) a sand pack; (**c**) example of the plotted recovery of a single-well response test conducted in a piezometer located in Wetland 109 in the St. Denis National Wildlife Area

Water level is monitored during the slug test using either a manual water-level sounder or a pressure transducer, depending on the rate of water-level recovery. For low-permeability settings, manual measurements can often be made quickly enough to capture the initial rapid phase of water-level recovery and can easily be made frequently enough during the slower phase of recovery. A pressure transducer is a far better choice for wells installed in sand or coarser sediments where the entire recovery can be completed in a matter of seconds. The transducer is suspended prior to the test at a depth greater than the reach of the slug to avoid damage to the transducer, and early enough that the well has recovered to the static water level following displacement of water during immersion of the transducer. The combined length of the slug and rope needs to be carefully measured to ensure that the slug does not slam into the pressure transducer as it is rapidly lowered into the well. If the slug is completely submerged during deployment, the known slug-displacement volume can be used to estimate the initial rise (or drop) of the water level during the test. Calculation of the maximum water-level change can then be compared with the measured value. A substantial difference between calculated and measured water-level change may indicate a procedural problem or a problem with the piezometer construction. It also is important to ensure that the piezometer water level does not go below the top of the screen or the top of sand pack during the entire test. For this reason, single-well response tests are not recommended for water-table wells.

The average (or bulk) hydraulic conductivity (K_b, m s^{-1}) of the material surrounding the piezometer screen (or sand pack, if present) can be estimated from the recorded water-level data:

$$K_b = \pi r^2/(FT_b) \tag{3.66}$$

where r (m) is the radius of the inside of the well casing, F (m) is a shape factor representing the dimension and geometry of the groundwater flow field around the screen, and T_b (s) is the basic lag time of the piezometer (see below for definition). The "sample volume" of this method is approximately equal to a sphere with a radius similar to the length of the well screen, L (m). F is a function of L and R, the radius of the outer surface of the well screen or the sand pack, if present. Numerous equations have been suggested to estimate F for different types of piezometers under different conditions (see Butler 1998). In most cases, if L/R is not substantially smaller than 4, the formula of Hvorslev (1951) as cited by Freeze and Cherry (1979:341) gives a convenient means to approximate F:

$$F = 2\pi L/\ln(L/R) \tag{3.67}$$

T_b is determined by plotting head versus time on a semi-logarithmic plot (Fig. 3.43c). For convenience, head is normalized as:

$$H/H_0 = (h - h_s)/(h_0 - h_s) \tag{3.68}$$

where h (m) is measured head, h_0 (m) is the water level immediately after the introduction of the slug, and h_s (m) is the static water level prior to introduction of the slug. Once a straight line is fitted to the data, T_b is determined as the time in seconds since the beginning of the introduction of the slug when H/H_0 equals 0.37 ($\cong e^{-1}$) (Fig. 3.43c).

Once the slug test data have been collected and entered in a spreadsheet, you should follow the procedure listed below:

1. Prepare a data table containing time in one column ($t = 0$ at the maximum h value following introduction of the slug) and h in the second column corresponding to each value of t.
2. Compute H/H_0 for each reading.
3. Plot H/H_0 versus t, using a logarithmic axis for H/H_0.
4. Fit a straight line to the data points, and determine the value of t where the straight fitted line crosses $H/H_0 = 0.37$. Em shows an example, in which $T_b \cong 1{,}930$ s.
5. From T_b and the dimensions of the piezometer, compute K_b.
6. In the example shown in Fig. 3.43c, the piezometer is constructed similarly to panel b and has dimensions of $L = 0.73$ m, $R = 0.075$ m, and $r = 0.016$ m. Substituting these values and T_b into Eqs. 3.66 and 3.67 gives $K_b = 2.1 \times 10^{-7}$ m s^{-1}. This test was conducted in a piezometer located in Wetland 109 in the St. Denis National Wildlife Area in Saskatchewan, Canada (see Hayashi et al. 1998 for details).

References

Butler JJ (1998) The design, performance, and analysis of slug tests. Lewis, Boca Raton

Freeze RA, Cheery JA (1979) Groundwater. Prentice-Hall, Englewood Cliffs

Hayashi M, van der Kamp G, Rudolph DL (1998) Water and solute transfer between a prairie wetland and adjacent uplands, 1. Water balance. J Hydrol 207:42–55

Field Activity 3: Installation of a Seepage Meter and Temperature Sensors

The use of multiple methods to determine flow between groundwater and surface water is always a good idea because it improves understanding of the physical setting and it provides independent values representative of multiple spatial scales. Field activities 1 and 2 demonstrated measurement of hydraulic gradients and hydraulic conductivity to determine Q. Field activity three provides two additional methods for determining Q. A seepage meter makes a direct measurement of Q, but over a very small portion of the wetland bed. The piezometer that we installed in the wetland can serve double duty if we suspend temperature sensors inside of the piezometer casing, allowing calculation of Q based on temperature gradients and attenuation of diurnal cycles in temperature with depth.

Seepage Meter Construction and Installation

What you will need:

- 208-L (55-gal) plastic storage drum
- Hand saw for cutting plastic drum
- Permanent marker
- Measuring device
- Power drill (battery-powered or electric)
- Drill bits appropriately sized for the hose-connection hardware
- Hose-connection fittings
- Rubber or cork stopper
- Plastic tub and lid to serve as a seepage bag shelter
- Plastic seepage bag (approximately 3–5 L)
- Tube and fittings to connect plastic bag to hose
- Hose to connect bag shelter to seepage cylinder
- Brick or suitable weight to place on top of seepage cylinder

A seepage meter can be made from many different readily available products. The standard "half-barrel" seepage meter is described as such because it was made by cutting the ends off of a standard 208-L (55-gal) storage drum (Lee 1977). Although many other cylinders have been used as seepage meters, such as coffee cans, cut-off trash cans, trash-can lids, even wading pools, the half-barrel meter is often used because it is rigid, durable, does not readily deform, covers a larger surface area than many of the other devices, is still quite inexpensive, and can be easily obtained from many industrial supply companies. A storage drum will be used in this exercise. First, obtain a storage drum from one of a large number of suppliers. Either metal or plastic drums can be used, but to simplify construction for this exercise, you should obtain a plastic drum. Be sure to order a closed-top drum to eliminate possibilities of leaks associated with an open-top drum where the top can be removed, and order the larger 208-L (55-gal) drum because it covers a larger surface area than the 114-L (30-gal) drum. You will make seepage cylinders from the top and the bottom thirds of the drum.

Mark the side of the drum a consistent distance from one end of the barrel; commonly, a length of 30–35 cm is used. Connect the dots (marks) by drawing a line along the circumference of the drum. Use the hand saw to cut along this line to remove one end from the drum. Repeat this process for the other end of the drum. If vegetation on the wetland bed is tall and dense, you may instead simply cut the barrel in half, essentially making two seepage cylinders, each approximately 45 cm tall. A cross-cut hand saw can be used to cut the plastic drums whereas a cutting torch or reciprocating saw (or a hack saw used with great persistence) are generally required to cut a metal barrel. Carefully measure the diameter or the circumference of the open end of the cut-off cylinder and calculate the area based on either measurement. This open end of the cylinder will equal the area of the wetland bed covered by the seepage cylinder. Most 208-L drums will cover an area of about 0.25 m^2.

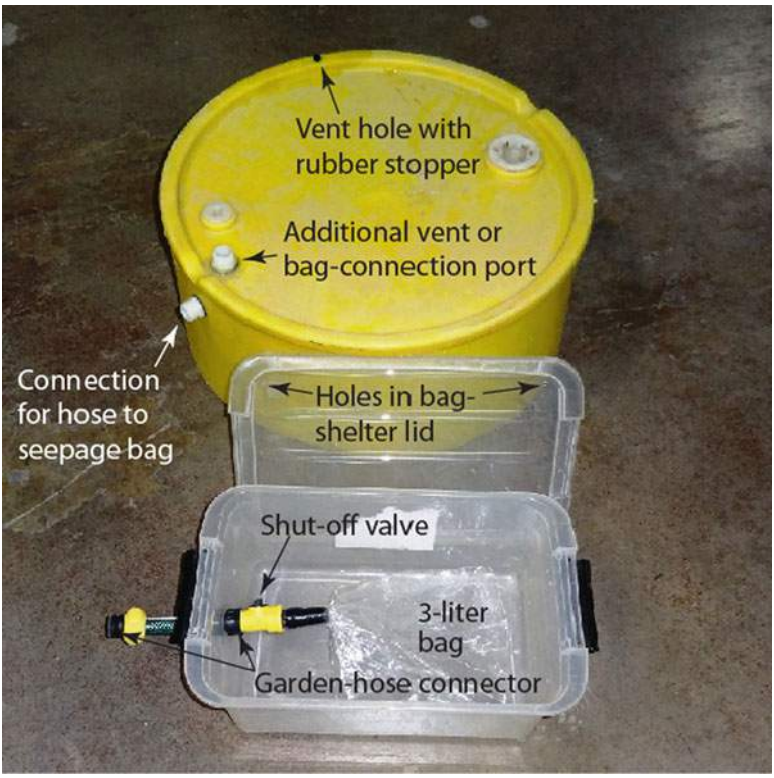


Fig. 3.44 Half-barrel seepage cylinder showing ports installed both in the top and the side of the cylinder. A section of garden hose with female garden-hose connectors on both ends (not shown) is used to connect the bag shelter to the seepage cylinder

Next, you will need to drill a hole in the side of the drum, near the drum end, to which a short hose will be attached (Fig. 3.44). The short hose will extend from the seepage cylinder to a seepage-bag shelter that will protect the bag from wind and waves, curious animals, and diving ducks (Fig. 3.45). The diameter of the hole will depend on the hardware that you use to attach the hose to the seepage cylinder. There are many different options available. Water flows through a seepage meter under very low pressure. The fitting should not leak under small pressures but you do not need to go to the expense of installing a water-tight bulkhead fitting either. Lastly, drill a small hole approximately 0.5–1 cm in diameter at the highest point of the seepage cylinder (vent hole identified in Fig. 3.1). This will be the vent for releasing any gas that is trapped during seepage-meter deployment. This hole will be open during installation of the seepage cylinder and then plugged with a rubber or cork plug during operation. If substantial amounts of gas are generated, a common situation in many wetland settings, you may need to install a vent tube that will extend above the water surface so that gas can be released to the atmosphere during seepage-meter operation (Lee and Cherry 1978).

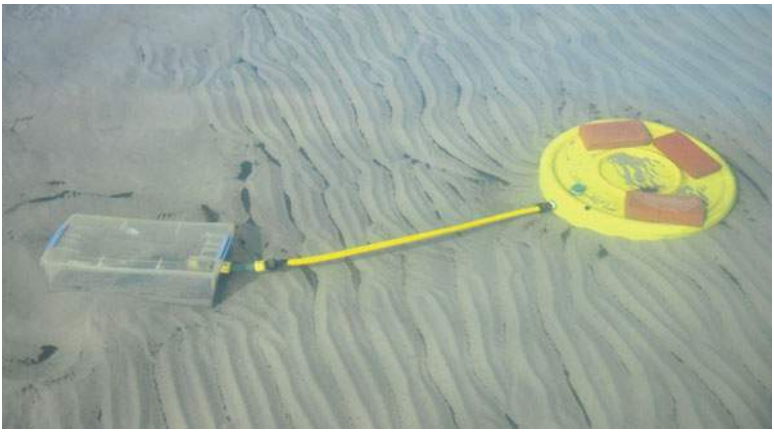


Fig. 3.45 Half-barrel seepage meter installed in sandy sediment. Note side port, to which the hose is connected, and top port with cap and vent hole with rubber stopper

The seepage bag, used to measure the volume of water that flows across the sediment-water interface covered by the seepage cylinder, also can be made from a variety of materials. A convenient bag volume is 3–4 L and thin-walled, flexible bags are preferable. Lightweight freezer-storage bags have often been used. Avoid using bags with thicker walls, such as medical intravenous (IV) bags or solar-shower bags; these bags have a substantial resistance to expansion and contraction in response to being filled or emptied. Use of these bags will substantially reduce the volume of water that otherwise would flow across the bed covered by the seepage cylinder. The opening of the bag can be gathered together around a hose and taped to the hose so the fitting does not leak. Another option is to weld or otherwise seal the bag opening and cut a small slit in one of the corners of the bag, through which you will insert a hose or tube and tape the bag to the hose or tube. As with the seepage cylinder, the bag and fittings should not leak under small pressures but the assembly does not withstand large pressures. It is convenient to install hardware that includes a valve that can be closed while the bag is being transported, attached or removed from the seepage cylinder, and during subsequent handling prior to being weighed or measured.

The bag should be placed in a shelter for several reasons: (1) to prevent the bag from being exposed to currents, (2) to maintain the bag in a proper orientation, and (3) to protect the bag from fish or mammals or waterfowl, a particularly important consideration in many wetland settings. Many different types of bag shelters have been used; examples are provided in Figs. 3.44 and 3.45. Design and build a bag shelter of your choosing, including a section of tubing or hose that will connect to the side opening on the seepage cylinder. The hose or tubing should be approximately 1–2 m long, which ensures that you will not disturb the seepage cylinder while attaching or removing the seepage-collection bag.

Seepage-Meter Installation

Select a location near the piezometer that you installed as part of field activity 1. Wade to the location, making sure to not step on the area that will be covered by the seepage cylinder. The bed should not be covered by any large rocks or debris (i.e., waterlogged sticks) that would alter seepage or prevent insertion of the seepage cylinder. Make sure the rubber plug is removed and the port on the side of the seepage cylinder is open; this allows water to escape as you are pressing the seepage cylinder into the wetland sediments. Press the cylinder into the sediment very slowly, allowing gas and water to escape through the top vent tube. You may need to twist the cylinder to aid in cutting through a vegetative mat, if one is present. If aquatic vegetation is very dense you may need to first cut a slit in the vegetative mat with a long knife to facilitate insertion of the cylinder. The bottom rim of the cylinder typically needs to penetrate the sediment approximately 5–10 cm to ensure a good seal with the sediment. However, if the bed surface is uneven, the insertion depth may need to be increased so no gaps are present beneath the edge of the seepage cylinder. You should probe with your fingers along the interface between the wetland bed and the seepage cylinder. If you can feel the bottom edge of the cylinder, then the insertion depth is not sufficient. In this case, press the cylinder deeper into the sediment until you can no longer feel the bottom edge of the cylinder. The meter also should be inserted with a slight tilt so that the vent hole is at the highest point, allowing any gas released from the sediment to escape. Once the meter is set, place a weight on the meter to counter the buoyant force of the plastic material. A concrete or masonry brick usually is sufficient. Plug the vent tube with the rubber stopper. The stopper will be removed later, prior to seepage measurement, to provide a relative guide for the volume of gas released from the sediment. If the volume is substantial, you will want to install a vent tube to release gas to the atmosphere. If unvented, gas released from the sediment will collect inside of the seepage cylinder, displacing water that will be routed to the seepage-collection bag.

Install the bag shelter and connect the shelter to the seepage cylinder. You may also need to place a small weight inside of the bag shelter to hold it in place and prevent movement in response to waves. The wetland bed has been substantially disturbed during meter installation and it is common for seepage rates to be larger than normal following meter installation. It is common practice to wait for hydraulic conditions at and near the bed to stabilize before measuring seepage. If your field schedule permits, wait until the next day before making the first measurement, or measure seepage directly after installation and compare those values with measurements made the following day.

Seepage-Meter Measurement

Since you do not know whether water is flowing into or from the wetland across the portion of the wetland bed isolated by the seepage cylinder, start your first measurement with the seepage bag approximately half filled with water. Place a known volume of water inside of the bag. Volume can be determined either with a

graduated cylinder or by weighing the water and the bag with an electronic scale. If using an electronic scale, knowing that the density of water is 1 g/cm^{3}and that 1 ml equals 1 cm^3 allows you to measure change in volume by recording change in weight of the seepage bag. Before making any measurements using an electronic scale, you should weigh the bag empty, then completely full, so you will know the range of volume that can be measured with the bag.

Once an initial volume of water in the bag has been measured (or weighed), you will need to remove all remaining air from inside of the bag prior to connecting the bag to the seepage cylinder. This is commonly called de-airing the bag. Close the valve on the bag that contains a measured volume of water, walk out to the bag shelter, suspend the bag vertically while holding onto the bag fitting, open the valve, and slowly lower the bag into the water, immersing the bag with the valve constantly pointing up and always above the water surface. This process will force air inside of the bag to leave via the open valve located above the water surface. Once the bag is pulled beneath the surface to the point where water inside of the bag is at the same level as the valve, close the valve. The bag is now de-aired and ready for deployment.

Carefully remove the bag-shelter lid and attach the bag to the threaded fitting inside of the bag shelter. Straighten the bag so the bag material is not twisted and the bag is oriented in a relaxed position inside of the bag shelter. Open the valve and record the time of opening. Your measurement has begun. Place the lid on the bag shelter very slowly to avoid forcing water out of the bag during the measurement. Now you wait. Since you do not know the seepage rate a priori, the wait time is somewhat of a guessing game. A half hour to an hour should be sufficient to allow a change in water volume that is large enough to allow you to know whether water is flowing to or from the bag. To remove the bag, repeat the process described above but in reverse. Remove the lid on the bag shelter very slowly, and close the valve on the bag being careful to not touch the bag. Record the time as you close the valve. Remove the bag and measure the final volume of water (or determine the final weight of the bag plus water if an electronic scale was used prior to bag attachment). By the gain or loss in volume or weight, you will know the direction of flow and have an initial assessment of the relative seepage rate. If the bag is full or empty upon removal, you waited too long and your next measurement should be conducted over a shorter period. If there is no measurable change in volume, your next measurement period should be increased. After one or two iterations, you should have a good estimate for the amount of time it will take to make a seepage measurement. Simply divide the change in volume by the time of bag attachment to get seepage results in ml/min. Divide that value by the area covered by the seepage cylinder to report your results in flux units (distance per time).

Installation of Temperature Sensors

Accurate measurements of temperature can be made easily with inexpensive instruments, making its use in quantifying exchanges between groundwater and surface water particularly attractive. Here we will make use of newer technology

Fig. 3.46 Nest of piezometers installed at different depths beneath the wetland bed with pressure transducers and temperature sensors installed in five of the seven wells. All sensors are connected to a digital datalogger positioned on shore to the left of the photo. Note also the four seepage meters, with bags attached directly to the tops of the seepage cylinders, installed near the wells. Attaching the bag directly to the seepage cylinder is sometimes acceptable where wind and currents are minimal

for measuring temperature, along with the concepts presented in Sect. 3.6, to determine a value for Q at the piezometer we installed earlier. This value can be compared to Q determined with the Darcy method described in Field activity 1.

Two basic types of electronic sensors are commonly deployed for this purpose. The thermocouple is a device that consists of two wires made of different metals that are connected together at both ends. A current is generated when two junctions of these wires are exposed to different temperatures. Copper and constantan wires are commonly paired for use in environmental applications. The method requires that one of the junctions be related to a known temperature. Therefore, a separate reference temperature sensor also is required to use this measurement method. The second commonly used sensor, and one that often is used as the reference thermometer for thermocouple installations, is the thermistor. A thermistor is basically a resistor that changes resistance in response to changing temperature. The choice of thermocouple or thermistor often depends on the number of temperature sensors required. If more than 5–10 sensors are required, it may be more cost effective to deploy thermocouples.

Two methods of deploying temperature sensors also commonly are used. One consists of a sensor connected to wires that transmit the signal to a nearby data-collection device (Fig. 3.46), and the other consists of the sensor and datalogger in a

single, self-contained unit. Recent versions of the latter device have become very small (e.g., 17 mm diameter) and can be inserted inside small-diameter piezometers.

Either type of sensor can be used for this installation. First, familiarize yourself with the electronic thermometer of choice, making sure that the sensor output is reasonable, that output changes in response to placing the sensor in a warmer or colder environment, and the sensor is logging data. For this application, collecting data at 15-min intervals generally is sufficient to monitor diurnal changes in temperature, although more frequent data collection is certainly acceptable.

Attach one sensor to the outside of the casing of the piezometer that is installed in standing water in the wetland. The sensor should be positioned just above the sediment-water interface. You may also wish to deploy an additional sensor to record changes in air temperature that drive changes in the wetland water temperature. Next, position one sensor at the bottom of the well and another one or two sensors at equal distances between the well bottom and the sediment-water interface. Only one sensor is actually required to be deployed inside of the well; additional sensors allow a determination of the degree of heterogeneity in hydraulic conductivity between the sediment-water interface and the bottom of the well. It is common to suspend sensors on appropriate lengths of string or fine wire from the top of the well (be sure to first check whether the sensors sink or float), or if a signal cable is involved, to affix the signal cable to the top of the well so the sensor hangs at the appropriate depth.

Collect data from the sensors for a period of one to several weeks. Retrieve the sensors, download the data, and plot the time series from all sensors on the same plot.

1. After viewing the data you have collected, is it likely that groundwater is discharging to the wetland or that wetland water is flowing vertically downward to become groundwater? Or is it not possible to make this determination based on your data?
2. Calculate the difference between the daily maximum and minimum temperatures for each sensor. Plot the differences versus time. If you have collected air-temperature data, include daily differences for air temperature as well. Can you make any determination regarding any potential change in the rate of flow across the sediment-water interface?

You can determine the rate of vertical flow across the wetland bed in either direction using the methods described in Short exercise 11. You will also need estimates of thermal conductivity, porosity, dispersivity, and heat capacity of the sediment. Since you also know the vertical hydraulic-head gradient based on measurements you made at this piezometer in field activity 1, you could use one of several methods described in Appendix B of Stonestrom and Constantz (2003) to determine Q. As an additional exercise, you are encouraged to use the free software described in Stonestrom and Constantz to calculate Q based on the temperature data you have collected.

References

Lee DR (1977) A device for measuring seepage flux in lakes and estuaries. Limnol Oceanogr 22:140–147

Lee DR, Cherry JA (1978) A field exercise on groundwater flow using seepage meters and mini-piezometers. J Geol Educ 27:6–20

Stonestrom DA, Constantz J (2003) Heat as a tool for studying the movement of ground water near streams: U.S. Geological Survey Circular 1260:96

Field Activity 4: Stream Gaging Techniques

Stream inflow or outflow may be the dominant component of a wetland water balance, in which case it is important to measure stream discharges as accurately as possible. The following field activities will provide values of stream discharge using three different methods. These measurements are ideally conducted in a relatively small stream with a well-defined channel that is safely accessible by observers.

First, identify a suitable stream reach that satisfies the conditions listed in the first paragraph of the "Discharge measurement" segment of Sect. 3.4. Following the procedures described in "Velocity-area-method" of Sect. 3.4, a measurement section perpendicular to the flow direction should be set up. One observer wades into the stream with a current meter and a device to measure the depth of water (e.g., a wading rod), while the second observer takes notes on the bank and also takes necessary precautions for the safety of the observer in the stream. Depending on the type of current meter used, the velocity is measured at a prescribed depth (e.g., six-tenth point for the Price-type meter), or averaged over the entire depth profile in a subsection. From the depth and velocity data for individual subsections, the total discharge is calculated using Eq. 3.22. Repeat the same measurement two or three times, preferably moving the cross section upstream or downstream by several meters, and compare the results to assess the repeatability and errors of the method.

Next, measure discharge in the same stream reach using the float method described in the section "Other methods of discharge measurement". This method usually is not as accurate as the velocity-area method, but it provides a useful alternative when a current meter is not available. Any floating objects that are clearly visible and are relatively unaffected by wind can be used. Subsections should be determined in a manner similar to the velocity-area method (but usually with coarser spacing of measurement points). Once points are determined, float-velocity measurements simply replace measurements made with a current meter. The profile-averaged velocity can be estimated by multiplying the surface velocity determined with the floats by 0.85.

The tracer-dilution method provides a third value of stream discharge at this stream reach. First, select a suitable location upstream of the measured cross section for release of the stream tracer. This location should be sufficiently far upstream to ensure complete mixing of the tracer solution. This may require preliminary release of tracer at several upstream locations, along with accompanying downstream measurements of tracer concentration at several locations, to confirm complete

mixing. You will want to select a tracer that can be released in small quantities but that will not be masked by the background concentration in the stream. The tracer also needs to be one that is not regulated by any stream-management authorities, or one for which you have a permit to release.

It may be convenient to use electrical conductivity (EC) as a surrogate for tracer concentration if a sufficient amount of tracer can be released to create an easily measured increase of the EC of the stream water. In streams that have very low background EC, a strong correlation between tracer concentration (e.g., chloride) and EC can be pre-established, and concentration can be estimated from the measurements of EC. If this is not feasible, water samples will need to be collected and analyzed with a field analyzer or in the laboratory. This will require a large number of samples for slug injection tests.

After the location for tracer release is selected, a choice must be made between the constant-rate injection (CRI) and the slug injection (SI) method. The CRI method requires a device for injecting tracer solution at a constant rate, but only three values of concentration are required (see Eq. 3.23). The SI method does not require a special device, but many concentration values are required to establish the time-concentration curve shown in Fig. 3.17. Here we describe the use of the CRI method. It is assumed that the background concentration is small enough that the tracer concentration can be estimated from the measurement of EC. To establish the relation between EC and tracer concentration, prepare a set of standard solutions from the tracer chemical and the stream water; for example, solutions of 0, 5, 10, 20, . . . 1,000 mg of sodium chloride in 1 L of stream water. The EC values of these solutions are plotted against concentration values to establish a calibration curve.

For successful application of the CRI method, the tracer solution should be released at an appropriate rate and concentration to ensure that concentration at the measurement section can be accurately measured relative to the stream background concentration, and that a sufficient volume of tracer solution exists in the tracer-injection reservoir to achieve steady state at the sampling location. The constant release rate of tracer solution can be maintained using a Mariotte bottle or a field-portable pump with controlled flow rate (see Moore 2004 for construction of a simple Mariotte bottle from readily available materials). Once a steady value of EC is established at the sampling location and tracer concentrations are determined, the observer can calculate discharge using Eq. 3.23.

In summary, the suggested field activities for stream gauging are the following:

1. Determine stream discharge using the area-velocity method. If time permits, determine the discharge at multiple locations and assess the errors and uncertainty of this method.
2. Estimate stream discharge using the float method at the same location, and compare the accuracy of this method with the area-velocity method.
3. Determine stream discharge using the tracer dilution method.
4. Compare the values of discharge obtained by all three methods and discuss their advantages and disadvantages for application at this particular location, as well as other possible locations and situations.

Chapter 4
Hydric Soil Identification Techniques

Lenore M. Vasilas and Bruce L. Vasilas

Abstract Conceptually, hydric soils are soils that formed under hydrologic conditions associated with wetlands. Identification of soils as "hydric" is critical to the identification and protection of wetlands. Conditions of saturation and anaerobiosis associated with wetland hydrology create morphological characteristics in soils that can be used to distinguish them from non-hydric (upland) soils. These distinctive morphological characteristics have been used to develop "indicators" to facilitate the rapid identification of hydric soils in the field without relying on chemical assays or long term monitoring. An understanding of how soils form and the soil properties related to hydric soil morphologies such as soil color and texture are needed to field identify indicators of hydric soils. This chapter emphasizes the proper application of field indicators of hydric soils, the process of describing soil morphology inherent to the use of hydric soil indicators, and approaches to address soils suspected to be hydric but do not meet a field indicator.

4.1 Introduction

To fully understand the material in this chapter it should be accompanied by *Field Indicators of Hydric Soils in the United States* (Version 7.0) (USDA, NRCS 2010a) and subsequent errata, the *Army Corps of Engineers Wetlands Delineation Manual* (Environmental Laboratory 1987) and approved Regional Supplements (U.S. Army, COE 2012), and the *Munsell Book of Color* (available from Munsell Color Company, Inc. Baltimore MD).

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Soil morphology refers to field observable soil characteristics that can be assessed visually or by touch. Morphological characteristics addressed in this chapter include horizonation or layers, color, texture, and structure. Soil morphology typically reflects long term hydrologic conditions. Therefore, the ability to identify, document, and interpret soil morphology is critical to many wetland investigations. Expertise in soil morphology and the interpretation of soil morphology assists in (1) determinations and delineation of wetlands subject to federal jurisdiction, (2) assessment of current or past wetland hydrology, and (3) assessment of changes to wetland condition.

Of particular importance for wetland determinations is the ability to apply Field Indicators of Hydric Soils in the United States (hereafter referred to as *Field Indicators*) properly. Hydric soils are routinely identified in the field through hydric soil indicators, which are sets of morphological patterns that are correlated with soils that formed under hydrologic conditions associated with wetlands. Hydric soils are one of three factors needed to identify an area as wetlands subject to federal jurisdiction under the Clean Water Act and Food Security Act. In this chapter, we present soil morphological concepts that are used in the application of the Field Indicators. These same concepts can be used to further characterize site-specific hydrology with respect to hydroperiod (the seasonal pattern of water table depth) and hydrodynamics (the direction and energy of hydrologic inputs).

The goal of this chapter is not to turn the reader into a soil scientist, but to give the individual enough expertise in soil science to allow for routine wetland determinations and delineations, as well as hydrologic assessment. Knowledge of soil morphology also allows the wetlands practitioner to identify difficult situations where a soil scientist should be called in for assistance.

4.2 Overview of Hydric Soils

4.2.1 What Is a Hydric Soil?

Soil is a natural body comprised of solids (minerals and organic matter), liquid, and gases that occurs on the land surface, occupies space, and is characterized by one or both of the following: horizons, or layers, that are distinguishable from the initial material as a result of additions, losses, transfers, and transformations of energy and matter, or the ability to support rooted plants in a natural environment. The upper limit of soil is the boundary between soil and air, shallow water, live plants, or plant materials that have not begun to decompose. Areas are not considered to have soil if the surface is permanently covered by water too deep (typically more than 2.5 m [~8 ft.]) for the growth of rooted plants. The lower boundary that separates soil from the nonsoil underneath is most difficult to define. Soil consists of horizons near the Earth's surface that, in contrast to the underlying parent material, have been altered

by the interactions of climate, relief, and living organisms over time. Commonly, soil grades at its lower boundary to hard rock or to earthy materials virtually devoid of animals, roots, or other marks of biological activity. For purposes of classification, the lower boundary of soil is arbitrarily set at 200 cm (~6.5 ft.) (Soil Survey Staff 1999).

The term *hydric soil* was first published in *Classification of Wetlands and Deepwater Habitats* (Cowardin et al. 1979). The initial purpose of the definition was to define a class of soils that were closely correlated with hydrophytic vegetation and to produce a list of soils that could be used with soil surveys to facilitate the development of National Wetland Inventory (NWI) maps. Conceptually, hydric soils are soils that developed under hydrologic conditions associated with wetlands. Because of the role of hydric soil identification in jurisdictional determinations of wetlands, very specific criteria/definitions are applied to distinguish hydric soils from non-hydric soils.

4.2.2 Hydric Soils and Wetland Regulation

Identification of soils as *hydric* is critical to the protection of wetlands under the Clean Water Act (CWA) (Federal Water Pollution Control 2008) and for conservation compliance under the Farm Bill. According to the *US Army Corps of Engineers Wetlands Delineation Manual* (hereafter referred to as the *Delineation Manual*) (Environmental Laboratory 1987), the presence of a hydric soil is one of three factors that must be met in order for an area to meet the definition of a jurisdictional wetland. The other two are the presence of hydrophytic vegetation and wetland hydrology. The use of the Delineation Manual and Regional Supplements (U.S. Army COE 2012) is required for all federal agencies involved in identification of wetlands that may be jurisdictional, as well as for most states that have environmental programs to protect wetlands.

A hydric soil as defined by the National Technical Committee for Hydric Soils (NTCHS) is *a soil that formed under conditions of saturation, ponding, or flooding long enough during the growing season to develop anaerobic conditions in the upper part* (Federal Register, July 13, 1994). For a soil to qualify as a hydric soil for regulatory purposes, it must meet the definition of a hydric soil. It is important to note that a soil meets the definition if it developed under the stated hydrologic conditions. If those hydrologic conditions are altered through drainage or protection (levees), the soil is still considered to be hydric *if the soil in its undisturbed state developed as a hydric soil*.

A hydric soil is defined in the National Food Security Act (USDA, FSA 1985) as *a soil that, in its undrained condition, is saturated, flooded, or ponded long enough during the growing season to develop an anaerobic condition that supports the growth and regeneration of hydrophytic vegetation*. While the definition is slightly different than the definition developed by the NTCHS, the methods (hydric soils

list, Field Indicators, and Hydric Soil Technical Standard) that can be used to identify a hydric soil are the same.

Important concepts in the definition to note are:

1. *in its undrained condition* means that the soil formed under wet conditions and the absence of a water table would not preclude the soil from still being considered hydric. In other words, it may be currently in the dry part of the season when it is being observed or it may have been artificially or naturally drained but if the soil formed when the water table saturated the upper part of the soil it is still hydric;
2. *saturated*, *flooded*, *or ponded* means that the soil must have water in an unlined bore hole in the upper part of the soil or the water must rise above the surface of the soil;
3. *during the growing season* means that the water must be present during the growing season as determined by the use of the Hydric Soil Technical Standard (NTCHS 2007);
4. *anaerobic condition* means the soil lacks oxygen and is a reducing environment.

If the soil meets all the above mentioned concepts, then it will support the growth and regeneration of hydrophytic vegetation. Hydrophytic vegetation, as defined in the FSA Manual means a plant growing in (A) water; or (B) a *substrate that is at least periodically deficient in oxygen during a growing season as a result of excessive water content* [16 U.S.C. 3801(a)(13)].

4.2.3 Hydric Soil Indicators

Nearly all hydric soils exhibit characteristic morphologies that result from repeated periods of saturation or inundation for more than a few days. Saturation or inundation, when combined with microbial activity in the soil, causes the depletion of free oxygen (O_2). This anaerobiosis (without O_2) promotes certain biogeochemical processes, such as the accumulation of organic matter and the reduction, translocation, or accumulation of iron (Fe) and other reducible elements. These processes result in distinctive characteristics that persist in the soil during both wet and dry periods, making them particularly useful for identifying hydric soils in the field.

Hydric soils are routinely identified in the field through use of the Field Indicators. Most hydric soils are readily identified by observing either a predominance of gray color with redoximorphic concentrations (formerly called "high chroma mottles") near the surface or an accumulation of organically enriched material on the surface. These features indicate that the soil has been chemically reduced and fits the standard saturated soil/wet soil morphology paradigm. These readily observable soil morphologies resulting from oxidation-reduction of principally Fe near the surface and accumulation of organic matter comprise the primary Field Indicators used for jurisdictional determinations of wetlands. The presence of one indicator is evidence that the soil meets the definition of a hydric soil.

The hydric soil indicators are "proof positive," i.e., the presence of an indicator is proof that the soil is hydric. The absence of an indicator does not prove that the soil is not hydric ("proof negative"). It is important to remember that a soil that does not contain a hydric soil indicator may in fact be hydric if it meets the definition of a hydric soil. In general, soil morphology reflects long-term hydrologic conditions, which is the basis for the Field Indicators. For a myriad of reasons, some of which are still poorly understood, there are some relatively small but significant areas that are, or appear to be, anomalies to the standard saturated soil/wet soil morphology paradigm. That is, not all hydric soils develop diagnostic redoximorphic features, and some soils have colors that suggest that the soils formed under saturated conditions when, in fact, they did not. It is these anomalous soil morphologies that are so difficult to interpret and are easily misinterpreted by the layperson that have become known collectively as *problem soils*.

Hydric soil lists, Field Indicators, and the Hydric Soil Technical Standard were all created to help identify those soils that meet the definition. If a soil meets the definition of a hydric soil, then it is hydric regardless of whether or not it is a soil series on a hydric soils list or meets an approved Field Indicator.

Currently there are not Field Indicators or soil series mapped that fit every hydric soil condition. These soils are considered problem soils for the purpose of hydric soil identification. Chapter 5 of the Regional Supplements has some suggested methods to assist in making hydric soils determinations in problem soils where Field Indicators may not adequately identify hydric soils. Ultimately, the Hydric Soil Technical Standard may need to be applied to collect data to make a hydric soils determination and/or to develop a field indicator that will work in a problem soil situation.

4.3 Soil Formation

4.3.1 Factors of Soil Formation

Soils develop as a result of the interactions of climate, living organisms, and landscape position as they influence parent material decomposition over time (the five soil-forming factors). Each of these five soil-forming factors also influence the development of morphological patterns on which the Field Indicators are based.

4.3.1.1 Parent Material

Parent material refers to the great variety of unconsolidated organic matter and mineral material in which soil formation begins. Certain parent materials such as red parent material or parent material that weathers to soils with high pH can be problematic because the hydric soils that develop in these parent materials often lack characteristic hydric soil morphologies.

4.3.1.2 Climate

Climate is a major factor in determining the kind of plant and animal life on and in the soil. It determines the amount of water available for weathering minerals. Warm, moist climates encourage rapid plant growth and thus high biomass production (primary productivity). The opposite is true for cold, dry climates. High primary productivity does not necessarily result in high soil organic matter levels as much of the fixed carbon (C) is sequestered in standing biomass. In addition, organic matter decomposition (and the demand for soil O_2) is accelerated in warm, moist climates. In saturated soils, partially decomposed organically enriched material may accumulate, such as in bogs, fens, and swamps.

4.3.1.3 Landscape Position or Topography

Topography in terms of landscape position causes localized changes in moisture and temperature. Even though the landscape has the same soil-forming factors of climate, organisms, parent material, and time, drier soils at higher elevations may be quite different from the wetter soils where water accumulates. Wetter areas may have reducing conditions that will inhibit proper root growth for plants that require a balance of soil O_2, water, and nutrients. Landscape position is an important soil forming factor for hydric soil development. A hydric soil is only going to occur in landscapes that allow for an excessive accumulation of water to cause soil saturation and reduction in the upper part.

Figure 4.1 illustrates a few common landscapes. Older terraces, or soils on second bottom positions, usually have developed B horizons (soil layers characterized by illuviated clay or organic matter). Recent soils deposited in floodplains or first bottom positions usually do not have a developed B horizon. Instead, they may have stratified layers varying in thickness, texture, and composition. Differences in climate, parent material, landscape position, and living organisms from one location to another as well as the amount of time the material has been in place all influence the soil forming process.

4.3.1.4 Organisms

Plants affect soil development by supplying upper layers with organic matter, recycling nutrients from lower to upper layers, and helping to prevent erosion. Microbial activity is the driving force behind the development of soil morphological features that are used as Field Indicators. Soil microbes have adapted to a wide range of soil conditions and are rarely a limiting factor in the development of hydric soil indicators.

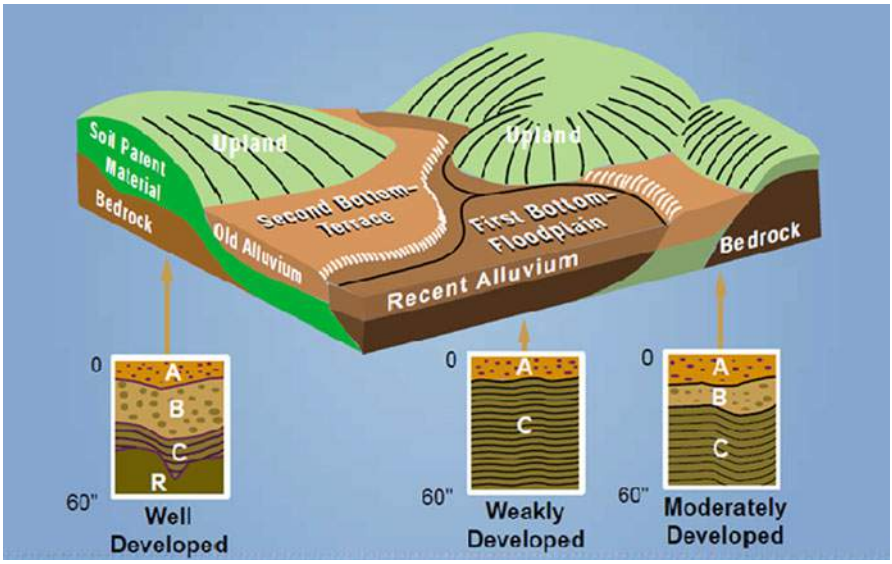

Fig. 4.1 Landscape position influences soil development (Published with kind permission of US Department of Agriculture, Natural Resources Conservation Service (2010b). Figure is public domain in the USA. All Rights Reserved)

4.3.1.5 Time

Time is required for horizon formation. The longer a soil surface has been exposed to soil forming agents like rain and growing plants, the greater the development of the soil profile. Soils in recent alluvial or windblown materials or soils on steep slopes where erosion has been active may show very little horizon development. Soils on older, stable surfaces generally have well defined horizons because the rate of soil formation has exceeded the rate of geologic erosion or deposition. Relatively young soils may lack typical hydric soil morphologies due to lack of time to allow for organic matter accumulation or redoximorphic feature formation.

4.3.2 Soil Forming Processes

The four major processes that change parent material into soil are additions, losses, translocations, and transformations.

4.3.2.1 Additions

The most obvious addition is organic material. As soon as plant life begins to grow in fresh parent material, organic material begins to accumulate. Organic matter gives a black or dark brown color to surface layers. Even young soils may have a

dark surface layer. Partially decomposed organic material may accumulate in saturated soils resulting in thick organic surfaces. These thick organic surfaces are one of the features that can be used to identify a hydric soil.

4.3.2.2 Losses

Most losses occur by leaching. Water moving through the soil dissolves certain minerals and transports them into deeper layers. Some materials, especially sodium salts, gypsum, and calcium carbonate, are relatively soluble. They are removed early in the soil's formation. As a result, soil in humid regions generally does not have carbonates in the upper horizons. Quartz, aluminum, Fe oxide, and kaolinitic clay weather slowly. They remain in the soil and become the main components of highly weathered soil.

4.3.2.3 Translocations

Translocation means movement from one place to another. In low rainfall areas, leaching often is incomplete. Water starts moving down through the soil, dissolving soluble minerals such as calcium carbonate as it goes. Saturation promotes the reduction of Fe which helps bridge clay particles together. Reduced Fe and the associated clays will move with the water. When the water stops moving these materials are deposited. Soil layers enriched with clays, calcium carbonate or other salts form this way. Translocation upward and lateral movement is also possible. Translocation is most apparent in seasonally saturated soils as minerals and clay move up and down with the water table.

4.3.2.4 Transformations

Transformations are biogeochemical changes that take place in the soil. Microorganisms that live in the soil feed on fresh organic matter and change it into humus. For example, ferric Fe (Fe^{+3}) commonly present in Fe oxides under aerobic conditions and is readily reduced to soluble ferrous Fe (Fe^{+2}) which is quite easily removed from the soil by leaching. The patterns in the soil as a result of Fe transformations is the most common feature used to identify hydric soils.

4.4 Soil Horizons

The factors of soil formation do not have a consistent impact with depth. For example, plant roots may not extend throughout the entire depth of the soil. Some soils contain more than one type of parent material. Anthropogenic disturbance (such as plowing) is usually restricted to the upper part of the soil. Soil moisture

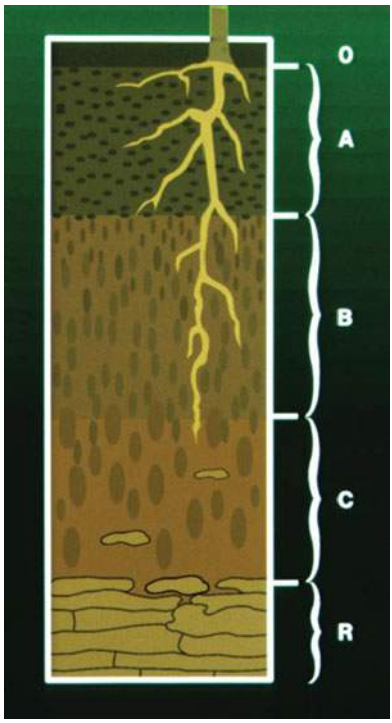

Fig. 4.2 A diagram of soil horizons (Published with kind permission of US Department of Agriculture, Natural Resources Conservation Service (2010b). Figure is public domain in the USA. All Rights Reserved)

typically increases with depth and diurnal fluctuations in soil temperature are minimized with depth. Because of this, soil forming processes are not uniform with depth. As a result, soil morphology typically changes with depth and displays distinct horizontal layers of soil called *horizons* (Fig. 4.2). Horizons can be composed predominately of organic matter (O horizons) or composed predominately of unconsolidated mineral materials (designated as A, E, B, and C horizons). Consolidated bedrock is designated as R.

O horizons form at the soil surface because they are composed primarily of plant roots and leaves in various stages of decomposition. O horizons are dark brown or black. A horizons, commonly called topsoil, are predominately mineral, but distinguished from other mineral horizons by organic matter enrichment. As a result, they tend to be dark brown or black in color. A horizons form at the soil surface or below an O horizon. E horizons represent zones of elluviation, the loss of soil components such as clay, Fe, or organic matter. B horizons represent zones of illuviation, the gain of soil components such as clay, Fe, or organic matter. C horizons display little of the soil forming processes and are similar in composition to parent material. Generally, the horizon you are in does not matter when identifying a hydric soil. However, it is important to understand when you are in an A or E horizon as the requirements for those horizons for some indicators are different than for other horizons.

A soil may lack one or more of these horizons or may have similar horizons at multiple depths. O or A horizons, which form near the soil surface, may be found deeper in the soil due to subsequent formation of horizons above them following

Fig. 4.3 Leaf litter and other recent debris should be removed and not included in the soil description

deposition. An individual horizon may display characteristics of two different types of horizons; its designation reflects this duality. For example, a horizon that is enriched with organic matter but depleted of clay would be designated AE. The presence or absence of specific horizons, and the vertical arrangement of these horizons are used by soil scientists to classify soils.

Soil descriptions document these morphological characteristics. Observations are typically made on a soil profile, a two dimensional vertical slice of soil. For assessing the presence of Field Indicators a slice to a depth of 45 cm (18 in.) is usually sufficient. Shallow soil slices are routinely extracted with a tiling spade hence the phrase *spade slice*. However, spade slices can be extracted with any shovel with a relatively flat blade. It is important to maintain the integrity of the spade slice during the extraction process. This may be facilitated by first digging a pilot hole and extracting and discarding the soil, then cutting a slice from the resulting hole. For deeper observations such as those to accompany monitoring well installment, soil samples are collected with a bucket auger. Samples collected by augering are laid out on the ground in sequence corresponding to the depth of extraction for each sample. Depth of each sample must be documented. A folding carpenter's tape works well for this purpose. Soil descriptions should start directly below the previous year's leaf fall or litter and organic material beneath the layer is considered to be part of the soil (Fig. 4.3).

4.5 Soil Color

4.5.1 Overview

Soil color is an important characteristic of soil morphology as it can be interpreted to provide information on soil mineral composition, distinguish between organic and mineral soil materials, and reflects long term hydrologic conditions. For example,

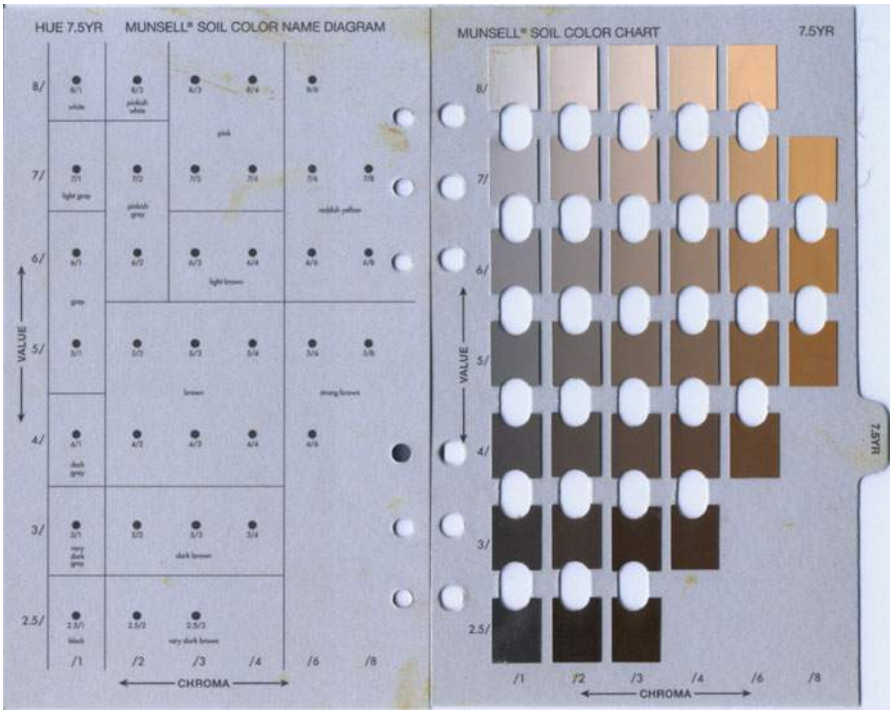

Fig. 4.4 Picture of a color chart in the Munsell™ Soil Color Chart (GretagMacbeth 2009)

in well drained soils Fe oxides usually give soils a yellow, orange, or red color. In soils that are saturated for extended periods, Fe oxides are reduced. The reduced (ferrous) form of Fe is easily removed from the soil by leaching. After the Fe is gone, generally the leached area has a grayish or whitish color. Repeated cycles of saturation and drying create a mottled soil (splotches of color(s) in a matrix of a different color). Part of the soil is gray because of the loss of Fe, and part is red or yellow where the Fe oxides remain.

Therefore, the ability to correctly identify and document soil colors and patterns of soil colors is critical to wetland investigations. Soil scientists rely on the Munsell System of Color Notation in part because it is standardized. The Munsell System includes the entire visible color spectrum using three components: hue, value, and chroma. Colors most commonly found in soils are arranged in books of color chips (Munsell™ Soil Color Charts). One of the Munsell™ soil color charts is presented in Fig. 4.4. Soil is held next to the chips (or better yet, underneath) to find a visual match and assigned the corresponding Munsell™ notation. The notation is recorded in the form: hue, value/chroma – for example, 5Y 6/3. All color chips correspond to an English name in the Munsell™ Soil Color Charts. An example color would be 10YR 4/6, which is called *dark yellowish brown*. 10YR, or 10 yellow-red, is the hue. Four is the value and 6 is the chroma. 10YR means that there are ten parts yellow to one part red.

4.5.2 Components of Soil Color

The four components that have the most affect on soil color are organic compounds (usually black or brown), manganese (Mn) oxides (usually black), iron (Fe) oxides (usually red, orange, or yellow), and the color of the mineral grains (usually clear or neutral gray). Soil color is determined by matching a moist soil to a chip in the Munsell™ Book of Color. Each chip has a specific hue, value, and chroma, identified on the printed page facing each page of chips.

4.5.2.1 Hue

Hue is the chromatic composition (color) of light that reaches the eye. Each Munsell™ page is a different hue that is printed on the upper right corner. Most soils in the Mid-Atlantic Region are on the 10YR (yellow red) page, with redder colors on pages to the left of 10YR and more yellow and grayer colors on pages to the right of 10YR. Additional hues are also used to describe soils on the gleyed pages. These hues include greens, blues, and neutral colors (white, gray, and black).

4.5.2.2 Value

Value is the degree of lightness or darkness of soil color. The value notations are found on the left margin of each page beside each row. The lower values have darker color, while higher values have lighter colors. Value is a continuous scale from 0 to 10. Whenever soil colors do not match a value chip exactly you can round the value to the nearest chip.

4.5.2.3 Chroma

Chroma is the strength or purity of color. The chroma notations are found on the bottom margin of each page under each column. The lower chromas have more neutral (often grayer) color, while highest chromas have the strongest expression of that particular hue. Technically, chroma has no upper limit to the scale, but typically the range found in soils is 0–8. Soil colors that do not match a chroma chip exactly, should be noted as falling between the two color chips. This can be done by estimating a decimal value (10YR 4/2.2) or by using a + (10YR 4/2+). Some Field Indicators require chromas of $\leq$x while others may require a chroma $<$x. So knowing whether the soil color meets a chroma or is in between a chroma is important.

4.5.3 Conditions for Measuring Soil Color

Ideally, soil color should always be read on a ped (clump of soil) interior, immediately after excavation, in a moist state and under direct natural light. Soil is not smeared prior to reading soil color. Hydric soils, especially when they are saturated, may change color quickly upon exposure to oxygen. Therefore, it is important to describe the colors soon after excavation. If the soil does change color with time, you should also record the color of the soil once it has changed and the amount of time that has passed since excavation.

Although it is best to describe soil color moist, often a hydric soil is saturated and thus it is impossible to acquire a moist sample while in the field. In this case documentation that the soil color was read under saturated conditions is made and a sample may be collected and let dry to a moist state before soil color is read again. A saturated soil may change color as it dries indicating a reduced matrix (Fe is reduced in situ). Changing moisture content may affect soil value, while a change due to oxidation or reduction of Fe will most likely produce a change in chroma and confirms a reduced matrix (reduced Fe was present). If the change in color is only due to moisture state and not Fe reduction, then the moist color only needs to be recorded. However, if it is in fact a reduced matrix both colors and the fact the matrix is reduced should be noted.

Soil color should be read under full natural light with the color book facing the sun at a 90° angle. It is best to do this during mid-day when the sun is high. If soil color is read in a forest, the color should be read in a spot where the sun is shining through the canopy. Morning and evening sunlight makes it much more difficult to distinguish between different colors, especially in the winter.

4.5.4 Describing Soil Colors

Multiple colors are often present in a single horizon or layer of a hydric soil. The color pattern is critical to many of the Field Indicators. Therefore, when describing soil colors it is important to document the pattern of colors according to the following parameters.

4.5.4.1 Matrix Color

The matrix color (dominant color) is the color that occupies the greatest volume of the layer (Fig. 4.5). If there are multiple colors that appear to be equally dominant, the soil is described as having a mixed matrix.

Fig. 4.5 The dominant color of this soil is *gray*. Therefore, the matrix color of this soil would be considered *gray* while the other splotches of color would be considered mottles. In this soil, the mottles are due to wetness in the soil and are a type of mottle called redoximorphic features

4.5.4.2 Mottling Versus Redoximorphic Features

Secondary zones of color less dominant in surface area to the matrix are referred to as mottles. Redoximorphic (redox) features are a type of mottling that is associated with wetness and form as a result of saturation and reduction of Fe and manganese (Mn).

4.5.4.3 Percentages

When assessing a soil layer with multiple colors care should be taken to accurately document percentages as they are critical to many of the Field Indicators. Some of these require a minimum percentage of the matrix and/or redoximorphic features. One example would be A11 Depleted Below Dark Surface, which requires a matrix color that has $\geq$60 % of the layer with a chroma of $\leq$2 starting within 30 cm (12 in.) of the soil surface. Another would be F6 Redox Dark Surface, which requires $\geq$2 % distinct or prominent redox concentrations (F6a) or $\geq$5 % distinct or prominent redox concentrations (F6b).

4.5.4.4 Contrast

Contrast refers to the degree of visual distinction that is evident between associated colors. Three categories of contrast are recognized as faint, distinct, and prominent. Contrast is an important consideration when using the Field Indicators as most indicators require redox concentrations to be either distinct or prominent. Note that currently the only Field Indicator to allow faint contrast is S6 Stripped Matrix. The upper threshold for faint contrast is presented in Table 4.1.

Table 4.1 Upper thresholds for faint contrast. Any feature above the upper threshold for faint features would be considered either distinct or prominent

Upper thresholds for faint contrast		
Δ Hue	Δ Value	Δ Chroma
0	2	1
1	1	1
2	0	0
Hue	**Value**	**Chroma**
Any	3	2

4.5.4.5 Type of Redoximorphic Features

Four classes of redoximorphic features are recognized as defined below. On the data sheet under the category *Type*, they should be noted by their abbreviation. Examples of redox concentrations and depletions are shown in Fig. 4.6.

1. Concentration (C): Bodies of apparent accumulation of Fe-Mn oxides.
2. Depletion (D): Bodies of low chroma (less than or equal to) having values of 4 or more where Fe-Mn oxides alone have been stripped out or where both Fe-Mn oxides and clay have been stripped out.
3. Reduced Matrix (RM): Soil matrices that have a low chroma color *in situ* because of the presence of Fe^{2+}, but whose color changes in hue or chroma when exposed to air as the Fe^{2+} is oxidized to Fe^{3+}. The change in color occurs within 30 min or less after the sample is exposed to air.
4. Masked Sand Grains (CS): This applies to particles masked with coats of organic material.

4.5.4.6 Location of Redoximorphic Features

When noting "Location" there are two categories, which are defined below:

1. Pore Lining (PL): Zones of accumulation are either coatings on a ped or pore surface or impregnations of the matrix adjacent to the pore or ped.
2. Matrix (M): Zones of accumulation that are impregnations within the matrix.

4.6 Soil Texture

Soil texture, or particle size distribution, is the numerical proportion of the mineral particles <2 mm (in.) in size (sand, silt, and clay) and is expressed as percent by weight. These mineral size classes are distinguished by size: sand, 0.05–2 mm; silt, 0.002–0.05 mm; and clay, <0.002 mm. Figure 4.7 shows the relative sizes of sand, silt, and clay. Almost everyone knows what sand and clay feel like, either from playing in a sandbox or sculpting with clay in an art class. Silt feels similar to talcum powder or flour. Typically, a sample of soil will contain all three components in various ratios. Therefore, soil textural classes were created to designate the ratios (Fig. 4.8). For example, a sandy clay contains 35–55 % clay,

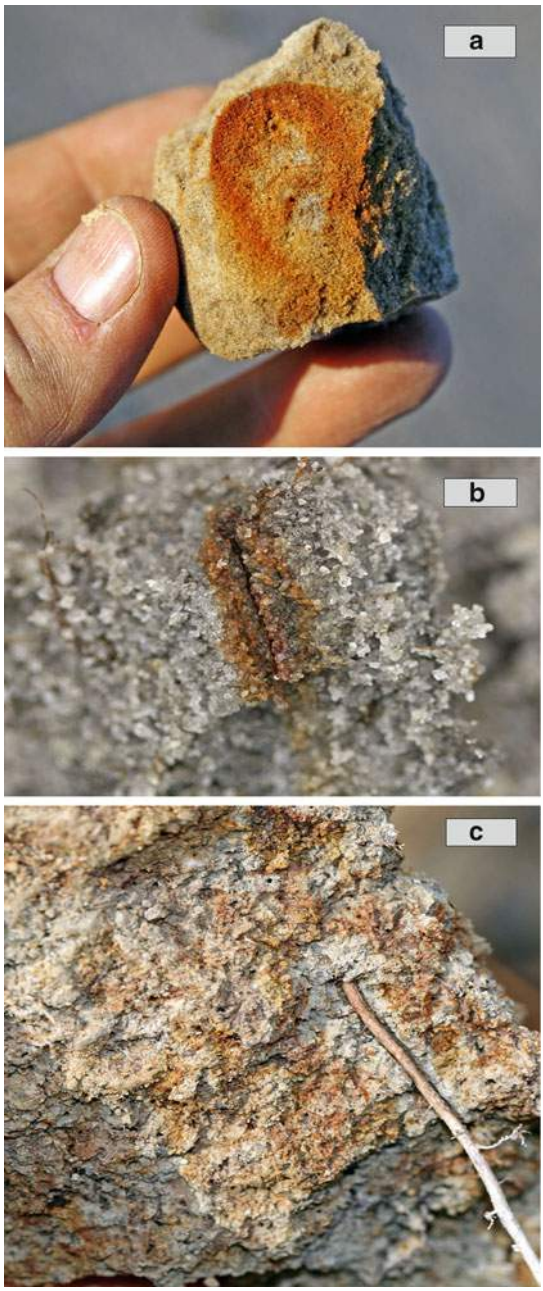

Fig. 4.6 Types of redoximorphic features. (**a**) is a redox concentration as a soft mass on the interior of a ped within the matrix. (**b**) is a redox concentration along a pore lining. (**c**) is a redox depletion adjacent to a plant root

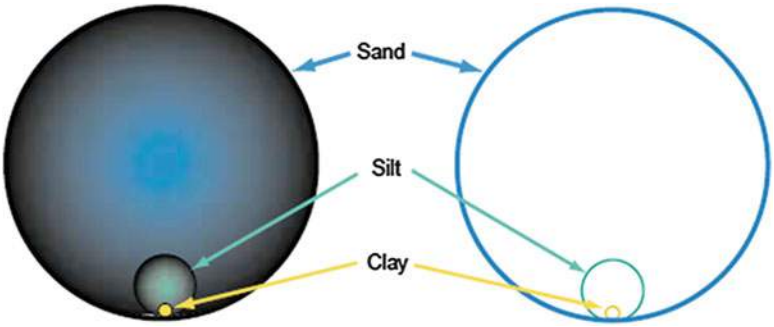

Fig. 4.7 Relative sizes of sand, silt, and clay particles (Published with kind permission of US Department of Agriculture, Natural Resources Conservation Service (2010b). Figure is public domain in the USA. All Rights Reserved)

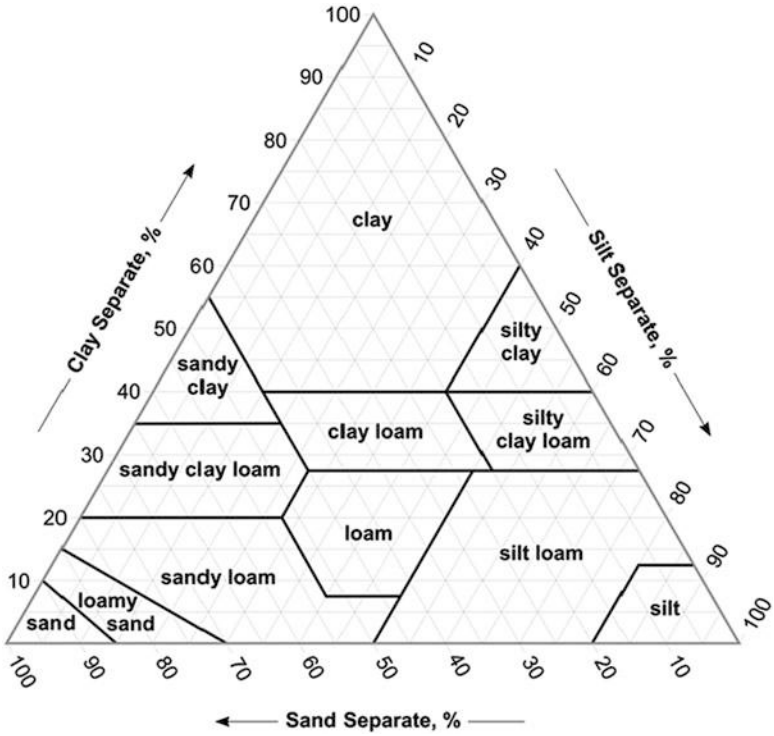

Fig. 4.8 An example of the USDA textural triangle for soils (Published with kind permission of US Department of Agriculture, Natural Resources Conservation Service from Schoeneberger et al. (2002). Figure is public domain in the USA. All Rights Reserved)

45–65 % sand, and 0–20 % silt. Textural classes can be identified in the field using a flow chart (Fig. 4.9). The process is difficult with very dry samples so it is helpful to moisten a dry sample with a spray bottle.

This flow chart is intended only for soil materials that are predominately mineral. For organic soil materials, see *Soil Organic Matter* later in this chapter. A difficult call, even for an experienced soil scientist, is for mineral soil materials enriched with organic material to the extent that it displays characteristics of organic material. In that case, the mineral texture is assigned a mucky modifier, for example, mucky-modified silt loam. This issue is also addressed in *Soil Organic Matter*.

The most important mineral soil texture separation is between loamy fine sand and loamy very fine sand as this break determines which field indicators can be used for identifying the soil as hydric. Based on the diagram, a soil that does not ribbon is generally sandy (loamy fine sand or coarser) and those that do form a ribbon are loamy or clayey (finer than loamy fine sand).

4.7 Soil Structure and Bulk Density

In general, mineral soil particles do not occur as independent units. Instead, multiple particles are grouped into secondary units called peds or aggregates. This aggregation is promoted by oxidized Fe, organic matter, and physical forces associated with wetting and drying cycles, freezing and thawing cycles, or vehicular traffic. Soil structure refers to the shape and distribution of peds and the resistance of peds to physical change. Examples of structural shape classes are presented in Fig. 4.10. Structural units are also rated for strength and are reported as weak, moderate, or strong.

Bulk density is defined as soil dry weight per unit volume. Sand has a higher bulk density than clay. Organic soil materials have a lower bulk density than mineral soil materials. O horizons have lower bulk densities than mineral horizons, and an A horizon generally has a lower bulk density than the underlying B horizon. Bulk density decreases as porosity (% pore space by volume) increases. Compaction is an increase in bulk density; it can be caused by vehicular traffic or by long-term inundation. Structural classes are associated with general ranges in bulk density. For example, granular structure is prevalent in A horizons and is promoted by organic matter; it is associated with low bulk densities. The single grain class is associated with sandy materials, C horizons, and high bulk densities. Blocky structure is associated with B horizons enriched with clay and intermediate bulk densities.

Use of the Field Indicators does not require familiarity with structure or bulk density. However, as addressed later, knowledge of these soil characteristics are critical to the installation of monitoring wells and the assessment of wetland hydrology as they can significantly impact the flow path of water in soil. Figure 4.11 shows the impact of structure on percolation of water through soil.

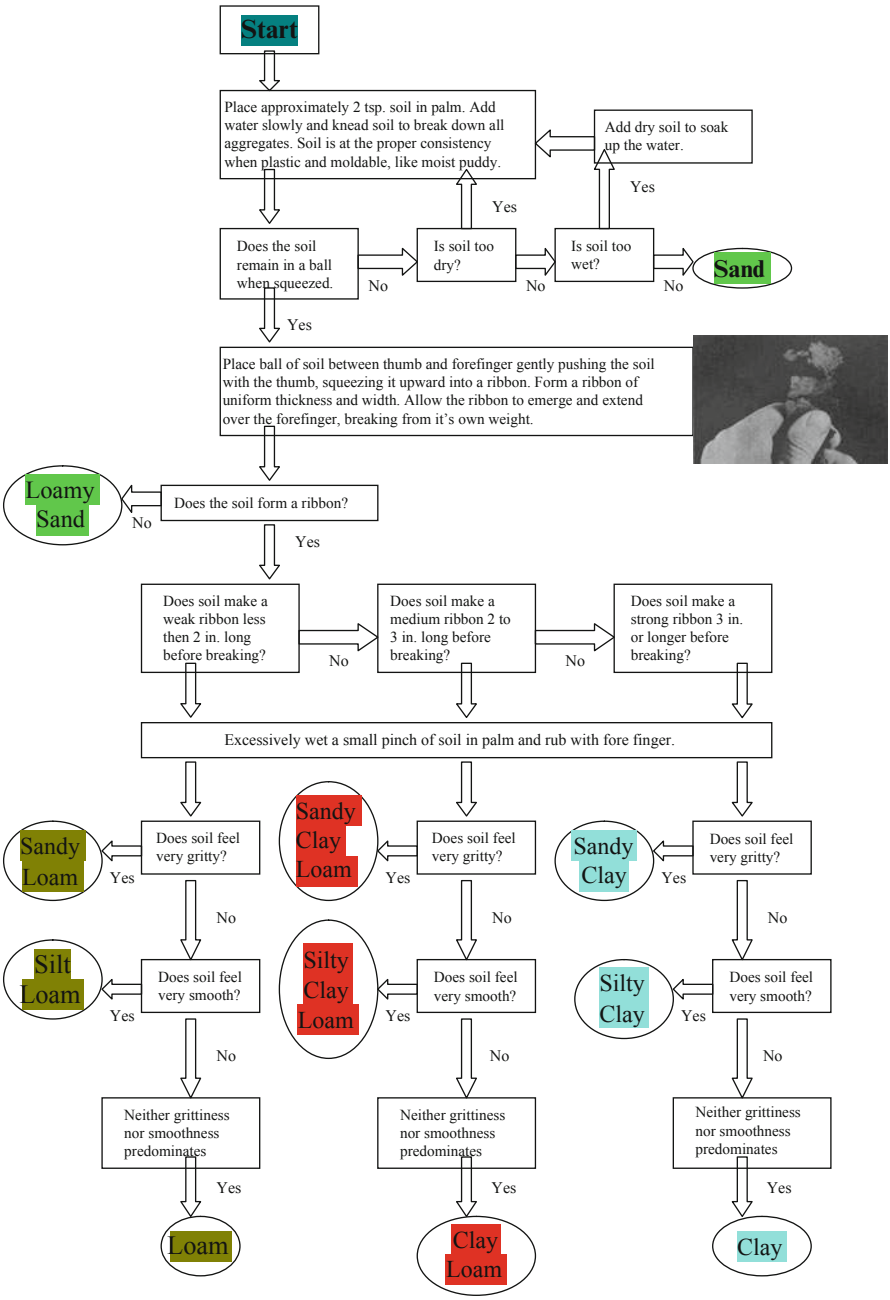

Fig. 4.9 Flow chart for determining soil texture (Modified from Thien (1979), p. 55. Published with kind permission of © American Society of Agronomy, 5585 Guilford Rd., Madison, WI 53711, 1979. All Rights Reserved)

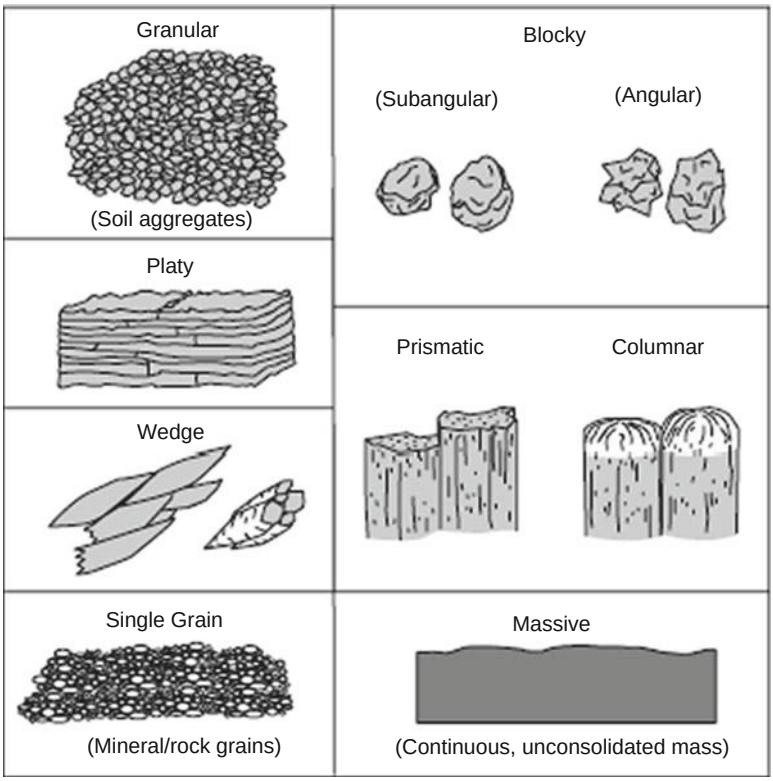

Fig. 4.10 A diagram of the types of soil structure (Published with kind permission of US Department of Agriculture, Natural Resources Conservation Service from Schoeneberger et al. (2002). Figure is public domain in the USA. All Rights Reserved)

4.8 Soil Organic Material

4.8.1 Overview

Soil microbes use carbon (C) compounds found in organic material as an energy source. However, the rate at which organic C is utilized by soil microbes is considerably lower in a saturated and anaerobic environment than it is under aerobic conditions. Therefore, soils that are saturated the entire growing season may accumulate partially decomposed organic material. The result in wetlands is often the development of O horizons of various thicknesses or dark organic-rich mineral surface layers. Three types of O horizons are recognized and distinguished by the level of organic material decomposition. Oa indicates highly decomposed organic material, Oi is slightly decomposed, and Oe is intermediate in decomposition.

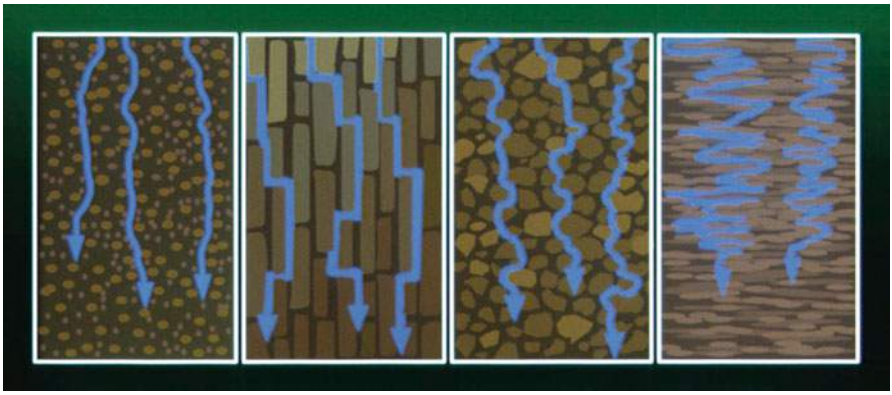

Fig. 4.11 Impact of structure on percolation of water through soil (Published with kind permission of US Department of Agriculture, Natural Resources Conservation Service (2010b). Figure is public domain in the USA. All Rights Reserved)

By definition, organic soil material is saturated with water for long periods or is artificially drained and, excluding live roots, has an organic C content $\geq$18 % with $\geq$60 % clay, or $\geq$12 % organic C with 0 % clay (Soil Survey Staff 1999). Soils with an intermediate amount of clay have an intermediate amount of organic C. Three types of organic soil materials are recognized and distinguished by the degree of decomposition: muck (highly decomposed), peat (very little decomposition), and mucky peat (intermediate decomposition).

To distinguish mineral soil materials that are highly enriched with organic matter, a "mucky" modifier is added to its mineral texture designation; for example, mucky sand. Mucky modified mineral soil with 0 % clay has 5–12 % organic C. Mucky modified mineral soil with 60 % clay has 11–18 % organic C (Soil Survey Staff 1999). Soils with an intermediate amount of clay have intermediate amounts of organic C. The Field Indicators in this category are: A7 5 cm Mucky Mineral, S1 Sandy Mucky Mineral, and F1 Loamy Mucky Mineral.

4.8.2 Field Characterization of Soil Materials High in Organic Carbon

Material high in organic C could fall into three categories: (1) organic, (2) mucky mineral, or (3) mineral. In lieu of laboratory data, the following estimation method can be used for soil material that is wet or nearly saturated with water. The first step is to determine whether the material is mineral or organic. If organic, the second step is to determine the type of soil organic material.

These soil material categories can be determined by gently rubbing wet soil material between forefinger and thumb. For sandy textured soils, if upon the first or second rub the material feels gritty, it is mineral soil material. If after the second rub

the material feels greasy, it may be either mucky mineral or organic soil material. If after additional rubbing (2–3×) the sample feels gritty or plastic, it is considered mucky mineral soil material; if it still feels greasy, it is organic soil material. Accumulation of silt residue on fingers after rubbing indicates that the sample is likely mineral or mucky mineral.

Another method is to take equal amounts of known mineral soil and the horizon in question. An organic soil material will be much lighter than equal amounts of mineral material. Mucky mineral would be slightly lighter than equal amounts of mineral material. The reason for the difference in weight is due to the greater bulk density of mineral material compared to organic matter. If the material is organic soil material, a further division should be made to identify the type of organic soil material. Organic soil materials are classified as sapric, hemic, or fibric which correspond to the organic texture classes muck, mucky peat, and peat, respectively. Organic texture class can be determined by rubbing a soil sample about ten times and then visually estimating the proportion of the sample comprised of fibers (excluding live roots). After rubbing, sapric material or muck will have less than 1/6 visible fibers; fibric material or peat will have more than 3/4 fibers; and hemic material or mucky peat will have between 1/6 and 3/4 fibers (Soil Survey Staff 1999).

4.9 Formation of Hydric Soils

Hydric soils are soils that developed under conditions of saturation close to the soil surface. Under saturated conditions, plant roots and microorganisms use O_2 faster than it can be replenished by diffusion from the atmosphere resulting in first anaerobic conditions and then reducing conditions. The change from aerobic conditions to anaerobic conditions causes a shift in the direction or rate of a number of biogeochemical processes, especially those that impact the accumulation or loss of Fe, Mn, sulfur (S), or C compounds. This results in distinct soil morphological characteristics that serve as the basis for the Field Indicators. For more information on this subject refer to Chap. 7 on *Wetland Biogeochemistry Techniques*.

4.9.1 Processes

4.9.1.1 Soil Saturation

A horizon is considered saturated when the soil water pressure is zero or positive (at or above atmospheric pressure). At these pressures, water will flow from the soil matrix into unlined auger holes. Three types or patterns of saturation are defined:

1. Endosaturation-ground water table. Soil is saturated in all horizons below the water table to a depth of 2 m.
2. Episaturation-perched water table. Soil is saturated in a horizon that overlies an unsaturated horizon, and the unsaturated horizon lies within a depth of 2 m from the surface.

Table 4.2 Element reduction sequence in inundated soils

Eh threshold, mv	Element	Oxidized form	Reduced form(s)
+350	Oxygen	O_2	H_2O
+220	Nitrogen	NO_3^-	N_2O, NO_2^-
+200	Manganese	Mn^{+4}	Mn^{+2}
+120	Iron	Fe^{+3}	Fe^{+2}
−150	Sulfur	SO_4^{-2}	H_2S
−250	Carbon	CO_2	CH_4

3. Anthric saturation-paddy soil with a created perched water table. Like episaturation but must occur under controlled flooding, for example, wetland rice or cranberries.

It should be noted that the term *water table* is not used in the definition of saturation. Also, horizons within the capillary fringe are technically not considered saturated since this contains soil water that has pressures less than atmospheric pressure. Under ideal circumstances, horizons that are saturated by the above criteria will have all their soil pores filled with water. However, for a horizon to be considered saturated it is not necessary that all pores be filled with water. Horizons that have soil water pressures of zero or positive are considered saturated even if they contain entrapped air in some pores. Saturation can occur at any time during the year.

4.9.1.2 Anaerobiosis

When aerobic conditions exist, bacteria decompose organic matter and consume O_2 in soil pores containing air. Under anaerobic conditions, bacteria decompose organic matter by consuming dissolved O_2 until it is gone. At this point, the soil water is reduced. The bacteria continue to consume organic matter, but at a slower rate. They produce organic chemicals that reduce nitrates (NO_3^-) and minerals, including Fe and Mn oxides. The sequence is shown in Table 4.2. While nitrate reduction is the first indication of anaerobic conditions, it does not leave a visible indicator that can be used for the easy identification of a hydric soil.

4.9.1.3 Iron and Manganese Reduction, Translocation, and Accumulation

Both oxidized Fe and Mn can be chemically reduced under certain soil conditions. Reduction occurs when oxidized forms of Fe (ferric, Fe^{3+}) or Mn (manganic, Mn^{3+} or Mn^{4+}) accepts electrons from another source such as organic matter to produce ferrous Fe (Fe^{2+}) and manganous Mn (Mn^{+2}). When these elements are reduced in a soil, several processes occur: (1) Fe and Mn oxide minerals begin to dissolve in water; (2) the soil colors change to gray; and (3) Fe^{2+} and Mn^{+2} ions diffuse through

or move with the soil water to other parts of the soil horizon or may be leached from the soil. When Fe and Mn are in their reduced form, they have much less coloring effect on soil than when they occur in their oxided forms. Of the two, evidence of Fe reduction is more commonly observed in soils.

4.9.1.4 Sulfate Reduction

Sulfur is one of the last elements to be reduced by microbes in an anaerobic environment. The microbes convert sulfate (SO_4^{-2}) to hydrogen sulfide (H_2S) gas. This results in a very pronounced "rotten egg" odor in some soils that are inundated or saturated for very long periods. In unsaturated or non-inundated soils, SO_4^{-2} is not reduced and there is no rotten egg odor. The presence of H_2S is a strong indicator of a hydric soil, but this indicator is found only in the wettest sites in soils that contain S-bearing compounds. It can sometimes be sensed by simply walking across these areas. This is indicator A4 Hydrogen Sulfide. Caution should be used when using this as an indicator so that other smells such as those associated with the decomposition of organic matter are not mistaken for a sulfidic odor.

4.9.1.5 Organic Accumulation

Soil microbes use C compounds found in organic material as an energy source. However, the rate at which organic C is utilized by soil microbes is considerably lower in a saturated and anaerobic environment than under aerobic conditions. Therefore, in saturated soils, partially decomposed organic material may accumulate. The result in wetlands is often the development of organic surfaces of varying thicknesses, such as peat or muck, or dark organic-rich mineral surface layers. These soils are typically saturated for very long periods of time.

4.9.2 Development of Redoximorphic Features

Redoximorphic features are those formed by the reduction and oxidation of Fe and Mn compounds in seasonally saturated soils. Fe oxide minerals give the soil red, brown, yellow, or orange colors depending on which iron minerals are present. Manganese oxides produce black colors. These oxides tend to coat the surfaces of the soil particles. Without the oxide coatings, the particles are gray. Areas in the soil where Fe is reduced often develop characteristic bluish-gray or greenish-gray colors known as *gley*. Ferric Fe is insoluble but Fe^{2+} easily enters the soil solution and may be moved or translocated to other areas of the soil. Areas that have lost Fe typically develop characteristic gray or reddish-gray colors and are known as *redox depletions*. If a soil reverts to an aerobic state, Fe that is in solution will oxidize and

become concentrated in patches and along root channels and other pores where oxygen enters or remains in the soil. These areas of oxidized Fe are called *redox concentrations*. Since water movement in these saturated or inundated soils can be multi-directional, redox depletions and concentrations can occur anywhere in the soil and have irregular shapes and sizes. Soils that are saturated and contain Fe^{2+} at the time of sampling may change color upon exposure to the air, as Fe^{2+} is rapidly converted to Fe^{3+} in the presence of O_2. Such soils are said to have a *reduced matrix* (Vepraskas 1994).

While indicators related to Fe or Mn depletion or concentration are the most common in hydric soils, they cannot form in soils whose parent materials are low in Fe or Mn. Soils formed in such materials may have low-chroma colors that are not related to saturation and reduction. For such soils, features formed through accumulation of organic C may be present.

4.9.3 Types of Redoximorphic Features

4.9.3.1 Iron and Manganese Depletions

Formation is similar for Fe and Mn depletions, and both may occur within the same or adjacent horizons. It is easiest to visualize these features forming around roots that grow along stable macropores. These are required so that features continue to enlarge as succeeding roots grow and die along the same macropore. Roots growing along a structural crack or channel provide an energy source, organic material, that is needed by the microbes for Fe reduction. When the root dies and the macropore is filled with water, the bacteria will consume the root tissue and utilize (reduce) O_2 in the water if soil temperatures are high enough for the bacteria to be active. The newly formed bleached layer where Fe and Mn have been removed along the channels is a redox depletion, specifically an Fe depletion due to its lower content of Fe and Mn.

4.9.3.2 Masses, Nodules and Concretions

When a horizon has been repeatedly saturated, reduced, and drained, Fe masses will form where air penetrates into the horizon slowly to oxidize reduced Mn and Fe ions. Nodules and concretions are believed to form when air penetrates quickly, perhaps at a point into the wet matrix containing Fe^{2+} and Mn^{+2}.

4.9.3.3 Reduced Matrices

A reduced matrix forms simply by the reduction of Fe in the soil. This requires that the soil horizon be saturated to exclude air for a long enough period of time such that Fe reduction occurs. Reduced matrices can only occur where soluble organic matter is present and microorganisms are active.

4.9.4 Location of Redoximorphic Features

Pore linings occur along ped surfaces as well as root channels. They are also found on the roots of living plants that can transport O_2 to their roots in saturated soils (oxidized rhizospheres). These form by diffusion of Fe^{2+} and Mn^{+2} ions toward aerated macropores, where the ions are oxidized adjacent to the macropores and even on root surfaces. If both Fe and Mn are in solution, the Fe tends to precipitate first because it will oxidize at a lower Eh value than will Mn. Therefore, pore linings may appear to consist of clearly separated Mn oxides (in the macropore) and Fe oxides (in the matrix).

In terms of location, reducing conditions will occur near the root channels if the soluble carbon source required by the bacteria comes from dead roots. If the organic compounds are dissolved and dispersed in the soil water, then reduction can occur at any place in a soil horizon where the pores are filled with water.

4.10 Using Field Indicators of Hydric Soils

4.10.1 Overview

To fully understand this section, we recommend that the reader downloads and prints out a copy of the most recent version of *Field Indicators of Hydric Soils in the United States* along with any subsequent errata from the NTCHS website at http://soils.usda.gov/use/hydric/ and a copy of Chapter 3 of the Corps Delineation Manual Regional Supplement for the area of interest at http://www.usace.army.mil/Missions/CivilWorks/RegulatoryProgramandPermits/reg_supp.aspx for reference. The publication *Field Indicators of Hydric Soils in the United States* is a comprehensive list of all the Field Indicators approved for use by the NTCHS. All Field Indicators listed in the Corps Regional Supplements are a subset of the NTCHS national list of indicators. The Regional Supplements also contain the appropriate data sheet for determination and delineation of wetlands and hydric soils in that region.

Not all of the Field Indicators are appropriate for each situation. The Field Indicators are regionalized, and each indicator is only valid in specific Land Resource Regions (Fig. 4.12). In addition, some indicators are restricted by soil texture. There are three categories of Field Indicators which are distinguished by soil texture: All Soils, Sandy Soils, and Loamy and Clayey Soils. *All soils* refers to soils with any USDA soil texture. Examples include A1 Histosol, A4 Hydrogen Sulfide, and A12 Thick Dark Surface, among others. Sandy soils have a USDA texture of loamy fine sand and coarser (sandier). Examples include S1 Sandy Mucky Mineral, S6 Stripped Matrix, and S10 Alaska Gleyed. The loamy and clayey soils category has USDA textures of loamy very fine sand and finer (more clay). Examples include F1 Loamy mucky mineral, F6 Redox Dark Surface, and F9 Vernal Pools.

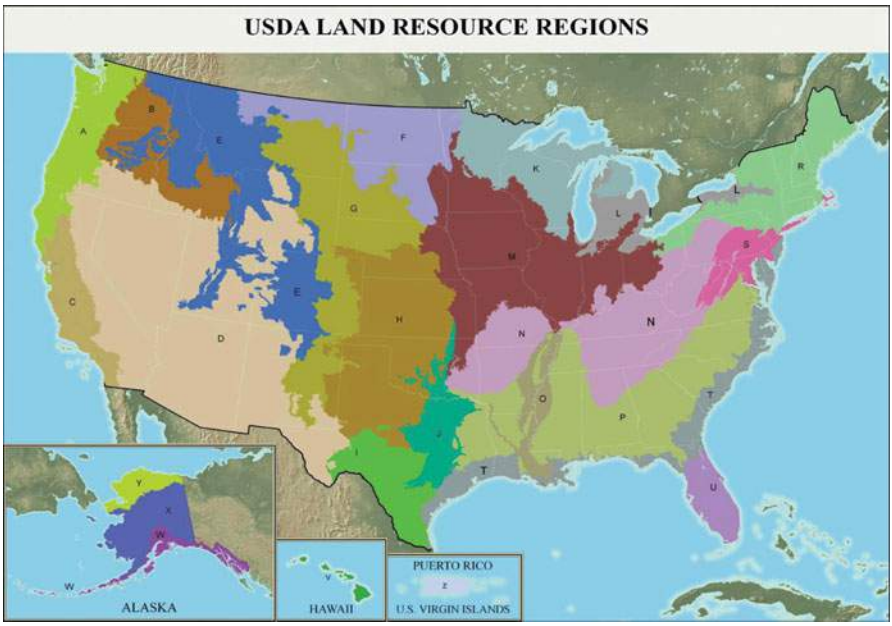

Fig. 4.12 Major land resource regions (Published with kind permission of US Department of Agriculture, Natural Resources Conservation Service (2006). Figure is public domain in the USA. All Rights Reserved)

The descriptions of the Field Indicators are structured as follows:

1. Alpha-numeric listing (A, S, or F Indicators)
2. Short name
3. Applicable land resource regions (LRRs)
4. Description of the field indicator
5. User notes

For example, *A2* is the second indicator in the "all soils" category; the short name is *Histic Epipedon*; the indicator is for use in *all LRRs*; the description is *a histic epipedon underlain by mineral soil material with chroma of 2 or less*. Helpful user notes are added.

4.10.2 Terminology

There are many important definitions that must be understood to properly use the Field Indicators. Many can be found in the glossary of the Field Indicators of Hydric Soils publication (USDA 2010a). Some of these definitions may be slightly different than the use of the same term for other purposes such as for use in soil taxonomy and soil survey. When a term that describes an indicator differs from other soil

science references, an asterisk (*) is placed next to the term in the Field Indicators glossary. Familiarity with the following terms (presented in the Field Indicators glossary) is necessary for the identification of Field Indicators.

1. Depleted matrix – This is an important concept used in many of the Field Indicators. Note that all depleted matrices must have values $\geq$4 and chromas $\leq$2, and depending on the value and chroma, may or may not need the presence of redox concentrations.
2. Gleyed matrix – The definition of a gleyed matrix for the purposes of the Field Indicators are different than the definition used in Soil Taxonomy. For the purposes of the Field Indicators, a gleyed matrix has colors found on the gleyed pages of the Munsell Book of Color and also must have a value $\geq$4.
3. Layer(s) – A soil horizon and a layer for the purposes of the Field Indicators are not synonymous. There can be multiple layers in the same horizon if that horizon meets all the requirements of two different layers and is thick enough to meet the combined thicknesses requirements of the both layers. There can also be multiple horizons that meet all the requirements of the same layer except thickness that can be combined to meet the thickness requirements. For a good explanation of combining horizons to meet the thickness requirement, see Chapter 3 of any Corps of Engineers Regional Supplement (http://www.usace.army.mil/Missions/CivilWorks/RegulatoryProgramandPermits/reg_supp.aspx).
4. LRR and MLRA – Field Indicators are regionalized. Identification of the LRR (and in some cases what MLRA the sites occurs in) for the site in question is critical as it limits the Field Indicators valid for that site. Figure 4.10 is a map of the LRRs. MLRA identification can be obtained from: http://soils.usda.gov/survey/geography/mlra/.
5. Organic soil material – Use of the Field Indicators require a distinction between organic soil material and mineral soil material. Some Field Indicators also require the distinction between the grades of decomposition (muck, mucky peat, or peat).
6. Within – When a Field Indicator states that a layer must start within a certain depth, if the layer starts at that depth it is considered to be within that depth.

4.10.3 Concepts and Rules

In addition, a clear understanding of the following concepts is inherent to the proper application of the Field Indicators:

1. The Field Indicators are proof positive. If a soil meets a Field Indicator, it is a hydric soil. If it does not meet an indicator, it is still a hydric soil if it meets the definition.
2. A soil must meet the requirements in the indicator description to meet that Field Indicator. User notes are provided to assist in the interpretation of those requirements.

3. The Field Indicators were developed to locate the hydric soil boundary. Wetter soils may not meet a Field Indicator.
4. Depths and thicknesses are critical in the upper 30–45 cm (12–18 in.) of the soil when using the Field Indicators. It is recommended that a spade, not an auger, be used to excavate the soil.
5. All soil (A) indicators can be used in any layer regardless of texture. Sandy soil (S) indicators can be used in layers that are loamy fine sand or coarser. Loamy and clayey (F) indicators can be used in layers that are loamy very fine sand and finer.
6. Layers are not synonymous with horizons. One horizon may consist of multiple layers or one layer may include multiple horizons.
7. If a soil meets all the requirements of multiple indicators except thickness, you can combine indicators by adding up the thicknesses of each layer that meets the requirements. You can then designate it as hydric if the thickness is as thick as the most stringent thickness requirement of the indicators it meets (see an explanation of this in the introductory information in Chapter 3 of your regional supplement for a more thorough explanation).
8. Chromas should not be rounded. If the chroma appears to be between color chips, indicate that by using a + or a decimal point. Some indicators require a chroma of x or less. Others require a chroma less than x. In the former, if the color is between the required chroma and a higher chroma, it does not meet the requirement. In the latter, if the color is between the listed chroma and the next lower chroma, it does meet the requirement.
9. In LRRs R, W, X, and Y, observations begin at the top of the mineral surface (underneath any and all fibric, hemic, and/or sapric material) except for application of indicators A1, A2, and A3, where observations begin at the actual soil surface. In LRRs F, G, H, and M, observations begin at the actual soil surface if the soil is sandy and for the application of indicators A1, A2, and A3; and at the muck or mineral surface for the remaining Field Indicators. In the remaining LRRs, observations begin at the top of the muck or mineral surface (underneath any fibric and/or hemic material) except for application of indicators A1, A2, and A3 where observations begin at the actual soil surface.
10. Except for indicators A16, S6, S11, F8, F12, F19, F20, and F21 (those indicators that do not require a chroma ≤ 2 to meet the indicator), any soil material above the indicator must be a chroma ≤ 2 or if the chroma is >2 it must be less than 15 cm (6 in.) thick.
11. Both the definition of a depleted matrix and a gleyed matrix require values ≥ 4. This is to separate redox colors from organic matter accumulation colors. A, E and calcic horizons require ≥ 2 % concentrations.
12. Remember to describe organic features such as type (peat, mucky peat, or peat), color, mucky modified mineral, and percent masking of sand grains.

It is critical that the practitioner be familiar with the general rules required for using the Field Indicators. There are situations where a soil may meet all the requirements of the Field Indicator, however, it is not a hydric soil based on that Field Indicator because it has failed one of the general rules. The most common

Fig. 4.13 Two hydric soils. The soil on the *left meets* Field Indicator S7 Dark Surface. The soil on the *right meets* Field Indicator F3 Depleted Matrix (Published with kind permission of US Department of Agriculture, Natural Resources Conservation Service (2010a). Figure is public domain in the USA. All Rights Reserved)

example is a soil that meets all the requirements of F3 Depleted Matrix, but the layer that meets the requirements starts below 15 cm (6 in.) and the matrix chroma above the layer is a chroma >2. While the soil does meet all the specific requirements of the Field Indicator, it fails the rule that any soil material above the indicator must have a matrix chroma ≤ 2 or, if the matrix chroma is >2 it must be <15 cm thick.

It is helpful for the practitioner to review the Corps Regional Supplement for the geographic area in question and identify those Field Indicators applicable in their LRR or MLRA. It may also be helpful to create a one page cheat sheet of the Field Indicators that only lists the indicator descriptions for those identified for the region in question. However, when learning to use the Field Indicators it is helpful to have the user notes and glossary handy for referral when attempting to use a Field Indicator. The sheer number of Field Indicators that that can be used in a region can be intimidating to the novice. However, with experience, the practitioner will find that a small number of them are used for the majority of hydric soil identifications. The remainder of Field Indicators are used in areas that are obviously wet and not near the hydric soil boundary or for areas or specific situations that did not have commonly used Field Indicators. Nationwide, the commonly used Field Indicators are A11 Depleted Below Dark Surface, F3 Depleted Matrix (Fig. 4.13), F6 Redox Dark Surface, S5 Sandy Redox, and S7 Dark Surface (Fig. 4.12).

4.11 Soil Surveys and Hydric Soil Lists

Soil surveys are available for most areas and can provide useful information regarding soil properties and soil moisture conditions. A list of available soil surveys is located at http://soils.usda.gov/survey/online_surveys/. Soil maps and data are available online at http://websoilsurvey.nrcs.usda.gov/. Soil survey maps divide the landscape into areas called map units. Map units usually contain more than one soil type or component. They often contain several minor components or inclusions of soils with properties that may be similar to or quite different from the major component. Those soils that are hydric are noted in the Hydric Soils List.

Hydric Soils Lists are developed for each detailed soil survey based on criteria to identify soil map unit components that are at least in part hydric (Federal Register [FR Doc. 2012–4733], 2012). These lists rate each soil component as either hydric or non-hydric based on soil property data. If the soil is rated as hydric, information is provided regarding whether the soil meets the definition due to saturation, flooding, or ponding; and on what landform the soil typically occurs. Hydric Soils Lists are useful to identify areas likely to contain hydric soils. However, not all areas within a mapping unit or polygon identified as having hydric soils may be hydric. Conversely, inclusions of hydric soils may be found within soil mapping units where no hydric soils have been identified.

Soil survey information can be valuable during preliminary data gathering and synthesis. Landscape relationships and other information that can help identify the location of the component of the map unit that is hydric vs. non-hydric is also helpful. Local Hydric Soils Lists are available from state or county NRCS offices and over the internet from the Field Office Technical Guide, Section 2 (http://www.nrcs.usda.gov/technical/efotg/index.html) or Soil Data Mart (http://soildatamart.nrcs.usda.gov/). Local Hydric Soils Lists have been compiled into a National Hydric Soils List and are available at: http://soils.usda.gov/use/hydric/.

4.12 Soils That Lack Hydric Soil Indicators

4.12.1 Overview

As stated earlier, the requirements of a hydric soil are those presented in the definition. Field Indicators were created to assist in identifying those soils that meet the definition. However, the Field Indicators do not replace or relieve any of those requirements. If it meets a Field Indicator, it has morphology that indicates that the soil meets the hydric soils definition and therefore is a hydric soil. However, if it does not meet a Field Indicator, it may still be a hydric soil if it meets the requirements in the defintion. Hydric Soils Lists and the Hydric Soil Technical Standard are two approaches that may lead to an assessment of a soil as hydric by definition even though it does not meet a Field Indicator.

4.12.2 Quick Identification of Soils That Lack Field Indicators of Hydric Soils

The following key was created to identify soils that will definitely not meet a Field Indicator. This key does not identify soils that are not hydric soils. However, the identification of soils that cannot meet a Field Indicator saves field time by eliminating the more tedious process of identifying a Field Indicator.

A quick way to identify these soils is to dig a hole to 6 in. (15 cm) and address the following questions:

1. Do organic soil materials or mucky modified layers exist?
2. Do chromas ≤ 2 exist?
3. Are there any distinct or prominent redox concentrations as soft masses or pore linings?
4. Is it a sandy soil with stripped zones?
5. Is there a hydrogen sulfide odor?
6. Are you in red parent material, a depression, on a floodplain, or within 200 m (656 ft.) of an estuarine marsh and 1 m (3.3 ft.) of mean high water?

If answer is no to all five questions, the soil will not meet an indicator. This does not mean the soil is not hydric. If the soil meets the definition of a hydric soil but fails this test it only means it will not meet a Field Indicator.

4.12.3 Problematic Soil Situations

There are many problematic soil situations that currently lack an appropriate Field Indicator. Chapter 5 of the Corps Regional Supplements suggests methods to assist in the identification of a hydric soil in these problematic situations. Also, at the end of Chapter 3 of each Regional Supplement is a list of test indicators that may help in problematic soil situations. Test indicators are Field Indicators that show potential but have not been approved by the NTCHS.

If no hydric soil indicator is present, the additional site information below may be useful in documenting whether the soil is indeed non-hydric or if it might represent a "problem" hydric soil that meets the hydric soil definition despite the absence of indicators. Addressing the following questions can aid in the identification of problematic soil situations.

1. *Hydrology* – Is standing water observed on the site or is water observed in the soil pit? What is the depth of the water table in the area? Is there indirect evidence of ponding or flooding? Is the site adjacent to a downcut or channelized stream? Is the hydrology impacted by ditches or subsurface drainage lines?
2. *Slope* – Is the site level or nearly level so that surface water does not run off readily, or is it steeper where surface water would run off from the soil?

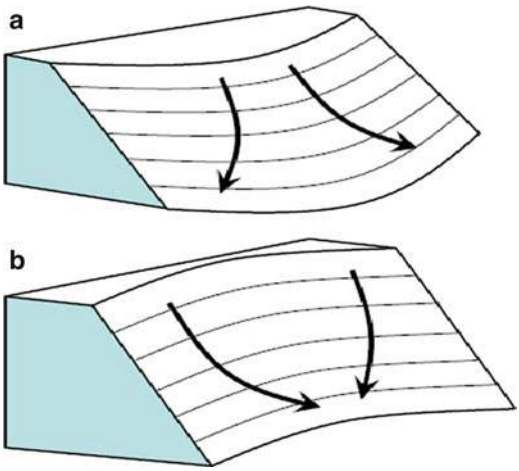

Fig. 4.14 Divergent slopes (**a**) disperse surface water, whereas convergent slopes (**b**) concentrate water. Surface flow paths are indicated by *arrows*

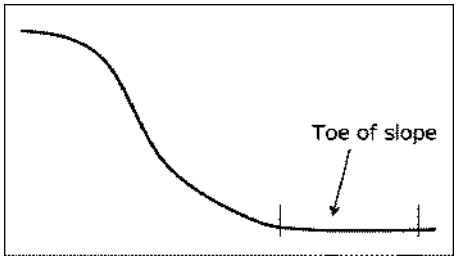

Fig. 4.15 At the toe of a hill slope the gradient is slightly inclined or nearly level

3. *Slope shape* – Is the surface concave (e.g., depressions), where water would tend to collect and possibly pond on the soil surface? On hillsides, are there convergent slopes (Fig. 4.14), where surface or groundwater may be directed toward a central stream or swale? Or is the surface or slope shape convex (e.g., dome shaped), causing water to run off or disperse?
4. *Landform* – Is the soil on a low terrace or floodplain that may be subject to seasonal high water tables or flooding? Is it at the toe of a slope (Fig. 4.15) where runoff may tend to collect or groundwater emerges at or near the surface? Has the microtopography been altered by cultivation?
5. *Soil materials* – Is there a restrictive layer in the soil that would slow or prevent the infiltration of water? This could include consolidated bedrock, compacted layers, cemented layers such as duripans and petrocalcic horizons, layers of silt or substantial clay content, seasonal ice, or strongly contrasting soil textures (e.g., silt over sand). Platy or prismatic soil structure may also result in restrictive layers. Is there relatively loose soil material (sand, gravel, or rocks) or fractured bedrock that would allow the water to flow laterally down slope?
6. *Vegetation* – Does the vegetation at the site indicate wetter conditions than at other nearby sites, or is it similar to what is found at nearby upland sites?

4.12.4 Hydric Soils Technical Standard

For a problematic site that requires monitoring to determine the presence of a hydric soil, the Hydric Soils Technical Standard (NTCHS 2007) is used as guidance for what data must be collected to satisfy the requirements of a hydric soil. In addition, the Hydric Soils Technical Standard can be used to:

1. evaluate the function of wetland restoration, mitigation, creation, and construction,
2. evaluate onsite the current functional hydric status of a soil, and
3. with appropriate regional data, modify, validate, eliminate, or adopt Field Indicators for the region.

The Hydric Soils Technical Standard includes requirements to determine that the soils are saturated, ponded, or flooded through water table monitoring and proof that the soils are anaerobic and reducing. Saturation (or inundation) and anaerobic conditions must be present for at least 14 consecutive days. It should be noted that the growing season is assumed to have started when the soil goes anaerobic since the conditions occur when soil microbes are active. Saturation is confirmed by the presence of free water in a piezometer installed to a soil depth of 25 cm (10 in.). Anaerobic conditions are confirmed by direct measurement of Eh, alpha, alpha-dipyridyl dye, or IRIS tubes. Refer to Chap. 7, *Wetland Biogeochemistry Techniques*, for more details on confirmation of anaerobic conditions. For more information on the use of the Hydric Soil Technical Standard see the NTCHS Technical Note 11 at ftp://ftp-fc.sc.egov.usda.gov/NSSC/Hydric_Soils/note11.pdf.

4.12.5 Normal Rainfall

Any data collected to evaluate hydric soils should be correlated to rainfall. Normal rainfall data, for wetland purposes, are available in NRCS National Weather and Climate Center WETS (wetlands determination) tables. WETS tables are produced for local weather stations throughout the United States. They can be accessed at http://efotg.sc.egov.usda.gov//efotg_locator.aspx. Pick your state and then county of interest. The Field Office Tech Guide menu tree will appear. Pick Section II from the drop down menu, then open the climate tab, and select AgCIS (Climate Information System). Select WETS as the product and then it will give you a list of weather stations that are available for that area. Select the weather station most appropriate for your location and then go and the WETS table will be generated.

To evaluate if a given year has had normal precipitation, local rainfall data (either from a local weather station or from an onsite rain gauge) are compared to data in the geographically appropriate WETS table. Rainfall is normal for any given month if the amount of rain falls between the values for that month in the columns “30 percent chance will have less than” and “30 percent chance will have more

than." Water table depths for a given time period are impacted not only by precipitation during that timeframe but also by precipitation in the preceding months; therefore, any evaluation of rainfall data for a given time period should also include consideration of the precipitation patterns prior to the time period of interest. For example, the NTCHS recommends the evaluation of precipitation data for the 3 months prior to the period when the soil in question is most saturated and reduced (NTCHS 2007).

4.13 Other Uses for Soil Morphology Information

4.13.1 Monitoring and Interpreting Wetland Hydrology

4.13.1.1 Field Indicators of Hydric Soils

All wetlands by definition experience saturation in the upper part of the soil for at least part of the year in a majority of years. However, wetlands display a wide range in hydroperiods, from peraquic moisture regimes (continuously saturated) such as tidal marshes to seasonally saturated wetlands such as many mineral soil flats. Some wetlands are inundated for extended periods in most years, others are rarely inundated. In addition, some wetlands such as groundwater driven slope wetlands display a static water table with a consistent depth. Others, such as precipitation driven mineral soil flats display a wide range in water table depths and multiple fluctuations in depth each year.

Soil morphology typically reflects long term hydrologic conditions. Field Indicators were developed to identify soils that developed under hydrologic conditions associated with wetlands. Hydric soils are as diverse as wetlands. Therefore, the Field Indicators represent a range in hydrologic conditions and individual indicators represent a more limited range in hydrologic conditions. Soil scientists recognize this relationship and associate specific indicators with certain hydroperiods. For example, F3 Depleted Matrix is based on the reduction and translocation of Fe, not the accumulation of organic matter. Conversely, A3 Black Histic is based on the accumulation of organic matter. Development of a histic epipedon or Histisols requires longer periods of saturation than the reduction and translocation of iron. For example, A3 Black Histic is found in wetlands that are inundated for extensive periods; whereas, F3 Depleted Matrix is found in wetlands that are rarely inundated and have a very dynamic water table. Therefore, the Field Indicators can be used not only to identify a hydric soil, but also to characterize wetland hydroperiods.

Soil colors can be used to distinguish between episaturation and endosaturation. By definition, episaturation is characterized by two layers of saturated soil separated by an unsaturated zone. Horizons that are saturated for extended periods are typically characterized by low chroma colors. In a peraquic moisture regime,

soils will have consistently low chromas with little change with depth. In an endosaturated soil that is rarely inundated, matrix chromas gradually decrease to ≤ 2 with depth. A horizon with chromas ≤ 2 directly above a horizon with high chromas indicates episaturation. In most cases, a physical distinction (change in texture or structure) between the two adjacent horizons will be apparent.

4.13.1.2 Monitoring Well Installation

A soil description should accompany the installation of monitoring wells or piezometers. Soil characteristics can impact the proper depth of well installation and may be needed to interpret the well data. For these purposes, the most important characteristics are color, texture, and structure. The identification of soil horizons that may restrict water movement is critical, and episaturation should be distinguished from endosaturation. A common scenario is a precipitation driven depressional wetland which maintains wetland hydrology through episaturation. At times, there will be two water tables-a perched water table and the deeper apparent water table. The water table may drop significantly during the growing season but the soil close to the surface may remain saturated. Installation of a well to a depth below the perched zone will result in misleading data as the wetland will appear to have a dry hydroperiod. If episaturation is suspected, it is best to install two wells, one above and one below the horizon that is restricting water flow. Water may perch directly above an aquitard; a soil layer that transmits water very slowly or not at all. Aquitards can often be identified by high bulk densities or by platy or prismatic structure. Perching can also be due to relatively small differences in texture in adjacent horizons. Free water below the aquitard may have positive pressure. If a well is installed through the aquitard, water will rise in the well to an elevation above the water table, again resulting in misleading data.

4.13.2 Assessing Changes in Wetland Hydrology

4.13.2.1 Field Indicators of Hydric Soils

As stated previously, soil morphology typically reflects long term hydrologic conditions. Draining a wetland will not cause rapid changes in morphology. Some organic matter decomposition will occur and subtle changes in redoximorphic features can occur. However, it is difficult to distinguish between drained and undrained versions of the same soil. This is a limitation to the Field Indicators. However, morphological stability of hydric soils can be used to determine if the hydrology of a wetland has been altered when used in conjunction with direct assessment of hydrology such as monitoring well data. For example, consider a

wetland with a static water table that is inundated continuously for several months yet has a soil meeting the Field Indicator F3 Depleted Matrix. Wetland hydrology during the monitoring period does not match the long term hydrology as the site is now wetter. Now consider a wetland with a dynamic water table displaying seasonal saturation and multiple fluctuations in water table depth but the soil meets the indicator A3 Black Histic. That site is now drier. In this process local precipitation data should be considered to distinguish between permanent changes in hydroperiod and unusual short term precipitation patterns.

4.13.2.2 Soil Structure and Horizonation

A number of soil morphological features are associated with dynamic water tables. Redoximorphic features were discussed previously. Subangular and angular blocky structure is believed to be caused by forces created by alternating periods of wetting and drying. We do not expect to see these structural types in soil that consistently stays wet. Similarly, argillic horizons, a B horizon enriched with illuviated clay, do not form in soils with a static water table near the surface as a fluctuating water table is required to transport clay vertically. Therefore, if hydrologic monitoring indicates a static water table near the surface, but the soil has an argillic horizon or strong blocky structure, the site is now wetter.

4.14 Additional Resources

4.14.1 National Technical Committee for Hydric Soils (NTCHS)

The NTCHS is chaired by NRCS and has representation from all federal agencies involved in wetlands work as well as university experts in hydric soils related issues. The NTCHS makes all decisions on issues related to hydric soils.

4.14.2 Documents

National Technical Committee for Hydric Soils 2007. The Hydric Soil Technical Standard. Hydric Soils Tech Note 11. (http://soils.usda.gov/use/hydric/ntchs/tech_Notes/index.html) – The standard required to collect long term data to

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4.14.3 Websites

http://soils.usda.gov/use/hydric/ – This is the NTCHS website and contains all official technical information regarding hydric soils and hydric soil issues.
http://soils.usda.gov/technical/ – This site is the NRCS Soil Survey Division technical resources website and contains all NRCS technical references pertaining to soil survey issues.
http://www.usace.army.mil/Missions/CivilWorks/RegulatoryProgramandPermits/reg_supp.aspx – This is the website to obtain copies of the Corps of Engineers Wetland Delineation Regional Supplements. For information on hydric soils, go to Chapter 3 in your local suppplement and Chapter 5 for problematic soil situations.
http://soils.usda.gov/education/resources/lessons/texture/ – This site is from the USDA, NRCS and includes the Textural Triangle & the Guide to Texture by Feel.

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Student Exercises

Classroom Exercises

Classroom Exercise #1: Sources of Hydric Soil Information

Objective: To familiarize students with sources of hydric soils information.

Procedures:

Take a moment to explore the following sources of hydric soils information derived from the soil survey for your selected site location.

1. Go to the NTCHS website (http://soils.usda.gov/use/hydric/) and look up the national Hydric Soil List and find the soil survey area your site is located in.
   (a) Note the soil survey area which is located in column B.
   (b) Column D contains the map unit symbol as it is mapped in the official soil survey.
   (c) Column E contains the map unit name as it is named in the official soil survey.
   (d) Column F contains the component name(s) that are hydric soil components of that map unit. Note that some components are major components that are named in the map unit name and others are minor components (inclusions) within the map unit.
   (e) Column G contains the percentage of that map unit that contains a hydric soil component.

(f) Column H contains the landform in which you will find the hydric component.
(g) Column I provides the criteria that was used to establish this component as hydric.

2. Now go to the Soil Data Mart (http://soildatamart.nrcs.usda.gov/) and look up the local Hydric Soil List for your survey area.

(a) Click on "select state" and select the state you are in.
(b) Click on "select survey area" and select the one you are interested in.
(c) Click on "generate reports" and click on "select all" to select all the map units in the survey area. This should highlight all the map units. You can also select individual map units if you do not want a list of the whole county.
(d) From the pull down menu of reports, select "hydric soils" and click on "generate report".
(e) It will take a moment for the hydric soil report to come up.
(f) This report gives you the same information that is presented in the national Hydric Soil List, however, this report is updated any time information is updated in Soil Data Mart, while the national Hydric Soil List is only updated about once a year. There may be differences if an update to Soil Data Mart was made after the national Hydric Soil List was generated. Soil Data Mart should be your source for official (and current) soil survey information.

3. Now go to Web Soil Survey (http://websoilsurvey.nrcs.usda.gov/)

(a) Click on "start WSS".
(b) Locate your area of interest. You can do this by: selecting state and county and then zooming in by clicking the icon at the top of the map with a magnifying glass with a + sign in the upper left of the map screen, soil survey area and zooming into your area of interest, typing in an address, using latitude and longitude, or any of the other methods listed (Use the help menu in Web Soil Survey for more detailed instructions.).
(c) Once zoomed in, click on either the area of interest rectangle icon or the area of interest icon that allows an irregular shape. These icons are located in the upper left of the map screen.
(d) Outline the area of interest on the map and then click on the "Soil Data Explorer" tab. You should now see a soils map overlaying your area of interest.
(e) Under the tab "Suitability and Limitations for Use" click on "Land Classification", then "Hydric Rating by Map Unit", and then view rating.
(f) This will produce a map that shows map units that are hydric, partially hydric (contain components that are both hydric and non-hydric), or not hydric.
(g) Click on "Add to Shopping Cart" in the upper right and then ok.
(h) Click on the "Shopping Cart" tab at the top, and then "Check Out" in the upper right and then ok.
(i) Now you have a customized .pdf soil survey report for your area of interest that contains the Hydric Soils List information from Soil Data Mart along with a map categorizing those map units that are hydric, partially hydric, or non-hydric. You can save this to your computer and/or print the file for use when you do your on-site investigation exercise a little later.

Classroom Exercise #2: Identification of Field Indicators of Hydric Soils

Objective: To familiarize students with the use of Field Indicators of Hydric Soils.

Proceedures:

Below are examples of data sheets that are completely filled out. Go through the latest version of Field Indicators of Hydric Soils and list all indicators that are met for each of the descriptions.

SOIL Sampling Point: 1

Profile Description: (Describe to the depth needed to document the indicator or confirm the absence of indicators.)

Depth (inches)	Matrix Color (moist)	%	Redox Features Color (moist)	%	Type[1]	Loc[2]	Texture	Remarks
0-8	10YR3/2	90	7.5YR5/4	10	C	PL	SL	
8-24	2.5Y6/1	88	7.5YR5/4	12	C	M	SL	

[1]Type: C=Concentration, D=Depletion, RM=Reduced Matrix, CS=Covered or Coated Sand Grains. [2]Location: PL=Pore Lining, M=Matrix.

Hydric Soil Indicators: (Applicable to all LRRs, unless otherwise noted.)

- ___ Histosol (A1)
- ___ Histic Epipedon (A2)
- ___ Black Histic (A3)
- ___ Hydrogen Sulfide (A4)
- ___ Stratified Layers (A5) (**LRR F**)
- ___ 1 cm Muck (A9) (**LRR F, G, H**)
- ___ Depleted Below Dark Surface (A11)
- ___ Thick Dark Surface (A12)
- ___ Sandy Mucky Mineral (S1)
- ___ 2.5 cm Mucky Peat or Peat (S2) (**LRR G, H**)
- ___ 5 cm Mucky Peat or Peat (S3) (**LRR F**)
- ___ Sandy Gleyed Matrix (S4)
- ___ Sandy Redox (S5)
- ___ Stripped Matrix (S6)
- ___ Loamy Mucky Mineral (F1)
- ___ Loamy Gleyed Matrix (F2)
- ___ Depleted Matrix (F3)
- ___ Redox Dark Surface (F6)
- ___ Depleted Dark Surface (F7)
- ___ Redox Depressions (F8)
- ___ High Plains Depressions (F16) (**MLRA 72 & 73 of LRR H**)

Indicators for Problematic Hydric Soils[3]:

- ___ 1 cm Muck (A9) (**LRR I, J**)
- ___ Coast Prairie Redox (A16) (**LRR F, G, H**)
- ___ Dark Surface (S7) (**LRR G**)
- ___ High Plains Depressions (F16) (**LRR H outside of MLRA 72 & 73**)
- ___ Reduced Vertic (F18)
- ___ Red Parent Material (TF2)
- ___ Other (Explain in Remarks)

[3]Indicators of hydrophytic vegetation and wetland hydrology must be present, unless disturbed or problematic.

Restrictive Layer (if present):

Type: None

Depth (inches): _____

Hydric Soil Present? Yes _____ No _____

Remarks:

Landscape: Mineral flat

SOIL Sampling Point: 2

Profile Description: (Describe to the depth needed to document the indicator or confirm the absence of indicators.)

Depth (inches)	Matrix Color (moist)	%	Redox Features Color (moist)	%	Type[1]	Loc[2]	Texture	Remarks
0-8	10YR4/3	90	7.5YR4/6	10	C	PL	L	
8-24+	2.5Y5/1	82	7.5YR5/4	18	C	M	SL	

[1]Type: C=Concentration, D=Depletion, RM=Reduced Matrix, CS=Covered or Coated Sand Grains. [2]Location: PL=Pore Lining, M=Matrix.

Hydric Soil Indicators: (Applicable to all LRRs, unless otherwise noted.)

- ___ Histosol (A1)
- ___ Histic Epipedon (A2)
- ___ Black Histic (A3)
- ___ Hydrogen Sulfide (A4)
- ___ Stratified Layers (A5) **(LRR F)**
- ___ 1 cm Muck (A9) **(LRR F, G, H)**
- ___ Depleted Below Dark Surface (A11)
- ___ Thick Dark Surface (A12)
- ___ Sandy Mucky Mineral (S1)
- ___ 2.5 cm Mucky Peat or Peat (S2) **(LRR G, H)**
- ___ 5 cm Mucky Peat or Peat (S3) **(LRR F)**
- ___ Sandy Gleyed Matrix (S4)
- ___ Sandy Redox (S5)
- ___ Stripped Matrix (S6)
- ___ Loamy Mucky Mineral (F1)
- ___ Loamy Gleyed Matrix (F2)
- ___ Depleted Matrix (F3)
- ___ Redox Dark Surface (F6)
- ___ Depleted Dark Surface (F7)
- ___ Redox Depressions (F8)
- ___ High Plains Depressions (F16) **(MLRA 72 & 73 of LRR H)**

Indicators for Problematic Hydric Soils[3]:

- ___ 1 cm Muck (A9) **(LRR I, J)**
- ___ Coast Prairie Redox (A16) **(LRR F, G, H)**
- ___ Dark Surface (S7) **(LRR G)**
- ___ High Plains Depressions (F16) **(LRR H outside of MLRA 72 & 73)**
- ___ Reduced Vertic (F18)
- ___ Red Parent Material (TF2)
- ___ Other (Explain in Remarks)

[3]Indicators of hydrophytic vegetation and wetland hydrology must be present, unless disturbed or problematic.

Restrictive Layer (if present):

Type: None

Depth (inches): _____

Hydric Soil Present? Yes _____ No _____

Remarks:

Landscape: Mineral flat

SOIL Sampling Point: 3

Profile Description: (Describe to the depth needed to document the indicator or confirm the absence of indicators.)

Depth (inches)	Matrix Color (moist)	%	Redox Features Color (moist)	%	Type[1]	Loc[2]	Texture	Remarks
0-8	10YR4/4	90	7.5YR4/6	10	C	PL	SiL	
8-18+	10YR5/6	100					SiL	

[1]Type: C=Concentration, D=Depletion, RM=Reduced Matrix, CS=Covered or Coated Sand Grains. [2]Location: PL=Pore Lining, M=Matrix.

Hydric Soil Indicators: (Applicable to all LRRs, unless otherwise noted.)

- ___ Histosol (A1)
- ___ Histic Epipedon (A2)
- ___ Black Histic (A3)
- ___ Hydrogen Sulfide (A4)
- ___ Stratified Layers (A5) **(LRR F)**
- ___ 1 cm Muck (A9) **(LRR F, G, H)**
- ___ Depleted Below Dark Surface (A11)
- ___ Thick Dark Surface (A12)
- ___ Sandy Mucky Mineral (S1)
- ___ 2.5 cm Mucky Peat or Peat (S2) **(LRR G, H)**
- ___ 5 cm Mucky Peat or Peat (S3) **(LRR F)**
- ___ Sandy Gleyed Matrix (S4)
- ___ Sandy Redox (S5)
- ___ Stripped Matrix (S6)
- ___ Loamy Mucky Mineral (F1)
- ___ Loamy Gleyed Matrix (F2)
- ___ Depleted Matrix (F3)
- ___ Redox Dark Surface (F6)
- ___ Depleted Dark Surface (F7)
- ___ Redox Depressions (F8)
- ___ High Plains Depressions (F16) **(MLRA 72 & 73 of LRR H)**

Indicators for Problematic Hydric Soils[3]:

- ___ 1 cm Muck (A9) **(LRR I, J)**
- ___ Coast Prairie Redox (A16) **(LRR F, G, H)**
- ___ Dark Surface (S7) **(LRR G)**
- ___ High Plains Depressions (F16) **(LRR H outside of MLRA 72 & 73)**
- ___ Reduced Vertic (F18)
- ___ Red Parent Material (TF2)
- ___ Other (Explain in Remarks)

[3]Indicators of hydrophytic vegetation and wetland hydrology must be present, unless disturbed or problematic.

Restrictive Layer (if present):

Type: None

Depth (inches): _____

Hydric Soil Present? Yes _____ No _____

Remarks:

Landscape: Closed depression subject to ponding.

SOIL Sampling Point: 4

Profile Description: (Describe to the depth needed to document the indicator or confirm the absence of indicators.)

Depth (inches)	Matrix Color (moist)	%	Redox Features Color (moist)	%	Type[1]	Loc[2]	Texture	Remarks
0-14	2.5Y 2.5/1	100					S	
14-18	10YR 4/6	100					S	
18-24+	2.5Y 6/1	100					S	

[1]Type: C=Concentration, D=Depletion, RM=Reduced Matrix, CS=Covered or Coated Sand Grains. [2]Location: PL=Pore Lining, M=Matrix.

Hydric Soil Indicators: (Applicable to all LRRs, unless otherwise noted.)

- ___ Histosol (A1)
- ___ Histic Epipedon (A2)
- ___ Black Histic (A3)
- ___ Hydrogen Sulfide (A4)
- ___ Stratified Layers (A5) (**LRR F**)
- ___ 1 cm Muck (A9) (**LRR F, G, H**)
- ___ Depleted Below Dark Surface (A11)
- ___ Thick Dark Surface (A12)
- ___ Sandy Mucky Mineral (S1)
- ___ 2.5 cm Mucky Peat or Peat (S2) (**LRR G, H**)
- ___ 5 cm Mucky Peat or Peat (S3) (**LRR F**)
- ___ Sandy Gleyed Matrix (S4)
- ___ Sandy Redox (S5)
- ___ Stripped Matrix (S6)
- ___ Loamy Mucky Mineral (F1)
- ___ Loamy Gleyed Matrix (F2)
- ___ Depleted Matrix (F3)
- ___ Redox Dark Surface (F6)
- ___ Depleted Dark Surface (F7)
- ___ Redox Depressions (F8)
- ___ High Plains Depressions (F16) (**MLRA 72 & 73 of LRR H**)

Indicators for Problematic Hydric Soils[3]:

- ___ 1 cm Muck (A9) (**LRR I, J**)
- ___ Coast Prairie Redox (A16) (**LRR F, G, H**)
- ___ Dark Surface (S7) (**LRR G**)
- ___ High Plains Depressions (F16) (**LRR H outside of MLRA 72 & 73**)
- ___ Reduced Vertic (F18)
- ___ Red Parent Material (TF2)
- ___ Other (Explain in Remarks)

[3]Indicators of hydrophytic vegetation and wetland hydrology must be present, unless disturbed or problematic.

Restrictive Layer (if present):

Type: None

Depth (inches): _____

Hydric Soil Present? Yes _____ No _____

Remarks:
Landscape: Mineral flat

SOIL Sampling Point: 5

Profile Description: (Describe to the depth needed to document the indicator or confirm the absence of indicators.)

Depth (inches)	Matrix Color (moist)	%	Redox Features Color (moist)	%	Type[1]	Loc[2]	Texture	Remarks
0-2	10YR 3/1	100					MK-PT	
2-14	2.5Y 3/1	100					SiL	
14-28+	2.5Y 6/1	90	7.5YR 4/6	10	C	M	SiL	

[1]Type: C=Concentration, D=Depletion, RM=Reduced Matrix, CS=Covered or Coated Sand Grains. [2]Location: PL=Pore Lining, M=Matrix.

Hydric Soil Indicators: (Applicable to all LRRs, unless otherwise noted.)

- ___ Histosol (A1)
- ___ Histic Epipedon (A2)
- ___ Black Histic (A3)
- ___ Hydrogen Sulfide (A4)
- ___ Stratified Layers (A5) (**LRR F**)
- ___ 1 cm Muck (A9) (**LRR F, G, H**)
- ___ Depleted Below Dark Surface (A11)
- ___ Thick Dark Surface (A12)
- ___ Sandy Mucky Mineral (S1)
- ___ 2.5 cm Mucky Peat or Peat (S2) (**LRR G, H**)
- ___ 5 cm Mucky Peat or Peat (S3) (**LRR F**)
- ___ Sandy Gleyed Matrix (S4)
- ___ Sandy Redox (S5)
- ___ Stripped Matrix (S6)
- ___ Loamy Mucky Mineral (F1)
- ___ Loamy Gleyed Matrix (F2)
- ___ Depleted Matrix (F3)
- ___ Redox Dark Surface (F6)
- ___ Depleted Dark Surface (F7)
- ___ Redox Depressions (F8)
- ___ High Plains Depressions (F16) (**MLRA 72 & 73 of LRR H**)

Indicators for Problematic Hydric Soils[3]:

- ___ 1 cm Muck (A9) (**LRR I, J**)
- ___ Coast Prairie Redox (A16) (**LRR F, G, H**)
- ___ Dark Surface (S7) (**LRR G**)
- ___ High Plains Depressions (F16) (**LRR H outside of MLRA 72 & 73**)
- ___ Reduced Vertic (F18)
- ___ Red Parent Material (TF2)
- ___ Other (Explain in Remarks)

[3]Indicators of hydrophytic vegetation and wetland hydrology must be present, unless disturbed or problematic.

Restrictive Layer (if present):

Type: None

Depth (inches): _____

Hydric Soil Present? Yes _____ No _____

Remarks:
Landscape: Mineral flat

Answers

Sampling point 1: This description meets A11 Depleted Below Dark Surface, F3 Depleted Matrix, and F6 Redox Dark Surface.

Sampling point 2: No indicator is met. Note that this description would meet F3 Depleted Matrix. However, it fails the general rule that you cannot have 15 cm (6 in.) or more of a matrix chroma higher than 2 above the depleted matrix (indicator).

Sampling point 3: F8 Redox Depressions. Note that this is one of the landscape specific indicators that allows matrix chromas higher than 2. In this case, it can only be used in soils that occur in closed depressions subject to ponding. Examples are vernal pools, playa lakes, rainwater basins, "Grady" ponds and potholes.

Sampling point 4: S7 Dark Surface. This is an example of where horizons and layers are not synonymous. To meet this indicator, you must have 10 cm (4 in.) with a value of 3 or less and a chroma of 1 or less. Immediately below the 10 cm (still in the same horizon), you meet the next layer requirement with a chroma of 2 or less. However, if you went to the next horizon instead of looking immediately below the 10 cm layer, you would not meet this indicator since it has a chroma higher than 2. S7 is not an approved indicator for all LRRs. When identifying indicators in a real world scenario you would need to identify that you are in an approved LRR before using this indicator.

Sampling point 5: Answer: A11 Depleted Below Dark Surface. Note that if you incorrectly start your measurements at the actual soil surface instead of the mineral surface (at 5 cm), the depleted matrix would be too deep to meet this indicator.

Laboratory Exercises

Laboratory Exercise: Description and Identification of Hydric Soils in the Field

Overview: The following field exercise is intended to allow you to use the skills you have learned to make a hydric soils determination in the field. If you are a novice to writing soil descriptions, you may want to seek out assistance from a soil or wetland scientist with more experience for assistance. Before you go to the field, you will need to gather the soil report you created earlier in Web Soil Survey for the area you will be using for the excercise, a copy of the *Field Indicators of Hydric Soils in the United States* (Version 7.0), a copy of Chapter 3 of your local Corps Regional Supplement, and the Key to Soils that Lack Field Indicators of Hydric Soils.

Objectives: To describe and identify hydric soils in the field

Materials and Equipment Needed:

1. Tiling spade or similar flat bladed shovel
2. Bucket auger if you think you may need to describe your soil to a depth greater than 45 cm (18 in.).

3. Measuring tape
4. Water for estimating soil texture
5. Knife or other tool for picking the soil surface
6. Munsell soil color chart
7. Clipboard
8. Pencil or pen
9. Data sheets
10. You may also want to contact your local Resource Soil Scientist to determine if there are any other tools you may need. For example, in areas where Mn may be used as a Field Indicator, it is useful to carry hydrogen peroxide to determine if dark mottles in the soil are in fact redox concentrations that contain Mn.

Procedures:

Once in the field, locate an area that you feel is on the wet side of the hydric/non-hydric soil boundary and fill out the soils portion of the wetland delineation data sheet completely. Go to the drier side and complete another data sheet filling out the soils portion completely. Make sure you describe all the information that you will need to identify the Field Indicators of Hydric Soils the soil might meet. For example, if you are in a sandy soil with dark matrices, it is important to record an estimate of masked vs. unmasked sand grains because this is a characteristic that can separate hydric soils from non-hydric soils. Once you have completed your descriptions, go through the Field Indicators in the field to identify all those indicators your soil meets. You should record all the Field Indicators met, although only one indicator is required. Once you become familiar with the Field Indicators, you can begin completing your descriptions in the field and going through the Field Indicators following the field visit, but for this exercise, you should attempt to determine the indicators that are met in the field in case you need to go back and identify features you may have forgotten to record. Note the importance of recording colors, soil textures, accurate depths, percentage and location of redoximorphic features, masked vs. unmasked sand grains, and, if it applies, the type of organic soil material (muck, muck peat, peat).

If you have gone through your exercise and were not able to identify a Field Indicator in a site that you feel should contain hydric soils, you may be in a problematic soil situation. The first step in identifying a problematic soil situation is to address the following questions:

1. Look at the big picture.
2. What landscape are you in?
3. Does the vegetative community make sense?
4. Are the soil characteristics what you expect?

Read the information on problematic hydric soils and then go back to the information you have gathered to determine if your site fits any of the problematic hydric soil situations described in your Regional Supplement and whether the information provided can assist you to identify the soil as hydric. If you have not already asked for assistance from the local Resource Soil Scientist, you may want to contact them to discuss whether it is likely that the site is problematic.

Chapter 5
Sampling and Analyzing Wetland Vegetation

Amanda Little

Abstract Effectively sampling and analyzing wetland vegetation is an important part of wetland science, as an indicator of wetland health and quality, and jurisdictional and mitigation success determinations. This chapter explains spatiotemporal vegetation sampling considerations by addressing key questions, such as which wetlands should be sampled and when and at what scale sampling should occur. It also plainly discusses the advantages and disadvantages of basic sampling techniques, such as different types of plot-based, plotless, and relevé systems. Methods of assessing different vegetation and environmental attributes, such as cover and functional groups are discussed in detail. The chapter then describes methods of analyzing wetland vegetation, including simple summary analyses and more complex multivariate methods, such as classification, ordination, and floristic quality indices. Explanations of different types of these analyses and their advantages and disadvantages are provided. Finally, both field and laboratory-based exercises in sampling and analysis are provided for faculty and students studying wetland vegetation.

5.1 The Importance of Wetland Vegetation

There are many reasons to investigate wetland vegetation. Aside from purely scientific interest, wetland vegetation has long been used as an indicator of wetland health and quality (U.S. EPA 2002), a basis of comparison between reference and restored or mitigated states (Matthews et al. 2009a), and as one of the three indicators of jurisdictional wetlands (Environmental Laboratory 1987). It also provides valuable ecosystem services as habitat for fish (Gabriel and Bodensteiner 2011), birds (Valente et al. 2011), amphibians (Hamer and Parris 2011), and insects (Molnar et al. 2009), and is a component of biodiversity in its own right. In addition,

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specific properties of wetland vegetation (e.g., carbon storage and uptake) are important in studies of ecosystem function. In this chapter, we will explore common techniques for sampling and analyzing wetland vegetation.

5.2 Considerations of Location and Timing – Which Wetlands and When Should They Be Sampled?

What a scientist finds depends upon where and when they look. This section will explore considerations of sample location at multiple scales: watersheds, wetlands, and zones or communities within wetlands. At each scale, randomization options and pseudoreplication considerations (discussed in Chap. 1) need to be carefully considered and applied.

5.2.1 Which Wetlands?

In many cases, the choice of study wetland is pre-determined by the goals and objectives of the study. However, if the goal is to compare wetlands or generalize about particular wetland types or conditions, the choice of study wetlands becomes the most important decision (Curtis 1959). It is best to begin by identifying the range of possible wetlands within the study's scope. Numerous free resources are available, including the National Wetlands Inventory (U.S. Fish and Wildlife Service), the U.S. Department of Agriculture Web Soil Survey (for locating areas of hydric soil), state or county-level wetlands inventories, and more detailed wetlands inventories created for specific management areas, such as cities, preserves, or forests. These inventories will provide information as to the type of wetland, its size, shape, and geographic location – typically associated with a geographic information system (GIS) map layer. The level of detail provided about wetland type ranges from basic information about the dominant strata and hydrology in the wetland (e.g., emergent, shrub, forested) in large-scale inventories (Fig. 5.1) to species-level and hydrology data provided in more small-scale inventories. In many cases, the inventory will be based upon remotely-sensed data and therefore subject to error, especially for forested wetlands, which are more difficult to detect remotely (Kudray and Gale 2000).

These inventories provide a range of possibilities for study. Practitioners who want to get an unbiased representation of different wetlands across an area could apply a stratified random sampling scheme (see Chap. 1), stratified upon type, to select study wetlands. If wetland-level attributes will be used as samples in statistical analyses, pseudoreplication should be avoided by ensuring that different wetlands are not hydrologically-connected closely within the same watershed. Alternatively, those seeking to identify representative or reference wetlands could

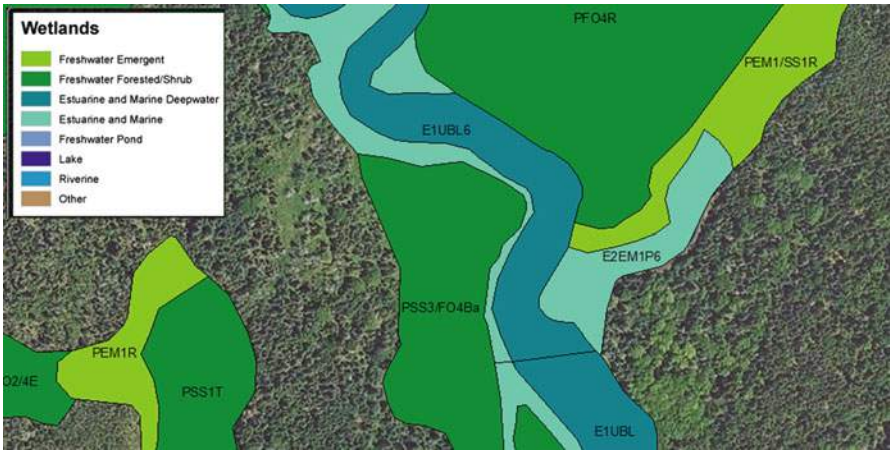

Fig. 5.1 The National Wetlands Inventory map of an estuarine system in Maine. Information includes dominant system (e.g., palustrine, estuarine), vegetation life form (e.g., shrub-scrub, emergent, forested), and limited detailed subclass modifiers (e.g., evergreen type, inundation permanence, and some water chemistry information)

simply use the inventory as a starting point for site visits. In any case, it is critical to keep the purpose of your study in mind while selecting wetlands and to avoid "reinventing the wheel" when possible.

Questions to ask yourself when choosing wetlands:

1. Is it important to have an unbiased sample of wetlands?
2. Is it important to identify representative or reference-type sites using professional judgment?

5.2.2 Where in the Wetland?

With some exceptions, (e.g., monocultures or some large peatlands), wetlands tend to have highly heterogeneous vegetation. This heterogeneity is often due to hydrologic differences at multiple different scales within the wetland. For example, some wetlands have distinctive bands, or zones of vegetation corresponding to large-scale hydrologic gradients (Fig. 5.2a). Each of these zones could be considered a different community within the wetland. Within each zone, vegetation can be further influenced by microtopographic features such as hummocks, tussocks, deep holes, or trees (Fig. 5.2b, (Ehrenfeld 1995)). Disturbance factors, such as herbivory, pollution, fire, or animal trails can lend further heterogeneity to vegetation, often in less predictable patterns.

Wetland vegetation heterogeneity can initially be explored using high-quality aerial photography, which can provide a good idea of large-scale patterns. This

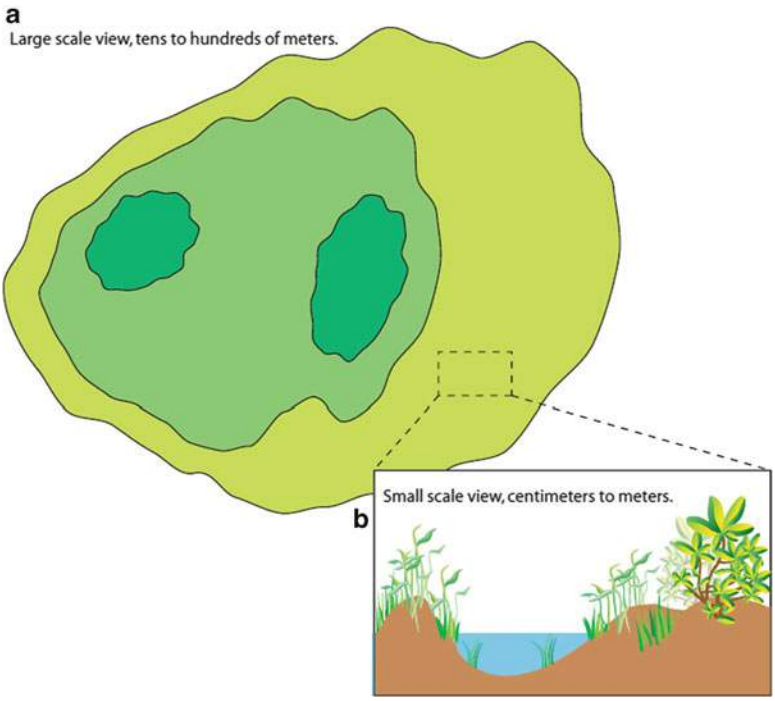

Fig. 5.2 Spatial heterogeneity created by multiple scales of environmental gradients in wetlands. (**a**) Large-scale zonation corresponding to large-scale elevation gradients. (**b**) Small-scale microtopographic heterogeneity corresponding to features like hummocks, pools, and coarse woody debris (Published with kind permission of © M. Kuchta 2014. All Rights Reserved)

photography can be obtained from multiple sources, including the National Wetlands Inventory (aerial imagery viewer), United States Department of Agriculture (USDA) Natural Resources Conservation Service (NRCS) Gateway (http://datagateway.nrcs.usda.gov), and the United States Geological Survey (USGS) EarthExplorer (http://earthexplorer.usgs.gov). Important information to gather from photos includes: wetland size and dimensions, type of strata (aquatic, understory, shrub, and/or tree), access points, and any obstacles to travel within the wetland (e.g. ponds, rivers). After locating and investigating the photography, a preliminary site visit can help identify microtopographic patterns to take into consideration, further obstacles to travel within the wetland, and unknown plant species to identify prior to intensive sampling.

Questions to ask yourself before choosing where to sample within the wetland:

1. Is it important to quantitatively describe the complete vegetation of the wetland or is a description of the vegetation in each community type sufficient?
2. Are communities easily recognizable and discrete, or do they grade into each other so that it is difficult to detect changes without quantitative data?
3. Is pseudoreplication an important consideration? That is, is it important to collect samples only from within one zone or microsite, or across many?

5.2.3 When?

Timing is extremely important in vegetation sampling. Just as there are multiple scales of spatial investigation and variability, temporal variability has multiple scales. The most common consideration in vegetation sampling is season of the year. Within the northern hemisphere, most wetland vegetation is sampled in mid to late summer in order to facilitate sedge, grass, and aster identification, and to capture vegetation at the peak of its growth. However, many early-season sedge, mint, and violet species may be difficult to identify in late summer, and several orchid species seem to disappear entirely after blooming. Within forested wetlands, spring ephemerals may be missed altogether. A good strategy is to visit the wetland site periodically early in the year in order to identify early-blooming and fruiting species and then apply this knowledge to a more comprehensive quantitative sampling later in the growing season when early-bloomers are in a non-flowering state. If a comprehensive species list for the wetland is desired, then returning to sample at multiple times during the year is necessary.

On a larger scale, interannual variability can also be important. When reporting results of vegetation study, include information about whether the climate was typical or unusual that year. Some plant species flourish or become more apparent during times of high or low water levels (Warwick and Brock 2003). Late spring freezes can temporarily eliminate a host of spring herbaceous species. If the sampling year is atypical, an additional year will probably be necessary in order to fully describe the wetland vegetation. Wetlands like prairie potholes or beaver (*Castor canadensis*) meadows experience larger-scale cycles of disturbance that can extend for decades or longer. In order to capture the full variability of these systems, it may be necessary to sample wetlands at different stages in the cycle. It is important to recognize and document the site history and temporal context of your sampling when reporting and applying results.

Questions to ask yourself before choosing when to sample:

1. Is it important to get a general description of the vegetation, or a comprehensive species list?
2. Does the system have a characteristic frequent disturbance regime that will make the results of your sampling only narrowly applicable?

5.3 Basic Vegetation Sampling Techniques

Entire books have been written about how to properly sample vegetation. What follows is an introduction to some basic techniques. For further information, the reader is encouraged to consult Greig-Smith (1983), Bonham (1989), Kent and Coker (1995), Elzinga et al. (1998), Krebs (1998), and Mueller-Dombois and Ellenberg (2003).

Before any sampling protocol can be created, the goals of the project must be clarified. Questions to ask yourself include:

- Is this a one-time assessment, or would you like to track changes in the vegetation over time?
- Are you more interested in one to a few different populations, the plant community as a whole, or in plants as a production component of an ecosystem model?
- What attributes of the vegetation are important to you? Presence or absence of species? Abundance of individual species? Vegetation cover, height or biomass? Dominance or importance of different species?
- Is it important to have quantitative data that can be used in a statistically-valid manner, or is a basic qualitative (e.g., a species list) description of the system adequate?
- Is characterizing the spatial pattern of the vegetation or having precise location information important to answering your question?

Once you have considered and answered these questions, you will be able to determine which techniques are best suited for your purpose.

5.3.1 Attributes of Vegetation

The basic building blocks of any sampling protocol are the attributes to be measured. Once a system for selecting and delimiting sample locations has been established, vegetation characteristics are assessed. Commonly-measured attributes include:

- **Presence:** Does the species occur within the plot or site? This measure can later be used to calculate the **frequency** of a species within a site (the number of plots in which the species occurred).
- **Abundance:** How much of the species occurs? This can be measured in different ways, such as by count of individuals or by visual percent-cover.
- **Production:** How much biomass is produced by different species in the plot? Root, shoot, and or total plant biomass can be measured.
- **Structure:** How tall is the vegetation or how many stems or branching points are produced? How much three-dimensional space is occupied by the plant? These types of measures can be particularly helpful when assessing habitat for animals.
- **Composition:** Which and how many different species occur within the plot?
- **Functional groups:** What is the abundance of species from different functional groups (e.g., perennial graminoid, annual forb, floating-leaf submergent)? Many attributes, such as invasive or wetland indicator status, can be assigned after sampling based upon available information (e.g., USDA Plants Database: http://plants.usda.gov).
- **Morphological characteristics:** What types of traits does each species have? Traits measured in the field include leaf number, leaf shape, specific leaf area, and flower number.

- **Dominance or importance:** Which species is most important in its influence on the community or ecosystem? This attribute is often assessed during the analysis stage by creating composite scores using different attributes, such as relative cover or density. However, attributes like height or basal area (calculated from tree diameters) may be important to collect in the field.
- **Spatial pattern:** Is the vegetation dispersed in a clumped, random, or regular pattern?

5.3.2 The Sample Unit: Plot-Based and Plotless Techniques

The choice of sample unit will depend upon the size of the plants, the resources available (time and money), and the ease of using the subsequent data to meet the specific goals for your analysis or report.

Plot-based techniques assess vegetation within an area of pre-defined size and shape. These techniques have the advantage of leading to relatively straightforward calculations of density and other summary attributes in the analysis stage, because they have a known area. The size of the plot typically relates to the size of the organisms studied. For example, a larger plot will be used to assess trees than to assess understory vegetation. In order to capture a wide variety of species in understory vegetation, plot sizes from 0.01 to 1.0 m^2 are typical and sometimes nested within each other (Elzinga et al. 1998). A common plot size for trees is 100 m^2. Plots can frequently be nested around a common point when investigating multiple strata (e.g., understory, shrubs, trees). Small plots are often called quadrats. As a general rule, the plot should be roughly twice the size of the largest organism in your sample (Greig-Smith 1983). Other researchers suggest that plots be as large as possible given time and effort constraints (Kenkel and Podani 1991). Organism spatial distribution should be considered. If plant populations are clustered, researchers should be sure that plots are not of a size that will result in numerous empty plots when they land in between plant clumps (Elzinga et al. 1998). Another good strategy for choosing plot size is to use previously-published studies. For a detailed discussion of plot size, see Elzinga et al. (1998).

The effects of plot shape have also been debated by ecologists (discussed in (Krebs 1998)). Some argue that a rectangular plot is most effective because it captures the most vegetation heterogeneity (with quadrats oriented to capture variability within plots as opposed to between). Others argue that circular or square plots are best because they minimize edge effects and require the least amount of subjective "in or out of plot" decisions on the part of the worker due to their low perimeter:area ratio. Choose the plot shape based upon the purpose of your study. If the purpose is to capture the maximum heterogeneity, a rectangular plot will be most suitable. If not, choose a square or circular plot in order to maximize accuracy and minimize edge decisions (U.S. EPA 2002).

Nested plots (where smaller plots are located within larger plots) are used for many reasons. The most common use of nested plots is to survey vegetation within

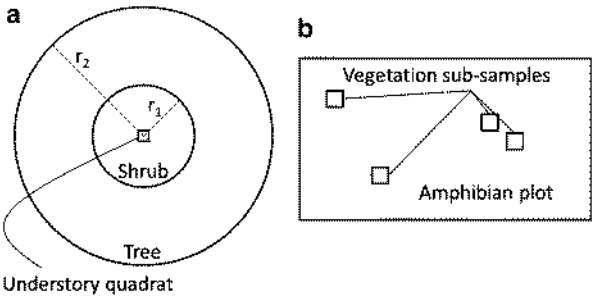

Fig. 5.3 Nested plots. (**a**) Vegetation of different strata can be sampled within differently-sized nested plots. (**b**) Different types of data can be sampled within nested plots, corresponding to the organism studied

Table 5.1 Example line intercept data sheet for a line 5.0 m long

Species	Intercepts (m)	Total length (m)	% intercept
Calamagrostis canadensis	0.5–0.8, 1.2–1.7, 2.9–3.5, 4.8–5.0	1.6	32.0
Chamaedaphne calyculata	0.7–1.3, 3.9–4.1	0.6	12.0
Myrica gale	0.0–0.6, 1.6–3.0, 3.4–3.8	2.4	48.0
Typha latifolia	0.0–0.2	0.2	4.0
Spiraea alba	4.0–4.9	0.9	18.0

different strata. In this case, understory vegetation is typically sampled in a small rigid frame at the center of the plot, and then successively larger circular plots are used to survey shrubs and then trees (Fig. 5.3a). Nested plots may also be effective in collecting different types of data at different scales. For example, individual plant count data may be collected in a subsample of small quadrats within a larger plot that is surveyed for trees or non-vegetation-related ecological attributes (Fig. 5.3b). One of the most complex uses of nested plots is to determine the rate of species accumulation over larger and larger areas (Barbour et al. 1998). In this use, the investigator first surveys a series of small quadrats, recording species. Then, within subsequently larger plots, repeats the procedure, adding any new species. The results can be used to determine the most effective plot size for characterizing community or population attributes (the plot size where there is little subsequent change in your estimates). Finally, nested plots can be used to assess the spatial pattern of a plant population or community (Dale 1998; Greig-Smith 1983).

The line-intercept technique is essentially a variation upon the plot technique in which the plot is one-dimensional (very long but with no width). A line is created with a meter tape, and vegetation is measured wherever it intersects the vertical plane created by the tape (both above and below). A data sheet for line-intercept data includes the species that intersect the line, and the distances at which they intersected it (Table 5.1). Line intercept data can be quicker to collect than quadrat data when there are few species and large areas to be covered. A challenge is comparing the intercept of a graminoid (grass-like) species to those of species that have more two-dimensional coverage (like shrubs). Additional challenges include deciding when to start and end the continuous intervals of intercept with plants that are not themselves continuous over the entire interval. The belt-transect method is a

combination of the line-intercept and plot-based methods in which field personnel traverse a long, linear plot, counting all individuals within the belt-like plot. This method is most effective for low density plant populations, shrubs, or trees.

Point-based sampling is another variation on the plot method, conducted with a plot of one dimension. A long rod is vertically placed in the ground, and each plant that intersects or touches the rod is recorded. This method is primarily good for assessing understory species presence, but could be combined with a canopy tube to assess tree presence at the point. The point-based method is relatively more objective in that few observer decisions must be made, but it also supplies less information per sample. Since most time in the field is spent moving between locations, it is relatively inefficient. In addition, the technique offers limited statistical flexibility because the wetland site as a whole is quantitatively described (by all of the points), but each point sample is not (no abundance information is collected at each sample). For these reasons, point-based sampling is rarely used to provide a general description of the vegetation in a wetland, although it is used by the Washington State Department of Transportation for wetland monitoring due to its more objective nature (WSDOT Environmental Services 2008).

Plotless methods are location-restricted, but do not have any distinct boundary within which to assess vegetation. They are most commonly used to assess tree populations or forest communities, or relatively rare plant populations. There are a wide variety of plotless methods (Bonham 1989). The point-quarter (PQM) and the Bitterlich methods are commonly used.

The PQM is a distance-based method that can be easily combined with nested plot sampling because it has a central point that smaller quadrats or plots can be centered upon. It also has the advantage of being usable by a single investigator; however, it is most appropriate in forested settings and not in wetlands with bands of herbaceous vegetation. These center points can be positioned randomly or regularly throughout the wetland area, just like other plots. Once the central point is determined, the surrounding region is divided into four 90-degree sectors (typically aligned with compass directions, but not always). The nearest tree or plant of interest within each sector is selected, measured, and the distance back to the center point is recorded (Fig. 5.4a). If trees are of interest, diameter-at-breast-height (DBH, 1.4 m from the ground) is typically measured. The PQM data can be used to calculate the density of different species, the mean basal area, and the frequency of different species in a forest (Barbour et al. 1998). In order to calculate density, a mean point-to-plant distance is calculated for all trees, with:

$$\text{Total density/ha}\left(10,000\,\text{m}^2\right) = 10,000/(\text{mean distance in m})^2$$

And the density (no./ha) for each species can be calculated by:

$$\text{Relative density of species A} = (\text{total no. of species A})/(\text{total no. of all trees}) \times \text{total density}$$

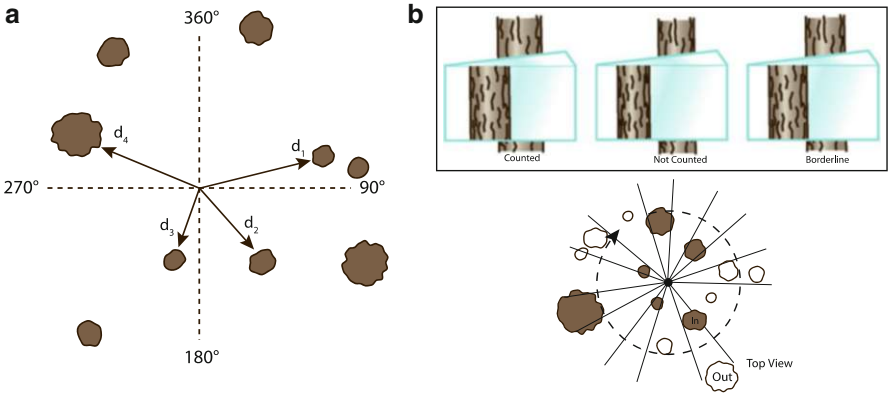

Fig. 5.4 Plotless sampling methods for trees. (**a**) The point-quarter method of dividing the plot space into four quadrants and measuring the distance (d_x) and diameter at breast height of the nearest tree. (**b**) The Bitterlich method in which a central figure counts trees that are "in" and "out" using a prism. If the prism trunk image overlaps actual trunk, the tree is counted. If there is no overlap, the tree is not counted (Published with kind permission of © M. Kuchta 2014. All Rights Reserved)

Basal area (BA) per hectare for each species can then be calculated by:

$$\text{BA/ha of species A} = (\text{Mean BA of species A}) \times (\text{Mean density of species A})$$

The Bitterlich method is used to very rapidly calculate the total basal area of trees in a forest, and so has the most application in forestry. Today, hand held glass wedges, called "prisms", are used to carry out Bitterlich sampling. By standing in a central location, the worker uses the prism, which is calibrated to a specific basal area factor (BAF, usually 0.929–1.858 m^2/acre) to determine whether surrounding trees are included in the sample or not. In general, a tree that is closer or larger is more likely to be included than a distant, small tree (Fig. 5.4b). The number of each species included in the sample is then multiplied by the BAF to obtain the number of m^2/acre for each species. Although the inability to obtain density information from this method is a definite drawback, it may be useful in wetlands applications where the primary interest is a general description of the overstory trees with a more specific focus on understory or shrub layers.

Another plotless method frequently used to assess the plant species present within a wetland or community is the timed meander search. The practitioner simply walks around the wetland or community, recording all species that they observe. The time limit means that one can be consistent in order to compare different wetlands to each other. However, the timing should be scaled according to wetland size. An advantage to this technique is that it can capture a larger number of species than plot-based sampling alone because the investigator is free to explore a larger diversity of potentially species-rich microsites wherever they occur. For this reason, it typically leads to larger species counts than plot-based sampling.

This type of sampling is also used to detect rare species (Goff et al. 1982). A disadvantage to this technique is that it can only be used effectively by skilled field botanists who know the likely habitats of different species and can identify them quickly. In addition, the practitioner must be careful to specifically check for the small plants that might be more easily detected using plot sampling.

5.3.3 Locating Samples Within a Wetland

As discussed above, locating sample points within a wetland system is not a simple matter due to the patterned heterogeneity of much wetland vegetation. There are two basic approaches widely used in wetland vegetation assessment today: (1) representative, more subjective sample placement or (2) systematic sample placement based upon a pre-defined objective scheme. These pre-defined schemes have been described in Chap. 1 as random, restricted-random, regular, or haphazard.

Representative sample placement is quick and efficient, but less defensible in scientific or legal settings than systematic sampling. Nonetheless, it is common practice for monitoring wetland mitigation sites and for rapid wetland delineations. Using a combination of on-the-ground and aerial reconnaissance, different plant communities are roughly delineated (frequently on a map or aerial photograph), and a sample is described from one to several representative locations within each community (Fig. 5.5a). Representative sampling is easier to practice when there are relatively distinct and homogenous communities. It is most commonly used when a rapid, general assessment is needed and or there is a high level of trust in the judgment of the practitioner.

Pre-defined sampling schemes frequently use a baseline plus transects, which define a grid system for sample placement within the wetland (Fig. 5.5b). The baseline is established parallel to the dominant hydrologic gradient of the wetland, and transects extend perpendicular to it and the gradient (Fig. 5.5b). Sample sites are then located at specific locations on the transects. Transects can be regularly or randomly arranged on the baseline, and sample sites can be located regularly or randomly on the transects. This method is generally perceived as more objective and accepted by the scientific and legal community in North America. However, it may be overkill in situations where only preliminary descriptions of vegetation are needed. In addition, the method can be difficult to implement in very large wetland complexes with complex or non-obvious hydrologic gradients or in wetlands with large areas of deep water in the middle. Consider the extreme example of placing one 1 m^2 quadrat every 200 m on a 2,000 m long transect. Clearly alternative strategies must be devised. For large complexes, subdividing the region into smaller representative subsections may be more practical. For wetlands ringed with vegetation around large central deep water, a better strategy might be establishing a baseline around the perimeter of the wetland and running transects in toward the center (Fig. 5.5c). However, this method risks over-sampling the wetland's center with more plots than the periphery.

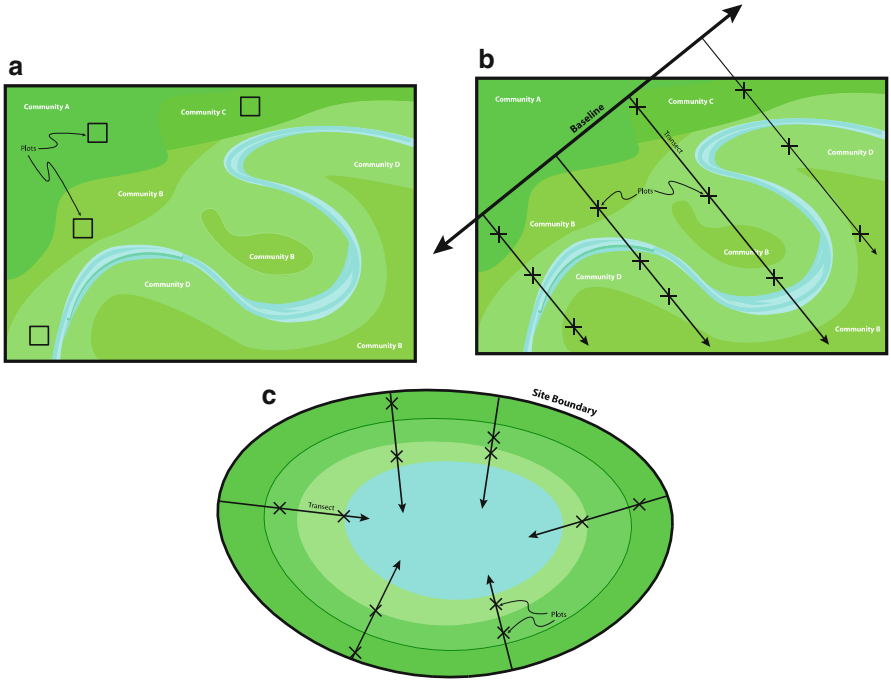

Fig. 5.5 Different methods of locating samples within a wetland. Different shades indicate different plant communities. (**a**) Representative sampling method; squares are samples. (**b**) Systematic sampling method with transects; crosses are samples. (**c**) Sampling with a perimeter baseline tends to oversample the wetland center (Published with kind permission of © M. Kuchta 2014. All Rights Reserved)

5.3.4 Relevé Systems

In general, the more objective system of systematically-placed plots using pre-defined sampling schemes has been adopted by United States ecologists, while the more subjective system of the relevé has been used by European ecologists. Relevés are becoming more common in the United States wetland monitoring community (see Minnesota Department of Natural Resources 2007; U.S. EPA 2002), however, due to the ability to obtain representative and detailed information in a relatively short amount of time. In the hands of a skilled practitioner with expertise in wetland plant identification, relevés can be very effective. The basic procedure involves establishing a 100 m^2 plot (400 m^2 for forested wetlands (Minnesota Department of Natural Resources 2007)) at a representative location within a wetland community. By walking through the plot, practitioners compile a species list, with cover estimates of both life form groups (e.g., evergreen, graminoid) and individual species. Multiple strata are assessed (tree, shrub, understory herbaceous), and heights and sociability (clustered, mat-forming, single) can be estimated. Size information can be taken for trees within the relevé plot to

provide a more comprehensive data set. Important species located just outside of the study plot can also be included.

In many ways, the relevé gives a richer picture of the wetland plant community than systematically-placed plots. However, skilled field botanists are needed to execute it properly. Relevé data are more difficult to statistically summarize at smaller spatial scales due to the smaller number of plots. Over large spatial scales, species presence and abundance estimates can be used to assemble solid community descriptions. Practitioners also use permanently-established relevés to detect community change over time. Relevés may not be the best solution in systems with numerous discrete plant communities concentrated within small spatial scales, because collecting data from a high number of relevés can be time-consuming (U.S. EPA 2002).

5.3.5 Number of Samples

The number of samples is always a compromise between the resources available and the desire to collect as much data as possible to adequately characterize the system of interest. Ideally, a pilot study should be conducted prior to implementing a sampling scheme. The pilot study reveals the amount of variability in the wetland vegetation and can therefore give an idea of how many samples are needed to adequately describe that variability. The practitioner will typically vary the number and/or size of samples in the pilot study. The data are then used to create species accumulation curves (for species richness: number of species) or performance curves (for other measures). From these curves, the investigator can estimate the point at which additional samples yield minimal additional information. This point optimizes the efficiency and accuracy of sampling. The pilot study data should also be used in statistical power calculations (see Chap. 1).

A species accumulation curve is obtained by comparing the mean cumulative number of species to the number of samples (or size of plot, Fig. 5.6a). The asymptote of this curve is the point at which an adequate number of samples has been collected to characterize the richness of the system. In practice, this value is tedious to compute (consider: calculating the mean number of species for sample size of one is quite easy, but what about all pairs of samples for sample size of two?). However, there are computer programs (such as EstimateS; Colwell 2009) that will calculate for you based on your data matrix of samples and species. Likewise, the performance curve is obtained by plotting the mean and variability of some attribute against the number of samples (or size of plot, Fig. 5.6b). When there is no further change in the mean (within acceptable limits), the number of samples is adequate.

If there are no resources available for a pilot study, the adequacy of sampling can be assessed post-hoc using these methods. Statistical methods have been developed to estimate actual species richness from inadequate samples (discussed later in this chapter). In the absence of a pilot study, an important rule of thumb is to collect at least 20 samples at each site to meet the demands of some statistical methods (such as linear regression).

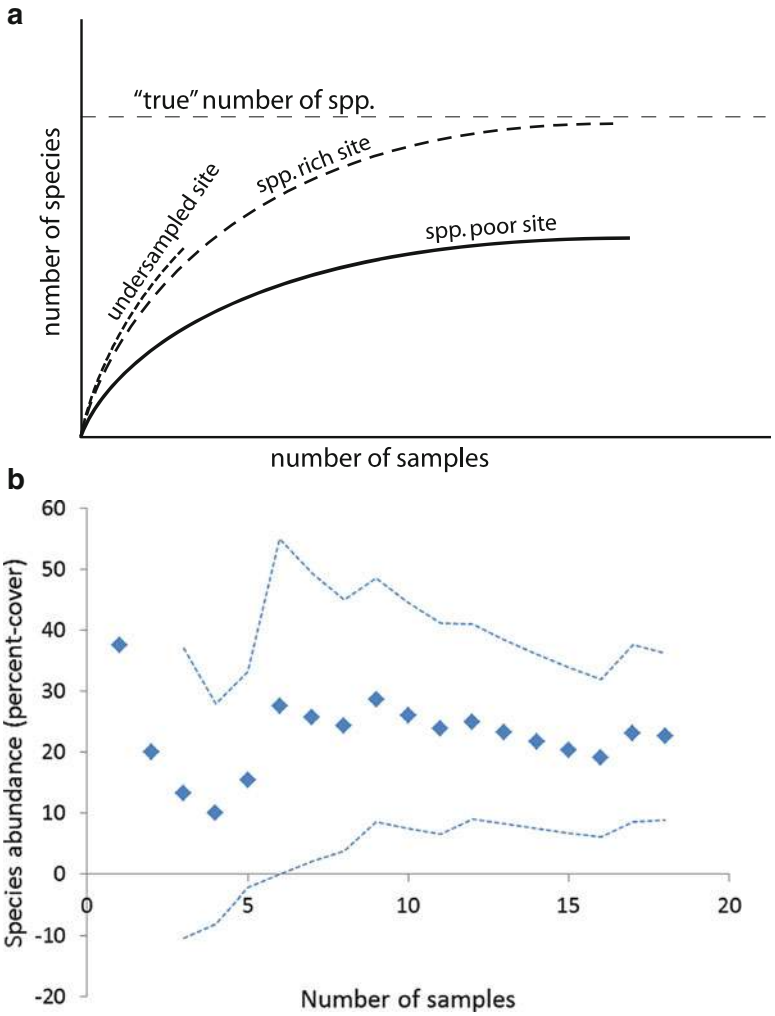

Fig. 5.6 Results from a pilot study. (**a**) Species accumulation curve, spp. = species. (**b**) Performance curve for species-abundance. *Dashed lines* are a 95 % confidence interval

5.3.6 Tracking Change Over Time

Permanent plots and photopoints are techniques for effectively monitoring vegetation change over time. If changes are substantial, it is possible to track changes using random sampling of the same sites at two different times, but maximum statistical power and confidence that change is real comes from sampling *exactly* the same place at different times. In order to install permanent plots, decisions must be made about location, plot type, and monumentation. The locations of permanent

plots could be random or representative, depending upon the number of plots and the study purpose. They should be located so as to minimize potential for human or animal disturbance.

Three different types of permanent plots are common: circular (marked by central point), square (marked by four corners), and transect (marked by two endpoints). Researchers should not expect enough accuracy from a compass to be able to mark only one endpoint of a transect. In addition, although global positioning system (GPS) technology has advanced to a stage where most practitioners can get sub-meter accuracy and occasionally sub-foot, this is not high enough accuracy to avoid leaving monuments in the field. GPS can help to narrow down plot location, but then permanent marking structures should be installed to identify exact plot location. Of course, leaving structures in the field brings risks that they will be disturbed by humans, animals, or acts of nature (flood, landslide, fire). Non-visible, ground-level markers minimize the chance that they will be disturbed, but also minimize chances of relocation in the future. The ideal marker is rugged and hard to remove (such as rebar), visible or detectable by metal-detector, and designed to minimize harm to or detection by passers-by. Elzinga et al. (1998) provides a detailed discussion of monumentation considerations. Within wetlands, it can be extremely difficult to use a metal detector in areas of thick litter and vegetation, because the vegetation dampens the signal.

A picture is worth a thousand words. Photopoints not only communicate change over time in a visible way, but they can also help locate permanent plots in the field. A picture is typically located at some type of permanent marker and associated with a compass direction. In order to associate a particular photopoint with a location, record photographs containing location information can be taken in between photographs of the plant community or population. It is also good practice to locate these points using GPS.

5.3.7 Assessment Techniques for Specific Attributes

Substantial research and invention has been invested into how to best assess different vegetation attributes. Provided here is a non-exhaustive list of some commonly-used techniques and considerations for their use.

- **Definitions:** Meaningful definitions are important to establish and consistently use throughout the study. In vegetation studies, understory is typically defined by its height (i.e., <breast height (1.4 m)), the shrub layer defined by its number of stems, height and diameter (i.e., >1.4 m tall, with DBH <4 cm), while a tree is defined by its diameter (i.e., >4 cm DBH). In some cases, definitions of functional groups will need to be created. When assessing plant traits, clear definitions of leaves, stems, and flowers are important.
- **Density:** The number of individuals per area can be estimated using the plot-based counts or plotless techniques described above. Information on density is

Table 5.2 Common cover class systems

Cover class	Braun-Blanquet (1965) (midpoint)	Daubenmire (1959) (midpoint)
+	<1 % (0.5 %)	
1	1–5 % (3 %)	0–5 % (2.5 %)
2	5–25 % (15 %)	5–25 % (15 %)
3	25–50 % (37.5 %)	25–50 % (37.5 %)
4	50–75 % (62.5 %)	50–75 % (62.5 %)
5	>75 % (87.5 %)	75–95 % (85 %)
6		95–100 % (97.5 %)

especially important in population studies, and can be more easily collected for shrubs and trees than other measures of abundance. Difficult decisions must be made when defining an individual, however. Many plants are connected to each other by vegetative reproductive structures (e.g., blanket-like clones of grasses, grass tussocks, branching tree trunks, or multiple shrub stems connected underground). What is an individual in these circumstances? Investigators typically choose a definition that will capture the influence of the individual on the system of investigation. Trees are often considered separate individuals if they are separate at breast height. Shrubs are frequently measured using stem counts with disregard for underground structures. Grasses can be counted using number of shoots, although it is highly tedious. Boundary decisions (is the plant inside or outside the plot?) can also be difficult. Elzinga et al. (1998) provides a good overview of boundary decisions. Comparisons of density between different species are most informative when species are of similar size.

- **Cover:** Percent-cover is a measure of abundance that describes the horizontal area that a plant species or individual occupies. All plot-based techniques can be used to measure cover. It is frequently faster to use than density for very dense populations, and it is more effective than density for mat-like, low-growing vegetation. Some also argue that it is a better measure of the species' influence on the community than density, because it takes the size of the individuals into consideration. A disadvantage of cover measurements is that they change dramatically over the course of a season, so consistency in time-of-year is important.

There are numerous methods of assessing percent-cover, and the boundary decision problem still applies here. One of the most common methods is visual percent-cover. The observer stands over the plot and assigns cover to each species present. Naturally, this process is highly subjective and can differ dramatically between different plots and different observers (Kercher et al. 2003). In order to minimize observer bias, cover classes are typically used (Table 5.2). Mid-points of cover classes are typically used when analyzing the data, although this introduces substantial uncertainty into the data (Podani 2006). Care must be taken when analyzing ordinal data derived from cover classes (Podani 2006).

In order to avoid the observer bias inherent in the visual-percent cover system, some practitioners use a pin-frame system. This is a metal or plastic grid frame with attached vertical pins (10–100 pins) that fits over the top of a plot. Just as in the

point-based method, any plant that hits a pin is recorded, and species cover for the plot is the number of pins that a given species intersects. This method is quite tedious, but less subjective. Other investigators use digital cameras suspended above the plot on a frame, which is also more objective and minimizes time in the field (except for camera-leveling). Digital image processing software is then used to differentiate different classes of ground cover (Luscier et al. 2006). The digital image technique is not as effective for plots with high species richness or multiple layers of vegetation. In addition, photographic images may make species identification difficult.

A consistent concern with cover measurements is that they mean different things for plants with different physiognomy. For example, mosses or mat-like plants have substantial horizontal cover, with little vertical structure. Graminoids, on the other hand, have substantial linear, vertical structure and can be under-represented by cover estimates.

- **Biomass:** Biomass assesses the amount of production of different species, and can indicate the above- and below-ground influence of the species on the population, community, or ecosystem. It is a destructive sampling technique, and so is difficult to justify when using permanent plots. Most biomass measurement techniques rely on harvesting the vegetation at the peak of its growth. Above-ground biomass is frequently assumed to represent plant allocation to growth, while below-ground biomass represents allocation to maintenance and mineral nutrient and water acquisition (Gurevitch et al. 2006). Above-ground harvest involves clipping at ground level, air- or oven-drying the harvest in paper bags, and weighing the sample. If species-specific biomass is desired, the species should be sorted in the field and bagged separately. Methods have been developed to visually-estimate biomass, based upon a calibrated clipped subsampling scheme (double-weight sampling (Interagency Technical Team 1996)). Below-ground biomass is substantially more difficult to assess, and typically involves excavating roots from a known and consistent volume of soil. In order to identify roots to the species level, above-ground parts must remain attached. Soil can be removed by washing prior to drying or sieving after drying. Core samplers have been devised for sampling below-ground biomass of submersed aquatic vegetation (Madsen et al. 2007).
- **Dominant Species:** Dominant species assessment can be important in classifying or differentiating wetland community types. Dominance is typically assigned to each vegetation layer (or stratum) separately. The concept of dominant species is complicated, because individual studies or methodologies have their own definitions of dominance (Barbour et al. 1998). A dominant species is one that has a large influence on the community or ecosystem due to its size or abundance. Frequently, this is determined after analyzing data back in the lab. Within forests, dominant tree species are those with the highest basal area. However, it is possible to determine and assign dominant species in the field if there is a clear definition based upon easily-measured attributes. Within a plot, for example, a dominant species could be that with the highest height $\times$ cover value. The height of dominant plant species can be used as an indicator of

wetland productivity or nutrient pollution (Little 2005). Analyzing the environmental tolerances of dominant plant species can also be helpful in modeling the dynamics of wetland plant communities (Squire and van der Valk 1992). The concept of dominance is also important in wetland delineation (Environmental Laboratory 1987). The wetland indicator status of dominant species, as determined by the "50/20" rule, determines whether a plot area is designated wetland.

- **Plant Functional Groups:** In order to effectively model plant communities, it is helpful to reduce the hundreds of species present into a smaller more manageable set. Species are assigned to groups based upon traits that reflect similar function in the ecosystem or community. Groups and traits are defined according to the application at hand. For example, Raulings et al. (2010) used plant response to flooding to create functional groups that they then modeled under varying flooding regimes. Other types of functional groups are based upon growth form (e.g., tussock, rhizomatous) or life history (e.g. annual, perennial) or combinations of these (Bouchard et al. 2007). The wetland indicator status used in wetland delineation (Lichvar and Kartesz 2011), is another example of a plant functional group scheme. Exploring the relations between functional groups and other organisms or environmental variables can yield interesting patterns that help us better understand and make predictions about wetland systems. Using established functional group definitions (such as the wetland indicator status or status from the U.S.D.A. Plants database) makes it easier to connect work to previously published studies, and is more acceptable to the scientific community.
- **Plant traits:** Plant traits are genetically-determined characteristics, like leaf shape, flowering time, seed number, or photosynthetic method that are inherent to the taxa, irrespective of the environment (Violle et al. 2007). They may also include genetically-determined responses to the environment, such as variation in specific leaf area based upon light availability and nutrient status. Relations can be drawn between plant traits and environmental attributes (e.g., carnivory and nutrient-poor wetlands). Practitioners also use plant traits to predict the behavior of individual species (e.g., invasiveness) or their response in wetland restoration settings (e.g., assembly rules, (Matthews et al. 2009a)). Plant traits can frequently be determined from the published literature after field work has been completed. However, if researchers are working with a novel trait-species combination, the trait parameters will need to be assessed in the field using adequate and representative sampling from the population. Use a performance curve to determine sampling adequacy.

5.3.8 Sampling Aquatic Vegetation

Many deep-water aquatic systems are not considered wetland, although they may be surrounded by or grade into wetland systems and so are of interest here. Many of

the basic techniques and attributes described above for emergent and terrestrial vegetation can be applied, with modification, to aquatic vegetation. If the submerged vegetation is very shallow, the techniques can be applied directly, but for deeper water, access to the plants can be a problem. In order to sample deep water vegetation, there are two solutions: go to the plants or bring the plants to you. Going to the plants involves SCUBA or snorkeling. Sampling can be accomplished using open-ended polyvinyl chloride (PVC) frames for plots to surround tall vegetation (Parsons 2001). A different method of "going to the plants" involves creating a "viewing tube" out of PVC and clear plexiglass that can be used from a boat. Unless this is permanently-attached to the boat, it must be limited in size in order to penetrate the water. One of the most common methods of sampling aquatic plants is using a simple garden rake to harvest plants from a point, and then estimating percent cover on the rake of different aquatic plant species that are brought up to the surface. If water is very deep, the rake can be attached to a rope instead of the rake handle (Parsons 2001). Wide landscaping rakes used to prepare lawns are often preferred, because they are relatively light and bring up a large quantity of plants. GPS units are essential for locating plots when using a boat to sample.

5.3.9 Practical Considerations

There are several common practices used in field studies that are worth discussing.

- **Trampling the vegetation:** Although this may seem a petty concern, the results of trampling are not petty. When establishing plots and transects, it is important to not trample the vegetation in the area that you will be sampling. Trampled vegetation is more difficult to identify, and visual percent-cover is far more difficult to estimate. Trampling vegetation within a permanent plot can also affect future growth. When walking transect lines, always walk on the side of the tape opposite the side you will be sampling. Always walk outside the plot that you are establishing.
- **Voucher specimens:** It is important to collect a sample specimen of each species in your study. These are pressed in a plant press, identified, and deposited in a local herbarium, where their identities can be verified. The purpose of a voucher collection is to increase the quality of the study so that future researchers can determine the plant species found in the study, even if the names have changed, decades into the future. In situations where there are multiple observers over multiple years, vouchers can ensure consistency in identification. In order to not affect composition and structure of sample plots, when at all possible, voucher specimens should be obtained outside the plot.
- **Site map:** Site maps allow future researchers to return to your site to replicate your study or locate important features, such as monitoring wells or access points. Of course, GIS maps of a site with an aerial photograph for background are the gold-standard in site maps, but even hand-drawn maps with important

features, like access points, labeled permanent plot locations, streams, or different plant community locations can be extremely helpful in the future, or when sharing data collection duties with other workers. They also help ensure that interpretations made in the field align with those assumed back in the lab or stored in the computer file.

- **Multiple observers:** If large amounts of data are collected, it is inevitable that multiple personnel will be involved in vegetation assessment. Working with multiple observers adds additional variation in (1) plot boundary decision interpretation, (2) visual percent-cover estimates, and (3) definitions of individuals, among other aspects. One way to minimize variability is to be clear and consistent about rules and definitions, and document them in standard operating procedures (SOPs). In order to minimize variability in cover estimates, calibrating teams until results are consistent between observers is important (Kercher et al. 2003). This calibration may have to be repeated on a daily basis. Different observers may also have differing levels of expertise in plant identification. If differences in species-richness estimates between individuals are observed, correction factors can be applied post-hoc.

5.3.10 Other Important Data

Some data describing the wetland environment on a small scale can be easily recorded during vegetation sampling.

- **Litter and peat:** Wetlands can produce copious amounts of litter, which eventually may become peat. This litter can potentially suppress plant growth, and so may be an important variable influencing the vegetation. Attributes such as litter depth, percent cover, and type can be easily measured by sampling at one to many locations within a plot. Peat depth and type may also be important in structuring wetland plant populations and communities. Depth is easily measured using >2 m marked plastic rod inserted into the ground. If peat is deeper than the rod, then chances are the extra depth is not biologically significant, and a dummy depth can be used for analysis purposes.
- **Bare ground:** Bare ground within a wetland could signify disturbance, available seed bed, or stressful conditions for plant growth. In any of these cases, it is biologically interesting, and can be easily assessed using the percent-cover method.
- **Elevation or water depth:** Water depth is critically important to wetland plant growth and community structuring. It can be easily measured from the middle of a plot (or subsampled) using a meter stick or tape measure. This type of local measurement is a good supplement to staff gauge or piezometer information, because it is at a smaller scale and may be more relevant to the plants. For more intensive studies, survey equipment (laser level, tripod, and stadia rod) can be used to assess the elevation of each plot. For smaller plots, a single measurement in the plot center is adequate. For larger plots, multiple readings may need to be taken.

- **Microtopography:** Microtopographic features in wetlands (i.e. hummocks, pools, stumps, or tussocks) can exert strong control of the local plant community (Peach and Zedler 2006). Depending upon the study purpose and scope, microtopography can be measured quantitatively (more intense, smaller scope) or qualitatively (less intense, larger scope). Quantitative measurements of high and low points within plots (associated with topographic breaks) can be measured with high-accuracy GPS units associated with local base stations (Werner and Zedler 2002) or using meter sticks to determine tussock height (Peach and Zedler 2006) or maximum height difference within the plot. For studies that are broader in scope, a plot can be assigned qualitative microtopographic scores corresponding to all types within a plot (e.g. stump, high hummock, hummock, low hummock, hollow, flat, or pool). For data analysis purposes, these can be transformed into ordinal scores, and plot microtopographic richness, mean score, or maximum difference can be calculated (Little et al. 2010).
- **Canopy cover:** Canopy cover is an important environmental variable to measure in forested wetlands, because many wetland understory species respond to shade. There are three common ways to measure canopy cover, increasing in accuracy: (1) canopy tube, (2) spherical densiometer, and (3) digital camera with fish eye lens and image-processing software. Canopy tubes are simply vertical tubes with a cross hairs and some type of leveling mechanism. These can be sophisticated tubes with mirrors, or home-made toilet-paper tubes with a dangling level inside. Visual percent-cover of canopy within the tube is recorded from the middle of the plot. Alternatively, the crosshairs can be used to determine presence/absence of canopy at a set of points per plot (Ganey and Block 1994). A spherical densiometer is a small, handheld gridded mirror with a leveling bubble. It is held above the plot, and the observer views how many grid cells are occupied with canopy cover by visualizing a series of dots within the cells (Lemmon 1956). The most sophisticated and accurate measurements of canopy cover use a fish eye (hemispherical) lens with digital image-processing software to calculate canopy cover and light transmission (Englund et al. 2000). However, these cameras are very expensive, and data can only be collected a certain times of day under specific weather conditions – limiting their utility.
- **Spatial data:** A detailed discussion of spatial data collection and autocorrelation is beyond the scope of this chapter, but practitioners should consider whether important spatial relations may exist within or between wetland systems of study. Landscape ecology approaches may be needed to assess relations between wetland sites (consult Turner et al. (2001) for ideas). Within sites, numerous workers have found interesting relations between hydrological features and plant communities using measures as simple as distance of plot from a feature (Grace and Guntenspergen 1999). Since hydrological and dispersal gradients often vary with distance in wetlands, it can be a helpful, and easily-measured surrogate for other variables.

5.3.11 Field Forms

Customized data sheets or files are frequently used in ecology to help streamline data collection and ensure that nothing is accidentally omitted. Data sheets can be as simple as a table on a single page (see Field Labs at the end of this chapter) to a complex multi-page and attribute form like is used in wetland delineation. By listing commonly-encountered plant species in the form in advance, then the data recorder does not have to write them in each time data is collected. In addition, if repeated sampling is planned, consistent data forms can ensure that the same data is collected each time. These forms can also be designed to simplify data entry once fieldwork is completed.

5.4 Basic Analysis Techniques Commonly Used for Vegetation Data

As with sampling techniques, entire books have been written about analyzing ecological and vegetation data. The reader is encouraged to explore McCune and Grace (2002) and Kenkel (2006) for more detailed discussion of multivariate techniques, their assumptions, and the data transformations needed to meet those assumptions.

5.4.1 Basic Calculations

Summarizing the basic attributes of a plant population or community is an important step in the initial stages of data analysis. Exploratory data analysis is critical to understanding the data structure in preparation for more advanced analyses (Kenkel 2006).

- **Frequency:** The number of plots or samples in which a species appears, based upon presence or absence. Frequency is a good measure of how common the species is across the site.
- **Density:** The number of individuals per area. Density measures can be quite variable, spatially. Measures of mean and variability are calculated.
- **Cover or Basal Area:** The areal cover of a plant. Basal area pertains to tree trunks, and is $\pi\left(\frac{d}{2}\right)^2$ where d is the diameter of the tree at breast height (typically measured with a special diameter tape).

Relative values of each of these measures can be calculated, and these are how much each species contributes (as a fraction or percent) of the total frequency, density, or cover of all species. Calculating relative values enables comparisons between sites with dramatically different total cover, for example.

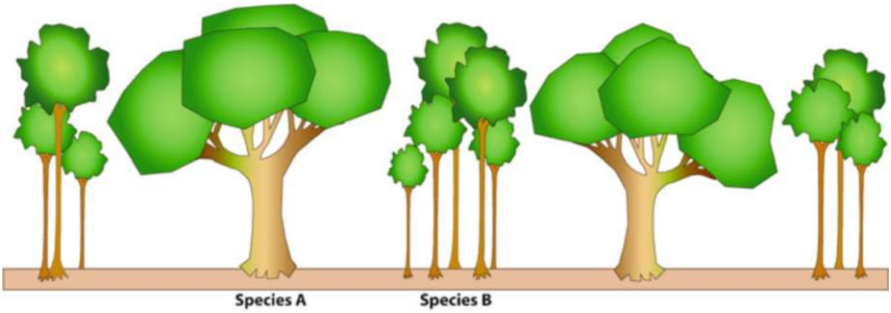

Fig. 5.7 Two species may have different roles in a community, but similar importance values (Published with kind permission of © M. Kuchta 2014. All Rights Reserved)

$$\text{Relative value of species A} = \text{Value of species A}/\text{Total value for all species}$$

Frequency, density, and cover each represent a different aspect of the role of a species in a plant community. If a measure of the overall importance of a species is desired, then composite measures, such as an importance value (Curtis 1959) can be calculated.

$$\text{Importance Value} = (\text{relative frequency} + \text{relative density} + \text{relative cover})/3$$

Practitioners must use caution when interpreting composite measures, however, because two species with very different biological roles in a community can have the same importance value. If species A has low density but high cover and species B has high density, but low cover, they could have the same importance value (Fig. 5.7).

5.4.2 Assessing Species Richness and Diversity

There are many reasons why it is important to assess wetland plant species richness and diversity. Levels of diversity can indicate the health or status of wetland systems (U.S. EPA 2002). Diversity assessments can be important in establishing conservation priorities. Sometimes practitioners are interested in the factors that contribute to high or low wetland plant diversity or richness (e.g., Bedford et al. 1999, Michalcova et al. 2011). In wetland plant science, richness refers to the number of species found in a defined area. Diversity is defined by a combination of species richness (number) and the evenness of the distribution of abundance among species. For example, a wetland site with 142 different species, but 97 - percent-cover of *Typha* spp. has high richness, but low evenness. Numerous mathematical formulas have been invented and used to describe how diverse such a system is. Regardless of the method used, both richness and diversity give no

information about species identity. A very rich community could contain a high number of invasive species, which would be negative from an ecological value perspective.

- **Species Richness:** The number of species is frequently easier to communicate to decision-makers than composite index numbers representing species diversity, and so is frequently used to describe wetland systems. Sampling for species richness can be difficult; pilot studies and species-accumulation curves should be evaluated prior to final sampling (see above). Raw, or observed, species richness straight from the field tends to underestimate the true species richness of a site or system. In order to correct for sampling deficiencies (bias or loss of precision), species richness estimators have been created. Most estimators use a process of generating numerous estimates from randomized resampling of data with different numbers of samples and calculating the mean estimate from the resampling (Michalcova et al. 2011). According to Magurran (2004), the richness estimators with the least bias and highest accuracy are the Chao2, Jack1 and Jack2 methods. Several statistical packages can be used to calculate estimators, including the R package vegan (Oksanen et al. 2011) and EstimateS (Colwell 2009). In practice, the data set with actual observations is entered, yielding an output with several estimates of species richness based upon different estimators. The user then must choose which estimator performs the best for the given data set by examining the output. Some estimators are more conservative than others or will better mirror the observed species accumulation curve.
- **Species Diversity:** Diversity indices are numbers generated from information about species richness and how evenly-distributed the abundance of different species is within the community. These indices provide more information, but they can be open to interpretation and difficult to communicate to decision-makers. There are three general types of diversity: alpha, beta, and gamma. Alpha diversity is the diversity of a single point or site, and is the type most commonly used. Beta diversity is the difference in community composition (change in species and their abundance) over a series of samples. Gamma diversity is the species pool, or the set of species present in the larger regional landscape, and can be important in determining the potential set of propagules available for a restored or disturbed wetland site. There are numerous published diversity indices for describing alpha diversity (see Magurran (2004) for a thorough review), although only a few are widely used. Each index is based upon the proportion of total abundance (p_i) that each species comprises within the community.
  - **Simpson's Index:** This widely-used metric is simply the sum of squares of all species proportions, where S = number of species, and p_i is the proportion of species i:

$$D = \sum_{i}^{S} p_i^2$$

Simpson's index of diversity is 1 – D, the probability that any two randomly drawn species will be different. Values range from one (high diversity) to zero (low diversity). It emphasizes common species and de-emphasizes rare species, which means that the measure is not dramatically affected by missing rare species during sampling. The effective number of species using the Simpson's index is 1/D.

- **Shannon-Wiener Index:** This index is also very popular in ecological studies. It is a measure of the "disorder" in a sample. The higher the disorder or uncertainty, the more diverse a system is. The higher the H′ value, the more diverse the site. The Shannon-Wiener index is the negative sum of the proportion of each species (p_i) times the log of p_i:

$$H' = -\sum_{i}^{S} p_i \ln p_i$$

Index values typically range from 1.5 (low diversity) to 3.5 (high diversity). This measure is more sensitive to rare species than Simpson's index, but less sensitive than plain species richness. That is, rare species count for more value in the Shannon-Wiener Index than in the Simpson. The effective number of species using the Shannon-Wiener index is $e^{H'}$, the exponent of H′.

- **Effective number of species:** This metric can be calculated from any diversity measure, and describes the equivalent number of equally common species for a data set. That is, if all species were of equal abundance, how many would there be? This number takes into account the evenness of the community, and will always be lower than the actual species richness (unless all species are equally abundant). Sites with higher numbers of effective species are more diverse than sites with lower values. Unlike the index values, using effective number of species makes intuitive sense to a lay audience.
- **Evenness:** An evenness value can also be calculated for each sample. Using the Shannon-Wiener index (H′) as a starting point,

$$J = \frac{H'}{\ln S}$$

where S is the species richness. This metric is "Pielou's J," and ranges from one (perfect equitability among species) to zero (no equitability). In terms of evaluating wetland plant communities, higher evenness may mean that a community is more diverse, and less dominated by a few highly competitive species.

5.4.3 Preparation of Multivariate Data

Users must be careful to understand the structure of their multivariate data before beginning. Many parametric data analysis techniques rely on normally-distributed

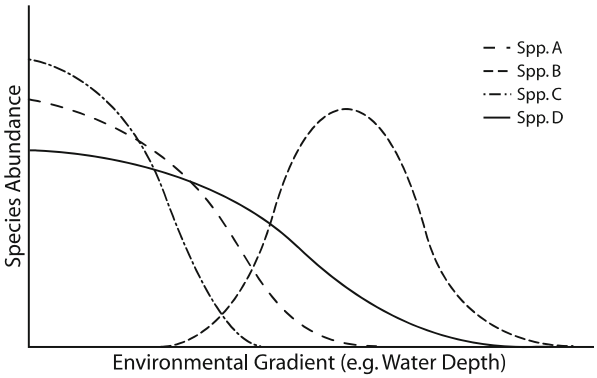

Fig. 5.8 Direct gradient analysis in which species distributions are plotted along an environmental gradient (Published with kind permission of © M. Kuchta 2014. All Rights Reserved)

data sets that have linear relationships with other normally-distributed data sets. Ecological data sets rarely have these characteristics. Species frequently respond to environmental gradients in a non-linear manner (Fig. 5.8), with low abundances at the extremes of their tolerances and high abundances in the center at their ideal conditions (called a Gaussian distribution). In addition, response curves can be solid: even in the most favorable environments, species may not be present due to dispersal restrictions or other factors and species abundances may range from zero to very abundant in the most favorable conditions. In addition, we have no information on species response to conditions beyond the range of their tolerances. Our data sets are truncated at zero, because it is impossible for a species to have a negative abundance (Fig. 5.8). Additional complications arise when species distributions are more skewed or peaked than normal. Finally, many species will exhibit shared absence in numerous sites, creating a species by site matrix that contains numerous zeroes. Just because two species are not present in the same site does not mean that they respond similarly to the same environmental factors. However, this mutual absence may produce a correlation artifact in the data. These characteristics of ecological data can be dealt with by with data preparation and transformation strategies that will minimize variation and maximize expressed data structure. These strategies are beyond the scope of this chapter, but are described in McCune and Grace (2002).

5.4.4 Classification of Wetland Plant Communities

Classifying wetlands, or putting them into categories, is important to effectively manage and restore them. It is typically a first step in any study of a novel system, essential to description. Classification facilitates conservation, and predicting future behavior in response to environmental change. There are numerous methods of classifying wetlands, such as the hydrogeomorphic classification system (Brinson 1993; see Chap. 2 in Vol. 3). Wetlands, or communities within wetlands, can also be classified on the basis of their vegetation. Classifying based upon vegetation can

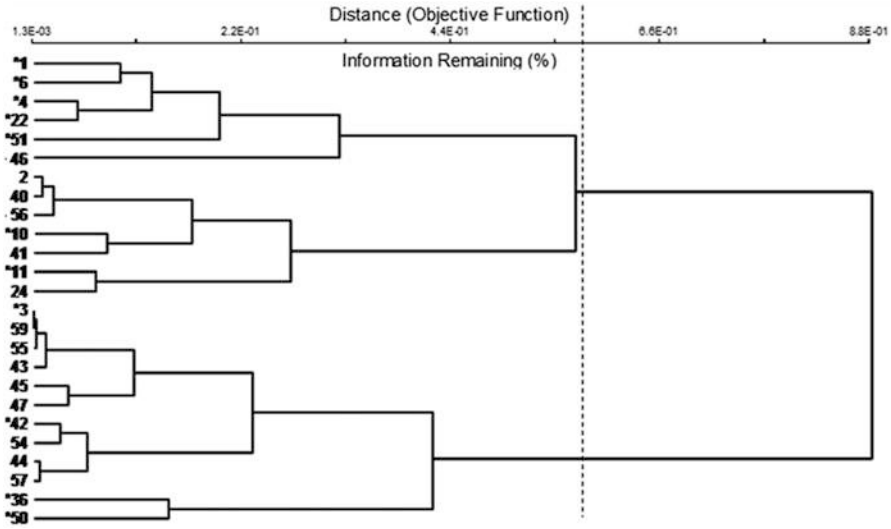

Fig. 5.9 Cluster dendrogram of 25 wetland sites based upon species dissimilarity. *Dashed line* indicates subjective cut-off to create two wetland groups

be useful, because vegetation integrates hydrological, edaphic, and biogeochemical signals (U.S. EPA 2002). It also responds more rapidly to anthropogenic and natural disturbances than hydrogeomorphic setting. On the other hand, this relatively short-term response to disturbance can be a disadvantage if one is attempting to discern response to underlying long-term signals, such as global warming or acid rain.

All classification methods attempt to form groups of like communities out of species data from multiple sites or samples. Early methods relied primarily on relevé descriptions and placement into associations based upon species tables (Barbour et al. 1998), (see Graf et al. (2010) for a recent application). For a while, TWINSPAN (two-way indicator species analysis) was a popular method of forming groups identified by indicator species. However, TWINSPAN should never be used except in simple cases of a single dominant environmental gradient (change in some environmental variable across sites or time (McCune and Grace 2002)). There are multiple methods of distinguishing vegetation groups out of multivariate data. Multivariate data are collected when workers collect multiple measurements (species or environmental variables) at a single sampling location (Kenkel 2006). Complete coverage of methods and their mathematical rationale is given in (McCune and Grace 2002). The most commonly used method in studies of wetland vegetation is hierarchical cluster analysis. Sites or samples can be placed into groups based upon their multivariate vegetation using dissimilarity indices (such as Sorenson or Euclidean distance). It is a hierarchical process, because smaller, more similar groups are combined into larger, less similar groups, with the smaller groups becoming sub-groups of the larger groups. The end product is a dendrogram showing the multivariate similarity between sites or samples (Fig. 5.9). Groups can

be defined post-hoc using subjective methods or more objective measures which assess the homogeneity or heterogeneity of groups (Sharma 1996). In general, the practitioner's knowledge of the study system is most important when defining groups that are helpful to modeling the system (not too many or too few groups for understanding). The process of non-hierarchical K-means cluster analysis is becoming more popular (Carr et al. 2010), in which the practitioner first determines the number of groups and then a computer program optimizes a statistical parameter within those groups (McCune and Grace 2002).

The non-parametric multi-response permutation procedure (MRPP) can be used to assess within group homogeneity and to test for significant differences between groups based upon multivariate data (McCune and Grace 2002). However, statistically significant differences are not always ecologically-meaningful.

Currently, classification for mapping purposes is more frequently accomplished remotely using vegetation reflectance from the visual and near infrared spectra. These remotely-detected pixel signals are frequently combined into groups using supervised or unsupervised classification with K-means clustering to identify the spectral signatures of different wetland plant communities (Zhang et al. 2011). Remotely-sensed and classified communities can be mapped very easily (Midwood and Chow-Fraser 2010), however, the level of detail in these classifications is necessarily limited. Numerous statistical packages can perform cluster analyses, including the freeware R package vegan (Oksanen et al. 2011) and PC-ORD (McCune and Mefford 2011).

5.4.5 Ordination

Typically, a practitioner will have multiple sites or samples, with each sample described in numerous ways (the abundances of multiple species, environmental characteristics, etc. . .). Ordination is a method of discovering patterns and underlying structure in this multivariate data (Kenkel 2006). Because ordination diagrams and the process of ordination itself can be confusing, ordination information is typically not directly presented to lay people or political decision-makers. However, that does not mean that it has no role to play in wetland conservation and management. Ordination has been used to assess the effects of management practices on wetland plant communities (Hall et al. 2008); compare damaged, restored, and reference plant communities (Rooney and Bayley 2011); assess the community-level effects of exotic species invasion (Mills et al. 2009); and generally understand how environmental degradation affects wetland systems (Carr et al. 2010). A complete discussion of ordination techniques, their assumptions, and mathematical background can be found in McCune and Grace (2002), Kenkel (2006), and Legendre and Legendre (1998).

There are several different types of ordination, but all involve reducing the variability in a large data set down to a few axes that express the primary patterns

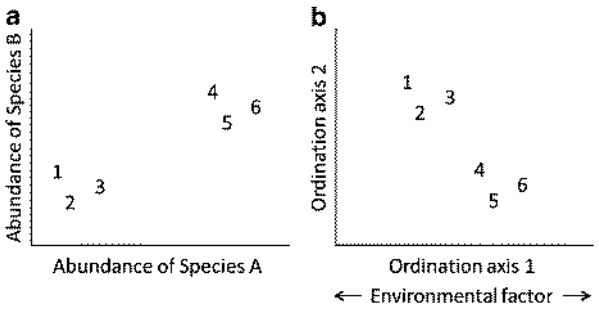

Fig. 5.10 Ordering sites according to their multivariate plant species composition using indirect gradient analysis (ordination). *Numbers* are wetland sites. (**a**) A data set consisting of only two species and six sites. (**b**) A data set consisting of numerous species in six sites. The underlying structure in the species information has been extracted by ordination methods into two axes of variation. Site ordination scores are related to environmental measurements using correlation

in the data using the correlation between the multiple variables (McCune and Grace 2002). Sometimes there is a particular environmental factor of interest. If this is the case, then direct gradient analysis, or positioning samples along axes determined by environmental measurements, is most appropriate (Kenkel 2006). The practitioner can then examine the relations between the species within those sites and the environmental factor of interest (Fig. 5.8).

Indirect gradient analysis does not assume the importance of any particular environmental gradient *a priori*, but rather lets the plant species data order itself. In multivariate speak: "arranging plots in species space." There are several ways of using ordination, but the most common is to plot sites along axes of species composition, and then correlate these axes with environmental variables to determine which environmental variables most strongly influence the plant communities. One can conceptualize this more easily by beginning with a system of multiple sites with only two species. It is easy to plot samples or sites within species space, with each axis representing the abundance of species A or B (Fig. 5.10a). In Fig. 5.10 sites 1, 2, and 3 are similar in their composition of species A and B, and sites 4, 5, and 6 are similar to each other. In a cluster analysis, these two groups would most likely cluster together. Ordination typically involves many more than two species, but it is very difficult to create a graph that contains 100 axes for each different species. In order to make meaning, ordination mathematically sorts through the variation in these different species to draw out the strongest patterns (based on correlation or similarity), and these patterns are reduced to usually two or three axes (Fig. 5.10b). Each site is assigned a score or position along each axis, based upon its species composition. Because each site has associated environmental measurements, the ordination score can be correlated with the environmental factors to determine how these factors affect the community as a whole rather than an individual species.

Various methods exist for indirect gradient analysis: PCA (principle components analysis), Bray-Curtis, NMS (nonmetric multidimensional scaling), CA

(correspondence analysis), DCA (detrended correspondence analysis), and CCA (canonical correspondence analysis). McCune and Grace (2002) and Kenkel (2006) thoroughly review the options and mathematical background behind each technique. In general, PCA should be used when there are few primary gradients that relate broadly-linearly to the scope of plant communities studied. CA is best used on categorical contingency table data (optimizing both sites and species variation simultaneously). DCA should be avoided. McCune and Grace (2002) argue that NMS is currently the method of choice because it makes no assumption of linear relations, and performs very well with high diversity data sets. However, Kenkel (2006) suggests using NMS as a last resort only after PCA and CA options have been exhausted. CCA is ordination constrained by environmental variables, and should be used when there are one to few strong environmental gradients of interest. All of these techniques are available in the R package vegan (Oksanen et al. 2011) and in vegetation-specific software, like PC-ORD (McCune and Mefford 2011). Roberts (2011) provides helpful online tutorials for using R to analyze vegetation data.

Ordination is extremely helpful for initial pattern detection and description of novel systems. Ordination using different transformations of the same species dataset (e.g., one using abundance, one using frequency, and one using presence or absence) can be helpful in differentiating between levels of organization in plant communities (Allen and Wyleto 1983). It is critically important to prepare the species dataset for ordination by removing outliers and performing data transformations to meet the assumptions of the technique (McCune and Grace 2002; Kenkel 2006). One criticism of ordination is that it cannot test hypotheses using the philosophy of inferential statistics, although the structure of ordinations themselves can be tested through bootstrapping (testing real data configurations against multiple randomized variations). However, structural equation modeling (SEM) provides a new way of statistically testing relations discovered through ordination (McCune and Grace 2002).

5.4.6 Classification and Regression Trees (CART)

Another method of analysis that addresses the question of how environmental variables affect plant populations or communities is CART (McCune and Grace 2002). Classification trees model which independent variables best differentiate pre-defined groups from each other (e.g., plant community groups from classification or occupied versus unoccupied sites). Classification trees have also been used to assess wetland condition (Cohen et al. 2005). Regression trees have continuous response variables. One advantage of CART is that it is a non-parametric method, meaning that it does not require the same assumptions of data normality that other methods require. The output is a predictive model that resembles a dichotomously-forking tree which separates pre-defined groups based on a threshold value of the best-differentiating environmental variable at each level. Like in hierarchical classification, the initial fork separates two relatively heterogeneous groups from each

other, and subsequent divisions can lead all the way down to individual sites or some pre-defined stopping value. The resulting model allows the user to determine, given the values of different environmental variables, the type of plant community likely to occur in a site or whether the site could be suitable habitat for a species. Using this technique, relationships sometimes emerge that are otherwise difficult to detect using other linear or even multivariate models. For more information, see De'ath and Fabricius (2000) and McCune and Grace (2002).

5.4.7 Mantel Test

A Mantel test is simply a method of correlating two similarity or distance matrices with each other. It is also especially helpful when evaluating the effect of spatial proximity on plant community similarity or the strength of plant community – environment relationships. For example, a goal of a project may be to determine whether plant communities respond to wetland disturbance or restoration in a similar fashion to macroinvertebrate communities across a set of wetlands. With the Mantel test, the similarity in response can be compared by correlating the plant and macroinvertebrate distance matrices. A Mantel test can also be used to assess the significance of the correlation between geographic distance and community distance (McCune and Grace 2002).

5.4.8 Indicator Species Development and Analysis

Although wetland plants can be used successfully as indicators of wetland health (U.S. EPA 2002) and wetland status (Environmental Laboratory 1987), this section does not focus solely on those particular applications. Indicator species analysis (Dufrêne and Legendre 1997) is a mathematical technique that can determine indicators for different groups of sites or plant communities. Therefore, it can be used to develop wetland condition indicators, but that is not its sole purpose. Once groups have been established either *a priori* or using the techniques described above, indicator species analysis determines how faithful a given species is to a particular group (whether it is always present), and how exclusive the species is to the group (never occurring in other groups, (McCune and Grace 2002)). A species abundance or presence data set is input, and the output is a table of indicator values (percent of perfect indication, with 100 % being perfect), and an associated *P*-value based on a Monte Carlo (randomization) test with a null hypothesis of no difference between groups. The R package labdsv (Roberts 2010) and PC-ORD (McCune and Mefford 2011) both calculate indicator species values.

Indicator species analysis has been used to better describe plant community groups (Rooney and Bayley 2011), differentiate wetlands invaded by non-native plant species (Johnson et al. 2010), and associate plant species with different

environmental conditions for wetland condition assessment (Johnston et al. 2007). The combination of cluster analysis, NMS ordination, and indicator species analysis is commonly used to describe and differentiate plant communities in the wetland literature.

5.4.9 *Floristic Quality Assessment Indices*

Floristic quality indices (FQAIs) are frequently used to determine the condition of a wetland based upon the ecological "conservatism" of the plant species (U.S. EPA 2002). Some plant species are more sensitive to human disturbance and therefore more conservative in terms of their growth requirements. These species are indicators of high quality systems. By sampling an area, one can assess its quality using the plant species scores (i.e., coefficients of conservatism). Species are ranked with values from one (not conservative) to ten (conservative and highly ecologically sensitive) by experts. The ranking needs to be done on a regional basis, because species behave differently in different regions. Therefore, one cannot apply a FQAI developed in the Upper Midwest to New England wetlands.

Once a wetland has been sampled, various formulas can be used to summarize the wetland conditions based upon the C of Cs (coefficients of conservatism) of the plants found there. One commonly used formula is

$$I = \sum_{i}^{S} CC_i \Big/ \sqrt{N_{native}}$$

Where I = the FQAI for the site, CC_i is the C of C for species i, and N is the total number of native species found at the site (Andreas et al. 2004). A simple mean C of C ($\bar{C}$) for all species can also be calculated:

$$\bar{C} = \sum_{i}^{S} CC_i / N$$

Both of these measures rely only on species presence data, which is an advantage in that it is faster to inventory species presence than abundance. A weighted average measure can incorporate species abundance, with relativized species abundance as the weights (see Exercise 4). Rooney and Rogers (2002) discuss some problems with FQAI and pose alternative calculations. Matthews et al. (2009b) does a thorough assessment of the performance of different vegetation indicators, including FQAIs, when tracking wetland restoration trajectories in comparison with reference systems. They warn that using any single metric to assess wetland restoration success provides an incomplete picture, that multiple methods should be used, and that there is no simple metric that adequately assesses restoration success (Matthews et al. 2009b).

5.4.10 Using the Wetland Indicator Status of Vegetation

One of the most frequent applications of vegetation sampling and analysis in a wetland setting is for wetland delineation purposes. Very specific sampling and analysis protocols are used, according to the 1987 U.S. Army Corps of Engineers manual (Environmental Laboratory 1987) and the newer Regional Supplements. In this system, plant species are assigned an indicator status (obligate wetland = 1, facultative wetland = 2, facultative = 3, facultative upland = 4, or upland = 5) for different regions according to expert opinion. The indicator status of species also can be used for other purposes aside from wetland delineation protocols. One application is to calculate a weighted average of indicator scores with weights based upon species importance value, cover, or frequency in order to track the relative wetness of a site. This application is especially helpful when conducting repeated studies to assess wetland mitigation success, for example (Atkinson et al. 1993).

5.4.11 More Resources

The subject of vegetation sampling and analysis has generated a vast and rich literature. This chapter is intended to expose the reader to a variety of sampling considerations and basic analysis techniques. The following excellent resources should be consulted for further information:

- Sampling and analysis for plant population studies: Elzinga et al. (1998)
- Plant community data analysis, especially of multivariate data: McCune and Grace (2002) and Kenkel (2006)
- Using vegetation as an indicator of wetland quality: U.S. EPA (2002)
- U.S. Fish and Wildlife Service National Wetlands Inventory: http://www.fws.gov/wetlands/
- U.S. Department of Agriculture Web Soil Survey: http://websoilsurvey.nrcs.usda.gov

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Field Labs

Field Lab 1: The Effect of Quadrat Shape on Plant Density and Spatial Pattern Estimates

Objectives: Be able to. . .

- Discuss how method of observation (quadrat shape) can influence your results.
- Establish a sampling grid for randomly-placed plots in the field using a tape and compass.
- Use a spreadsheet program to summarize your data.
- Use a statistical program to analyze your data.

Questions

- Which quadrat shape will have more variation between quadrats, leading to a higher variance:mean ratio?
- Do different quadrat shapes yield significantly different plant population density measurements?

Hypotheses

Write down hypotheses pertaining to the questions above. Think about how the quadrat shape relates to plant shape and any environmental variation in the site.

Study system: This exercise is best conducted in a setting that has easily-recognizable plants with somewhat aggregated (clumped) distributions. Alternatively, sampling could include two different plant species, each with a different spatial pattern (clumped, randomly, or regularly-dispersed). In any case, even if it is a clonal plant, you will be counting individual stems (ramets). These stems should be easily-recognizable for all students in the class, so choose the species with care.

The Set-up: Students will be collecting data at randomly-placed points within a grid. Plan enough space for a 10 × 10 m plot for each pair of students in the course, with a buffer in between each plot (Fig. 5.11). Students will establish a grid in the field using meter tapes and a compass. Plant flags or stakes every 1 m to demarcate the grid. Students can either identify pairs of points from a random number table and work within their own plot for the lab, or they can be assigned sets of random numbers (0–10 or 0–20 if using ½ m spacing), and sample all the plots in the class using those same numbers. The lower left corner or center of the frame should be placed at the random grid coordinates. Boundary decisions (how to deal with plants on the edge of the quadrat) should be made and consistently applied within the class. Each of the quadrats in Fig. 5.11 has a total area of 1 m^2. Quadrats of ½ m^2 could also be used and an exploration of quadrat size effect could also be made.

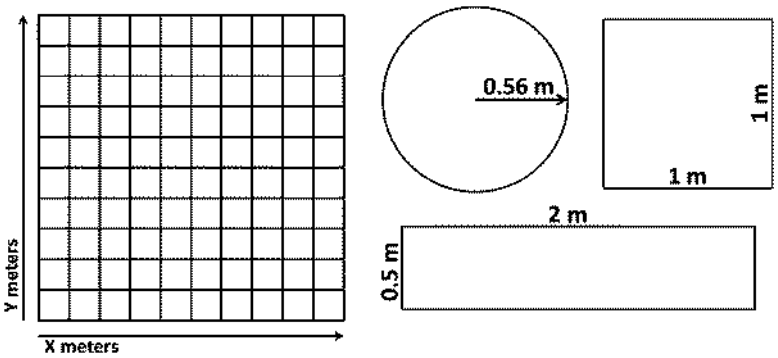

Fig. 5.11 Random sampling grid and quadrats of different shapes (each 1 m^2 in area)

At each set of random coordinates, assess the number of stems of each species within quadrats of all shapes, and record in the table below. As always, remember to avoid stepping within the plots.

Plot	Quadrat shape	Random Coordinates	Number of Stems Species 1	Number of Stems Species 2	Comments

Data Summary and Analysis

1. There were two response variables in this exercise: plant population density and spatial pattern. What was the independent variable?
2. Using the class data and spreadsheet program, prepare a table that contains the means, standard deviations, and standard errors for number of stems/m^2 for each of the shapes. Also include the sample size that you are using in each calculation. Discuss whether these results seem to match expectation. If you assessed two species, then create two different tables. Do not forget to include units!

Shape	Mean	Standard Deviation	Standard Error	Variance	n
Square					
Circle					
Rectangle					

3. Using a statistical package and coded data (e.g. 1 = square, 2 = circle, 3 = rectangle), conduct a one-way ANOVA to determine if the different shapes yielded different mean densities.

Shape code	Density

4. In order to detect differences in population spatial pattern, calculate the Variance: Mean Ratio (VMR) of the plant density in quadrats of different shapes. If variance is high compared to the mean, then the population is clumped in pattern. If variance is low compared to the mean, then the population is regularly-distributed. If the VMR ≈ 1, then the population is randomly-dispersed. Does spatial pattern change with quadrat shape?

Shape	Mean	Variance	VMR	Spatial pattern
Square				
Circle				
Rectangle				

5. If your statistical package allows, conduct a non-parametric Levene's test to determine whether the three different quadrat sizes gave significantly different variances.
6. Interpret your results. Did quadrat shape significantly affect plant population density or spatial pattern estimates? Why do you think this is?
7. What type of quadrat shape would you use in future studies and why?

Field Lab 2: Tree Populations & Succession

Objectives

- Use plotless methods for assessing population size.
- Use a compass to establish transects.
- Map plant populations using GPS and GIS.
- Interpret population data in order to predict future successional trends.

Background

In this lab, we will assess a forest stand containing interacting populations of trees, which form a community, in order to determine how it will change in the future. This skill is important to many natural resource agencies, which need to predict the future composition of the land.

- A population is a group of individuals of the same species in the same place at the same time. At any moment in time, a population has the attributes of population size and spatial distribution.
- A community contains interacting species in the same place at the same time. The species composition of communities can change over time – a process called succession.

One of the fundamental parameters of interest to ecologists is the density of organisms in a given area. However, in nature it is either impossible or impractical to count all organisms, and so we *estimate* density. For relatively small, immobile organisms, quadrat sampling is used to estimate density. For large, immobile organisms, remote-sensing, plot-based, or plotless techniques can be used. For mobile organisms, ecologists use mark-recapture techniques.

Factors controlled by the investigator that can affect the density estimate:

- the experience of the observer
- method of observation (instrument or chosen sampling technique)
- the number of samples taken

Factors beyond the control of the investigator:

- organism density
- organism spatial arrangement

Plot-based techniques frequently rely upon frames to isolate a sample area. These frames are called quadrats: arbitrarily-sized and -shaped sampling units. There are alternative techniques that are especially useful for large plants (trees). These are commonly called plotless sampling methods. During this laboratory, you will use the plotless Point Quarter Method (PQM) to estimate tree density and basal area

Regular Sampling Scheme

It takes time to establish a random grid and locate plots on it. Although totally random plot placement is the statistical "gold standard," it may be infeasible due to resource constraints. In addition, sometimes you want to ensure an even distribution of plots across a site, in which case totally random sampling may not be appropriate.

Regular sampling consists of using a set spacing between plots. Like random sampling, it typically precludes intentional and unintentional observer bias.

Although not technically statistically sound, ecologists often ignore statistical assumptions in favor of a more representative sample. Sampling schemes including combinations of regular and random sampling are typically favored by ecologists.

In this exercise, we will implement regular sampling with a random start so as not to bias our samples and save time.

Global Positioning Systems (GPS)

GPS allows ecologists to locate their position on the earth. It relies upon a network of 30+ satellites that encircle the planet, sending signals down to GPS receiver antennas. The receivers differ in quality, some capable of sub-foot accuracy. You will use GPS units to map the center of each plot by establishing waypoints. Be careful to wait until you get roughly 10 m accuracy before plotting a waypoint. Label your waypoint with the plot number. Later, you may download your points into a GIS according to instructor-provided instructions.

Number of Plots

Each group will sample along transects in one of the forests using meter tape and a compass. Take point measurements (as described below) every 20 m until you have sampled at least five points.

Tree Identification

Your instructor will provide you with a tree identification guide and a list of common trees and their abbreviations.

The Point Quarter Method

At each point, divide the surroundings into four quarters along the principal compass directions (N, S, E, W). Use the data sheets provided to record the distance (d, expressed in meters) from the center point to the nearest tree that has a DBH (diameter at breast height) >4 cm in each of the four quarters (Fig. 5.12). Also record the DBH (in cm) and species of each of the four trees. These four measurements constitute data for one point sample. Do not count dead trees. Trees that have multiple trunks, but are separated at breast height are considered multiple trees.

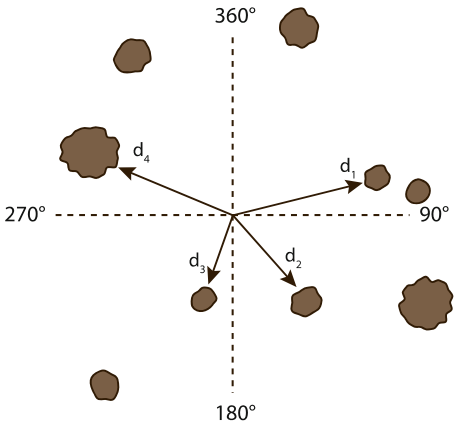

Fig. 5.12 Sampling trees using the Point Quarter Method. The area around a central point is divided into four quadrants, and the closest tree within each quadrant is sampled for distance from point and DBH

Site: _____ ***Group:*** _____ ***Date:*** _____
Compass bearing:_____ ***Plot distance apart:*** _____

Tree Layer

Plot	Tree 1	Tree 2	Tree 3	Tree 4
	Sp, Dist-m, DBH-cm	Sp, Dist-m, DBH-cm	Sp, Dist-m, DBH-cm	Sp, Dist-m, DBH-cm

Notes:

Lab Part 2: Analyzing Point-Quarter Data

Objectives

- Analyze point-quarter data using MSExcel.
- Interpret population data in order to predict future successional trends.

Analysis of Point-Quarter Data

The final product of your calculations should be a table that looks like this (Table 5.1):

Species	Frequency (no. of plots)	Relative frequency	Density (trees/ha)	Relative density	Mean basal area per tree (m^2)	Mean basal area/ha (m^2/ha)	Relative basal area
Total							

Use the questions and formulas below to fill in the table using the class data.

How common is each species?

1. We can answer this question by simply looking at the number of points that each species occurs in.

$$\text{Frequency} = \text{no.points that the species occurs at}$$

How frequent is each species relative to the total?

2. If you counted 40 plots total, and 4 of these had white pines, white pines would represent 4/40, or 0.10 of the total points.

$$\text{Relative frequency} = \text{no. of plots containing species A/total no. of plots}$$

What was the total density of all trees in the site?

3. The first step in analyzing point quarter data is to determine the mean point-to-plant distance for all of the trees on each transect. This value represents the mean distance between trees in the site. Compute this value and write it here:

$$\text{Mean point-plant distance for ALL trees} = \_\_\_\_\_ \text{ m}$$

4. Next we need to compute tree densities. The mean point-to-plant distance squared (d^2) gives the mean area per tree.

$$\text{Mean area per tree over all species} = \_\_\_\_\_ \text{ m}^2$$

By knowing the mean area per tree, we can figure out how many of them are contained in a defined area (usually a hectare (ha), which contains 10,000 m^2).

The average tree density (in trees per ha) on each site = 10,000 m^2 per ha/(mean m^2 per tree)

Mean tree density over all species(total density) = _____ trees/ha

What was the mean density of each different tree species?

Mean density for Species A = (no. of trees of Species A)/(total no. of trees) × total density

5. If the total tree density on the site was 800 trees/ha, then the density of white pine trees would be 0.10 × 800/ha = 80/ha. Compute the density for each tree species.

Are some species bigger than others?

6. Foresters are often concerned with how big each tree is and how much wood is on each site as a measure of profitability. Ecologists care about this, because bigger trees can potentially exert more influence on an ecosystem. Tree size is often represented by basal area, which is the cross-sectional area of each tree (usually at breast height).

   Calculate the basal area for each tree by using $BA = \pi r^2$. Use the diameter at breast height (DBH) data to determine the radius (r) of each tree. Once you have computed the basal area of each tree, find the mean basal area per tree of each species on the site.
7. Next, compute the total basal area per hectare of each tree species. This is:

$$\text{Mean basal area per tree}\left(\text{in } m^2\right) \times \text{no. of trees per ha(density)}$$

   For example, if the mean cross-sectional area of a white pine tree was 2,000 cm^2 you would first divide this by 10,000 to convert it to 0.2 m^2. Then multiply this by 80 trees/ha (the density of white pines that we calculated above) to find the total basal area. In this case it is 16 m^2/ha. A high basal area can be achieved by either having a high basal area per tree or a high density of trees.
8. Finally, compute the relative basal area of each species by dividing that species' basal area per tree by the total basal area per tree for the site.

Questions

Use the data in your tables to answer the following questions in complete sentences:

1. What tree species is present in the highest density and lowest density?
2. What tree species is present in the highest basal area and the lowest basal area?

3. How do species rankings by density compare to rankings by basal area?
4. Draw a forest stand in which species A has high density and low basal area, while species B has low density and high basal area.
5. In order to determine the importance or overall magnitude of a species impact on an ecosystem, we sometimes calculate importance values (IVs). IVs combine all aspects of a species influence into a single number.

$$\text{IV} = \text{relative density} + \text{relative frequency} + \text{relative basal area}$$

Relative values are simply the value of the species divided by the total for all species (taken from Table 5.1). Create a second table of importance values for the different species in your site:

Species	Relative Density	Relative BA	Relative Freq.	IV

6. Use the data in Table 5.2 to answer the following questions:
   A. Which species had the highest importance value?
   B. Which species had the lowest IV?
7. Draw a forest stand in which Species A has a very high IV and Species B has a very low IV.
8. If two species have the same IV, does that mean that they influence the ecosystem in the same ways? Why or why not?

Size-Class Distributions

One way to investigate successional trends in a forested wetland or any forested system is to construct size-class distributions for the different important species. Size-class distributions can be graphically represented by plotting the number of trees in different size classes (e.g., 1, 2, 5, 10 cm classes, Fig. 5.13).

9. Create size-class distribution plots for the three species with the highest IVs.
10. What do these size-class distribution plots tell you about the future of the forest?

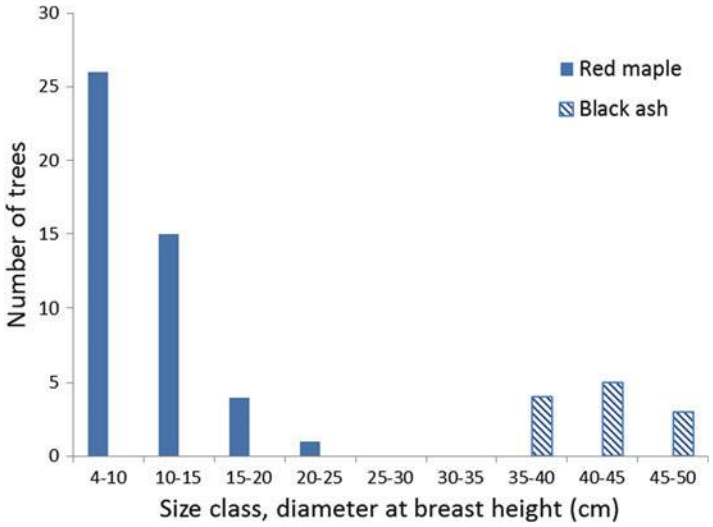

Fig. 5.13 Size-class distribution for red maple and black ash in a forested wetland site

Homework

Exercise 1: Devise a Sampling Strategy

Your goal is to construct a sampling scheme based upon a pilot study (in the case of the provided data set, this is reed canarygrass (*Phalaris arundinacea*)). Using your own data or the data provided below, devise a sampling strategy based upon the (1) species accumulation curve and (2) performance curve of abundance of the species of interest. If you plan to use your own data, download the free program EstimateS (Colwell 2009), to calculate your own species accumulation curve.

Provided data set (calculate a performance curve):

Sample	*P. arundinacea* percent-cover	Cumulative mean percent-cover	95 % Confidence Interval
1	37.5		
2	2.5		
3	0		
4	0		
5	37.5		
6	87.5		
7	15		
8	15		
9	62.5		

(continued)

(continued)

10	2.5		
11	2.5		
12	37.5		
13	2.5		
14	2.5		
15	0		
16	0		
17	87.5		
18	15		

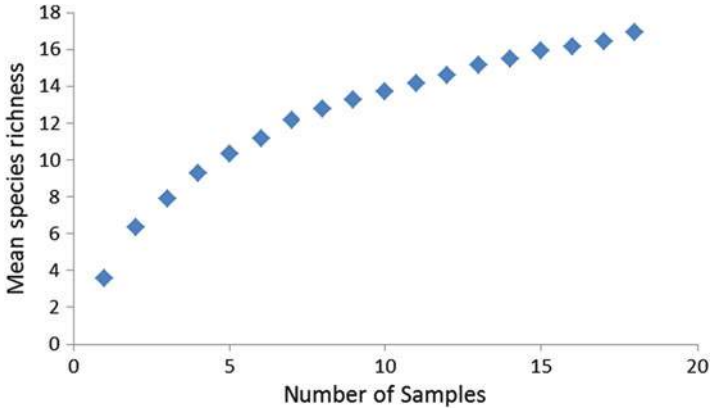

Fig. 5.14 Species accumulation curve

1. Create a performance curve + 95 % confidence interval using the *P. arundinacea* data above and calculating a cumulative mean.
2. Did you collect enough data to accurately estimate the abundance of *P. arundinacea*? Why or why not?
3. If you were trying to maximize efficiency while estimating an accurate abundance of *P. arundinacea*, how many samples would you collect at this site and sites similar to this?
4. According to the species accumulation curve (Fig. 5.14), was the sampling adequate to characterize species richness at this site?
5. How many samples would you need to collect to most accurately and efficiently estimate species richness at this site and sites like it?

Exercise 2. Species Diversity Assessment

Compare the two plant communities below using diversity statistics. Determine which statistics are most helpful, and why.

Community data

Species	Community 1 abundance (percent-cover)	Community 1 abundance (percent-cover)
A	30	12
B	30	12
C	15	12
D	15	12
E	2	12
F	2	12
G	2	12
H	2	12
I	2	4
Total	100	100

1. Simply by inspecting the data, compare the two communities in terms of their species richness and your opinion of their evenness.
2. Calculate Simpson's Index

Species	Comm1 p_i	Comm1 p_i^2	Comm2 p_i	Comm2 p_i^2
A				
B				
C				
D				
E				
F				
G				
H				
I				
Total		D =		D =

Simpson's diversity index = 1 − D: Comm 1: _____ Comm 2: _____
Effective number of species = 1/D: Comm 1: _____ Comm 2: _____

3. Calculate Shannon-Weiner Index

Species	Comm1 p_i	Comm1 ln p_i	Comm1 $p_i \times$ ln p_i	Comm2 p_i	Com21 ln p_i	Comm2 $p_i \times$ ln p_i
A						
B						
C						
D						

(continued)

(continued)

E						
F						
G						
H						
I						
Total			$H' = -$			$H' = -$

Shannon-Wiener index = H': Comm 1: _____ Comm 2: _____
Effective number of species = $e^{H'}$: Comm 1: _____ Comm 2: _____

4. Compare the interpretation of the Simpson's and Shannon-Wiener diversity indices. (A) Which seems to be more effective at distinguishing between the two communities and why? (B) If you were trying to communicate your results to a lay audience, which statistic is easier to interpret and why?
5. Inspect the effective number of species derived from the Simpson's and Shannon-Wiener indices for the two communities. (A) Do the results from the two communities make sense to you? Why or why not? (B) Is there a difference between the Simpson's and Shannon Wiener effective number of species? Why do you think this is?
6. Calculate Pielou's evenness from the Shannon-Wiener index. (Recall that $J = H'/\ln(S)$ where S is the species richness.

   Pielou's J: Comm 1: _____ Comm 2: _____

7. Do the evenness statistics make sense given the initial data? Why or why not?

Exercise 3. Calculating an FQAI

Using either data that you collected yourself, or the data provided below, calculate the floristic quality index and mean C of C for the site. If using the provided data set, refer to the University of Wisconsin – Stevens Point herbarium http://wisplants.uwsp.edu/namesearch.html for the coefficient of conservatism (the wetland site is located in Wisconsin). After entering the species name, select the "more information" link for the species C of C.

Provided data set:

Species	Mean abundance (percent-cover)
Agrostis gigantea	15
Carex atherodes	42
Carex lacustris	13
Carex utriculata	8
Eupatorium perfoliatum	21
Phalaris arundinacea	52
Typha latifolia	10

Calculation table (use your own or provided data set). A typical FQAI does not include abundance data, but only species presence. However, you may have abundance data that you may want to use to weight your findings.

Species	Coefficient of conservatism	Mean abundance	Relative abundance	Weighted C of C (CC'_i)
Sum	A	B	1.00	D

$A = \sum_i^S CC_i$
$B = \sum_i^S x_i$ where x_i is the mean abundance of species i
Relative abundance of species i $= x'_i = x_i/B$
Weighted C of C for species i $= CC'_i = x'_i \times CC_i$
D (Weighted C of C of site) $= \sum_i^S CC'_i$

1. FQAI = _____
2. Mean C of C = _____
3. Weighted C of C of site = _____
4. What does the FQAI tell you about the quality of the wetland site?
5. Do the mean C of C or the weighted C of C provide similar or different interpretations to the FQAI? How are they similar or different?

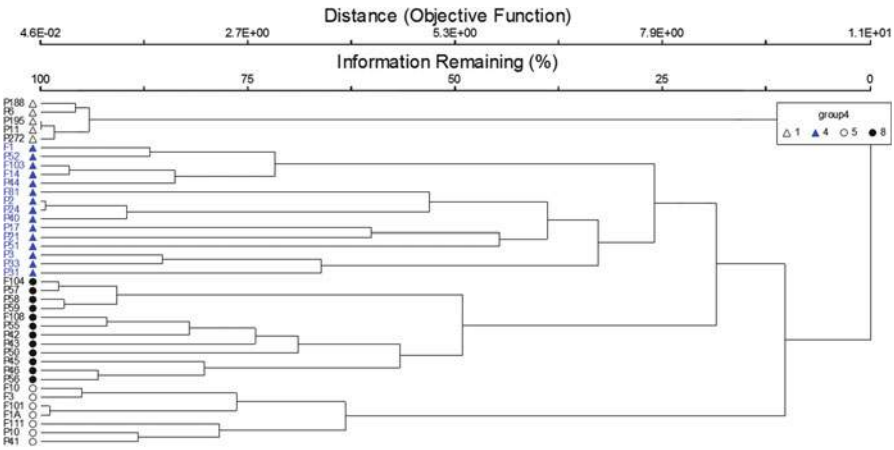

Fig. 5.15 Cluster dendrogram of 39 wetlands based upon *Sphagnum* community dissimilarity. Four groups have been constructed based upon interpretability

Exercise 4. Interpreting Multivariate Data

The following figures are output from a multivariate data analysis of 25 *Sphagnum* species found in 39 different wetlands. Wetlands were clustered into groups based upon their species dissimilarity using hierarchical cluster analysis (Fig. 5.15) and were ordinated within *Sphagnum* species abundance space using non-metric multidimensional scaling (Fig. 5.16).

1. Draw a line on the cluster dendrogram where the group cut-off occurs. What percent of information is remaining at this point?
2. If you were to divide the black circle group into four sub-groups, which wetlands would be included in each group?
3. Which wetland group is the least tightly clustered in this ordination diagram?
4. The red/gray lines are correlations of axes with environmental data collected in each wetland. Which wetland group contains the oldest wetlands? Which wetland group contains wetlands with the highest average groundwater specific conductivity?
5. Which group of wetlands is closest to the centroid for *S. inundatum* on the ordination diagram?
6. What species (three letter abbreviation) is most negatively correlated with Axis 3? Which species is most positively correlated with Axis 3?
7. Which wetland sites (numbers) most likely have the most *S. flavicomans*?

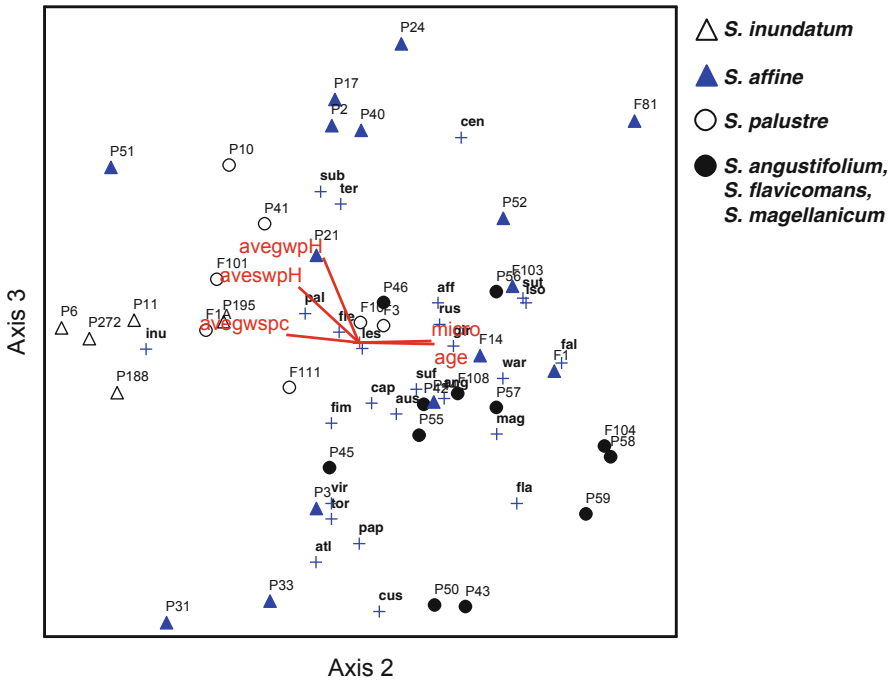

Fig. 5.16 NMS ordination of wetlands (labeled P or F) within *Sphagnum* species space. Wetlands were classified into four groups, named after group indicator species. Centroids of species abundance are labeled by *crosses*, with the three-letter species abbreviation (e.g., cap *S. capillifolium*). *Lines* are vectors of correlation with environmental variables; *longer lines* indicate stronger correlation. Micro microtopographic score, age time since most recent beaver inhabitation, avegwpH and aveswpH are groundwater and surface water pH, respectively, and avegwspc is mean groundwater specific conductivity

Exercise 5. Indicator Species

The table below contains data about the distribution of two species in degraded and non-degraded wetlands. Given these data, which species would be a better indicator of degradation and why?

	Mean abundance in group/ Mean abundance overall		% of sites within group in which species occurs	
	Degraded	Non-degraded	Degraded	Non-degraded
Typha angustifolia	0.92	0.08	100	10
Alnus incana	0.45	0.55	60	80

Chapter 6
Physical and Chemical Monitoring of Wetland Water

Joseph R. Bidwell

Abstract The physical and chemical attributes that comprise the "water quality" of a wetland have a significant influence on the system's biotic structure and function. Assessments of wetland water quality can be used for reference-based monitoring in the development and implementation of wetland water quality standards and to provide ancillary information in support of biotic surveys. While the methods used to evaluate water quality in wetlands are generally the same as those used for other surface waters, wetlands may differ in their dominant source of water, often have greater heterogeneity in habitat types, and can exhibit significant variability in water permanence. These characteristics can lead to spatial and temporal variability within and among wetlands that can make it difficult to use water quality data to detect human impacts. This chapter reviews some of the major sources of variability in wetland water quality and discusses approaches and general sampling considerations for characterizing basic water quality in wetland monitoring studies.

6.1 Introduction

The "water quality" of a wetland encompasses a range of physical and chemical attributes that largely determine its biotic composition and function. This chapter focuses on some of these physical and chemical variables (Table 6.1), with the goal of highlighting issues that should be considered when collecting these data. Even though a significant amount of "chemistry" goes on in wetland soils as they become anoxic after inundation (see Boon 2006; Mitsch and Gosselink 2007), the emphasis here will be on the water column as chemical and physical elements of the water are more commonly measured in routine monitoring of wetlands. This discussion will also primarily focus on inland wetlands, although coastal/tidal systems may exhibit significant variability in water quality as well.

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DOI 10.1007/978-94-007-6860-4_6,

Table 6.1 Overview of common water quality parameters measured in wetland monitoring studies

Parameter	General notes/reasons for measuring the parameter	Potential sources of variability to consider when monitoring wetlands
Temperature	Temperature has a major influence on chemical and biological processes including reaction rates, saturation constants of dissolved gases, water density, and the metabolic rates of animals	*Hydrology*: Water input from overland flow, precipitation or groundwater can increase or decrease temperature *Internal and other processes*: Diurnal and seasonal variation in water temperature occurs due to changing incident thermal radiation Differences in thermal regimes can exist between wetlands due to position on the landscape (e.g., shading effects, exposure to prevailing winds) Spatial variability within the wetland can occur due to differences in water depth, presence of macrophytes, and shading from shoreline vegetation Vertical stratification is possible in open water zones
Dissolved oxygen	As the terminal electron acceptor in aerobic respiration, the concentration of dissolved oxygen in water can influence the breakdown of organic matter in wetlands and the presence and distribution of many aquatic organisms that inhabit these systems Oxygen has a major affinity for electrons and determines the status of oxidation/reduction reactions that influence the chemical form of elements in biogeochemical and other chemical processes in wetland soils and water When coupled with other chemical indicators such as levels of nutrients, reduced oxygen levels in wetlands can be indicative of eutrophication	*Hydrology*: Water input from overland flow, precipitation or groundwater can influence dissolved oxygen levels either directly or indirectly by influencing temperature or introducing nutrients (overland flow and groundwater) into the system that stimulate biological productivity and ultimately biological oxygen demand *Internal and other processes*: Diurnal and seasonal variation in water temperature due to changing thermal regimes influences oxygen solubility Diurnal fluctuations in dissolved oxygen levels can result from the combined effects of photosynthesis and respiration Spatial variability in oxygen levels may result from differences in biological activity between stands of macrophytes and open water Vertical stratification of dissolved oxygen levels may also occur in open water zones

pH	pH is a measure of the hydrogen ion (H^+) concentration of water with values less than neutral (pH = 7) indicating acidic conditions and those above 7 indicating alkaline conditions. The pH of natural waters is significantly influenced by the carbonic acid equilibrium that develops when carbon dioxide dissolves in water: $CO_2 + H_2O \rightleftharpoons H_2CO_3 \rightleftharpoons HCO_3^- + H^+ \rightleftharpoons CO_3^{-2} + 2H^+$ The implications of this are that uptake of CO_2 by photosynthesizing plants can cause pH to increase while release of CO_2 due to respiration causes pH to decrease pH can influence chemical solubility and biogeochemical cycles. For example, acidic pH can increase the solubility and toxicity of sediment-bound metals, while increases in pH above 8 can lead to the loss of ammonia nitrogen (Catallo et al. 1995; Boon 2006) pH has been used as a water quality modifier in wetland classification systems (see Cowardin et al. 1979; Wetzel and Likens 2000)	*Hydrology*: Water input from overland flow, precipitation or groundwater can influence pH- systems with precipitation as the primary water source often have a pH <7 because precipitation tends to be slightly acidic Depending on basin geology and soil type, groundwater input could be acidic or alkaline, with overland flow similarly influenced *Internal and other processes*: Diurnal and spatial variability in pH can result from those same factors that influence dissolved oxygen Seasonal variation is also possible due to temperature effects on respiration
Alkalinity	Alkalinity of water is a measure of the capacity of constituent dissolved chemicals to accept and neutralize protons and is mostly attributed to the presence of carbonates (HCO_3^- and CO_3^{-2}) Alkalinity can indicate the geochemistry of the wetland basin and catchment since it commonly results from carbon dioxide and water attacking limestone or dolomite formations Alkalinity levels in wetlands can influence reactions associated with nutrient cycling (e.g., oxidation of ammonium (Parkes et al. 2007) and so monitoring this parameter can be important in assessing treatment wetland efficiency and understanding wetland nutrient profiles	*Hydrology*: Baseline alkalinity levels in wetlands are influenced by source water and local geochemistry *Internal and other processes*: Diurnal and seasonal variation in wetland water alkalinity has been associated with iron and sulphate reduction, denitrification, and assimilation of ammonium and nitrate (Eser and Rosen 1999; Sisodia and Moundiotiya 2006)

(continued)

Table 6.1 (continued)

Parameter	General notes/reasons for measuring the parameter	Potential sources of variability to consider when monitoring wetlands
Total dissolved solids	Total dissolved solids (TDS) is a measure of those solids in water that pass through a 2.0 μm filter and include ions such as sodium, chloride, and calcium While effects of elevated TDS on water quality are mostly related to impacts on domestic or industrial use, there could be physiological implications for freshwater organisms if levels become sufficiently elevated TDS and salinity are often used interchangeably and conversion factors are available to estimate one value from the other (see salinity)	*Hydrology*: Water source is the major source of between-wetland variability in TDS. For example, groundwater sources often have higher TDS levels due to contact with geologic material while wetlands that fill mostly via precipitation usually have lower TDS levels than those receiving groundwater or other surface input Temporal/seasonal variation in TDS could result from dilution effects due to precipitation events or evapoconcentration associated with water drawdown at the end of the hydroperiod (Charkhabi and Sakizadeh 2006) Spatial differences in TDS could occur within a wetland due to differences in water quality of tributaries flowing into the system.
Salinity	The sum of all dissolved ions in water determines the 'salinity', with the salinity of most inland waters dominated by Ca^{+2}, Mg^{+2}, Na^{+}, K^{+}, CO_3^{-2}, HCO_3^{-}, SO_4^{-2}, and Cl^{-} Salinity has been identified as a "keystone" chemical parameter because it differentiates freshwater from brackish and marine systems and so significantly influences plant and animal assemblages and wetland structure and function Salinity has been used as a water quality modifier in wetland classification systems (see Cowardin et al. 1979), although moderate variation in salinity common in most inland wetlands may not represent a significant influence on wetland organisms (see Mendelssohn and Batzer 2006)	*Hydrology*: See influences for TDS

Specific conductance (Conductivity)	Conductivity is a measure of the electrical resistance in an aqueous solution and depends on the total concentration of dissolved electrolytes Conductivity can serve as a general indicator of productivity of freshwater systems (with highly productive systems usually having higher conductivity than less productive systems) and water source Conductivity has been used as a surrogate for total dissolved solids (Trebitz et al. 2007) and factors for converting between the two parameters are available. (e.g., Conductivity × 0.67 ~ total dissolved solids (mg/L) (Dickerson and Vinyard 1999). More complex equations are available to estimate salinity from conductivity (APHA, AWWA and WEF 2005)	*Hydrology*: See influences for TDS *Internal and other processes*: Seasonal variation in conductivity has been associated with increased biological activity and chemical reactions that result from changing oxidation/reduction characteristics (Eser and Rosen 1999; Stratford et al. 2004) Increased biological activity may also lead to spatial differences within the wetland (although this is not extensively reported in the literature)
Total Hardness	Total hardness is a measure of divalent cations in water, primarily calcium and magnesium. Total hardness is not often evaluated in wetland monitoring studies although may be a useful modifier for some wetland classification schemes (e.g., Warner and Rubec 1997) or to establish wetland water quality criteria for metal contaminants (Gordon et al. 1997; Nimmo et al. 2006)	*Hydrology*: Water source and associated basin and catchment geochemistry represent the dominant source of between-wetland variation in water hardness Also see influences for TDS
Total suspended solids	Total suspended solids (TSS) is a measure of those solids in water that are retained on a 2.0 μm filter and include both organic and inorganic and living and dead material Elevated levels of suspended material in the water column can reduce light availability and lead to elevated water temperatures due to increased absorbance of thermal radiation. Light availability and temperature affect bacterial, algal, and zooplankton physiology, as well as the feeding and movements of macroinvertebrates and fish. Wetlands may reduce TSS of through-flowing water by facilitating settling of particulate material (Evans et al. 1996) When coupled with other chemical indicators such as levels of nutrients, TSS levels in wetlands can be indicative of eutrophication	*Hydrology*: Water source can influence TSS levels due to the import of particulate matter associated with overland flow and tributaries entering the wetland Temporal/seasonal variation in TSS could result from dilution effects due to precipitation events or evapoconcentration associated with water drawdown at the end of the hydroperiod (Boeckman and Bidwell 2007) *Internal and other processes*: Blooms of phyto and zooplankton can lead to increased levels of suspended solids and result in spatial differences in TSS within a wetland

(continued)

Table 6.1 (continued)

Parameter	General notes/reasons for measuring the parameter	Potential sources of variability to consider when monitoring wetlands
		Internal generation of solids occurs through fragmentation of detritus and litter, algal cells, and bioturbation by fish and invertebrates can further contribute to the concentration of wetland TSS (USEPA 2008a)
		Wind-driven mixing of wetland water can cause suspension of particulate material and result in elevated TSS. As such, landscape topography and wetland position on the landscape may also lead to differences between wetlands
Turbidity	Turbidity is a measure of water clarity and results from particulate matter (living and non-living) and dissolved color. See additional discussion for total suspended solids (TSS)	*Hydrology*:
	Trebitz et al. (2007) found that turbidity could serve as a possible surrogate for total suspended solids, although state that determining water transparency through the use of a "Secchi transparency tube" would also serve as a surrogate for TSS	See influences discussed for TSS, although color and dissolved organic carbon in source water can influence turbidity readings as well (Trebitz et al. 2007)
		Internal and other processes:
		See influences discussed for TSS. Biological activity and seasonal variation of dissolved organic carbon in the wetland water column (e.g. Waiser and Robarts 2004) could also influence turbidity readings
Nutrients	Nutrients include the macronutrients (e.g., phosphorus, nitrogen, sulphur, potassium, magnesium, and calcium) and micronutrients or trace elements (e.g., iron, copper, silicon). Nitrogen and phosphorous are often measured in routine water quality monitoring since they may be limiting and so could have a significant impact on productivity of the system. Nitrogen and phosphorus can both occur in the water column as organic and inorganic forms, with each having dissolved and particulate fractions. Different fractions of phosphorous are also designated based on reaction with molybdate (see Wetzel (2001) and APHA, AWWA and WEF (2005) for a more extensive description of these categories). Generally, the dissolved inorganic forms (soluble reactive in the case of phosphorous) are most readily available to plants, but particulate forms can indicate the overall available pool in the system.	*Hydrology*:
		Recent precipitation events may result in increased nutrient loading from overland flow while flooding from rivers may decrease or increase nutrient loads them depending on the nutrient status of the river (Weilhoefer et al. 2008)
		In seasonal wetlands, nutrient levels in the water column may increase shortly after inundation due to release from sediments. Nutrients may also increase during wetland drawdown due to evapoconcentration
		Internal processes:
		Seasonal variation in nutrient levels may occur due to temperature effects on biota and nutrient fluxes influenced by changing oxygen levels (Trebitz et al. 2007; USEPA 2008a)

Important chemical forms:

Nitrogen:

- *Dissolved inorganic nitrogen*: Ammonium: NH_4^+–N, Nitrate: NO_3^-N, Nitrite: NO_2^-–N
- *Total nitrogen (TN)*: All digestible forms of nitrogen (dissolved and particulate organic and inorganic nitrogen)
- *Total Kjehdahl Nitrogen (TKN)*: The sum of organic nitrogen and ammonium nitrogen. Commonly measured as part of permitting requirements for wastewater treatment plant discharges

Phosphorous:

- *Reactive phosphorous*: Phosphates that react with molybdate without preliminary digestion-mostly orthophosphate, PO_4^{3-}. *Total reactive phosphorous* is derived from water samples that have not been filtered, while dissolved or *soluble reactive phosphorous* (SRP) is derived from samples passed through a 0.45 μm filter
- *Dissolved inorganic phosphorous*: orthophosphate– basically the same as SRP
- *Total phosphorous (TP)*: All digestible forms of phosphorous (dissolved and particulate organic and reactive phosphorus)

The forms of nitrogen and phosphorous most commonly measured in water quality monitoring are total nitrogen and phosphorous, ammonium, nitrate, and orthophosphate/soluble reactive phosphate. However, due to the potential for rapid uptake of dissolved forms of inorganic nutrients, measurement of total fractions has been proposed as a better option for monitoring and characterizing nutrient dynamics (Norris et al. 2007; Trebitz et al. 2007)

Wetland sediments play a critical role in the cycling of nitrogen and phosphorous and the reader is referred to Boon (2006) and Mitsch and Gosselink (2007) for excellent overviews of wetland biogeochemistry

Spatial variation within a wetland may be observed between macrophyte beds and open water zones due to differences in biological activity (Rose and Crumpton 1996)

Vertical profiles may be observed with higher levels of ammonia near bottom sediments and release of orthophosphate from anoxic sediments (Ryder and Horwitz 1995; Glińska-Lewczuk 2009)

Unless indicated otherwise, information for general notes was derived from Wetzel and Likens (2000), Wetzel (2001) and APHA, AWWA and WEF (2005)

6.2 Uses of Water Quality Data in Wetland Monitoring

One of the more common approaches to assess wetland water quality is to compare measurements of current conditions with that of an ecologically similar but undisturbed or less-disturbed reference site (Norris et al. 2007). Brinson (1988) used this reference-based concept to help define "water quality", stating that good water quality represents the normal unaltered chemical condition with departures representing a deterioration in quality. The biogeochemical assessment module for wetlands developed by the United States Environmental Protection Agency (USEPA 2008a) is an example of a reference-based application of water quality data for wetland monitoring. A number of states in Australia have also incorporated various physical and chemical water quality parameters as a component of the reference-based *Framework for Assessing River and Wetland Health* (Norris et al. 2007; Alluvium Consulting 2011).

A second approach that uses chemical data to evaluate water quality is the comparison of specific chemical concentrations derived from site measurements with those deemed to support designated uses of the water or habitat (Norris et al. 2007; Chapelle et al. 2009). In the United States, this forms the basis for chemical-based wetland water quality standards as mandated by Section 303 of the Clean Water Act (USEPA 1990, 2008b; ELI 2008; Kusler 2011a, b). As stated in USEPA (1990), the original objective for US states was to have either narrative or chemical-based wetland water quality standards in place by 1993. However, as of 2011, only 14 states had adopted standards that were specific to wetlands (Kusler 2011a; also see ELI (2008) for a broader discussion of state-based wetland protection regulations in the US).

Physical and chemical data collected from wetlands may also provide important ancillary information to help understand the distribution of organisms in both basic and applied wetland studies. Water quality has a major influence on what organisms occur in a wetland (e.g., Dunson et al. 1997; Batzer et al. 2004; Euliss et al. 2004; Longcore et al. 2006; Ginocchio et al. 2008; Chen et al. 2011; Bojkova et al. 2011), with parameters such as salinity considered to be "keystone" variables that control plant and animal assemblages and drive wetland structure (Mendelssohn and Batzer 2006). Temperature, pH, dissolved oxygen, and alkalinity also dictate the extent and rate of important wetland functional processes such as nutrient transformations (Kadlec 1999). Physicochemical data have also been used to modify classification systems for wetlands (e.g., Cowardin et al. 1979; Warner and Rubec 1997).

6.3 What Makes Wetlands Different from Other Surface Waters When Monitoring Water Quality?

Mitsch and Gosselink (2007) state that while no particular biogeochemical processes are unique to wetlands, their flooding frequency (permanent or intermittent) can make certain chemical processes more dominant in wetlands than in other types of aquatic systems. In particular, the anaerobic conditions that prevail in wetland sediments can lead to the reduction of chemicals that influences their ultimate fate in the system.

Fluctuations in the presence and depth of water can also lead to greater variability in the physical, chemical and biological attributes of wetlands as compared to other surface waters. For example, Reeder (2011) identified the generally shallow nature of the wetland water column as a key driver of temporal and spatial variability in parameters such as dissolved oxygen. This can pose a challenge for wetland monitoring programs aimed at detecting anthropogenic disturbance since natural variation in biotic and abiotic assessment metrics can make it difficult to detect human effects. Within and between-site variability has also been identified as a key challenge in developing wetland water quality standards (Trebitz et al. 2007; Kusler 2011b).

6.4 Key Factors That Determine Wetland Water Quality

An understanding of the factors that influence the physical and chemical nature of wetland water can help identify sources of variation in these characteristics and assist in the design and implementation of wetland monitoring studies and interpretation of data derived from those studies. For the purposes of discussion, these factors are considered separately, although it is important to realize that these often work together and influence each other. As such, landscape effects on wetland water quality can occur due to effects on the chemistry of water entering the wetland as well as localized effects on biological processes within the system.

6.4.1 Hydrologic Influences

Surface water quality of a wetland integrates the combined influences of geologic setting, hydrology, presence and activity of biota, and human activity within or near the system (Carter 1996; Boon 2006; Azzolina et al. 2007). Of these, hydrology stands as the dominant factor that establishes the "baseline" levels of dissolved and particulate materials in wetland water. As presented by Bedford (1996) and further discussed by Boon (2006), climate and hydrogeologic setting establish key "hydrologic variables" that drive biogeochemical properties of wetlands. These variables include water source, the mineral and nutrient status of that water, and the spatial and temporal dynamics of water in the system.

Wetland water sources include precipitation, groundwater, and overland flow and, as would be expected, the mineral and nutrient status of each is often quite different. For example, vernal pools and other depressional wetlands that fill largely from precipitation commonly have low levels of dissolved solids and are more acidic (pH <7) (Whigham and Jordan 2003; Colburn 2004), while wetlands receiving mostly groundwater input may have variable pHs and dissolved ion levels based on underlying basin and catchment geology (LaBaugh 1989; Bedford 1996; Carter 1996; Winter et al. 2001; Whigham and Jordan 2003; Cabezas et al. 2009; Nelson et al. 2011). Groundwater-fed wetlands may also have lower temperatures

than those fed by precipitation and run-off (Korfel et al. 2010). The influence of local geology on groundwater and associated wetland water quality is nicely illustrated by the pH and nutrient content of fen wetlands which vary from quite alkaline (pH 8.4) and nutrient rich (rich or minerotrophic fens) to acidic (pH 3.5) and nutrient poor (acidic or poor fens) depending on the nature of the glacial deposits the source water comes in contact with (Bedford and Godwin 2003; Kolka and Thompson 2006; Nelson et al. 2011). Similarly, wetlands that receive significant input from overland flow often have water quality characteristics that reflect the soil properties of their catchments. This can be observed on a seasonal basis in some coastal wetlands of Lake Huron (Laurentian Great Lakes) that have coloured, acidic water with low conductivity and elevated phosphorous in spring due to input from upland watersheds (deCatanzaro and Chow-Fraser 2011).

In many cases, the chemical profile and associated variability in wetland water is driven by multiple water sources and/or by seasonal changes in water source. In the Lake Huron coastal wetlands mentioned above, upland inflow decreases in summer while input from seiche-derived lake water increases. This leads to increased alkalinity, higher conductivity, and reduced phosphorous levels as compared to springtime when inflow from upland catchments dominates (deCatanzaro and Chow-Fraser 2011). Euliss et al. (2004) state that while precipitation is generally the most significant water source for prairie pothole wetlands, input from groundwater can increase the concentrations of solutes and other dissolved materials. The relative solute concentration in these wetlands ultimately represents the combined effects of groundwater and precipitation. The scenario is further complicated by the length of groundwater flow paths between wetlands that may be influenced by precipitation patterns (Euliss et al. 2004). A comparable interplay between groundwater and surface water sources has been observed in some Australian wetlands (Boon 2006; Jolly et al. 2008). In some cases, groundwater input to depressional wetlands may also serve to buffer increases in dissolved solutes caused by evaporative water loss (Rains et al. 2006; Korfel et al. 2010). Variation in water source and connectivity through groundwater can have important implications for jurisdictional regulation of depressional wetlands that are considered isolated because they lack a clear connection to surface water (see Whigham and Jordan (2003) for further discussion).

Riparian wetlands that are subject to pulse flooding by rivers can have very distinct water quality profiles between the times they are flooded by the river and when they are isolated from it (Gell et al. 2002; Weilhoefer et al. 2008). Flood water can dilute levels of dissolved constituents in floodplain wetland water as observed by Weilhoefer et al. (2008) who report lower levels of conductivity, total phosphorous, and total nitrogen during and just after a flood event. They also concluded that the magnitude and duration of the flood was an important determinant of how much the wetland water quality changed and that flooding could increase nutrient levels in a wetland depending on the nutrient status of the river. Cabezas et al. (2009) also studied the water quality of floodplain wetlands and found that the seasonal chemical profiles of the systems were largely influenced by the relative inputs of river and groundwater. Wetlands receiving major input from the river during flooding had lower conductivity but higher turbidity and nitrate levels, while those receiving mostly groundwater had higher conductivity and lower turbidity.

Wetland hydroperiod (seasonal wetting and drying) has an obvious influence on the abiotic and biotic features of wetlands (Brooks 2000; Jackson 2006) and differences in water between wetlands at different stages of the hydroperiod could be a significant source of between-site variability. Oxidative metabolism of organic matter in the substrate of a dry wetland can result in a pulse of nutrients to the water column when the wetland refloods (Euliss et al. 2004), although Boon (2006) states that nutrient release in newly-flooded wetlands may be derived from other sources as well. Regardless, a succession of chemical reactions occurs in saturated wetland soils as oxygen becomes depleted and new substrates are used as electron acceptors by respiring microorganisms (Boon 2006; Mitsch and Gosselink 2007). Concurrent changes in variables such as dissolved oxygen, pH, dissolved organic carbon, conductivity and water clarity would be expected to occur in the wetland water column as photosynthetic and respiratory processes become established and suspended material begins to settle out, although few studies have evaluated this with sufficient sampling frequency to effectively document the changes that do occur.

Most discussion of wetland hydroperiod and water quality focuses on how water loss influences water quality parameters. Euliss et al. (2004) discuss the "drought" versus "deluge" phases in prairie pothole wetlands in reference to changing solute concentrations that can influence wetland biota. Similarly, evaporative water loss and associated concentrating effects were used to explain stable isotope signatures and increased summertime cation levels in New Zealand peatlands (Chague-Goff et al. 2010), seasonal variation in salinity of arid and semi-arid zone wetlands in Australia (Jolly et al. 2008), and spatial differences in parameters such as conductivity, dissolved organic carbon, and levels of specific dissolved ions between different wetland zones of the Okavango delta (Mackay et al. 2011). Increases in other water quality parameters including total nitrogen, total phosphorous, pH, alkalinity, and hardness have also been associated with evapoconcentration as seasonal wetlands proceed through the hydroperiod (Gell et al. 2002; Boeckman and Bidwell 2007). Water loss and associated evapoconcentration effects on water quality variables can also vary considerably between wetlands based on factors such as basin size, localized landscape position of the wetland, and the presence of plants which can enhance water loss by evapotranspiration (Euliss et al. 2004; Jackson 2006).

Interestingly, increases in the levels of water quality variables due to evapoconcentration may have only limited influence on biotic communities in inland wetlands unless these exhibit extremely elevated conditions as has been described for prairie pothole wetlands and some other systems (Euliss et al. 1999; Mendelssohn and Batzer 2006). Batzer et al. (2004) report variation of up to two orders of magnitude in a suite of water quality parameters they measured in a series of forested wetland ponds that differed in hydroperiod and landform and found these factors had a relatively minor influence on macroinvertebrate assemblages. Babbitt et al. (2003) used plant assemblages and site visits to group a series of forested depressional wetlands according to hydroperiod duration and found permanently inundated wetlands were slightly warmer and had slightly higher pH, higher dissolved oxygen, and lower conductivity than wetlands with the shortest

hydroperiod. Based on ordination analysis, dissolved oxygen was the only water quality parameter found to have a strong influence on amphibian assemblages in the wetlands. As such, in some cases, differences in plant and animal assemblages between wetlands with different hydroperiod durations may be driven more by the actual presence of water and available habitat rather than changing water quality (see Jackson (2006) for further discussion).

Finally, water movement through a wetland can establish important within-site spatial differences in water quality, with contrasting levels of parameters such as dissolved oxygen, suspended solids, nutrients, and contaminants observed between the inflow and outflow zones (Ibekwe et al. 2007; Diaz et al. 2012). Trebitz et al. (2005) evaluated the influence of hydrology and geomorphology on water quality (temperature and dissolved oxygen) and other habitat parameters in Lake Superior coastal wetlands and in some cases, found within-wetland spatial differences due to seiche action and tributary inputs that were as large or larger than differences between wetlands.

6.4.2 *Landscape Influences*

Attributes of the landscape in which a wetland occurs can influence water quality through effects on hydrology, characteristics of water entering the system, and localized effects on wetland microclimate. A number of studies have linked differences in parameters such as levels of suspended solids and nutrients between wetlands with landscape-level differences in agricultural intensity, human population density, and point source pollution (Trebitz et al. 2007; Morrice et al. 2008; Cabezas et al. 2009). In their study, Trebitz et al. (2007) combined individual water quality parameters via principal components analysis into a wetland water quality metric that was more responsive to agricultural intensity than any single parameter alone. Physical characteristics such as size and drainage slope of watersheds that feed wetlands have also been found to significantly influence chemical parameters such as pH and levels of suspended solids and nutrients (deCatanzaro and Chow-Fraser 2011). Differences in parameters such as temperature and pH between forested depressional wetlands have been found to result from subtle differences in landscape topography, forest canopy cover, and tree age and size (Batzer et al. 2000; Skelly and Freidenburg 2000; Hossack and Corn 2008). In an investigation of what constitutes appropriate buffer zones adjacent to wetlands, Houlahan and Findlay (2004) found that water column nutrient levels in the systems studied were influenced by forest cover quality at over 2,000 m from the wetland edge.

6.4.3 *Internal Influences*

While the hydrologic variables discussed above are significant determinants of wetland water quality, it is important to also consider the effects of internal biological processes on chemical parameters and the physical influences that factors

such as temperature and light availability have on these processes. This is clearly illustrated by the differences in wetland water quality that can exist between different habitat types in a single wetland. For example, due to the link between CO_2 and the carbonic acid buffering system in water (Wetzel and Likens 2000, also see Table 6.1), uptake of CO_2 by photosynthesizing aquatic plants increases pH while release of CO_2 by respiring organisms decreases it. The competing effects of photosynthesis and respiration similarly influence dissolved oxygen levels which can have effects on other important parameters such as the oxidation and reduction (redox) potential in water and sediments.

Stands of aquatic macrophytes in particular can have a major influence on commonly measured water quality variables. Dense mats of floating plants and high levels of algal biomass have both been associated with localized increases in water temperature (in some cases by as much as 11 °C, see Reeder (2011) for discussion), while shading from emergent vegetation may locally reduce water temperature (Rose and Crumpton 1996). High microbial respiration associated with decaying plant biomass and reduced diffusion of atmospheric oxygen often leads to near anoxic conditions and reduced pH of the water surrounding beds of emergent and submergent plants as compared to open water zones (Chimney et al. 2006; Rose and Crumpton 2006).

Distinct vertical profiles in water quality variables due to thermal stratification of the water column are well described for deeper ponds and lakes, and may also be observed in the wetland water column. Ryder and Horwitz (1995) report significant differences in temperature, pH, conductivity, dissolved oxygen and redox potential between the surface and bottom of the permanently inundated zone of a depressional wetland in Australia. This stratification was observed in water less than 1.5 m deep, was most pronounced near stands of macrophytes, and exhibited a diel pattern of formation from early afternoon to early evening. Diel cycles of thermal stratification and associated vertical profiles of dissolved oxygen and dissolved methane were also observed in a shallow Australian floodplain wetland (Ford et al. 2002). In this study, surface water oxygen levels were highest in late afternoon and would sometimes be near zero by morning due to high respiratory demand. Boeckman and Bidwell (2007) also observed summertime thermal stratification across a maximum depth of 30 cm in an Oklahoma depressional wetland that resulted in vertical profiles of dissolved oxygen, pH, and suspended solids. The shallow nature of wetlands can make stratification of the water column quite transient as it is easily disrupted by wind (Boeckman and Bidwell 2007). However, stratification may still have an influence on wetland functional processes as indicated by Ryder and Horwitz (1996) who attributed reduced leaf processing in certain areas of the wetland they studied to limitations on microorganisms and invertebrates imposed by the diurnal stratification of the water column.

6.4.4 Temporal Influences

Temporal changes in wetland water quality can be driven by changing hydrologic conditions (see previous discussion of wetland water sources and hydroperiod) and

biological activity in the system. Temperature, dissolved oxygen, and pH are among those parameters most prone to diurnal fluctuations as a result of the combined effects of solar heating, radiant cooling, photosynthesis, and respiration, although diurnal variation in parameters such as conductivity and alkalinity have also been observed (Ryder and Horwitz 1995; Stratford et al. 2004; Sisodia and Moundiotiya 2006; Tuttle et al. 2008; Reeder 2011). Cornell and Klarer (2008) report dissolved oxygen levels in a Lake Erie coastal wetland varied between 20 and 150 % saturation over the course of the day and comparable dissolved oxygen fluctuations have been observed in stands of emergent vegetation in restored and natural floodplain wetlands (Boon 2006; Reeder 2011). High levels of photosynthetic activity can also lead to significant increases in pH, although these may be attenuated by other chemical characteristics of the system. Boon (2006) discusses studies of an Australian wetland that exhibited daily pH changes of up to 2 pH units and observed that this fluctuation could influence nitrogen dynamics in the system if the pH were to rise above 8 and convert ammonium to the more volatile ammonia. In contrast, Reeder (2011) did not observe a significant influence of primary productivity or respiration on the pH of restored floodplain wetlands, and attributed these results to buffering by divalent cations in the sediments and/or reduced effects of microbial respiration due to low levels of sediment organic matter.

The intensity of diurnal fluctuations in wetland water quality variables may vary based on habitat type and water source. As compared to open water zones, diurnal fluctuations in temperature may be dampened near stands of emergent vegetation, although daily fluctuations in dissolved oxygen and pH may be greater in beds of both emergent and submergent plants (Rose and Crumpton 1996; Chimney et al. 2006; Reeder 2011). Diurnal fluctuations in temperature and dissolved oxygen were reduced during hydrologic pulses in created riparian wetlands (Tuttle et al. 2008), while groundwater input in vernal pools has also been found to reduce daily temperature fluctuations (Korfel et al. 2010). Daily fluctuations in conductivity and water color in Great Lakes coastal wetlands was attributed to seiche-induced inflow of lake water that increased levels of dissolved ions and diluted color (deCatanzaro and Chow-Fraser 2011).

In temperate zones, seasonal changes in thermal input can lead to significant seasonal variation in wetland water temperature, with differences of 30 °C or more between minimum and maximum temperatures not uncommon (Black 1976; Boeckman and Bidwell 2007). These temperature differences drive seasonal changes in biotic and abiotic processes that influence other water quality parameters. For example, dissolved oxygen levels are often higher during cooler months owing to higher gas solubility and lower respiration (Boeckman and Bidwell 2007). Diel fluctuations in dissolved oxygen, pH, and alkalinity may be greater in summer due to increased rates of photosynthesis and respiration and reduced solubility of oxygen and CO_2. Levels of nitrate and orthophosphate in wetland water may be lower in summer due to greater uptake by plants or, in the case of nitrate, increased rates of denitrification (Mitsch and Reeder 1992; deCatanzaro and Chow-Fraser 2011). However, Eser and Rosen (1999) report seasonal maxima for nitrate and ammonium in late summer which was attributed

to nutrient release from the breakdown of organic matter coupled with reduced plant uptake as the peak growing season begins to trail off. Summer increases in wetland orthophosphate levels have also been observed and related to possible release of the nutrients from anoxic sediments (Glińska-Lewczuk 2009). Increased water concentrations of other elements, including potentially toxic metals such as cadmium, have been reported in some temperate wetlands and have been attributed to release of these chemicals from sediments due to increasing temperatures and changing redox conditions in spring (Olivie-Lauquet et al. 2001).

6.5 The Role of Wetlands in Improving Water Quality

Discussions related to wetlands and water quality often focus on "improving water quality" as one of the key services wetlands provide and studies on both natural and created wetlands have demonstrated clear effects on the chemical and physical characteristics of water moving through these systems (see Barnes et al. 2002; O'Geen et al. 2010; Dotro et al. 2011 for basic examples). Some of the proximate mechanisms that underlie these water quality effects have been reviewed by Hemond and Benoit (1988) and Verhoeven et al. (2006). However, there are a few general qualifiers that are worth keeping in mind when considering the "water quality" function of wetlands. First, some wetlands (e.g., some types of vernal pools) have little influence on water quality due to limited hydrological linkage to other waters (Rains et al. 2006). Second, the capacity of wetlands to enhance water quality may ultimately depend on the area of wetland available relative to the total catchment area. For example, Verhoeven et al. (2006) state that wetlands can significantly affect catchment water quality if wetland habitat makes up at least 2–7 % of the catchment area. Finally, in some cases, activities in the surrounding landscape and wetland alterations can actually lead to wetlands becoming sources of sediments, nutrients, and toxic chemicals that could actually degrade downstream water quality (Brinson 1988; Whigham and Jordan 2003; Verhoeven et al. 2006).

6.6 General Study Design and Approaches

6.6.1 Study Design

The objectives and questions to be addressed in collecting water quality data from a wetland should be clearly defined prior to the start of the study since this will inform the study design and associated statistical analyses. USEPA (2002a) discusses some of the key objectives that should be considered in a wetland monitoring study and provides guidance on site selection and sampling designs. Common sampling designs used for wetland water quality monitoring include stratified random sampling,

targeted/tiered sampling, and before-after, control-impact (BACI), with the choice of approach dependant on the project objectives (USEPA 2002a). The number of replicate measurements or samples to be taken for analyses of specific wetland water quality parameters is also an important consideration, although often appears to be arbitrarily determined in monitoring studies. An excellent discussion of approaches to determine effective sample number based on study design is available in USEPA (2002b).

6.6.2 Wetland Classification

Wetland classification is aimed at controlling some of the natural variability in the monitoring data derived from wetlands by grouping them according to common physical or biological characteristics such as hydrology, hydrogeomorphology, and/or vegetative assemblages (e.g., Cowardin et al. 1979; Brinson 1993; Reinelt et al. 2001; Jackson 2006). Water quality has also been used to group wetlands or modify classification systems (e.g. Cowardin et al. 1979; Warner and Rubec 1997). Brinson (1988) provides a conceptual discussion of how landscape position and associated flow characteristics, both of which are attributes used for geomorphological classification of wetlands, would influence elemental cycles and wetland water quality.

Studies that have specifically evaluated the extent to which classification helps control variability in water quality parameters among wetlands are limited, although those that are available indicate that broad scale classification may not be particularly effective in this regard. For example, Trebitz et al. (2007) and Morrice et al. (2008) found that grouping Great Lakes coastal wetlands according to relatively coarse hydromorphic types or biogeographic region did not enhance their ability to relate wetland water quality to land use and Trebitz et al. (2007) concluded that finer hydrologic classification would have been desirable in their study. Similarly, Azzolina et al. (2007) were unable to detect significant differences in surface water quality between wetlands grouped according to a modification of Brinson's hydrogeomorphic (HGM) classification (specifically, the LLWW approach as described by Tiner (2003) and Tiner and Stewart (2004)). Euliss et al. (2004) state that while regional landscape position often explains many of the chemical and biological properties of wetlands, finer-spatial scale and temporal influences are also significant determinants of wetland water quality. This is clearly supported by studies that have demonstrated how localized landscape factors and basin characteristics can influence water quality in individual wetlands (e.g., Skelly and Freidenburg 2000; Batzer et al. 2000; Hossack and Corn 2008). The development of regional wetland subclasses as described in the HGM approach (Brinson 1993) or even finer-scale modifiers of wetland classes may therefore be necessary to effectively reduce natural variation in water quality among wetlands.

6.6.3 *Temporal and Spatial Considerations*

As described above, wetland hydroperiod could influence water quality and lead to divergence between sites due to differences in water loss and associated effects of evapoconcentration. Ideally, water quality would best be compared between wetlands at comparable stages of their hydroperiod. Jackson (2006) provides a discussion of approaches that could be used to characterize wetland hydroperiod, including regular site visits, and use of devices such as staff gauges, automatic water level monitors and piezometers. Unfortunately, this type of intensive surveying may not be possible for studies with limited resources or that have the goal of sampling a large number of wetlands. Species composition of resident plant assemblages has also been used to provide a general indication of the duration of wetland hydroperiod (Babbitt et al. 2003; Sharitz and Pennings 2006) and this may assist in either developing finer-scale groupings of wetlands or as ancillary data to help with interpretation of observed patterns in water quality data.

If the objective of a wetland monitoring study is to obtain water quality data that represent the system as a whole, potential spatial differences between different habitat types should be considered (Fig. 6.1). This could be addressed by first mapping the wetland to identify major habitat types and tributaries running into the system, and then using these areas as strata in a stratified random sampling approach. Vertical stratification of open water zones of the wetland may also need to be considered, particularly if functional parameters such as carbon processing are being assessed (e.g. Ryder and Horwitz 1996). Representative values for the wetland as a whole could then be generated by simply calculating the arithmetic mean of the data derived from each habitat type or by using weighted averages to provide proportional representation for each habitat. Another alternative is to develop composite samples of water derived from different wetland strata and conduct all chemical analyses on those samples. If representative sampling of different habitats in the wetland is not possible, limiting sampling to only the major habitat type may be the next best option. Some effort should also be made to record water quality measurements from comparable habitat types in each wetland visited. Unfortunately, published wetland studies that include collection of water quality data often fail to indicate the type of habitat the data were derived from or if potential habitat variability in the parameters being measured was considered in the study design.

Diurnal variation in water quality driven by changing temperatures, photosynthetic activity and respiration may best be addressed by sampling wetlands within a defined time period (e.g., 10 AM to 2 PM) and some published studies do indicate the time interval during which water quality measurements were made in wetlands (e.g., Trebitz et al. 2007; deCatanzaro and Chow-Fraser 2011). However, this may again pose a challenge for studies aimed at sampling a large number of wetlands or in cases where travel time between sampling sites is significant. This is a good reason to include the time of day the wetland was sampled on field data sheets since this information could be used in later data analysis to determine if any diurnal trends in the data are apparent. Routine water quality monitoring of wetlands often

Fig. 6.1 A freshwater wetland with open water, emergent and floating vegetation. These different habitat zones may differ significantly in water quality parameters such as temperature, dissolved oxygen, pH, and alkalinity

occurs during warmer summer months, although some consideration should be given to the seasonal sampling window since differences in key biological processes that influence water quality could exist depending on when in the growing season individual wetlands are visited.

6.7 General Methodology

The methods used to collect chemical and physical water quality data from wetlands are largely the same as those used to collect these data from other surface waters. Detailed treatments of the chemistry of running and standing waters can be found in any number of basic freshwater ecology or limnology textbooks (e.g., Wetzel 2001), while Mitsch and Gosselink (2007) provide specific overview of wetland biogeochemistry. Similarly, excellent texts that provide in-depth discussions of water sampling and analytical methods are available. These include *Limnological Analyses* by Wetzel and Likens (2000) and *Methods in Stream Ecology* by Hauer and Lamberti (2007). *Standard Methods for the Examination of Water and Wastewater* (APHA, AWWA and WEF 2005) is also a critical reference for those interested in evaluating abiotic and biotic conditions in natural waters. Additional sampling methods and analytical procedures for wetland water quality are available in USEPA (2011).

The range of chemical and physical variables that could be measured in wetland water is extensive and ultimately depends on the objectives of the study. Those listed in Tables 6.1 and 6.2 were selected because they include those variables commonly measured in wetland monitoring programs and because the methodology for most involve basic equipment such as electronic water quality meters or relatively simple wet chemistry procedures. A number of suppliers also offer "environmental laboratory" kits that provide step-by-step methodology using pre-packaged reagents for the analyses of a number of these variables in the field. In most cases, the methods employed in these kits are derived from protocols described in APHA, AWWA and WEF (2005) and listed in Table 6.2. If water quality monitoring data are to be used for regulatory purposes, attention should be paid to whether analytical techniques are acceptable to state or federal regulatory agencies. Some of the methods presented in Table 6.2, such as those for nutrient determinations, have also been adapted for laboratory-based autoanalyzers that allow high through-put of samples.

Other water quality parameters such as forms of organic carbon and individual anions and cations are also often measured in wetland studies but are not discussed in detail here since current methods for their determination involve more expensive types of instrumentation. This is also the case for metals and organic contaminants. A general overview of this instrumentation is provided by Wetzel and Likens (2000). APHA, AWWA and WEF (2005) also provide general overviews and basic methodology for analyzing selected metals and organic contaminants including collection and processing of samples.

In addition to the sampling considerations already discussed, other issues must often be considered when collecting water quality data from surface waters including wetlands. Electronic water quality meters can greatly enhance data collection from the field, with some models allowing the determination of a number of parameters at one time. However, if these instruments are to provide reliable data, attention must be paid to their calibration and maintenance. Without proper calibration, a water quality meter can become an expensive random number generator. As such, all personnel using the device should become acquainted with the manufacture-prescribed frequency for calibration and the calibration procedure. Maintaining records on when the meter was calibrated and by whom is also an important element for data quality assurance. Similarly, regular inspection of the probes of the meter for damage or fouling can help ensure reliable output during use. Measurements taken with a water quality meter often rely on the passage of gasses or ions across an electrode membrane and it may be necessary to provide gentle agitation of the probe to enhance this exchange. Some probes are fitted with small impellers to maintain this water flow. In addition to the handheld water quality meter, a range of other electronic devices are available to collect water quality data from surface waters including temperature loggers and multi-parameter sondes that can be left on site to facilitate data collection over longer time frames. These instruments are particularly useful for characterizing diurnal patterns, although their use may be limited in shallower wetlands.

When using a water quality meter to take measurements from the wetland water column, care must be taken to avoid stirring bottom sediments or allowing the probe

Table 6.2 Summary of methods for the determination of water quality parameters commonly measured in wetland monitoring studies

Parameter (common units)	Example instrumentation/method	Collection container	Minimum sample size (mL)	Preservation	Maximum storage time	Method reference/ discussion[a]
Temperature (°C)	Thermometer, Temperature sensor/Water quality meter, data logger	*In situ*, plastic, glass	N/A	Analyze immediately	NA	1, 2
Dissolved oxygen (mg/L or % saturation)	Oxygen electrode/Water quality meter, Winkler titration	*In situ*, glass	300	Winkler titration-acidification step	Immediately if by electrode, Winkler titration- 8 h after acidification	1, 3
pH	pH electrode/Water quality meter	*In situ*, plastic, glass	50	Analyse immediately	NA	1, 4
Alkalinity (as mg $CaCO_3$/L)	Acid titration with indicator solution	Plastic, glass	200	Refrigerate	24 h	1, 8
Total dissolved solids (mg/L)	Sample filtration, drying and weighing	Plastic, glass	200	Refrigerate	7 days	10
Salinity (g/L)	Some water quality meters derive salinity from conductivity measurements, hand-held refractometers can provide a general indication	Glass	240	Analyze immediately	NA	1, 6
Specific Conductance (microSiemens/ cm = µS/cm)	Conductivity probe/Water quality meter	*In situ*, plastic, glass	500	Refrigerate	28 days	1, 5
Total Hardness (as mg $CaCO_3$/L)	EDTA titration	Plastic, glass	100	Add HNO_3 or H_2SO_4 to pH <2	6 months	1, 7
Total suspended solids (mg/L)	Sample filtration, drying and weighing	Plastic, glass	200–depends on level	Refrigerate	7d	11
Turbidity (Nephelometric turbidity units, NTU)	Nephelometer/Turbidity meter	Plastic, glass	100	Refrigerate in dark	24 h	1, 9

Ammonia (mg/L)	Phenate method, Salicylate method, Ammonia selective electrode	Plastic, glass	500	Reduce pH <2 by addition of H_2SO_4 and refrigerate	7 days	1, 12, 13
Nitrate (mg/L)	Cadmium reduction method	Plastic, glass	100	Refrigerate	48 h	14
Nitrite (mg/L)	Diazotization colorimetric method	Plastic, glass	100	Refrigerate	48 h	15
Total Nitrogen (mg/L)	Persulfate digestion followed by nitrate determination	Plastic, glass	500	Refrigerate	48 h	16
Total phosphorous (mg/L)	Persulfate digestion followed by ortho-phosphate determination	Plastic, glass	100	Reduce pH <2 by addition of H_2SO_4 and refrigerate	28 days	17
Orthophosphate (mg/L)	Ascorbic acid method	Acid washed glass	100	For dissolved– filter immediately and refrigerate	48 h	18

All information on sample collection, sample size, preservation, and storage from APHA, AWWA and WEF (2005). Method summaries based on those listed here or alternative procedures can also be accessed on the USEPA website at http://water.epa.gov/scitech/methods/cwa/methods_index.cfm

[a]Manufacturers manual for meters; 1: Wetzel and Likens (2000); 2: Hauer and Hill (2007); 3: APHA, AWWA and WEF (2005) Method 4500-OA-G; 4: APHA, AWWA and WEF (2005) Method 4500-H^+; 5: APHA, AWWA and WEF (2005) Method 2510 A-B; 6: APHA, AWWA and WEF (2005) Method 2520B; 7: APHA, AWWA and WEF (2005) Method 2340C; 8: APHA, AWWA and WEF (2005) Method 2320; 9: APHA, AWWA and WEF (2005) Method 2130; 10: APHA, AWWA and WEF (2005) Method 2540C; 11: APHA, AWWA and WEF (2005) Method 2540D; 12: APHA, AWWA and WEF (2005) Method 4500-NH_3; 13: Reardon et al. (1966); 14: APHA, AWWA and WEF (2005) Method 4500-NO_3^- E; 15: APHA, AWWA and WEF (2005) Method 4500-NO_2^- ; 16: APHA, AWWA and WEF (2005) Method 4500-N C; 17: APHA, AWWA and WEF (2005) Method 4500-P B,E; 18: APHA, AWWA and WEF (2005) Method 4500-P E

Fig. 6.2 A long-handled sampling pole with a collection bottle on the end can be used to avoid disturbing sediments and/or facilitate collecting water samples from deeper areas of the wetland

to contact sediments since this can significantly influence readings for parameters such as dissolved oxygen. This can sometimes be challenging in shallower systems or if it is necessary to wade into the wetland to access open water. If the wetland is large and deep enough, accessing open water by boat can help avoid these issues although attention must still be paid to where the water quality probe is located in the water column. The use of a long-handled sampling pole with a collection bottle on the end (Fig. 6.2) can be used to pull grab samples of water that can be analysed on shore with a water quality meter or transferred to appropriate containers for determination of wet chemistry parameters. In wetlands with deeper basins, depth-specific samples for wet chemistry analyses can also be collected using a commercial sampling device such as a Van Dorn sampler (Wetzel and Likens 2000). Basic quality assurance for wet chemistry procedures includes following appropriate sample storage and preservation techniques and conducting the analyses within the prescribed time frame to ensure sample viability (Table 6.2). The use of properly cleaned glassware to avoid inaccurate readings due to contamination is also a key issue if conducting sample analyses in house. Appropriate sample labelling and tracking methods are also important considerations. Guidance on analytical quality assurance and data handling is provided by Briggs (1996) and APHA, AWWA and WEF (2005)

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Student Exercises

Laboratory Exercises

Laboratory Exercise #1: Spatial Variation in Water Quality

1. Select a study wetland with different habitat types that can be accessed to determine water quality variables and develop a hand-drawn habitat map of the wetland to delineate major habitat types and structural features (e.g., inlets and outlets, stands of emergent, submergent, floating plants, open water). These habitat types will be used to develop hypotheses and as sampling strata in Step 2.
2. Based on the different habitat types in the wetland identified in Step 1 and an understanding of how wetland processes may influence water quality parameters, develop a series of hypotheses or predictions related to how the water quality variables being measured would be expected to differ between the habitat types. For example, how would you expect temperature, dissolved oxygen, and pH to compare between open water and stands of macrophytes?
3. Using a calibrated multi-parameter water quality meter, determine temperature, dissolved oxygen, pH, and conductivity for each wetland habitat. Other water quality parameters (e.g., turbidity, nutrients) may also be determined depending on availability of equipment**. If the wetland has an open water zone that is accessible by boat, use the water quality meter to generate a vertical profile of the water quality parameters from just above the sediment surface to just under the water surface. Space the measurements so that data are collected from at least three depths.
4. Provide a brief summary of whether the data collected supported your hypotheses and a brief discussion of the basis for the results observed.
5. Questions to consider:

    - What are some reasons for any observed differences in the water quality parameters measured between the wetland habitat types?
    - How would you determine if any relationships exist between the parameters measured? (i.e., dissolved oxygen vs. temperature vs. pH)
    - If relationships between parameters are observed, what is the basis for these?

**Water samples may also be collected in clean 1-L plastic bottles to take back to the laboratory for determination of "wet chemistry" parameters such as nitrate, orthophosphate, and alkalinity.

Laboratory Exercise #2: Temporal Variation in Water Quality

1. Select two or three different habitat types within a wetland and develop a sampling schedule that allows collection of water quality data (temperature,

dissolved oxygen, pH, and conductivity) using a calibrated water quality meter at 3 h intervals from dawn until dusk.
2. Develop a graph for each variable measured by plotting the level of the parameter measured against time.
3. Questions to consider:

   - Explain the basis for any observed fluctuations. What biotic and/or abiotic processes underlie the observed changes in water quality parameters over time?
   - Do any relationships exist between the water quality parameters measured? If so, what are they? Explain the basis for these relationships.

Laboratory Exercise #3: Land Use Influences on Water Quality

Select a series of depressional or other wetland class that exist across different land use types. For example, crop versus pasture, urban versus park land.

1. Use available resources (topographic maps, digital orthophotographs, etc.) to characterize the major land use types within 1 km of each wetland. Based on this analysis, develop a series of testable hypotheses related to land use and water quality of the wetlands. For example, how would nutrient levels or other water quality parameters of a wetland surrounded by crop land or within a golf course be expected to differ from systems within less disturbed landscapes?
2. What key considerations should be addressed when attempting to compare water quality data between different wetlands?
3. Construct basic habitat maps for each wetland when on site. Also determine the existence of any undisturbed buffer zones around each wetland.
4. Use a calibrated water quality meter and/or collect water samples in clean 1 L bottles for later analysis to obtain water quality data from representative habitats in each wetland. Use these data to address the hypotheses/predictions about the differences between wetlands.
5. Questions to consider:

   - Do any observed differences in water quality parameters match what you would expect based on land use types? If so, briefly explain the basis for these differences.
   - If no real differences are detected, what are some reasons for this?
   - What internal wetland processes may influence wetland water quality and potentially mask differences related to land use effects?

Chapter 7
Wetland Biogeochemistry Techniques

Bruce L. Vasilas, Martin Rabenhorst, Jeffry Fuhrmann, Anastasia Chirnside, and Shreeam Inamdar

Abstract Biogeochemistry is the scientific discipline that addresses the biological, chemical, physical, and geological processes that govern the composition of the natural environment, with particular emphasis placed on the cycles of chemical elements critical to biological activity. Biogeochemical assays may measure a specific elemental pool, determine the rate of a pathway, or address a surrogate of a biogeochemical process or an elemental pool. In this chapter, we have attempted to emphasize field techniques; however, some of the techniques have relatively standard laboratory components that are beyond the scope of this chapter. This chapter is not meant to be all inclusive. We have chosen to emphasize the cycling of carbon, nitrogen, phosphorous, sulfur, manganese, and iron. Some of these techniques are not appropriate for all types of wetlands, or may be appropriate for a seasonally saturated wetland only during part of the season. Some of the techniques are simple and rely on equipment available to most wetlands practitioners. Others, which utilize isotopic methodologies, require expensive sophisticated equipment. Some techniques, such as soil organic matter determination by loss on ignition, have been accepted as standard methods for decades. Others, such as the determination of dissolved organic matter represent recent advances in a rapidly evolving field of ultra-violet and fluorescence technology. Some techniques rely solely on direct field measurements; others rely on the

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incorporation of published data with field data. Apparent strengths and weaknesses of the various approaches, and wetland scenarios that would preclude the use or compromise the accuracy of a given technique are addressed.

7.1 Overview of Techniques

Biogeochemistry is the scientific discipline that addresses the biological, chemical, physical, and geological processes that govern the composition of the natural environment. Particular emphasis is placed on the study of the cycles of chemical elements such as carbon (C), nitrogen (N), and phosphorous (P) which are critical to biological activity. Biogeochemical assays may measure a specific elemental pool (e.g., soil organic carbon), determine the rate of a pathway (e.g., denitrification), or address a surrogate of a biogeochemical process or an elemental pool. The surrogate approach is popular for rapid assessment to characterize ecosystem health, functional capacity, nutrient loading, or water quality. In each case the practitioner must be aware of the exact nature of the parameter in question as well as limitations to the method. Attempts to quantify individual pools of C or N at best, produce representative estimates. On a wetland scale, it is not realistic to believe that the pool can be quantified with 100 % certainty. There is too much variability in the field and input sources which cannot be completely accounted for. Accuracy is compromised due to precision limits inherent to the technique and due to field variability. Results are often expressed on a per area basis (e.g., m^{-2}). Extrapolation of the values to a larger spatial area or to represent an entire wetland further increases the error. Therefore, the practitioner should consider these methods to be estimates. They are most useful for comparing wetlands, not for deriving absolute values. Also, the wetland concept encompasses a wide variety of ecosystems. So these techniques will be most reliable when comparing wetlands within a given class (e.g., piedmont slope wetlands). This chapter is not meant to be all inclusive. We have chosen to emphasize the cycles C, N, P, sulfur (S), manganese (Mn) and iron (Fe). Since many of these processes are microbially mediated or there is an exchange between the water column and the soil, there is inherent overlap with other chapters.

Some of these techniques will not be appropriate for all types of wetlands, particularly with respect to hydroperiod class. Nitrification levels will be difficult to detect and quantify in a permanently-inundated freshwater marsh as nitrification is an aerobic process. However, nitrification certainly could be measured in a seasonally saturated mineral soil flat as long as the measurements are not taken during a wet phase. Conversely, methane (CH_4) emissions could be detected in a marsh but not in a mineral soil flat. In addition, because some of these processes are strictly aerobic and others are strictly anaerobic, care must be taken to determine the time of season to run a field assay as the target process may not be occurring at detectable levels. This is primarily an issue with seasonally saturated wetlands where the practitioner must take into account seasonal variability in hydrologic conditions.

We have tried to emphasize field techniques. However, some of the techniques have relatively standard laboratory components that are beyond the scope of this chapter. In some cases the reader will be advised to check additional documents for the laboratory techniques. In addition, there are a number of commercial labs or university soil testing labs that will perform some of these assays at cost. In some cases, both a field assay and a lab assay are available to measure the same process. The field assay may be presented in this discussion, but the reader will be referred to documentation that covers the lab assay if the latter is considered to be more accurate.

7.2 Quality Control

7.2.1 Sample Collection

Many of the biogeochemical assays require collection of a field sample which is subsequently analyzed in the laboratory. The quality of the lab data is inherently limited by the quality of the field sample. Whether monitoring wetlands for regulatory purposes or for research studies, it is important to have a sampling program that employs proper field monitoring techniques and accurate laboratory analytical procedures. There are many publications that outline the proper methods for environmental monitoring. The National Wetlands Research Center (http://www.nwrc.usgs.gov) of the United States (U.S.) Geological Survey is a great source of information on wetland assessments. Many states have developed wetland monitoring guidelines; therefore, it is important to check with each state's environmental department for the most current monitoring strategies.

Samples collected for wetland assessment must be representative of the environmental variability that occurs both spatially and temporally within an ecosystem. The Environmental Monitoring Systems Laboratory of the U.S. Environmental Protection Agency's Office of Research and Development has published a guideline on soil sampling of any site under investigation (Mason 1992). Today, different geostatistical evaluations are used to design the monitoring approach and to evaluate the collected data in such a way as to minimize the inherent variability found within soils. Random selection techniques should be used to determine the actual location where the soil samples will be taken. Areas that should be considered before sampling include (1) maximizing the accuracy and precision of collection; (2) selecting sample locations that represent the wetland under study; (3) determining when, how often and how deep to sample; and (4) considering how the size of the wetland will affect the accuracy of sampling. Many field guides describe how these issues are addressed (Barth et al. 1989; Barth and Mason 1984; Brown 1987).

Collection and preservation of samples is dependent on the type of sample required and on the analytical procedures that will be performed on the sample. Each method of analysis requires specific collection methods, sampling containers

and storage requirements. These requirements are designed so that no significant changes in the composition of the sample occur before the tests are performed. When sampling for organic compounds and trace metals, special precautions are needed to ensure detection. Often these parameters are present at such low concentrations, that they may be totally or partially lost if the proper procedures are not used. Standard Methods for the Examination of Water and Wastewater (Eaton et al. 2005) is a comprehensive reference book that covers all facets of water and wastewater analytical techniques. Standard Methods is a joint publication of the American Public Health Association, the American Water Works Association, and the Water Environment Federation. The Soil Science Society of America (SSSA) has published the book series, Methods of Soil Analysis, containing the following volumes: Part 1: *Physical and Mineralogical Methods*, Part 2: *Microbiological and Biochemical Properties*, Part 3: *Chemical Methods*, Part 4: *Physical Methods* and Part 5: *Mineralogical Methods*. This series is one of the primary references on methodology in soil science. Part 4 contains information on sampling procedures. Another good reference on soil sampling is a SSSA Special Publication entitled *Soil Testing: Sampling, Correlation, Calibration, and Interpretation* (Brown 1987).

Sampling plans determine the type of sample required for each particular project. The objective of the sampling plan is to ensure that the number and type of samples collected is representative of the "population" under study. The plan designates how many samples are needed, the locations of the samples and the sample depth at each location. The plan may include simple random samples, stratified random samples, systemic samples, sub-sampling and composite samples. The best plan must consider the overall cost and precision (lack of error) of sampling.

Once in the field, it is important to document the sampling operation. Field log books are used to record all information pertinent to field sampling such as: the purpose for sampling; location of the sampling point; date and time the sample was collected; name and address of field contact; procedure for field decontamination of sampling tools between samples to prevent cross contamination, field measurements, and any observations worth noting. A chain-of-custody record should also accompany each sample or group of samples. This record includes: the sample number; signature of the collector; date, time, and address of collection; sample type; signatures of persons involved in the chain of possession; and inclusive dates of possession. Once collected, most samples need to be kept on ice until delivered to the analytical laboratory.

7.2.2 Quality Control and Detection Limits

Analytical laboratories have quality assurance and quality control plans that ensure data accuracy and precision. Quality assurance (QA) is the system that uses procedures and assessments that ensure reliable data. The plan ensures that the best available sample preparation, handling, preservation and storage methods

are used as recommended by the appropriate authority. In addition to the above, QA also includes control of the following: calibration and standardization of instruments, preventive and remedial maintenance, proper instrument selection and use, quality laboratory water, clean laboratory environment, replicate analysis, spiking of samples, holding facilities for samples, responsible evaluation of data, and recording and maintaining a quality control (QC) database.

Quality control is the system of practices and procedures that provides the measure of precision, accuracy, detection limits and completeness of the testing facility. Precision measures the degree of agreement among replicate analyses of a sample. It quantifies the repeatability of a given measurement. The precision is calculated as relative percent difference of duplicates. The precision for three or more replicates is estimated by calculating the relative standard deviation (RSD) as

$$RSD = 100\frac{s}{\overline{X}}$$

where: s = standard deviation of replicate analysis, and X = mean of replicate analysis.

The best mechanism to evaluate precision is the examination of relative percent difference of duplicate samples in the analytical run. This is expressed in the formula:

$$\mathrm{RPD} = 100[(\mathrm{X1} - \mathrm{X2})/\{(\mathrm{X1} - \mathrm{X2})/2\}]$$

where: RPD = relative percent difference, X1 = first observation of unknown X, and X2 = second observation of unknown X.

In analytical chemistry, the detection limit (also called the lower limit of detection or limit of detection) is the minimum concentration of a substance that can be determined with a given level (usually 99 %) of confidence. That is, the true concentration of the substance in question is greater than zero. There are several types of detection limits. For example, some detection limits are set by the manufacturer of a specific piece of analytical equipment. The method detection limit (MDL) is unique in that it is designed for each individual laboratory. A sample containing a known amount of the compound being measured is analyzed by the laboratory seven or more times and the standard deviation of those measurements is determined. The MDL is calculated according to the formula: MDL = Student's t value × the standard deviation.

Sample unknowns are duplicated based on the assay, commonly at the rate of 1 per every 10–20 unknowns, depending on the standard operating procedure. Relative percent differences of 10 % are expected at levels of ten times the method detection limit (MDL) and above. Another mechanism to evaluate precision involves a comparison of a check sample run daily with each batch of samples. If the check sample is run several times during the analytical run, then an estimate of replicability of the run can be obtained. The standard deviation of these results is an estimate of daily precision. The repeatability of the procedure over time can be

evaluated by the comparison of the results of this check sample on a day-to-day basis. The pooled standard deviation of the check sample over many days and analyses gives an evaluation of the precision of the method over time.

Accuracy measures the bias in a measurement and can be defined as the degree of agreement of a measurement, X, with an accepted or true value, T. It is usually expressed as the difference between the two values, or as a percentage of the reference value 100 (X − T)/T. Accuracy of laboratory measurements are usually defined as percent recoveries of the analyte of interest from matrix spikes, or spike reference material introduced into selected samples of a particular matrix, or by the use of appropriate internationally certified materials. For many projects, percent recoveries of the spiked samples and the laboratory control standards are set at 80–120 %.

The method detection limit is the analyte concentration derived from the method that yields a signal which is large enough to be considered significantly different from the blank with a statistical 99 % probability. The method detection limit is determined by analyzing reagent water fortified at a concentration considered to be two to three times the estimated detection limit. At least seven replicates of this fortified blank are analyzed by the same procedure followed in the determination of unknown samples. The MDL is then calculated using the equation MSDL = (t) × (S), where t = 3.14 (for seven replicates) and S = the standard deviation of the replicate analysis.

Completeness refers to the percentage of valid data received from actual analyses performed in the laboratory. Completeness (C) is calculated as follows: C = 100 (V/T); where V = number of measurements judged valid, and T = total number of measurements.

7.3 Background

7.3.1 Characteristics of Wetlands That Promote Biogeochemical Processes

Wetlands are diverse ecosystems and variation is found in topographic position (e.g., slope vs. depression), substrate (organic soils or mineral soils), plant community composition, dominant water source, and hydroperiod. Each of these characteristics affects biogeochemical cycles. One characteristic common to all wetlands is the presence of a water table close to the soil surface for at least part of the growing season. The shallow water table leads first to anaerobic soil conditions and then to reduced soil conditions. A number of biogeochemical pathways proceed only under anaerobiosis or reducing conditions. These pathways play a greater role in biogeochemical cycles in wetlands than in uplands. Many freshwater wetlands display significant temporal variability in water table depth so that anaerobic or reduced soil conditions are present for only a portion of the growing season. In these

wetlands, the dominant pathways switch during the year from anaerobic processes to aerobic processes. This temporal variability in soil oxygen (O_2) content promotes some processes such as denitrification as explained below. Water source and landscape position influence inputs and outputs. Sediment loading is a dominant process in riverine wetlands subject to frequent overbank flooding and much of the P inputs will be in particulate form as opposed to ground water driven slope wetlands in which most of the P inputs will be soluble orthophosphates. Hydrodynamics and surface roughness dictate water resonance time. Sedimentation is a more dominant process in a depressional wetland subject to surface runoff than a groundwater driven slope wetland. Mineral soil flats are associated with seasonally saturated hydroperiods and alternating periods of aerobic and anaerobic conditions.

7.3.2 Role of Plants and Microbes

Plants are the dominant source of organic C which supplies the energy for microbially-mediated processes. Microbes are critical to the decomposition of detritus and leaf litter and mediate pathways in the C, N, S, Fe, and Mn cycles. Microbial populations vary with respect to total numbers and species composition according to soil depth and distance from plant roots. These differences are primarily in response to a gradient of available C. The rhizosphere refers to the zone of soil close to and impacted by plant roots. The rhizoplane is the surface of plant roots. Microbial numbers are substantially higher (10- to 100-fold) (Paul and Clark 1996) in the rhizosphere than in bulk soil and are inversely proportional to distance from the roots. The highest microbial numbers, by far, are on the rhizoplane. Plant roots supply most of the C that drives microbial activity in soils. Up to 90 % of fine roots may die and decompose annually in forest soils. In addition, dead root cap cells slough off and supply organic C, and exudates from live roots include readily available C sources (sugars, organic acids), a readily available source of N (amino acids), and growth promoting (and sometimes inhibiting) compounds (Vasilas and Fuhrmann 2011).

7.3.3 Importance of Wetting and Drying Cycles

Soil microbes are critical to the development of anaerobic and reducing conditions in wetland soils. Their activity in turn is impacted by soil moisture conditions. Following the onset of soil saturation, respiration by plant roots and microbes produces anaerobic conditions. Further respiration by microbes produces reducing conditions. For purposes of this discussion we consider reducing conditions to be present when ferric iron (Fe^{3+}) is reduced to ferrous iron (Fe^{2+}). Increased microbial numbers and activity subsequent to rewetting a dry soil are commonly observed and are thought to reflect a temporary increase (pulse) of readily available organic C

(Butterly et al. 2009). The C pulse is thought to result from both the presence of dead microbial cells that accumulated during soil desiccation and the release of previously unavailable organic C sources that resided in the interior of soil aggregates and similarly protected areas. The C released is typically readily available to soil microorganisms and results in increased microbial respiration. Provided O_2 diffusion is restricted as a result of rewetting and sufficient nitrate (NO_3^-) is present, these C pulses can produce sharp spikes in respiratory denitrification (Myrold 2005). Rates of denitrification drop rapidly once C or NO_3^- availability decreases or O_2 availability increases. In fact, N removal from soils due to denitrification is typically greatest when alternating aerobic and anaerobic soil conditions occur frequently. This is because the nitrifying bacteria responsible for converting NH_4^+ to NO_3^- are active only under aerobic conditions, whereas denitrification is dependent on NO_3^- availability (produced during aerobic conditions), presence of easily decomposable C compounds, and lack of O_2 (Vasilas and Fuhrmann 2011). Therefore, the practitioner must be cognizant of these when designing exercises to quantify soil N or C pools, or to quantify rates of processes that contribute to these pools. It is recommended that soil redox potential (Eh) be measured when conducting investigations on biogeochemical processes affected by Eh. The methodology for measuring Eh is presented in *Oxidation-Reduction Processes in Soils*. If nothing else soil moisture conditions during the field assay period should be noted.

7.4 Carbon

7.4.1 Overview

There are six principal C reservoirs in wetlands: plant biomass C, microbial biomass C, soil C (both organic and inorganic), particulate organic C in the water column, dissolved organic C, and gaseous C compounds such as carbon dioxide (CO_2) and methane (CH_4). Often, C in microbial biomass and C in soil organic matter are combined into the category soil organic C (SOC). Carbon is also a major constituent of sedimentary rocks such as coal and limestone. In minerals, it is found predominantly as carbonates, salts of the carbonate ion (CO_3^{2-}) such as calcite ($CaCO_3$). Significant quantities of free carbonates may accumulate in high pH soils in arid climates. In some soils, extensive quantities of C are stored as carbonates (CO_3^{2-}). Public awareness of the C cycle has recently increased due to concerns over global warming which is attributed to the atmospheric increase in greenhouse gases including CO_2 and CH_4. Wetlands can serve as both a source and a sink for C (Kayranli et al. 2010) depending on their age, type, and condition. Some wetlands produce CH_4 (see *Methane Emissions* below). However, most wetlands are characterized by a net retention of organic matter and plant detritus (Mitsch and Gosselink 2000). As such, a critical wetland service is C sequestration-the removal

of C (primarily CO_2) from the atmosphere and subsequent long-term storage in a reservoir such as soil organic matter. Disturbance to a wetland, especially in the forms of artificial drainage or deforestation, reverses the net C flow so that disturbed sites initially serve as a source of CO_2.

Because of the impact of O_2 availability on the direction or rate of many biogeochemical reactions, some of the C processes are compartmentalized in specific zones in the soil or water column. For example, CH_4 oxidation occurs in aerobic zones, while methanogenesis is restricted to anaerobic zones (Knight and Wallace 2008). Furthermore, since many of these processes are driven by microbial activity, compartmentalization is further promoted by the availability of organic C as an energy source. For example, the highest decomposition rates are found in close proximity to the wetland surface where there are high inputs of fresh litter and recently synthesized labile organic matter (Sherry et al. 1998) and the highest duration of aerobic conditions.

7.4.2 Primary Productivity

7.4.2.1 Overview

Carbon sequestration refers to the removal of C from the atmosphere and subsequent storage in C sinks such as oceans, forests, and soils. Primary production is the production of organic compounds from CO_2 (atmospheric or aquatic) principally through the process of photosynthesis. Therefore, photosynthesis is integral to C sequestration. The primary producers in wetlands are mainly plants and algae. Net photosynthesis (gross photosynthesis-respiration) can be approximated by assessing biomass. In this section we present methods for determining above-ground biomass for trees and herbs, abscised leaves, and fine roots. Conversion of biomass to C requires a C content value which is obtained from the literature or by chemical analysis of the sampled biomass. Direct chemical analysis will be more accurate as published values will represent averages across species and may not reflect the specific growing conditions of the individual plants in question. Chemical analysis for C content of plant tissue is not presented here. We also do not address biomass and C assessment of shrubs. For this topic we refer the reader to Chojnacky and Milton (2008).

7.4.2.2 Tree Biomass-Allometric Equations

Direct calculations of tree biomass to determine primary productivity or C sequestration is not an option as it requires destructive sampling, determination of dry weight, and chemical analysis for C. However, there are indirect methods that allow for the estimate of tree biomass and C. Above-ground tree biomass can be estimated using a single field measurement, published data, and simple allometric equations

frequently in the form of "$M = aD^b$", where M = dry weight of the biomass component, D = diameter at breast height (dbh) (see *Diameter at Breast Height* below), and "a" and "b" are parameters whose specific values are presented in a number of publications. So the practitioner needs to determine dbh and plug its value into the equation with the appropriate parameter values obtained from the literature. Two of the more extensive sources for allometric equations and the parameters are Ter-Mikaelian and Korzukhin (1997), and Jenkins et al. (2003). Ter-Mikaelian and Korzukhin (1997) presented biomass equations for 65 North American tree species based on a literature review. Furthermore, they present equations that address the following biomass components: foliage, branches, stem wood, stem bark, total stem (wood + bark), and total aboveground biomass. The geographic region that generated the data from which the parameter values were derived is presented. Therefore, the practitioner has several equations available for each tree species and should select the equation most closely associated with the site of interest. These equations were developed primarily for timber species, and as such, some wetland tree species may not be represented. It is also likely that the relation between dbh and biomass will be different between an upland situation and a wetland situation. So it should be understood that these indirect methods will give approximate values. However, they are useful for comparing sites.

Biomass can be converted to C either by determining C content on specific samples or by using published values of C content. Carbon analysis should be conducted on tree cores taken at breast height. If bark represents a significant proportion of aboveground biomass, bark should be partitioned from bole wood both in the C analysis and in allometric equations.

7.4.2.3 Diameter at Breast Height

Diameter at breast height is a standard method of expressing the diameter of the trunk or bole of a standing tree. Tree trunks are measured at the height of an adult's breast, which is defined differently in different countries. In the U.S., breast height diameter is measured at a height of 1.4 m. On slopes or in wetlands with pit and mound topography, the soil surface reference point to determine the 1.4 m above ground sampling height may not be obvious. In those situations, the reference point can be set as the highest point on the ground touching the trunk, or set as the average between the highest and lowest points of ground. A consistent approach to setting the reference point is critical.

Diameter at breast height is measured with a diameter (or girthing) tape or calipers. A diameter tape measures the circumference (girth) of the tree; it is calibrated in divisions of π centimeters (3.14 cm) and gives a directly converted reading of the diameter. To determine tree diameter, the tape is wrapped (diameter side facing user) around the tree. Tree diameter is indicated by the alignment of the number "0" aligns with the rest of the tape. Calipers consist of two parallel arms; one is fixed, the other slides along a scale. Calipers are held at right-angles to the trunk with the arms on either side of the trunk. Diameter is directly calculated as the

distance between the two arms. Diameter at breast height can also be measured with a Biltmore stick. Although not as accurate as a diameter tape, it is quicker to use. Diameter at breast height is measured by holding the stick at a set distance, usually 64 cm, from the eye, and at breast height. The left side of the stick is lined up with the left side of the tree. The number on the stick that lines up with the right side of the tree is the approximate dbh.

7.4.2.4 Tree Age Determination

In some cases it is of interest to determine the annual rate of woody biomass production. This requires an estimate of tree age. Many tree species increase trunk diameter by producing a single layer of wood each year between the previous year's growth and the bark. In a horizontal cross section cut through the trunk of a tree, these growth bands appear as concentric rings, referred to as growth rings, tree rings, or annual rings. Each ring represents 1 year of growth, so that the tree's age can be determined by counting the rings. The least invasive way to see the tree rings is with an increment borer which takes a small (5 mm diameter) straw-like radial core sample from the tree. An increment borer consists of three parts: handle, steel shaft (core tube or auger), and extractor. Increment borers come in different sizes; the length of the shaft should be at least 75 % the diameter of the tree you are boring. Tree rings should be counted near the base of the tree. For consistency, the boring is commonly taken at breast height (1.4 m). To extract a tree core, the screw tip of the shaft is pressed against the tree and the handle is turned clockwise until the screw bit reaches the center of the trunk. This action forces the core of wood into the tube. The core is extracted by first breaking the core with a counterclockwise one-half turn of the handle. The extractor is then fully slipped through the tube. The core will be removed with the extractor.

There are situations that reduce the accuracy of using ring counts to determine tree age (Avery and Burkhart 2002). One year's growth includes both spring wood (rapid-growing, lighter colored wood) and summer wood (slower-growing, dark colored wood). The method is most reliable when there is a sharp contrast between spring wood and summer wood and for fast-growing coniferous species in northern temperate zones. In tropical or southern temperate zones, tree growth generally does not produce distinctive rings. Some deciduous species produce limited contrast between spring wood and summer wood. Adverse growing conditions, such as drought, result in very narrow rings that are difficult to distinguish. Conversely, a period of favorable growing conditions following a drought can result in *false rings* which represent small growth spurts.

7.4.2.5 Coarse Root Biomass

Destructive sampling of coarse roots (>10 mm diameter) is labor intensive and requires heavy equipment. As such it is not conducive to most wetland investigations. We recommend an alternative approach to estimating coarse root

biomass using published allometric equations that depend on dbh. Sources of these equations include Whittaker et al. (1974) and Vadeboncoeur et al. (2007). Carbon content can be determined from coarse roots excavated on site or from core samples taken with increment corers from roots leaving the tree base.

7.4.2.6 Fine Root Biomass

Fine roots (<2 mm diameter) typically contribute <5 % of the total tree biomass (DeAngelis et al. 1981; Vogt et al. 1996). However, it has been estimated that fine root production constitutes about 30–50 % of the C being cycled annually through forest ecosystems (Grier et al. 1981; Vogt et al. 1996). Therefore, fine roots constitute a small but functionally critical fraction of ecosystem biomass. Hertel and Leuschner (2002) consider fine root production and root exudation as the least known processes of the C cycle of forests. However, both processes supply organic C to microbes critical to biogeochemical cycling. In addition, fine root production and turnover may be a sensitive indicator of changing soil environments, and therefore, ecosystem health (Bloomfield et al. 1996).

The sequential root coring method is commonly used to collect fine root biomass data. In this approach, roots are collected from soil cores taken sequentially throughout the year, typically at 1–2 month intervals. It employs a metal tube sharpened on one end which is manually driven into the ground to collect the soil cores. There is no set size for the corer. It should be at least as long as the depth of the soil to be sampled. The wider the corer, the easier it is to extract the soil from the corer while maintaining the integrity of the soil core. This is critical if the soil core is to be divided into sub-samples based on depth. Vogt and Persson (1991) used corers with a diameter of 33 mm and a length of 150 mm. Persson (1978) used a corer with an internal diameter at the hardened steel cutting edge of 6.7 cm while the upper part of the tube had an internal diameter 2 mm larger which facilitates removal of the soil cores by inverting the corer. Typically, the soil core is divided into sub-samples based on horizon. Horizon thickness should be noted prior to sampling so that a sampling volume can be determined. Horizon thickness is determined from an adjacent spade slice as core insertion can compact the soil. This is especially a problem with organic horizons. Thickness of the sub-samples is also determined to allow for the correction of volume lost due to compression when calculating root density. Soil sub-samples are transferred to plastic bags, sealed, and transported to the laboratory for processing. Prior to processing, samples are stored at 4 °C. Live roots can be distinguished from dead roots (necromass) under a dissecting microscope after staining. The necessary techniques are presented by Hertel and Leuschner (2002) and Knievel (1973).

Combining the sequential root coring method with the ‘minimum–maximum method’ (SC – MM) allows for an estimate of biomass production without distinguishing between biomass and necromass (Edwards and Harris 1977). With the minimum–maximum method, biomass production is calculated as the difference between the minimum value and maximum value of fine root biomass and

necromass obtained in the measuring period. Hertel and Leuschner (2002) evaluated four methods that assessed fine root production in a *Fagus* spp. and *Quercus* spp. forest and compared the results with C budget data. Twenty samples each were collected at 4 week intervals over 1 year. The sequential coring or minimum–maximum approach showed the best agreement with the C budget data with an overestimation of 25 %.

7.4.2.7 Quantifying Litterfall

Leaf cages are used to catch abscised leaves from deciduous woody plants so that an estimate of foliage biomass or C can be obtained. The cages can also be utilized to collect abscised flowers and fruit. Techniques for quantifying branchfall are presented by Bernier et al. (2008). It should be understood that senescing leaves typically export simple carbohydrates prior to abscission, so that biomass or C estimates based on abscised leaves will not necessarily equate to foliage biomass or C. Foliage biomass can also be estimated for trees by using allometric equations (see *Tree Biomass* above). Litterfall can also be estimated using the cohort layered screen method (see below). However, the leaf cage approach will provide an estimate of C returned to the forest floor via leaf abscission annually. Commercially available laundry baskets can be used as leaf cages. Holes are drilled into the bottom of the basket and the basket is elevated above the soil surface to promote drainage. In addition, depending on the target species, fine mesh screening may need to be used to cover openings on the sides of the basket. An alternative design consists of an open wooden frame holding window screening. Screening is stapled to the frame to form a box shape open at the top and with four wooden legs. The legs are hammered into the ground to keep the cage in place. The screen bottom is positioned above the soil to allow for air flow and to prevent the leaves from picking up moisture from the soil. The height of the sides should be great enough to prevent leaves from moving out of the frame via wind. Rainfall can leach out soluble carbohydrates from the leaves so that samples should be collected at least weekly or before significant rainfall events. Variability in leaf fall increases with distance from the cage to the target canopy. For shrubs, two cages each halfway between the main stem and the drip line are usually sufficient. For trees, more cages are needed; the appropriate number depends on canopy diameter, and the contents of each cage should be analyzed separately to determine a measure of variability. Sampling designs are addressed by Bernier et al. (2008). The samples are placed in pre-weighed paper or mesh bags and dried to a constant weight at 80 °C. Pre-weighing the bags allows sample dry weight to be obtained without removing the sample from the bag.

Implicit to this technique is the assumption that leaves collected from the cage represent the portion of the canopy equivalent to the surface area of the cage bottom. Therefore, total canopy biomass can be estimated by: (biomass of collected leaves) × (horizontal surface area of canopy) ÷ (surface area of cage bottom). A major limitation to this method is the error associated with extrapolating to a

larger area. Wind currents can transport leaves for a considerable distance from the parent tree. The number of cages needed to give a relatively accurate estimate of leaf biomass will depend on the spatial area of the target site and the number of woody species present in the tree and shrub strata. To increase the accuracy, the value for each cage can be weighted by the percentage of the site area shaded by that particular tree or shrub associated with each cage.

7.4.2.8 Herbaceous Biomass

Herbaceous shoot biomass samples can be obtained by clipping at ground level all of the plants within a delineated area. The sample area can be delineated with a square or rectangular frame built from 2.5 cm diameter polyvinyl chloride (PVC) pipe; each piece is connected to the next by a right angle PVC connector. The frame is light in weight and can be easily constructed and broken down in the field. If the site is inundated, the frame should be raised to the surface of the water. This can be facilitated by using three-way connectors and attaching PVC 'legs' to the frame. Another option is to mark the plot with pin flags, but that method does not provide a continuous delineation edge. The samples should be placed in pre-weighed paper or mesh bags and dried to a constant weight at 80 °C. The bags should be pre-weighed so that sample dry weight can be obtained without removing the sample from the bag. Fresh harvested plant tissue will continue to respire and lose weight so samples should be kept on ice until they can be placed in a dryer.

7.4.3 Soil Organic Matter

7.4.3.1 Loss on Ignition

Loss on ignition (LOI) is a relatively simple method of determining soil organic matter (SOM) content as follows (Nelson and Sommers 1996). Pyrex beakers (20 ml) are heated in a muffle furnace at 400 °C for 2 h and then weighed to determine tare weight of the beaker. Air-dried soil samples are ground to pass a 0.4 mm screen. One to 3 mg of dried and ground sample are placed into a tared beaker and heated at 105 °C for 24 h in a drying oven. The beaker is cooled in a dessicator over $CaCl_2$, then weighed to determine the dry mass of the sample. The samples are ignited in a muffle furnace at 400 °C for 16 h, cooled in a dessicator over $CaCl_2$, then weighed to determine weight of the beaker and the ignited sample. All weights should be taken to within 0.1 mg. Organic matter content (%) is calculated as 100 $(Wt._{105} - Wt._{400})/Wt._{105}$, where $Wt._{105}$ is the sample weight after heating at 105 °C, and $Wt._{400}$ is the sample weight after ignition.

The high temperatures used in this method can cause the loss of structural water from inorganic soil constituents such as hydrated aluminosilicates resulting in weight losses in excess of organic matter content (Nelson and Sommers 1996).

This source of error is most pronounced in subsoils high in clay but low in organic matter (Howard and Howard 1990). This error can be corrected using a series of samples of known C content as described by Nelson and Sommers (1996). For purposes of this discussion, we recommend restricting the LOI method to organic horizons and topsoil where the error is not as large.

The determination of C content of SOM requires complex laboratory assays and or expensive equipment. Several methods are presented by Nelson and Sommers (1996). In lieu of running these assays, the practitioner has two options. Carbon content of SOM has been reported to range from 52 to 58 % (Sparks 1995), but is generally considered to be 58 % (Wolf and Wagner 2005). Therefore, a rough estimate of SOC can be obtained by multiplying SOM values by 0.58. The conversion factor can be fine-tuned by submitting selected samples to a commercial lab for organic C determinations. The selected samples should represent the range in SOM found in the entire suite of samples. It should be clear that the lab analysis is for organic C and not total C as soils formed from calcareous parent materials under arid conditions may contain large quantities of inorganic C (e.g., carbonates).

7.4.4 Organic Matter Decomposition

7.4.4.1 Overview

Biological oxidation of plant tissue and SOM is the principal process that returns terrestrial fixed C to the atmosphere. In the absence of anthropogenic influence, decomposition of plant material before burial is the major pathway for the return of nutrients to the water column. Therefore, decomposition not only returns C to the atmosphere but also supplies nutrients to macrophytes. In this section, we present two approaches (three methods) for assessing organic matter decomposition. One approach is to directly measure biomass losses in leaf litter over time (litter bags and the cohort layered screen method). A second approach (cotton-strip assay) is to measure the loss in tensile strength of cotton fibers. Although an indirect assay, this method has the benefit of using a standardized substrate that allows for the detection of soil effects independent of differences in substrate characteristics. Organic matter decomposition rates are much lower under anaerobic conditions than under aerobic conditions. Therefore, when utilizing any of the following methods, we recommend that soil Eh be documented as temporal differences or spatial differences may solely be a soil moisture effect.

7.4.4.2 Litter Bags

Litter bags are commonly used to determine decomposition rates of plant tissue in terrestrial ecosystems (Aber and Melillo 1980; Wieder and Lang 1982). Plant material of a known mass is placed in mesh bags which are placed in the field

and randomly retrieved at predetermined intervals. Bag size is commonly 20 cm × 20 cm. Nylon mesh or fiberglass screening material is used so the bags themselves are not subject to decomposition. Harmon and Lajtha (1999) recommend fiberglass bags for light intensive sites where UV light can degrade nylon. Litter bags typically have a mesh size of 1–2 mm (Robertson and Paul 1999), although mesh sizes from <1 mm to >10 mm have been used. If the mesh is too small, access to some macroinvertebrates may be denied. Large mesh sizes facilitate the loss of small particulate matter. One option is to use a small mesh, but staple the edges of the bags at relatively large intervals (e.g., 5 cm) to provide openings along the periphery for access of macroinvertebrates. Regardless of mesh size, low molecular weight organic compounds can be lost through leaching. The practitioner must be aware of implications to leaching losses and the bag deployment period should reflect that. Decomposition rates will be highest when the soil is moist but not saturated. Leaching losses will be greatest during periods of soil saturation. Therefore, the bags should not be deployed in continuously saturated soils.

In general, at least five replicate litter bags are collected at each sampling interval during year 1 of the study (Karberg et al. 2008). Sites displaying significant microclimate variability may require more replicates. For example, wetlands with pit and mound topography exhibit spatial variability in soil moisture and soil temperature and would require greater replication. Certainly, the researcher has the option of adjusting replicate number in subsequent years. Sample material is chopped into 2–5 cm lengths and a known amount of fresh plant tissue is placed in the bags. Subsamples of the plant material are dried (70 °C, 48–72 h) to obtain their water content. The organic matter of interest (e.g., leaves or fine roots) is placed in nylon mesh bags. The bags are then placed where the organic material would normally be found. For example, abscised leaves would be placed on the soil surface, fine roots would be placed in organic soil horizons or in the topsoil, and detrital tissue would be placed on the soil surface or in the detrital layers. One advantage of this system is that material can be collected from the site in question and returned to its natural environmental conditions. The filled bags should be returned to the site soon after sample collection to ensure representative environmental conditions. Bags are pre-weighed (tared) so that sample weight can be determined in the bag. Loss of biomass due to decomposition is calculated as the difference between initial biomass and remaining biomass. All values are expressed on a dry weight basis. Average rate of decomposition (per day) is determined by dividing biomass loss by the incubation period. However, since biomass decreases over time, a more accurate estimate of decomposition rate is produced with exponential decay equations (see Karberg et al. 2008). If a chemical analysis is conducted on the tissue before and after the incubation period, N and P mineralization rates can also be determined.

The litter bag method also can be used to estimate fine root decomposition rates (Fahey et al. 1988). Fine roots can be collected by the sequential coring method (above) or with a spade. Soil residues are removed by rinsing, and root samples (2 g fresh weight per bag) are placed in litter bags (nylon, 10 × 10 mm, mesh size 1.2 mm). Subsamples of the root material are dried (70 °C, 48 h) to obtain the water

content of the fresh roots. Bags are typically left in the field for 2–3 months, although the incubation period will depend on soil and air temperature, soil moisture, and biomass composition. After collection, bags are transported to the laboratory and the samples are gently rinsed to remove soil residues and fungal hyphae, and then dried and weighed.

There are two general approaches to analyzing the data. If the intent is to compare treatments such as litter species composition, or to compare sites, an analysis of variance is performed on the percentage of initial dry mass remaining at time *t*. If the intent is to determine decomposition rate constants, mathematical models are fitted to the data to describe biomass loss over time. Both single exponential decay models and double exponential decay models have been frequently used to describe organic matter decomposition. The single exponential decay model is based on the assumption that the relative decomposition rate remains constant over time. The double decomposition decay model is based on the assumption that litter has two distinct components, an easily decomposed (labile) fraction and a more recalcitrant fraction. Therefore, each fraction requires a separate decay rate constant. Wieder and Lang (1982) present a critique of these analytical methods.

7.4.4.3 Cohort Layered Screen

The cohort layered screen method is an inexpensive approach to assessing long-term ($\geq$3 years) litter decomposition. As presented by Karberg et al. (2008) aluminum or fiberglass window screening (1 m $\times$ 1 m, 2–3 mm mesh) is placed over the forest floor following the major annual litterfall. An additional layer of screen is placed over the screen from the previous year following each annual litterfall. The litter is held in place by the screens and decomposes *in situ*. Sections of the screen are cut out to supply samples of the decomposing litter which is then dried and weighed. Dry weights are then compared to stand level estimates (see *Quantifying Litterfall*) for the year in question. One benefit to this approach is that the litter sample is naturally representative of the site, as opposed to the litter bag method where the practitioner chooses the litter sample. The cohort layered screen method does have several limitations in common with the litter bag method. Certain macrofauna may be denied access to the litter which can alter decomposition rates. Also both techniques do not allow for the separation of true decomposition from losses attributed to leaching and comminution. Leaching losses are especially a concern in wetlands. Since the cohort layered screen method is intended for long-term deployment, it is not appropriate for wetlands that exhibit long-term periods of inundation.

7.4.4.4 Cotton-Strip Assay

The decomposition of cellulose strips has been used extensively as a surrogate for plant organic matter decomposition including in a variety of wetlands (Newman

et al. 2001). The assay quantifies cellulose decomposition on the basis of the reduction in tensile strength of cellulose fibers, referred to as cotton tensile strength loss (CTSL), of a standardized cotton fabric. Since the assay uses a standardized cotton fabric (97 % holocellulose; Shirley Institute Test Fabric, Didsbury, England) (Latter and Harrison 1988), it provides a method for normalizing substrate quality between sites (Harrison et al. 1988).

Cotton strips can be inserted vertically into the soil with a flat spade or sharpshooter shovel that is at least as wide as the strip. One end of the strip is trapped between the blade edge and the soil surface. The spade is then inserted into the soil, pulling the strip with it. Two sets of strips are used. One set is inserted and removed immediately. These serve as control or reference strips. The remaining strips are left in the soil for one to several weeks, depending on the expected rate of decomposition. Upon removal from the soil, strips are immediately washed in freshwater to remove debris and soil and then washed again in deionized water. The sample strips are dried at room temperature and then stored in plastic bags. The strips are cut into 3 cm wide horizontal segments and reduced to 2 cm segments by fraying. Segments are used to accommodate soil variability with depth that may impact decomposition rates. Tensile strength is measured from each segment with a tensometer (e.g., Monsanto Type-W) equipped with 7.5 cm wide jaws adjusted to 3 cm spacing. Temperature and humidity affect the results so all measurements should be are carried out at 18–22 °C and 100 % relative humidity (facilitated by soaking the strips in deionized water). Individual losses in tensile strength are calculated relative to the reference strips for each site.

Walton and Allsopp (1977) presented the benefits of this assay: (i) cellulose is a major component of plant remains; (ii) the decomposition of dead plant remains is a major biological process; (iii) cellulose provides a major food source for a wide variety of soil organisms; (iv) cotton is a natural substrate; and (v) degradation of any organic material begins with bond breaking, leading to changes in tensile strength. However, different litter constituents do not decompose at the same different rate (Minderman 1968) and Howard (1988) concluded that the rate of breakdown of pure cellulose added to soil cannot reflect litter decomposition rate. Walton and Allsopp (1977) concluded that this technique is best employed for comparative assessments of biological activity in different soils.

7.4.5 Soil Respiration

7.4.5.1 Overview

Soil respiration, or more accurately soil surface CO_2 efflux, is the release of CO_2 from the soil surface to the atmosphere. It results primarily from respiration by plant roots and soil microorganisms and may comprise 50–80 % of ecosystem respiration (Davidson et al. 2002; Giardina and Ryan 2002). Conceptually, respiration reflects substrate decomposition in soils and in many texts soil respiration is included in

SOM decomposition sections. Also, soil surface CO_2 efflux has been used to compare decomposition rates in different soils. But, soil surface CO_2 efflux is an index of respiration by soil organisms and plant roots (Zibilske 1994). As such, it is not directly comparable to SOM degradation.

There are three main approaches to assessing soil respiration-closed chamber systems, open chamber systems, and flux gradient sensors. Chamber-based approaches are simple, economical, and portable. A chamber is placed over the soil to create a headspace of air, which can be sampled repeatedly over a short time period. Closed chamber systems are the most commonly used and commercially available. They are classified as "closed" because there is no exchange of air between the chamber and the outside atmosphere during measurements. Closed chamber systems may be "active" or "static". In dynamic systems, air is continuously circulated between the chamber and an infrared gas analyzer. In static systems, air samples from the chamber are collected with a syringe for laboratory analysis or CO_2 is absorbed by soda lime in the chamber. In open chamber systems, air is exchanged between the chamber and the outside atmosphere. In the flux gradient approach, infrared sensors are inserted into the soil at various depths. The CO_2 concentration gradient over soil depth and additional soil characteristics are used to calculate CO_2 diffusivity. Bradford and Ryan (2008) present an evaluation of the relative benefits and challenges to each system. In this section, we present the soda lime method, a common and relatively simple method of measuring soil respiration that utilizes a static closed chamber approach. In the section *Methane Emissions*, we include methods that can also be used for quantifying CO_2 fluxes.

7.4.5.2 Carbon Dioxide Detection by Soda Lime Absorption

In this method, an open dish containing soda lime is placed on or just above the soil surface and covered with a container to restrict airflow between the soda lime and the atmosphere. Carbon dioxide and water vapor released by the soil microorganisms during decomposition are absorbed by the soda lime. After drying to remove water, the gain in soda lime dry weight during the exposure period reflects the amount of CO_2 evolved. The following specifics were detailed by Zibilske (1994).

The soda lime jar must consist of oven-safe glass with air-tight screw caps. A 5.5 cm diameter jar is suggested. Jar supports (for stability) consist of 12 cm square pieces of galvanized mesh, bent down at each corner to form four 2-cm legs. For jar covers, they used cylindrical cans (28 cm diam., 25 cm height), open at one end. The exact dimensions are not critical, but each container should have an opening at least 600 cm^2 and a total volume of at least 15,000 cm^3. To prevent the jar covers from direct exposure to sunlight in the field, they can be painted white, covered with aluminum foil, or by placing a 50 cm^2 flat board on top of them in the field. The amount of soda lime (6–12) mesh needed for each jar should be slightly greater than 0.06 g for each cm^2 of soil surface area covered by the jar cover container.

Each jar with the cap on is weighed. Soda lime is placed in the jars and dried to a constant weight in a drying oven at 100 °C for 24 h. The jars are re-capped, cooled, and reweighed to determine the amount of dried soda lime in each jar. A minimum of five replicate chambers are used at each location to accommodate spatial variability. Any vegetation or debris that would interfere with the formation of a tight seal between the cover and the soil surface is removed, but without disturbing any leaves and the soil surface under the chamber. The wire mesh legs are pressed into the soil to produce a stable surface. After opening, each soda lime jar is immediately placed on a mesh stand and covered. The lip of the cover is forced into the soil with a twisting motion. A weighted object such as a rock may be placed on the cover to keep it in place and maintain the soil surface seal. Controls are needed to account for any CO_2 absorbed during this part of the procedure. To construct controls, a jar of soda lime is left open for the same amount of time required to deploy the sample jars (from opening to covering) and then covered. For every ten sample jars, two control jars are used. The control jars will be used to produce blanks. Incubation is commonly for 24 hours (h). Retrieved sample jars are tightly and quickly capped; then dried without caps in a 100 °C oven for 24 h, capped, cooled, and weighed. The absorption of CO_2 generates water which is removed during drying of the soda lime. To account for this, the weight gain determined after drying is multiplied by 1.4. All necessary calculations are presented by Zibilske (1994). The data are commonly expressed as mass per unit area per unit time (e.g., g $CO_2/m^2/h$).

Absorption of water vapor is needed to activate the soda lime after drying. However, it should not come in direct contact with surface water. For this reason, this technique is not appropriate for sites with deep inundation. If need be, the jar supports can be constructed with larger legs to keep the jars above surface water. In addition, to compare soils they should have similar water content when sampled as soil moisture content impacts respiration. Therefore, deployment should coincide with a period of stable soil moisture conditions. We recommend that soil moisture content (dry weight basis) be determined for topsoil adjacent to each sampling point. Take four soil samples equally spaced from each other and 0.5 m from each sampling point and combine for soil moisture determinations. Temperature also affects respiration so air temperature should be taken at a height of 0.5 m and soil temperature should be taken at a depth of 5 cm.

7.4.6 Methane Emissions

7.4.6.1 Overview

Methane is of environmental concern as it has been implicated in global warming. Methanogenesis is the utilization of CO_2 as a terminal electron acceptor to produce CH_4, sometimes referred to as “swamp gas”. Methane is produced by a distinct group of obligate anaerobic bacteria (methanogens) only under very reduced (redox

potential less than −100 mV) conditions. Methane is oxidized to CO_2 by a group of aerobic bacteria (methanotrophs). Although the two competing microbial processes require extremely different redox potentials, both can occur simultaneously in the same soil. For example, CH_4 produced in an anaerobic subsurface horizon can diffuse into an aerobic surface horizon where it is converted to CO_2. Therefore, field attempts to quantify CH_4 emissions in actuality measures net CH_4 fluxes.

Many of the techniques for assessing gaseous emissions from wetlands work equally well for CO_2 as for CH_4. Most of the techniques depend on chamber-based approaches introduced in *Soil Respiration* above. One major difference is in the options for measuring the concentrations of the respective gases in the sample. Portable infrared gas analyzers are commercially available for the instantaneous measurement of CO_2 in the field. Equivalent technology is not readily available for CH_4, so gas samples must be stored and transported to the lab for analysis of the gas through gas chromatography.

7.4.6.2 Static Closed Chambers

The method presented here employs a static-chamber approach to measure CH_4 evolution from soils. It is discussed in detail by Weishampel and Kolka (2008). This approach may be used for other gases (e.g., CO_2) evolving from soil. However, there are subtle differences in the design, construction and deployment of chambers depending on the target gas. The reader is referred to Livingston and Hutchinson (1995) for a discussion of these factors. It is conducive to spatially intensive sampling exercises in wetlands or uplands. This method employs a static enclosure system comprised of collars that are permanently installed in the ground and portable chambers designed for syringe sampling that fit over the collars. The permanent nature of the installed collars maintains a tight seal with the soil surface and minimizes disturbance effects associated with collar installation during periods of gas flux measurement. Both collars and chambers are made of 25 cm diameter, schedule 40 PVC pipe. Methane flux is calculated from the change in concentration during the incubation period (period of chamber deployment in the field). The required calculations are presented by Weishampel and Kolka (2008).

It is critical that sediments are not disturbed during instrumentation or sampling as it can impact gas fluxes so wooden pallets may be needed to accommodate foot traffic. This is especially a concern for saturated or organic soils. In some instances, the collars are hammered partially into the ground. However, hammering can cause significant compaction to organic soils or very wet soils and should be avoided in those instances. Collars should also be installed at least 1 week prior to sampling to minimize any impact from soil disturbance.

Altor and Mitsch (2008) described a static chamber design for measuring CH_4 and CO_2 emissions from freshwater marshes. Chambers were constructed of PVC chamber frames and circular, high density polyethylene (HDPE) bases (0.27 m^2), and transparent 4-mil (0.1 mm thick) polyethylene bags. The frame consisted of a circular top and three legs. Frame heights were 50 cm for sampling points

without macrophytic vegetation and 150 cm for sampling points with macrophytic vegetation. Polyethylene bags were in place only during the sampling process. The bags were pulled down over the chamber frames and attached to the base with elastic straps. The bags should be constructed so that they fit snugly over the frame so that the volume of air sampled is consistent. The top of each bag was equipped with a butyl rubber sampling septa and a 3 m Tygon vent tube (1.6 mm inside diameter [i.d.]).

7.4.6.3 Scaling CH_4 Fluxes

Measurements of CH_4 fluxes from wetland soils typically reveal high spatial and temporal variability. The number of chamber measurements needed and the labor required to carry out this degree of sampling may preclude this approach to characterizing seasonal CH_4 fluxes on a landscape scale. Another approach is to rely on modeling or remote sensing methodologies that link CH_4 fluxes to more easily measured ecosystem properties or processes. For example, soil temperature, water table depth and range, community structure, and net primary productivity have been used in CH_4 flux models for wetlands (Potter 1997; Potter et al. 2006).

7.4.7 Dissolved Organic Matter

7.4.7.1 Overview

Dissolved organic matter (DOM) is operationally defined as the fraction of organic matter that passes through a 0.45 μm filter and is a heterogeneous mixture of compounds including carbohydrates, proteins, lignins, organic acids and other humic substances (Herbert and Bertsch 1995; Kalbitz et al. 2000). The fractions of DOM that contain functional groups with C and N molecules are generally classified as dissolved organic C (DOC) and N (DON). Wetlands have typically been identified as the largest sources of DOM in watersheds (Aitkenhead-Peterson et al. 2003; Mulholland 2003). Concentrations of DOC for wetlands have been observed to range from 3 to 400 mg/L with an average of 30 mg/L (Thurman 1985). Among wetland types, bogs (3–400 mg/L) have been found to yield the highest DOC values, while marshes represent the lower range (3–15 mg/L) (Thurman 1985). The elevated contents of DOM in wetlands can be attributed to a variety of factors including: (a) high primary productivity of wetlands compared to upland and aquatic ecosystems (Thurman 1985); low decomposition rates of organic matter in wetlands due to acidic conditions and anaerobic or low O_2 contents of wetland soils and surface waters (Kalbitz et al. 2000); reducing redox conditions that result in reductive dissolution of Fe and aluminum (Al) oxides that could otherwise have served as sorption sites for DOM (Kalbitz et al. 2000); flooding and hydrologic conditions that facilitate the slow continuous leaching of DOM; and

surficial hydrologic flowpaths that bypass mineral rich sorption surfaces present in subsurface soil horizons (Inamdar et al. 2011, 2012).

Past studies have generally focused on determining the bulk concentrations of DOC and DON in wetland soils and watershed runoff (Hinton et al. 1997; Inamdar and Mitchell 2006; Raymond and Saiers 2010). While these observations have been important in highlighting the significant role of wetlands for DOM, bulk DOM concentrations provide little information on the reactivity, bioavailability, molecular size, and mobility of DOM. To get an idea of these ecologically relevant characteristics of DOM, we need to know the functional groups or the individual constituents such as carbohydrates, proteins, carboxylic acids, lignins that make up DOM. For example, labile fractions of DOM that are easily consumed by microbes are found to be rich in carbohydrates and proteins (Benner 2003). Aromatic and humic-rich fractions of DOM play a preferential role in the complexation and transport of metals such as cadmium, arsenic, and mercury. Similarly, aromatic compounds of DOM are predisposed to forming carcinogenic disinfection by-products when water is chlorinated for drinking purposes (Nokes et al. 1999). Hydrophobic DOM compounds are preferentially sorbed on Fe and Al oxides in soils while hydrophilic DOM molecules remain in solution and move with runoff waters (Jardine et al. 1989; Kaiser and Zech 1998; Ussiri and Johnson 2004). Thus, to truly understand the fate and transport of DOM in watersheds and its implications for terrestrial and aquatic ecosystems we need to move beyond bulk determinations to characterizing the chemical constituents of DOM.

7.4.7.2 Characterizing Dissolved Organic Matter Using UV and Fluorescence Spectroscopy

In the past, DOM composition or characterization of functional groups have usually been performed using traditional chemical techniques that are labor-intensive, time-consuming, involve high analytical costs, and require large sample volumes. These challenges have precluded the routine characterization of DOM composition for many ecosystem and watershed studies. The recent advances in ultra-violet (UV) (Weishaar et al. 2003) and fluorescence technology (Coble et al. 1990; McKnight et al. 2003), however, overcomes some of these challenges and promises to be a useful tool for characterizing DOM chemistry, especially for studies that generate a large number of DOM samples.

Ultra-violet and fluorescence techniques rely on the property of DOM that different organic molecules absorb and reflect light at differing wavelengths. Thus, investigation of the absorption and fluorescence spectra can provide critical insights into the composition of DOM. It needs to be emphasized here that these spectrofluorometric procedures do not provide information on the concentration or chemical structure of the DOM functional groups but the proportion of fluorescence contributed by specific DOM moieties can be determined through post-processing of the spectra. Furthermore, only a small of fraction of the DOM pool responds to UV and fluorescence measurements (McKnight et al. 2003). Despite these

constraints, UV and fluorescence approaches to characterizing DOM have exponentially increased in recent years (Cory and McKnight 2005; Fellman et al. 2009; Helms et al. 2008; Inamdar et al. 2011, 2012; Jaffé et al. 2008; Miller and McKnight 2010; Wilson and Xenopoulos 2009) including some excellent reviews on the subject (Cory et al. 2011; Fellman et al. 2010).

The UV absorption spectra for DOM is generally obtained using a standard spectrophotometer equipped with a 1 cm path-length quartz cuvette (volume of 4 ml) over the 190–1,100 nm wavelength range at 1-nm intervals. Prior to the sample spectra, the instrument is set up and corrected for scattering and baseline fluctuations by using deionized (DI) water. The absorption spectrum for DOM follows an exponential pattern with a decrease in absorption with increasing wavelength. Some of the key UV metrics that are derived from this spectrum and which have been used to characterize DOM are reported in Table 7.1. The UV metric that has been most commonly reported is the specific UV absorbance (SUVA) which is computed by dividing the decadic UV absorbance at 254 nm by the concentration of DOC (mg C/L) (Weishaar et al. 2003). SUVA has been found to be strongly and positively correlated with aromatic content of DOM as determined by ^{13}C-NMR (Weishaar et al. 2003). SUVA values can however be influenced by the pH, nitrate and dissolved iron (Fe) content of the sample and appropriate screening and corrections need to be applied (Weishaar et al. 2003). Since Fe absorbs light at 254 nm, elevated concentrations of Fe (>0.5 mg/L) in the DOM sample can lead to incorrect (high) SUVA values (Weishaar et al. 2003). A metric similar to SUVA, the absorption coefficient at 254 nm (a_{254} in m) is also calculated by using the naperian UV absorption coefficient (Green and Blough 1994). The a_{254} also provides a measure of aromaticity but without normalization to DOC (Helms et al. 2008). Another UV index, the spectral slope ratio, S_R, is calculated as the ratio of the slope of the shorter UV wavelength region (275–295 nm) to that of the longer UV wavelength region (350–400 nm) (Helms et al. 2008) and is obtained using linear regression on the log-transformed spectral ranges (Yamashita et al. 2010). The spectral slope ratio, S_R has been found to be inversely related to the molecular weight of DOM (Helms et al. 2008).

In fluorescence spectroscopy, DOM samples are exposed to light in a fluorometer for a range of excitation wavelengths and the corresponding emitted wavelength and light intensity is recorded (Lakowicz 1999). The matrix of the fluorescence intensities that is generated is referred to as the excitation-emission matrix (EEMs), an example of which is illustrated in Fig. 7.1. Prior to generating fluorescence scans and deriving meaningful indices from the EEMs a number of important steps need to be performed such as correcting for instrument bias, diluting samples with high absorbance values (e.g., A254 $\geq$ 0.2) and applying corrections to account for inner-filter effects (McKnight et al. 2003). Once the EEMs are generated a variety of fluorescence indices can be generated by using the ratios of fluorescence intensities from specific regions (wavelengths) of the EEM matrix. In addition, EEMs can be further analyzed using rigorous multivariate statistical tools such as parallel factor analysis (PARAFAC, Stedmon et al. 2003) that decomposes the EEMs matrix into chemically and mathematically distinct components with the

Table 7.1 Selected UV and fluorescence metrics that have been commonly used to characterize the composition of dissolved organic matter (DOM)

UV and fluorescence indices	Reference	Definition and significance
Specific UV absorbance ($SUVA_{254}$) [L mg C^{-1} m^{-1}]	Weishaar et al. (2003)	UV absorbance at 254 nm divided by DOC concentration in mg C/L; provides a measure of aromaticity of DOM. High values of SUVA indicate more aromatic material
Absorption coefficient a_{254} [m^{-1}]	Green and Blough (1994)	(UV absorbance at 254 nm) × 2.303 × 100 Measure of aromaticity of DOM
Slope ratio S_R	Helms et al. (2008)	Ratio of the slope of the shorter UV wavelength region (275–295 nm) to that of the longer UV wavelength region (350–400 nm); Can be used as a proxy for molecular weight (MW) S_R decreases with increasing MW
Fluorescence Index (FI)	McKnight et al. (2003)	Ratio of fluorescence intensities at 470 and 520 nm at excitation of 370 nm; Used to distinguish between terrestrial and microbial sources of DOM; Terrestrial or allochthonous DOM: 1.2–1.5; Microbial or autochthonous DOM: 1.7–2.0
Humification Index (HIX)	Ohno (2002)	$HIX = aI435 - 480/(aI300 - 345 + aI435 - 480)$ Used to characterize humification status of DOM; Ranges from 0 to 1 and increases with increasing degree of humification
Freshness Index (β:α)	Wilson and Xenopoulos (2009)	Ratio of emission fluorescence intensity at 380 nm by the maximum emission fluorescence intensity observed between 420 and 435 nm, calculated at excitation wavelength of 310 nm. A measure of recently produced DOM, where β represents DOM of recent origin and α represents more decomposed DOM
Redox Index (RI)	Miller et al. (2006)	$Q_{red}/(Q_{red} + Q_{ox})$, where Q_{red} is the sum of the reduced components and Q_{ox} is the sum of the oxidized components; provide a measure of the redox state of DOM
Humic-like fluorescence	Coble et al. (1998)	Excitation <260; Emission 448–480 Indicates DOM from vascular plant sources, high molecular weight and humic in nature
Protein-like fluorescence	Coble et al. (1998)	Tyrosine-like: Excitation <270–275; Emission 304–312; Tryptophan-like: Excitation <270–280; Emission 330–368; Indicates protein-like DOM moieties, bioavailable DOM (Fellman et al. (2009)), and DOM of microbial origin

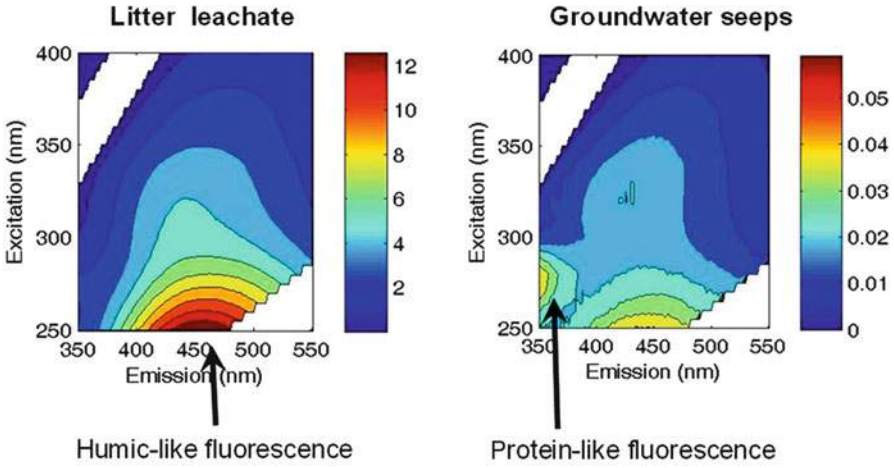

Fig. 7.1 Fluorescence excitation-emission matrices (EEMs) for dissolved organic matter (DOM) from two mid-Atlantic, USA watershed sources (forest floor litter leachate and groundwater seeps) illustrating humic-like and protein-like fluorescence peaks (Modified from Inamdar et al. (2012). Published with kind permission of © Springer Science + Business Media B.V 2012. All Rights Reserved)

relative contribution of each component to the total DOM fluorescence (Cory and McKnight 2005).

Some of the more common fluorescence metrics and PARAFAC components that have been used in recent studies are reported in Table 7.1. The fluorescence index (FI) is calculated using the ratio of fluorescence emission intensities at 470 and 520 nm at an excitation wavelength of 370 nm (Cory and McKnight 2005). McKnight et al. (2003) have used the FI to differentiate between DOM derived from vascular plants (FI: 1.3–1.4) versus microbial or planktonic sources (FI: 1.7–2.0). Another fluorescence metric, the humification index (HIX) was defined (Zsolnay et al. 1999) as a ratio of the peak integrated area under the emission spectra 435–480 nm by the peak integrated area under the emission spectra 300–445 nm; obtained at excitation wavelength 254 nm. This definition was later modified by Ohno (2002) to the integrated areas under the emission spectra 435–480 nm divided by the peak integrated area (300–345 nm + 435–480 nm); again obtained at excitation wavelength 254 nm. The revised equation of Ohno (2002) constrained HIX values to the range of 0–1 with higher values indicating more humified material. Another metric referred to as the freshness index (β:α; Wilson and Xenopoulos 2009) is computed as the ratio of emission fluorescence intensity at 380 nm by the maximum emission fluorescence intensity observed between 420 and 435 nm, calculated at excitation wavelength of 310 nm. The freshness index is a measure of recently produced DOM where β represents DOM of recent origin and α represents more decomposed DOM. Miller et al. (2006) proposed the redox index (RI) to explain the oxidation states of DOM fluorescence

defined as $Q_{red}/(Q_{red} + Q_{ox})$, where Q_{red} is the sum of the reduced components and Q_{ox} is the sum of the oxidized components. Obviously, high values of RI represented DOM components of reduced origin while low values represented oxidized DOM components. In addition to these indices, EEMs have also been used to identify fluorescence peaks in specific regions indicating humic-like and-or protein-like fluorescence (Coble et al. 1990; Cory and McKnight 2005). The humic-like fluorescence is typically assumed to represent DOM from vascular plants, with high molecular weight and aromatic in nature (Coble et al. 1998). In contrast, the protein-like fluorescence is assumed to represent DOM of low molecular weight, representative of amino acids like tryptophan and tyrosine (Yamashita and Tanoue 2003), composed of DOM that is more bioavailable (Fellman et al. 2009), and DOM that may be of microbial origin (Hood et al. 2009).

7.5 Nitrogen

7.5.1 Overview

Inorganic forms of N prevalent in wetland biogeochemical cycles include dinitrogen gas (N_2), ammonium (NH_4^+), ammonia (NH_3), nitrate (NO_3^-), nitrite (NO_2^-), nitric oxide (NO), and nitrous oxide (N_2O). Wetlands receive N inputs via atmospheric deposition, transport in the water column, and biological dinitrogen fixation. Dinitrogen fixation is the conversion of N_2 to NH_3 which is then rapidly converted to organic forms. Dinitrogen fixation occurs in algae, free living bacteria, and bacteria in symbiosis with macrophytes such as legumes. Nitrogen mineralization is the conversion of organic forms of N to inorganic forms (primarily NO_3^- and NH_4^+) during SOM decomposition. The reverse process is referred to as immobilization. Nitrification is the conversion of NH_4^+ to NO_3^-. Inorganic N is removed from wetlands via leaching losses, lateral transport in the water column, and gaseous losses to the atmosphere via denitrification and NH_3 volatilization. Denitrification is the microbial reduction of NO_3^- to gaseous products (primarily N_2O and N_2) which are returned to the atmosphere. Denitrification represents a significant path of N loss from wetlands and it is considered to be one of the more important wetland functions as it contributes to water quality by removing nitrates. Under high pH conditions (pH >8) NH_4^+ is converted to NH_3 which may be volatilized.

Many of the N techniques presented below contain a laboratory component to determine the N content of soil, water, or plant tissue. These chemical assays are beyond the scope of this chapter. The reader is referred to the following references: Bremner (1996), for total N; Mulvaney (1996), for inorganic N; and Stevenson (1996), for organic N.

7.5.2 Stable Nitrogen Isotopes

Many elements of biological interest, including C, H, O, N, and S have at least two or more stable (non-radioactive) isotopes. For a given element, the lightest isotope is present naturally in much greater abundance than the others. Nitrogen isotopes ^{15}N and ^{14}N have similar chemical characteristics and therefore behave almost identically in biological systems. The mass differences, however, result in partial separation of the two isotopes during chemical reactions and during physical processes such as diffusion. This separation of isotopes is referred to as isotope fractionation and results in a higher $^{15}N/^{14}N$ ratio in soils and in water systems than in the atmosphere. Levels of ^{15}N are commonly expressed as atom % excess relative to a standard or baseline level. The background level or baseline for each isotope is usually considered to be equal to atmospheric levels and expressed on a atom percent (At.%) basis: At.%$^{15}N = 0.3663$, At.%$^{14}N = 99.6337$. Atom % excess ^{15}N is any quantity of ^{15}N above this background level. For example, soil organic matter with an At.% ^{15}N value of 0.4773 would have an At.% excess ^{15}N value of 0.1110 (0.4773–0.3663). Individual components of the N cycle can be labeled by enriching it with ^{15}N. This allows one to trace the fate of N from individual pools. Fertilizer enriched with ^{15}N is commercially available. Adding ^{15}N-enriched fertilizer to the soil will label the soil N pool. The addition of organic matter with a high C/N ratio or the addition of sucrose along with the fertilizer will quickly result in a labeled soil organic N pool. Plants grown on the labeled substrate will also become labeled. Labeled plant tissue and labeled soil N can then be used to determine N_2 fixation or N mineralization rates.

7.5.3 Dinitrogen Fixation

7.5.3.1 Stable Isotope Techniques

Dinitrogen fixation in macrophytes such as legumes can be determined by two techniques that rely on the presence of the two stable N isotopes. The natural abundance technique is based on the naturally ^{15}N enriched soil N pool. With the isotope dilution technique, the soil N pool is labeled by additions of ^{15}N enriched fertilizer or ^{15}N enriched organic materials (Vasilas and Ham 1984). With both methods dinitrogen fixation is calculated by comparing the $^{15}N/^{14}N$ ratio in the fixing species with the $^{15}N/^{14}N$ ratio in a control (non-fixing) species. In the fixing species N derived from the soil is diluted with respect to ^{15}N by N derived from the atmosphere via N_2 fixation. The isotope dilution technique is considered to be more sensitive (Weaver and Danso 1994). However, it is not appropriate for non-managed ecosystems where control of the soil N pool is not an option. In wetlands, a further drawback to the isotope dilution method is the presence of a

periodically high water table which would result in significant losses of the fertilizer N through denitrification.

In both methods, tissue samples from both species (N_2-fixing and non-fixing) are collected, dried, and ground. Nitrogen in the ground tissue subsamples is converted to NH_4^+–N through acid digestion which is then oxidized to N_2. The N isotope ratio of the N_2 sample is determined by mass spectrometry or emission spectrometry. There are commercial labs that will run N isotope analyses on ground tissue samples. The only data needed from the control species is the N isotope ratio. To determine quantities of N fixed, biomass and total N content must be determined for the fixing species. Because the differences between the $^{15}N/^{14}N$ ratios in the atmosphere and that found in soils are very small, there are procedural differences between the isotope dilution method and the natural abundance method (Weaver and Danso 1994). First, in the natural abundance method, isotopic composition is expressed as $\delta^{15}N$ instead of At.% ^{15}N, where $\delta^{15}N = [(R_{sample} - R_{standard})/R_{standard}] \times 1{,}000$, and $R = {}^{15}N/({}^{14}N + {}^{15}N)$, so that 1 $\delta^{15}N$ unit = 0.00037 At.% excess ^{15}N. Second, it is recommended that a member of the fixing species be grown hydroponically on N-free medium to provide an estimate of discrimination that occurs during N_2 fixation. Third, multiple non-fixing plant species should be sampled to provide a mean of the isotopic composition of the soil. With the natural abundance method, the percentage of plant N derived from fixation (%Ndfa) is calculated as follows: $\%Ndfa = (x - y)/(x - f) \times 100$, where $x = \delta^{15}N$ of the non-fixing plants; $y = \delta^{15}N$ of the fixing plant grown in soil; and $f = \delta^{15}N$ of the fixing plant grown hydroponically.

7.5.3.2 Acetylene Reduction

Nitrogenase, the enzyme that reduces N_2 to NH_3 (N fixation), will also reduce acetylene (C_2H_2) to ethylene (C_2H_4). The acetylene reduction method has been used to provide a point in time assessment of N_2 fixation in both symbiotic systems (e.g., nodulated legumes) and free living organisms. The following method was presented by Carpenter et al. (1978) to assess N_2 fixation by free living bacteria and algae in tidal marshes. They used surface cores taken from vegetated marsh areas to target algae and sediment slurry samples taken from pannes to target cyanobacteria. A plastic corer is used to take a surface core (0.5 cm^2 dia., 0.25 cm deep); a pipette is used to remove 1 ml of slurry. Each sample is placed in a 6.5 ml wide mouth serum bottle, the bottles are capped, and injected with 1 cc high-purity C_2H_2 gas. The samples are gently rotated to facilitate solution of the C_2H_2, and incubated for 1 h. A gas sample is than removed and analyzed for C_2H_4 by gas chromatography.

Limitations to the acetylene reduction method include the point in time nature of the assay and the indirect nature of the assay which makes extrapolation to the amount of N_2 fixed questionable. Seasonal rates of N fixed cannot be determined from a point in time measurement because the rate may vary over time. Quantifying N_2 fixed from the amount of C_2H_2 reduced requires a conversion factor. Hardy et al. (1968) first suggested a theoretical conversion factor of 3 mol C_2H_2 reduced to

1 mol N_2 reduced (equivalent to 1.5 mol C_2H_4 per mol NH_3). More commonly, a conversion factor of 4 mol C_2H_2 reduced to 1 mol N_2 reduced is used (Boddey 1987). Hardy et al. (1973) suggested that a conversion factor should be experimentally determined for each system.

7.5.4 Nitrogen Mineralization and Nitrification

7.5.4.1 Overview

Nitrification is an aerobic process and N mineralization is more rapid under aerobic conditions than under anaerobic conditions. Field assays for estimating net rates for N transformations were developed for upland conditions. However, they are appropriate for some types of wetlands if care is taken so that the incubation period does not correspond to large fluctuations in water table depth. The results will be most reliable if soil moisture content is relatively constant during the incubation period. They are not appropriate for situations of continuous soil saturation as the rates of these transformations will be too low to be accurately measured.

Precise estimates of N mineralization or nitrification in the field are difficult to obtain and beyond the scope of most field exercises. Nitrate can be removed from the soil by plant assimilation, leaching, and dissimilatory nitrate reduction including denitrification. Plant uptake can be eliminated by using some type of soil containment device that keeps roots away from the sampled soil. Denitrification and other dissimilatory nitrate reduction effects can be minimized by avoiding saturated soil. Leaching effects can be minimized by preventing percolation during the incubation period. However, N mineralization and immobilization occur simultaneously. Therefore, most reported figures for N mineralization or immobilization reflect net gains or losses to a soil N pool. For example, net N mineralization $= (NH_4^+\text{–}N + NO_3^-\text{–}N)_{t+1} - (NH_4^+\text{–}N + NO_3^-\text{–}N)_t$, and net nitrification $= (NO_3^-\text{–}N)_{t+1} - (NO_3^-\text{–}N)_t$ (Hart et al. 1994). Gross rates of these processes can only be determined through isotopic techniques.

7.5.4.2 Buried Bag Method

The buried bag method is a simple technique for quantifying net N mineralization and nitrification in the field. It is most appropriate for assessing surface soils; for the assessment of subsurface soils the reader is referred to the closed-top, solid cylinder method (Hart et al. 1994). Intact soil cores are taken with a coring device (PVC or metal tubes), placed in polyethylene bags, sealed, returned to the original hole, and incubated in the field for 1–2 months. The plastic bags used in this procedure are permeable to gases but not to liquids (Gordon et al. 1987). One advantage to this method in comparison to lab incubation assays is that the soil samples are subjected to on-site temperature regimes. However, since the bags are impermeable to water,

the technique integrates the on-site soil water dynamics only if the soil water content at the beginning of the incubation period is representative of the entire incubation period. The following specifics to the assay were drawn from Hart et al. (1994).

Soil cores are extracted with thin-walled PVC or metal cylinders, sharpened at 1 end to provide a cutting surface. Recommended cylinder dimensions are 5 cm i.d. and 12 cm length. Longer cylinders allow for deeper soil samples but are more difficult to remove intact cores from the cylinders and require larger incubation bags. Cylinders are traditionally handled in groups of three. Cylinders are hand driven into the soil until 2 cm of the cylinder is above the soil surface. Three cylinders should be used for each 50 m^2 of area; these serve as replicates. Cylinders are removed and the soils are removed from the cylinders resulting in a 0–10 cm sampling depth. Soils from three cylinders are composited for pre-incubation analysis. These will serve to provide baseline (time 0) data for initial concentrations of NH_4^+ and NO_3^-. Additional soil samples are taken (pre- and post-incubation) to determine gravimetric soil water content. As the laboratory analysis is conducted on field moist soil, soil moisture content is needed to express the data on a dry weight basis. Each of the remaining soil samples are enclosed in polyethylene bags (15–30 μm thick), the bags are sealed with plastic ties, and returned to their respective bore holes. The bags are covered with leaf litter, if present. If not, the bags should be covered with a small amount of similar soil to prevent temperature extremes. Bags are removed after a 1 or 2 month incubation period. Samples with perforated bags should be discarded. Consecutive buried bag incubations will provide seasonal patterns of net N mineralization and nitrification. Microbial activity will continue after the bags are retrieved. Therefore, the samples should be kept cool (2–5 °C) during short-term storage ($\leq$2 d) and frozen for long-term storage. Inorganic N is extracted from soil subsamples with 2*M* KCl in the lab. See Hart et al. (1994) for the sample preparation and extraction procedures. Filtered extracts are analyzed for NH_4^+–N and NO_3^-–N (see Mulvaney 1996). Net N mineralization is calculated as the change in inorganic N (NH_4^+–N and NO_3^-–N) content over the incubation period; net nitrification is calculated as the change in NO_3^-–N content over the incubation period. Net P mineralization can also be determined with this method.

7.5.4.3 Resin Core Method

The *in situ* intact-core resin-bag method was developed to measure N mineralization and nitrification under the existing field soil conditions with respect to temperature and water content (Distefano and Gholz 1986). This system consists of an intact soil core within a PVC or metal cylinder with an ion exchange resin bag at each end. This allows water but not ions to flow through the soil core and eliminates the static moisture regime inherent to the buried bag method. No resin bag is needed at the top of the cylinder if the soil core does not receive leachate from overlying soil horizons or is exempt from inundation.

Wienhold et al. (2009) presented a modified version of the ion resin exchange technique in which they use a resin bag at the bottom only. A metal cylinder

(4.75 cm i.d.) was inserted 17 cm into the soil and immediately extracted, encasing the soil core. Two centimeter of the encased soil were removed from the bottom of the cylinder and replaced with a nylon resin. The bottom of the cylinder was then covered with sturdy nylon cloth to prevent root entry and the cylinder assembly was reinserted into the original hole. They found that incubation periods greater than 60 days resulted in resulted in loss of inorganic N from resins, and recommended 28- to 40-day incubations. However, they conducted their study in uplands; loss of inorganic N from resins would probably be greater under saturated conditions. They also cautioned against compacting the soil below the cylinder during installation as that can impede water movement through the soil core.

7.5.5 Denitrification

7.5.5.1 Overview

Several approaches have been used to estimate denitrification, including *in situ* measurements using natural N isotopic abundances (Søvik and Mørkved 2008), *in situ* measurement of N_2O evolution without an C_2H_2 block (Jordan et al. 2007; Whalen 2000; Wray and Bayley 2007), *ex situ* measurement of N_2 and-or N_2O evolution from intact cores without an C_2H_2 block (Horwath et al. 1998; Wray and Bayley 2007), *ex situ* measurement of N_2O evolution from intact cores using an C_2H_2 block (Bohlen and Gathumbi 2007; Horwath et al. 1998; Hunt et al. 2003), *ex situ* measurement using intact cores coupled with isotopic approaches (Racchetti et al. 2011; Rückauf et al. 2004), and *ex situ* measurement of denitrifying enzyme activity (DEA) in homogenized soil with or without using an C_2H_2 block and optimized incubation conditions (Bruland et al. 2009; Hunt et al. 2003, 2007; Jordan et al. 2007; Sirivedhin and Gray 2006). The methods perhaps best suited to routine estimations, and the two considered here, are (1) *ex situ* measurement of N_2O evolution from intact cores using an C_2H_2 block, and (2) *ex situ* measurement of DEA in homogenized soil with using an C_2H_2 block and optimized incubation conditions.

Using C_2H_2 block eliminates problems with measuring N_2 evolution against high atmospheric background levels of the gas or the necessity of using a non-N_2-containing atmosphere during incubations. Acetylene blocks the conversion of N_2O to N_2, thus simplifying analyses and providing a more sensitive assay. Intact cores can arguably give more realistic estimates of denitrification than does DEA, but are typically more variable both spatially and temporally. In comparison with using intact cores, estimates of denitrification based on DEA using homogenized soil are generally less variable but may give unrealistically high values, especially if the usual practice of optimizing incubation conditions is used (i.e., anaerobic atmosphere, presence of excess C and NO_3^-). It is for this reason that the values obtained are commonly considered potential rates of denitrification and are viewed as representing primarily maximum relative rather than absolute estimates. Thus, the investigator will need to decide which of the two approaches best meets the needs of the situation at hand.

We have included a third method, water column analysis for inorganic N, which represents a very different approach. Nitrogen removal from the water column can be calculated by comparing the inorganic N level in the input water from the N level in the output water. This is an indirect approach; it does not directly assess denitrification and there are other pathways for inorganic N removal (i.e., macrophyte assimilation). Furthermore, total N loss can only be determined if a hydrologic budget is created. However, relative differences in water quality services capacity between wetlands can be characterized by this approach.

7.5.5.2 Measurement of Denitrification in Intact Soil Cores using an Acetylene Block

This method is based on Bohlen and Gathumbi (2007), modified from Horwath et al. (1998) and Mosier and Klemedtsson (1994). A number of 2-cm-diameter soil cores are taken with a hammer corer fitted with cylindrical plastic inserts to the desired depth, the number of cores dictated by the resources available and the expected experimental variability. The plastic insert should be only partially filled, leaving room for sampling headspace gasses. The bottom end of the plastic insert is immediately sealed with a rubber stopper, and the cores kept at 4 °C until analyzed (<24 h). The cores are allowed to equilibrate briefly at ambient or some standard temperature. The top of the plastic cylinder is sealed with a stopper fitted with a rubber septum and C_2H_2 (See comments) is added to a concentration of ~10 kPa using a syringe. The C_2H_2 is distributed throughout the cylinder by repeated pumping of the headspace gasses with a large (~60-ml) syringe. Samples of headspace gasses are taken at intervals (e.g., after 2 and 6 h; see comments) and placed in evacuated vials fitted with rubber septa. Gas samples, including blanks and standards, are analyzed by gas chromatography using a Poropak Q (or equivalent) packed column in combination with an electron capture detector. For examples of specific chromatographic run conditions, refer to Hunt et al. (2007), Jordan et al. (2007), and Sirivedhin and Gray (2006). The amount of N_2O produced between selected sampling times is used to estimate denitrification.

7.5.5.3 Denitrification Enzyme Activity (DEA)

The following method is taken from Jordan et al. (2007) as modified by Tiedje et al. (1989). A weighed amount of soil (~25 g) is placed in a 125-ml flask containing 25 ml of a solution of 1 mM glucose, 1 mM KNO3, and 1 g/L chloramphenicol. The chloramphenicol is used to prevent *de novo* synthesis of denitrification enzymes. The flask is flushed with N_2 or another inert gas such as Ar and closed with a rubber stopper fitted with a gas-sampling septum. The flask is injected with 11 ml of C_2H_2 and incubated for a standard period of time (e.g., 2 h; see comments) at ambient or some standard temperature. Samples of headspace gasses are then taken by syringe and transferred to evacuated vials fitted with rubber septa. Gas samples, including blanks and standards, are analyzed as described for intact cores.

7.5.5.4 Comments

Core dimensions, soil sample size, and sampling times are given as guidelines only and can be modified to meet specific needs and situations. Acetylene used for the C_2H_2 block should not be obtained from commercial (e.g., welding) tanks, as there are reports that these can contain acetone (Jordan et al. 2007). Analytical grade C_2H_2 and C_2H_2 produced by reacting CaC_2 with water have been used successfully. Regarding measurement of DEA specifically, some investigators have chosen to agitate the assay flasks during incubation rather than using static flasks (e.g., Hunt et al. 2007). Additionally, some investigators (Hunt et al. 2003, 2007) used DEA approaches under both optimized (added NO_3^- and glucose) and non-optimized conditions and with and without an C_2H_2 block, in an effort to obtain estimates of denitrification having varying degrees of relatedness to true denitrification rates; the authors conceded that the values they obtained even under non-optimized conditions may still have been overestimates because an O_2-free atmosphere was used during the assay incubations (Hunt et al. 2007).

7.5.5.5 Water Column Analysis

Critical to this approach is the assessment of the hydraulic gradient to identify hydrologic input sources and outputs for sampling. For constructed wetlands, identification of hydrologic inputs and outputs is simple and straightforward. For natural wetlands, the identification of hydrologic input and output sources is more problematic and may require the expertise of a soil scientist. For groundwater driven slope wetlands or depressions, a hydraulic gradient is easily characterized. Slope inputs are easily identifiable as seeps (side slope or toe slope). For depressions, a soil investigation is required to identify the dominant hydrologic source. Often a gravel or sand lens representing a zone of high hydraulic conductivity will extend from the upland into the depression. Slope wetlands usually have apparent outlets; depressions may or may not have outlets. For mineral soil flats, elevations should be shot with the assumption that surface elevations correspond to water table elevations. For surface water driven systems, a flume is needed to collect overland flow.

Inflow points and outflow points are the water sampling points. At each point, a water sampling well is installed. Slotted PVC pipe (i.d. 5 cm) is inserted into a bore hole which is then backfilled with coarse sand or pea gravel and capped with bentonite (see Chap. 3). In some cases, it may be desirable to separate surface water from sub-surface water, or to separate two discontinuous water columns (i.e., episaturation). In those cases, two sampling wells must be installed at each sampling point.

Chemistry of water in the well casing may not be identical to that of the water column. Prior to sampling water, the wells are purged to remove stagnant water. Wells are emptied and the water discarded prior to sampling the wells after they refill. This is known as “purging the wells”. As a general guideline, 3–5 casing

volumes are purged, where a "casing volume" is defined as the volume of water standing in the well (Barackman and Brusseau 2004). In some cases, recharge of the well is so slow that the water sample must be collected a day after purging. A number of portable battery driven pumps are available for purging and sampling the wells. These pump the water through tubing inserted into the well. To avoid contamination, deionized water should be pumped through the tubing between samples. This method is not appropriate for all wetlands, and it should not be used to compare different types of wetlands. For seasonally saturated wetlands, there may not be sufficient well water to sample during parts of the year.

7.6 Phosphorous

7.6.1 Overview

Phosphorous is generally considered to be the major limiting nutrient in most freshwater wetlands (Mitsch and Gosselink 2000) determining the rate of primary production and decomposition. However, when present at very high concentrations, P stimulates macrophyte and algae growth and the resulting increase in the rates of decay of the vegetation depletes O_2 levels in the water. Wetlands represent both a source of P and a sink for P. Furthermore, an individual wetland can switch from a P sink to a P source as its trophic state changes. The trophic state of an aquatic ecosystem is defined by its rate of primary production, the concentration of nutrients and minerals and the rate of supply of organic matter (determined by the balance of primary production and decomposition) (Correll 1998). Oligotrophic systems have low rates and concentrations, while eutrophic conditions exist when the ecosystem is over-enriched with nutrients and minerals. As the flux of P into the wetland increases, primary production and trophic state increase while dissolved O_2 and biodiversity decrease (Correll 1998). These changes in ecosystem processes interfere with the wetland's ability to retain P and wetland soils can become saturated with orthophosphate (OP), after which P may be released to the water column resulting in eutrophication of receiving surface waters (Champagne 2008; Vepraskas and Faulkner 2001).

The major pool of P in natural wetlands is found in the soil sediments (fixed mineral P) and the litter (organic P) comprising approximately 80–90 % of the P within the wetland (Champagne 2008; Richardson and Craft 1993; Vepraskas and Faulkner 2001). The remaining fraction of P is found in the water column and the pore water as soluble OP, dissolved organic P and total dissolved P (Vepraskas and Faulkner 2001). Phosphorus in wetlands can be found as soluble and insoluble complexes in both organic and inorganic forms. The principle inorganic form is OP, which is present in the pH-dependent ionic species; $H_2PO_4^-$ ($2 < pH < 7$), HPO_4^{2-} ($8 < pH < 12$) and PO_4^{3-} ($pH > 12$) (Kadlec and Knight 1996). However, OP can react with soil constituents to form both insoluble and soluble compounds. In acidic soils (4–7 pH) under aerobic conditions, OP forms insoluble precipitates with

ferric Fe and aluminum oxides and hydroxides. As these soils become first anaerobic and then reduced, dissolution of the precipitates releases P into the soil pore water and water column. In alkaline soils (pH >7), precipitation of insoluble Ca or Mg phosphates or OP adsorption to carbonates occurs (Richardson and Craft 1993; Vepraskas and Faulkner 2001). The soluble inorganic OP is considered the biologically available form of P. Organic forms of P are found in plants, partially decomposed plant tissue, and found as OP bound in organic matter.

Phosphorous has a complex role in wetland ecology. Because of its dual role as both a limiting nutrient and as a major pollutant, especially with its role in eutrophication of surface waters, a number of approaches have been developed to assess different forms of P or different P pools in wetlands. Most of the assays are specific to soil or water. Sequestration is addressed by assays for soil P content. To address P as a limiting nutrient, two approaches are commonly used-a chemical extraction scheme that separates soil P into various fractions representing a range in bioavailability and a mineralization assay. A similar fractionation approach is used to evaluate water quality. As the major eutrophic surface water nutrient, both soil and water assays have been developed that take a bioavailable P approach and target OP. We will not go into detail on the wet chemistry techniques for assessing P. These methods are presented in detail by Kuo (1996) for soils and Eaton et al. (2005) for water. Kovar and Pierzynski (2009) (http://www.sera17.ext.vt.edu/Documents/P_Methods2ndEdition2009.pdf) present methods for soil and water including methods specific to runoff water and flooded soils.

7.6.2 Soil Phosphorous Content

7.6.2.1 Total P

Many methods have been developed to extract and analyze total P in soil; four of the more common include: sodium carbonate (Na_2CO_3) fusion, perchloric acid ($HClO_4$) digestion, sulfuric acid-hydrogen peroxide-hydrofluoric acid (H_2SO_4–H_2O_2–HF) digestion, and sodium hypobromite (NaOBr) oxidation followed by H_2SO_4 dissolution. All four methods convert soil organic P to inorganic P (Kuo 1996), but the Na2CO3 fusion method is considered to be the most reliable (Syers et al. 1967). It is believed that the acid digestion methods fail to extract P from apatite inclusions or imbedded in the matrix of silicate minerals (Syers et al. 1967) and therefore underestimate total P. Three of the methods (Na_2CO_3, H_2SO_4–H_2O_2–HF, and $HClO_4$) are described by Kuo (1996).

7.6.2.2 Total Inorganic P

Inorganic P is extracted with 1 M hydrochloric acid (HCl) or H_2SO_4. Not all of the mineral forms of P may be solubilized with this extraction leading to an underestimation of inorganic P.

7.6.2.3 Total Organic P

Three methods are commonly used to determine total organic P in soils (Reddy and DeLaune 2008) as follows. The methods for quantifying P in extracts are found in Kuo (1996).

1. Difference method: Organic P is calculated as the difference between total P and inorganic P. This approach is best suited for soils high in organic matter; it tends to overestimate organic P in samples high in mineral forms of P.
2. Acid-alkaline extraction method: First P_i is extracted with acid (1 M HCl or H_2SO_4); this is followed by an extraction with alkali (0.5 M NaOH). The alkali extracts are digested to solubilize organic P. This method may underestimate organic P as the extraction procedure may not be sufficient to solubilize all forms of organic P.
3. Ignition method: Organic P is removed by ashing the sample at 550 °C. Inorganic P in the residue is extracted with acid. Organic P is calculated as the difference between total P in the unignited soil and inorganic P in the residue.

7.6.3 Soil Phosphorous Fractionation Schemes to Assess Availability

Orthophosphorus is considered the most bioavailable form of P and it constitutes the bulk of P found in soil pore water. Orthophosphate is readily adsorbed onto surfaces of clay particles and organic matter, and OP can also substitute for silicate within clays (Mitsch and Gosselink 2000). Adsorption of OP onto wetland soils and organic matter is considered long-term P retention. Phosphorus availability is characterized by the concentration of P in the soil solution and by the P buffering capacity that governs the distribution of P between the solution and solid phases. Some of the adsorbed P can be released (desorbed) into soil solution where it is readily bioavailable. Therefore, it is useful to separate exchangeable P, which is potentially bioavailable, from the non-exchangeable P. Because of the complex processes that govern the distribution of P between the solution and the solid phase within wetland soils, P tests have been developed in order to evaluate P availability.

A sequential chemical extraction scheme is commonly used to determine soil P pools in soil to approximate bioavailabilty. This approach is based on differential solubilities in a series of chemical extractants. A number of extraction schemes have been developed for soil P. We offer a simplified scheme below that was presented by Reddy and DeLaune (2008). Most are more complex than the example presented here but they allow for further fractionation. For example, some schemes utilize a filter to separate soluble P from particulate forms of P. Also, extraction schemes have been developed to target soils with specific characteristics (e.g., calcareous soils or organic soils). In each step the resulting extract is assayed for OP. Details on these analytical procedures are presented by Kuo (1996).

1. Extraction with salt solutions (e.g., KCl): This extraction removes soluble and exchangeable P which are considered bioavailable.
2. Extraction with alkali: Residue from Step 1 is treated with 0.1 M NaOH which removes P bound to Fe and Al, and some organic P. This P pool is generally considered to be unavailable but Fe bound P may become available under anaerobic conditions. A more complex extraction sequence can be used to separate Al-P from Fe-P and organic P.
3. Extraction with acid: Residue from Step 2 is treated with 0.5 M HCl which removes Ca and Mg bound P which is considered to be unavailable. The residue contains organic P and stable forms of inorganic P that are considered unavailable.

Soluble P in water and soil extracts can be determined without pretreatment colorimetrically or through instrumentation methods including inductively coupled plasma (ICP) spectrometry and ion chromatography. Ion chromatography measures OP; ICP measure inorganic P and organic P and it is used for soil extraction extracts. Colorimetric methods measure primarily OP. Most commonly used is the ascorbic acid method which measures OP and a small fraction of the condensed phosphates; standard methods calls them "reactive phosphorous". In the ascorbic acid method (Eaton et al. 2005), OP reacts with ammonium molybdate and potassium antimonyl tartrate in an acid medium to form phosphomolybdic acid, which is reduced to an intensely colored molybdenum blue by ascorbic acid. The intensity of the blue color is measure on a spectrophotometer and the concentration is determined from an individually prepared calibration curve.

7.6.4 Phosphorus Fractionation Methods for Reduced Soils

Phosphorous fractionation methods were developed primarily for agricultural soils in an attempt to estimate fertilizer needs. Wetland soils may present an additional challenge because P chemistry in soils and sediments is strongly influenced by the oxidation-reduction status (redox potential). Ferric and manganic oxides and hydroxides provide major adsorption sites for P under oxidized conditions. In addition, ferric and manganic phosphate minerals which form and persist under oxidized conditions become unstable under reduced conditions releasing soluble P into the soil solution (Moore and Reddy 1994). Oxidation of anaerobic sediments results in the rapid conversion of Fe^{2+} to Fe^{3+}. Within minutes, $Fe(OH)_3$ precipitates out of solution and has a tremendous capacity to sorb P, causing soluble P levels in the porewater to be reduced by orders of magnitude. Therefore, it is critical that samples collected from reduced soils and sediments are kept reduced during the sampling procedure, transport to the lab, storage, and the initial phases of P fractionation. Moore and Coale (2009) present methods that specifically address these concerns.

7.6.5 Phosphorous Mineralization

Chemical extraction procedures produce an estimate of available P at one point in time. P mineralization studies produce estimates of available P over time. Phosphorous mineralization rates can be assessed either *in situ* with litter bags (see *Litter Bags* above) or in the lab using an incubation approach. In the incubation approach, soil samples are kept under controlled climate conditions in the lab and leached at set time intervals. The two mineralization approaches would not be expected to produce the same results. The benefit to the litter bag approach is that mineralization is assessed under natural fluctuating conditions of moisture and temperature. It does not totally reflect real life conditions as the substrate is usually fresh plant material and the material is physically separated from the soil. Incubation studies typically use soil samples, a more realistic substrate, but employ optimum temperature and moisture conditions. In addition, soil structure is disturbed as the soil samples are typically mixed with sand to facilitate drainage during leaching. The benefit of the incubation approach is that it allows the comparison of different soils under identical environmental conditions so that inherent differences in mineralization potential can be determined.

Bridgham et al. (1998) utilized the following incubation procedure to estimate P mineralization potential for a series of wetland soils. One advantage to their approach is that they could estimate mineralization potential under both aerobic and anaerobic conditions. This allowed them to determine maximum and minimum mineralization rates for the given temperature. Field moist soil was mixed with acid washed sand to promote drainage during leaching. Samples were placed in 150 ml Falcon filter units and incubated at 30 °C. For aerobic incubations, the samples were exposed to ambient air. For anaerobic incubations, the duplicate samples were placed in filter units which in turn were placed in 500 ml Mason jars filled with water. Samples were leached with 0.001 M $CaCl_2$ at ten dates ranging from 2 to 59 weeks. Both the leachate and the Mason jar incubation water were analyzed for PO_4^{3-}.

7.6.6 Phosphorous in Water

There are many tests for P in water, most use a chemical or physical fractionation scheme as presented below. The P fraction analyzed in each step is based on whether the sample is digested and/or filtered, and the nature of the digestion. In each step, P concentration is determined by the ascorbic acid method. Particulate P is separated from the dissolved (or soluble) fraction by passing the water sample through a filter, typically a 0.45 μm cellulose (Millipore) filter. In each step, the resulting extract is assayed for OP by the ascorbic acid method. Phosphorous measured on undigested samples is considered to be inorganic, predominantly OP. Acid hydrolysis digestion converts inorganic P (primarily condensed phosphates) to OP. Some of the

phosphate from organic compounds found in the water may be released but this can be reduced by selecting the right acid strength, the right boiling temperature and the hydrolysis time. Oxidative digestion converts both organic and inorganic forms of P to OP. Organic P is calculated as the difference between total P and inorganic P. Analytical details for this scheme are presented in Eaton et al. (2005).

1. Dissolved acid hydrolyzable P (DAHP): OP determined on filtered samples after acid hydrolysis.
2. Total acid hydrolyzable P (TAHP): OP determined on unfiltered samples after acid hydrolysis.
3. Total dissolved P (TDP): OP determined on filtered samples after oxidative digestion.
4. Total P (TP): OP determined on unfiltered samples after oxidative digestion.
5. Dissolved organic P (DOP): DOP = TDP − DAHP.

7.6.7 Biologically Available Phosphorous

7.6.7.1 Overview

The term "biologically available P" (BAP) is used in a number of disciplines without a standardized meaning. However, in part due to the recognized importance of P in eutrophication of surface waters, BAP has been operationally defined as "the amount of inorganic P a P-deficient algal population can utilize over a period of 24 h or longer" (Sonzogni et al. 1982). The amount of BAP in soil, sediment, and water has been routinely quantified by algal assays or chemical extractions (see Sharpley (2009) for a review of methods). Algal assays require long incubation periods and chemical extraction times. Results from the chemical extraction approach are impacted by the nature of the extractant. Weaker chemical extractants (e.g., NH4F and NaOH) approximate P that is bioavailable under aerobic conditions, whereas stronger extractants (citrate-dithionite-bicarbonate) represent P that may become bioavailable under reducing conditions (Sharpley 2009). Because of these limitations, there has been increased interest in a different approach-one that utilizes P sinks.

The P-sink approach to assessing BAP is based on a chemical sink that attracts only P in forms that would be available to macrophytes and algae. In theory because the P sinks continuously remove dissolved P from the soil solution (Kuo 1996) which creates a concentration gradient, they simulate P removal from soil or water by plant roots and algae. This is not a completely accurate analogy because a chemical sink cannot approximate the rhizosphere influence as root exudates and mycorrhizal fungi can alter P availability. Two of the more common P-sink methods utilize anion exchange resins and Fe oxide impregnated filter paper. These are addressed below.

7.6.7.2 Anion Exchange Resins

Anion exchange resins represent the most common P-sink approach for soils. Anion exchange resins remove dissolved phosphates from the soil solution via surface adsorption (Kuo 1996). The rate of P adsorption is a function of solution P concentration. The resin promotes a low level of solution P concentration to maintain P desorption from the soil. Anion exchange resins can be added directly to soil suspensions, placed in a polyester bag, or the resin may be impregnated onto a plastic membrane. Typically, the procedure utilizes chloride-saturated resin, bicarbonate (HCO_3^-) saturated resin, or a combination of the two. The soil sample and resin are shaken together in deionized water or weak electrolyte 16–24 h. Adsorbed P is extracted with HCl and the P concentration is determined by the ascorbic acid method. For details on this procedure see Kuo (1996).

7.6.7.3 Iron Oxide Method-Runoff

The FeO method is a unique approach to assess the potential for runoff to increase fresh-water eutrophication. The use of iron-oxide (FeO) coated paper to test soil was first reported by Sissingh (1983), who sought to develop a P test to estimate plant-available P in tropical soils without mobilizing other forms of phosphates. Sissingh (1983) created a strip of filter paper impregnated with iron hydroxide which adsorbed mobile P from solution. The main advantage of this approach over standard soil P tests is that the FeO paper functions as an ion sink and doesn't react with soil as will chemical extractants. It has been shown that P extracted by this method (FeO-P) from runoff sediment is correlated to algal growth (Sharpley 1993a, b). A major benefit of the FeO method is its capability to differentiate between soluble inorganic P from FeO-P in sediment of runoff. Sediment FeO-P is considered bioavailable particulate P (BPP) and is calculated as follows: BPP = total BAP − SP; where total BAP is total FeO-P from unfiltered runoff, and SP is soluble inorganic P in filtered runoff (0.45-μm filter).

The FeO method has a stronger theoretical justification for estimating P availability of runoff for plants and algae than do chemical methods (Sharpley 1993a) as chemical methods may mobilize additional forms of P which are not available to plants or algae. Although algae uptake of P is restricted to OP, organic forms of P can undergo mineralization and become available (Correll 1998) and thus be considered a latent source of BAP. There has been discussions focused on methods to restrict hydrolysis of organic P adsorbed onto FeO paper (Robinson and Sharpley 1994). However, adsorption and hydrolysis of organic P is not considered a problem with the FeO method as organic P may be classified as latent BAP which may be readily mineralized, thereby becoming available for algal use.

The FeO paper is made from filter paper immersed first in $FeCl_3 \cdot 6H_2O$ and then in NH_4OH. Paper preparation is described in detail by Myers et al. (1997). The water of interest is sampled and taken to the lab. FeO paper is enclosed within

two polyethylene screens to maintain integrity during shaking. The polyethylene screens hold the FeO paper in a fixed orientation during shaking. This helps to prevent contamination from soil particles lodging in the pores of the paper (Myers et al. 1995, 1997) and prevents the paper from sticking to the walls of the shaking vessel, which could reduce adsorption effectiveness of the paper. The FeO paper-screen assembly is inserted into a bottle containing the runoff sample. Both deionized water and a $CaCl_2$ solution (0.01 M) has been used as the shaking matrix for the FeO paper and soil. Deionized water can disperse soil, causing it to lodge in the pores of the filter paper (Sissingh 1983), leading to errors in P analysis (Myers et al. 1995). The need for $CaCl_2$ probably depends upon the clay content of the sediment, the P content of the clay, and the amount of sediment in the runoff. The bottles are shaken on a reciprocating shaker to facilitate adsorption of P to the paper. Phosphorous is extracted from the papers by the addition of H_2SO_4 and further shaking. An aliquot of the H_2SO_4 solution is analyzed for P after neutralization of acidity. For further details on this procedure, see Myers et al. (1995, 1997).

7.7 Sulfur

7.7.1 Overview

Sulfur exists in both organic and inorganic forms. Inorganic forms can be classified as gaseous (e.g., sulfur dioxide-SO_2, and hydrogen sulfide-H_2S), reduced (e.g., elemental S-S°, sulfide-S^{2-}), or oxidized (e.g., sulfate-SO_4^{2-}, and sulfite-SO_3^{2-}). Common S containing minerals are gypsum ($CaSO_4$) and pyrite (FeS_2). In wetland soils, S is found mainly in organic forms (Reddy and DeLaune 2008). Under highly reduced conditions (Eh $<$100 mV), obligate anaerobes use SO_4^{2} as a terminal electron acceptor to produce H_2S. Sulfate reduction dominates anaerobic decomposition in brackish marshes, inhibiting CH_4 production and regulating soil C storage (Megonigal et al. 2003). Soluble S^{2-} at levels found in estuarine marshes can be detrimental or toxic to many organisms, and it has been shown to limit the growth of common marsh grasses including *Spartina alterniflora* Loisel (Koch et al. 1990; Mendelssohn and McKee 1988).

Detrimental environmental impacts occur when reduced S compounds (like sulfides) are oxidized to form sulfuric acid (H_2SO_4). Acid mine drainage refers to the outflow of acidic water from mines. The acidity, due to oxidation of metal sulfides like FeS_2, and subsequent dissolution of heavy metals can cause fish kills and loss of vegetation. Acid sulfate soil is the common name given to soils and sediments exhibiting these problems. Soils containing Fe sulfides (the most common being FeS_2) are referred to as potential acid sulfate soils, but when they oxidize and begin to generate acidity, they are referred to as active acid sulfate soils (Fanning et al. 2010).

Potential acid sulfate soils form under continuously anaerobic conditions and are commonly found in coastal wetlands. When these wetlands are drained, exposure of the soil to O_2 results in the formation of H_2SO_4 creating very acidic conditions (pH <2) and often releasing toxic quantities of Fe, aluminum and heavy metals such as arsenic. Because of the toxic effects of free S^{2-} and the potential for detrimental effects upon oxidation of FeS_2, the rapid assessment of soluble S^{2-} in soil pore water is valuable to wetland scientists and managers dealing with estuarine systems. Below we present two approaches to this assessment.

7.7.2 Measurement of Soluble Sulfide in Marsh Pore Water

7.7.2.1 Sippers and Peepers

Sulfide in pore water is usually measured using two basic methods that collect pore water in the field with sippers or peepers and analysis of water S^{2-} content in the lab using an ion-selective electrode and a set of standards (Eaton et al. 2005). Pore water extractors or "sippers" are inserted into the marsh soil, suction is applied and the pore water is collected in a syringe (e.g., Marsh et al. 2005; Keller et al. 2009). This approach is relatively rapid, but provides poor resolution as the sample is drawn from a soil volume of uncertain dimensions. A second approach uses equilibrium dialysis samplers or "peepers" (Hesslein 1976). In this method, a device containing a vertical series of chambers is filled with deoxygenated, distilled water, covered with a semi-permeable membrane, inserted into the marsh soil, and allowed to equilibrate. When the soluble constituents in the pore water reach equilibrium with those in the chambers, the device is extracted and the water in the chambers is analyzed in the laboratory. Both sippers and peepers are used to obtain vertical profiles of pore water chemistry within the sediment column. By virtue of their placement in the sediment, sippers allow for some vertical segregation of data. Peepers are designed for the collection of discrete water samples at a smaller spatial resolution by preventing vertical mixing of adjacent water masses. Peepers have superior vertical resolution to the sippers (1–2 cm vs. 5–10 cm), but are limited by a relatively long equilibration period (usually 1 week or longer) (Teasdale et al. 1995).

7.7.2.2 IRIS Panels

IRIS tubes were developed to assess the presence of reducing conditions in soils (Castenson and Rabenhorst 2006; Jenkinson and Franzmeier 2006) (see *Techniques to Measure Eh/Assess Reducing Conditions* below). IRIS tubes are PVC tubes coated with an Fe oxide paint. When inserted into soil, the paint dissolves when reduced exposing the bare PVC pipe. It was inadvertently discovered that when IRIS tubes were exposed to S^{2-}, Fe monosulfide (FeS) coatings formed on the tubes

(Stolt 2005) as evidenced by a dark gray to black coating. Rabenhorst et al. (2010) modified the IRIS tube technology to produce IRIS panels for assessing soluble S^{2-} in brackish marshes. The technology can be used for a qualitative assay to detect the presence of soluble S^{2-}, or a quantitative assay to determine S^{2-} concentrations. A simple color change indicates the presence of soluble S^{2-}, and the tube shape is sufficient. A quantitative assay requires a set of standards and image analysis capabilities. The standards correlate the intensity of the color change to a known S^{2-} concentration. Image analysis is used to quantify the surface area of the tube or panel that represents an individual intensity of color change. Rabenhorst et al. (2010) decided to use large flat panels rather than the cylindrical tubes to facilitate quick recording of the images with a flatbed scanner. In contrast to the pore water sampling approach (sippers and peepers), this new technology shows the potential for generating quantitative information on S^{2-} concentrations with millimeter-scale spatial resolution. An additional benefit is the time required to obtain data is restricted to a couple of hours.

Using specially prepared Fe oxide paint (Rabenhorst and Burch 2006), the lower 50 cm of PVC panels or tubes (usually 60 cm in length) are painted. These are then inserted into the marsh soil for periods of 5 or 60 min. The FeS phase is unstable under oxidizing conditions, and the dark color fades over a period of minutes to hours when exposed to the air. Therefore, collection of the images (either by scanning or by photography) must be done quickly. Images are then compared with standard images prepared from painted PVC chips placed into Na_2S solutions (adjusted to pH 7.5) of known concentration (range: 3–300 mg/L S^{2-}) for set periods of time (usually 5 or 60 min.) For more detail, the reader is referred to Rabenhorst et al. (2010).

7.8 Oxidation-Reduction Processes in Soils

7.8.1 Overview

The formal definition of wetlands explicitly refers to the prevalence of "saturated soil conditions" (U.S. Army Corps of Engineers Environmental Laboratory 1987). These "saturated soil conditions" of wetlands are specifically known as hydric soils, which are defined as having "formed under conditions of saturation, flooding, or ponding long enough during the growing season to develop anaerobic conditions in the upper part" (Federal Register 1994). Organic materials in soils (mostly plant remains) are routinely decomposed by heterotrophic microorganisms as an energy source, and during this oxidation reaction where electrons are lost from the oxidized C, some other compound must serve as an electron acceptor and receive or "gain" the electrons (thus, being electrochemically reduced). When available in the soil solution, O_2 is usually the preferred electron acceptor. In saturated soils, however, dissolved O_2 can become depleted during microbial oxidation of organic

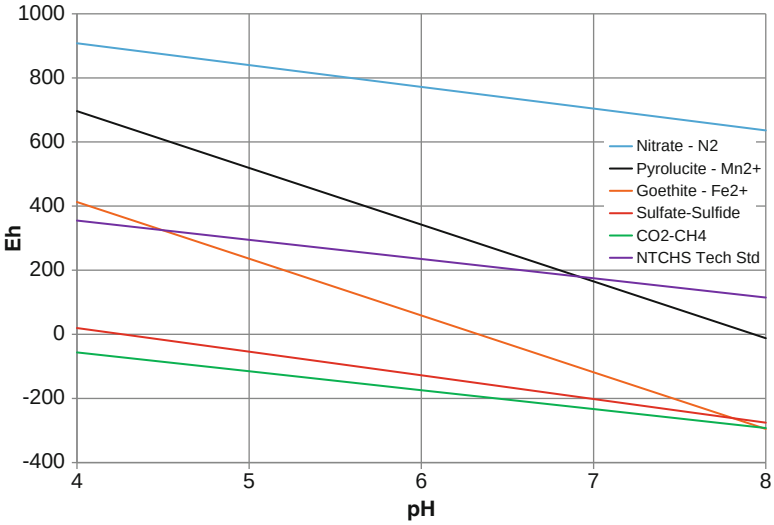

Fig. 7.2 Eh-pH stability phase diagram illustrating the redox conditions under which various common electron acceptors in soils become reduced

matter because the diffusion of O_2 into and through liquid water is quite slow. Such conditions where O_2 has become depleted are referred to as anaerobic.

In order for microorganisms to continue to oxidize organic materials under anaerobic conditions, they must use some alternate electron acceptor to O_2. Chemical thermodynamics determine which compounds that are common in soils can most easily accept electrons and be reduced. Those chemical ions and compounds readily found in soils that function as electron acceptors once O_2 is depleted are: NO_3^-; manganic manganese (Mn^{4+}); Fe^{3+}; sulfate (SO_4^{2-}); and CO_2. These are listed in order of their ease of being reduced which means that the soil environment must become increasingly reduced in order to proceed through this list of electron acceptors. This is sometimes shown in an Eh-pH stability diagram such as Fig. 7.2 where higher Eh values represent more oxidizing conditions and lower Eh represents more reducing conditions. Each line in Fig. 7.2 represents a "redox pair" and above the line, the oxidized form is stable and below the line, the reduced phase is stable. Thus, as the soil becomes progressively more reducing (lower Eh), various reduced phases would be predicted to be stable. First, NO_3^- would be expected to be reduced (function as an electron acceptor), and then Mn oxides (such as pyrolusite), Fe oxides (such as goethite), SO_4^{2-} and eventually (under highly reducing conditions) CO_2 can be reduced to methane.

Although this stepwise change or progression seems very systematic and orderly, soil systems are typically highly complex and also highly variable. Not all soils contain all these compounds. Also, the various proportions among these compounds can be very different among soils or ecological settings. Small scale variability in redox potential (Eh) in soils can be very great over short distances

(even millimeters). Nevertheless, if one can measure soil Eh and pH, graphs such as Fig. 7.2 can be useful in helping to predict which electrochemical phases might be stable and which phases might be expected to be reduced or oxidized. For such reasons, soil and wetland scientists are often interested in documenting or measuring the redox status or condition of a soil.

7.8.2 Techniques to Measure Eh/Assess Reducing Conditions

7.8.2.1 Direct Assessment: Pt Electrodes

One approach to documenting soil redox status is to measure the voltage generated between a Pt electrode and a reference electrode (such as a calomel or Ag-AgCl) inserted into the soil and then correcting the measured voltage for the difference between the reference electrode and the standard hydrogen electrode, which is reported as Eh (Bohn 1971; James and Bartlett 2000; Patrick et al. 1996). The Pt electrode is placed into the soil at the depth where one intends to measure the Eh; the reference electrode can be placed nearer the soil surface if that is more convenient. If measurements are to made at soil depths of more than a few cm, it may be useful or necessary to make a pilot hole with some instrument prior to inserting the Pt electrode to the desired depth. Best results are obtained when the soil is moist or wet. Otherwise, a salt bridge may be used to ensure good electrical contact between the reference electrode and the soil (Veneman and Pickering 1983). This may be particularly important if upper part of the soil is not wet at the time of measurement.

It is of particular importance to make the voltage measurement using a device with a high internal resistance (Rabenhorst et al. 2009). This can either be done by using a research grade voltmeter, or by using a low cost multimeter that has been modified to effectively increases the internal resistance to >200 Gohms (Rabenhorst 2009). When an unmodified multimeter (resistance approximately 10 Mohms) is used to measure the voltage, too large a current is permitted to flow during the measurement which alters the electrochemical environment in the vicinity of the Pt electrode causing substantial drift and an unreliable measurement. Also, because there is a great deal of small scale microsite variability in soils in the field, multiple (often as many as five or more) replicate measurements are made. If multiple measurements are to be made over the course of a field season, electrodes are sometimes left in place and then removed at the end of the field season and checked to be sure they are still functioning properly. Automated data loggers may also be programmed to collect repeated measurements. In other situations where it would be impractical to leave the electrodes installed in the field, they may be removed and reinstalled each time measurements are made.

7.8.2.2 Indirect Assessment: IRIS Tubes

During the last decade, another approach has been developed to assess reducing conditions in soils known as IRIS tubes (Indicator of Reduction In Soils) (Castenson and Rabenhorst 2006; Jenkinson 2002). In this approach, an Fe oxyhydroxide paint is applied to (usually 60 cm) sections of 1.25 cm schedule 40 PVC tubing and allowed to dry. After making pilot holes in the soil, the tubes are inserted into the soil for a period of approximately 1 month. If the soils are saturated and microbes are actively oxidizing soil organic matter, some of the Fe oxide paint functions as an electron acceptor, becomes reduced and solubilized, and stripped from the PVC tubing. The degree to which the paint is removed from the tubes is an indication of the degree or magnitude of reduction in the soil (Castenson and Rabenhorst 2006; Rabenhorst 2008). As expected, soil temperature has also been shown to affect the rate at which paint is removed from the tubes with less removal occurring during periods when soil temperatures are low and approaching biological zero (Rabenhorst 2005; Rabenhorst and Castenson 2005). Although the paint is sometimes referred to as being comprised mainly of ferrihydrite (Jenkinson 2002; Jenkinson and Franzmeier 2006), it is important in the synthesis of the Fe oxides to ensure that 40–60 % of the ferrihydrite has been converted to goethite, as otherwise, the paint will not adhere well to the PVC tubing (Rabenhorst and Burch 2006).

One of the perceived benefits of using IRIS tubes for assessing soil reduction is that it integrates the conditions of the soils over the period during which it is installed. So while Eh measurements or indicator dyes can tell you what is happening at the moment, the IRIS tubes provide a better indication of what the redox status of the soil has been over the course of several weeks. They also have the benefit of not requiring specialized lab equipment such as electrodes and volt meters. IRIS tubes are available commercially, and one possible limitation to their use might be the costs involved in purchase of the devices. However, the paint for making the tubes can also be manufactured fairly easily in a laboratory following published methods (Rabenhorst and Burch 2006).

Because the soil redox properties are considered to be quite variable, normal protocols for using IRIS tubes call for using five replicate tubes (Rabenhorst 2008). If a majority (three-fifth) of the tubes exhibits a minimum level of paint removal, the soil is considered to be reducing. To meet the requirement for reducing conditions that is specified in the Technical Standard of the NTCHS, at least 30 % of the paint must be substantially removed from a 15 cm zone, somewhere along the upper 30 cm of the tube (representing the upper 30 cm of the soil) (National Technical Committee for Hydric Soils 2007). However, other work has suggested that a less dramatic removal of the paint might still indicate reducing soil conditions (Castenson and Rabenhorst 2006; Rabenhorst 2008).

7.8.2.3 Indirect Assessment: Alpha-Alpha Dipyridyl Dye

Various dyes can be used as indicators of particular soil chemical conditions and alpha-alpha dipyridyl dye can be utilized for demonstrating the presence of Fe^{2+} in

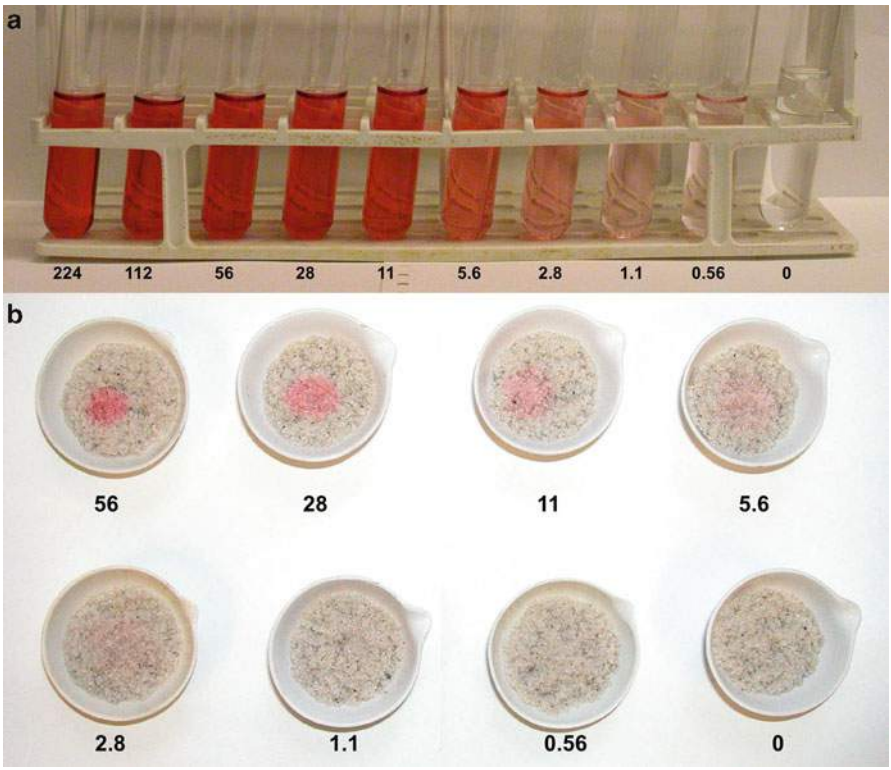

Fig. 7.3 *Pink color* developing in solutions and in saturated sand to which alpha, alpha dipyridyl dye has been added. Numbers represent concentrations of Fe^{2+} (μg/ml). In the *top figure* (**a**), 0.5 ml ααd dye was added to 6 ml of solution. In the *lower figure* (**b**), one drop of ααd dye was added to the saturated sand

the soil solution (Childs 1981). Except for some very unusual acid sulfate soils where oxidation of pyrite can (at least temporarily) form Fe^{2+} under oxidizing conditions, the presence of in soils usually indicates that the soil has been saturated and sufficiently reducing to cause Fe^{3+} oxides to be reduced to Fe^{2+} (Fig. 7.2). When alpha-alpha dipyridyl dye is applied to soils containing ferrous iron, a pink color develops, the intensity of which is function of how much Fe^{2+} is in solution. In clear solutions, as little as 0.5 μg/ml Fe^{2+} can be detected using the dye (Fig. 7.3a). However, when applied to soils, usually a minimum of 3–5 μg/ml is required for the pink color to be detected against the background soil color, and may require even higher concentrations to be seen if the soil is dark colored (Fig. 7.3b).

To test a soil using the dye, a core must usually be extracted and the dye is then applied to an interior broken face of the core. Care should be taken not to apply the dye to portions of the soil that have been in direct contact with a steel auger or spade as reduced iron from the tools has sometimes give a false positive reaction to the dye. Another way that the dye can be used is to extract some fresh soil solution from a saturated soil using a piezometer or suction lysimeter and adding a few drops of the alpha-alpha dipyridyl dye to a few ml of the extracted solution.

7.9 Additional Soil Characteristics

7.9.1 Soil pH

Soil pH refers to the pH of water in equilibrium with soil. pH is defined as the negative logarithm of the hydrogen ion activity in solution: $pH = -Log\ (H^+)$. Activity and concentration (moles/L) are nearly equivalent in dilute solutions, so that concentration is often used instead of activity. Soil pH impacts many of the chemical reactions in soil, determining the rate and or direction of a reaction. pH tends to move toward the neutral value as a soil becomes anaerobic (Mitsch and Gosselink 2000). Soil pH is routinely measured on a soil slurry (1:1 soil to water ratio) in the lab with commercially available pH meters and combination electrodes. The method described below was presented by Thomas (1996).

Ten g of air-dry soil is weighed out in a 50 ml beaker. After adding 10 ml of deionized water, the slurry is mixed well. Let stand for 10 min. After gently swirling the suspension the electrode is inserted. The sample may or may not be stirred during the readings. If stirred, the electrode is inserted into the suspension. If not stirred, the electrode is inserted either into the sedimented soil or into the supernatant above the soil. Readings taken in the supernatant are generally slightly higher than readings taken in the stirred suspension (Thomas 1996). All 3 approaches are acceptable; the important thing is that a consistent approach is used. The value should be recorded as pH_w. The electrodes should be rinsed with distilled water between readings.

Alternative methods for measuring soil pH include pH papers and test kits. Both are colorimetric methods; contact of the soil sample with the pH paper or, in the case of test kits, reagents causes a color change. That color is compared to a reference chart that equates pH values to a specific color. The primary limitations to these methods are a lack of accuracy and a limited pH range. For example, the Hellige-Truog soil pH test kit has long been used for routine soil investigations. It has a pH range of 4.0–8.5 and is accurate to within 0.5 pH units. Relatively inexpensive, rugged pH meters are now available for field work and are more appropriate for biogeochemical assessment of wetland soils.

7.9.2 Soil Oxygen Demand

Biochemical oxygen demand (BOD) is a measure of the quantity of O_2 used by microorganisms in the oxidation of organic matter. Eutrophication is the process where water bodies receive excess nutrients that stimulate excessive plant growth. Traditionally, BOD has been used to assess eutrophication in surface waters but a similar approach can be used to assess nutrient loading in wetlands. For wetlands that do not exhibit continuous inundation, soil O_2 demand (SOD) can provide an indication of nutrient loading. Soil oxygen demand can be easily measured in the lab using a soil incubation procedure and an O_2 electrode. A weighed wet soil

sample (5–10 g) is placed into a biological oxygen demand (BOD) bottle (250 ml) containing a stir bar. An additional soil sub-sample is weighed, dried, and re-weighed to determine water content. The bottle is placed on a magnetic stirrer and filled with deionized distilled water while stirring continuously. The contents are stirred for 15 min and dissolved O_2 (DO) is measured with an O_2 electrode. The bottle is sealed with a glass stopper and incubated in the dark for 24 h at 25 °C. Then, DO is again measured under continuous stirring. The analysis should be repeated with smaller sample size if DO levels decrease by more than 50 % (U.S. EPA 2008). Soil O_2 demand is calculated as follows:

$$\begin{aligned}\mathrm{SOD(mg/kg\text{-}hour)} = {} & [(\mathrm{DO_{t0}, mg/L}) - (\mathrm{DO_{t24}, mg/L})] \\ & \times (\text{water volume, L})/\text{dry weight of soil, kg}\end{aligned}$$

where: DO_{t0} and DO_{t24} = DO at time 0 and DO after the 24 h incubation period, respectively.

7.9.3 Bulk Density

7.9.3.1 Overview

To convert from SOM (or soil N, P, etc.) on a percentage basis to a weight basis, the bulk density (dry weight per unit volume) of the soil must be determined. This entails extracting a known volume of soil and determining the dry weight. Obtaining an accurate estimate of the volume of soil is the most difficult step in this exercise. Bulk density can be determined on intact soil cores obtained at known depth and volume (volume is determined from core diameter and depth), soil clods pulled from a pit (volume is determined by water displacement in the lab), or on unconsolidated materials obtained through excavation (volume obtained by filling the excavation hole with water). The soil is weighed after drying at 105 °C to a constant weight. Bulk density (g/cm^3) is calculated as "dry weight (g)/volume (cm^3)".

Depending on the intended use of the bulk density values, the contribution of rocks in the sample to sample weight and volume may or may not be accounted for. In most cases, the objective is to determine the bulk density of "fine earth" representing particles with diameters less than 2 mm, so coarser material is removed by screening. Mineral particles larger than very coarse sand (>2.0 mm) are considered rock fragments. If the bulk density value is to be used to extrapolate another parameter value to a soil volume or area basis, rocks are left in the sample. Otherwise, rocks in the sample should be accounted for by removing them, weighing them, and determining their total volume through water displacement. The rocks are placed in a graduated cylinder containing a known volume of water. The change in water volume after adding the rocks equals the volume of the rocks. The weight and volume of the rocks is then subtracted from the total sample values before calculating bulk density.

7.9.3.2 Excavation Method

The excavation method is most appropriate for sampling topsoil. Leaf litter is removed from the soil surface. The top of the excavation hole is leveled by removing soil with a trowel or knife and checking with a carpenter's level. Soil is removed with a trowel. The hole is lined with a single piece of Saran wrap or similar waterproof material. The hole is then filled with water from a graduated container. The excavated volume is determined by subtracting the final water volume in the container from the initial water volume. Another approach, which does not require Saran wrap, is to fill the hole with dried sand of a known bulk density. Sand is poured into the hole from a container with a known weight of sand. After filling the hole, the container of sand is weighed again. The volume (cm^3) of the soil cavity is calculated as "difference in sand weight (g)/sand bulk density (g/cm^3)".

7.9.3.3 Can Core Method

With a core method, a cylindrical metal sampler with a known volume is used to extract soil samples for bulk density determinations. Core samplers may be a hydraulically driven probe mounted on a vehicle (e.g., Giddings Probe), a double-cylinder, hammer-driven core sampler, or a simple metal can open at both ends. Use of the first two types of samplers is discussed by Blake and Hartge (1986). In this section, we discuss use of a small can. The can dimensions are not critical (except in calculations) but the diameter of the can should exceed the height of the can. Horizons should be sampled individually as they can vary significantly in bulk density. Tall cans make it more difficult to sample individual soil horizons. A representative size is 6.5-cm diameter and a 4.5-cm height. Three samples are collected from each horizon starting with the surface horizon and working down. At the sampling point, leaf litter is removed from the soil surface which is then smoothed to create a flat horizontal surface. The can is placed end down onto the soil surface and pressed into the soil by hand. This step can be facilitated by laying a board across the top of the can and tapping it lightly with a rubber mallet or hammer. However, care should be taken to avoid compacting the core. For horizons thinner than the height of the can, care must be taken to preclude simultaneously sampling multiple horizons. Dig out the inserted can plus a little of the surrounding soil and cut off the excess soil so the soil sample is flush with the top of the can. Empty the contents of the can into a sample bag. In each case, the thickness of the sampled horizon and the thickness of the core sample must be recorded. To sample subsurface horizons, dig down to the top of the target horizon and repeat the steps presented above.

7.9.3.4 Clod Method

The clod method is the most appropriate method for soil samples deep in the profile where it is difficult to determine volume *in situ*. In this method, clods are coated

with a water-repellent substance (e.g., Saran solution) and weighed twice, first in air and then in water. For a complete description of this procedure, the reader is referred to Blake and Hartge (1986). A major limitation to this approach is that typically the clods are extracted from a pit face which is not always feasible in wetland settings.

7.10 Sediment Gains and Losses

7.10.1 Overview

Because of ambiguity in terminology, it is important to preface this section with working definitions for the processes addressed. Accretion is the accumulation of sediment at a particular point; it could be sediment transported from an upslope point or sediment derived at the point itself. The accumulation of detrital material is commonly referred to as organic matter accretion, while sediment accretion refers to the accumulation of both organic and inorganic materials (U.S. EPA 2008). Deposition refers to sediment accumulation at a particular point due to sediment dropping out of flow when the amount of sediment in runoff exceeds the transport capacity of runoff. Erosion is the removal of sediment from a particular point. Subsidence is the decrease in the sediment surface elevation caused by an increase in the bulk density of sediment or a loss of sediment mass through oxidation of organic matter. Loss of sediment through erosion is not considered to be subsidence.

For the most part, the techniques presented in this section were developed for use in tidal marshes where significant sediment gains and losses are inherent to ecological functioning. However, with at most minor modification, they could be used in less hydrodynamic wetlands. The approaches inherent to all of the techniques presented here are based on either sampling and weighing sediment, or measuring the deviation in elevation relative to a fixed elevation point. All of the methods except for Cs137 techniques require multiple visits to the field sites.

7.10.2 Artificial Marker Horizons

Marker horizons provide an easy way to assess marsh accretion over a period of months to years (Cahoon and Turner 1989). White feldspar clay is thinly spread over the marsh surface in replicated plots creating an artificial horizon. Over time sediments accumulate above the artificial horizon through the process of accretion. The depth of sediment accumulation is determined by collecting a core from the sample plot and measuring the distance from the current marsh surface to the artificial horizon. This method is less expensive than isotopic techniques, the cores are simple to collect and process, and sampling success or failure is known at collection time (Cahoon and Turner 1989). White feldspar clay is the marker horizon

material of choice because it is conducive to submerged systems, and its bright white appearance is readily distinguishable from the surrounding sediment.

Typical plot size is 50 cm × 50 cm, with a layer thickness of 5 mm. Care should be taken during spreading the clay to ensure uniform horizon thickness. Plots are marked with pipes or rods sufficient in height to rise above the water column or vegetation to facilitate future sampling. Samples can be collected with a thin-walled core tube. Collected cores are refrigerated and taken to the laboratory in a vertical position. If processing is delayed, the cores are stored in the freezer. Melting can compromise core integrity, especially those cores high in organic materials. Therefore, the cores should remain frozen during processing. The frozen core is sectioned to determine the thickness of the material above the feldspar marker. The thickness of the sediment located above the feldspar marker is measured with calibrated calipers. Sectioning the core facilitates sampling for bulk density measurements and organic matter determinations.

A mass estimate of accretion can easily be determined by collecting a known volume of the accreted material and determining bulk density (see *Bulk Density* above). This estimate can be further refined into organic accretion and inorganic accretion by subjecting the sample to LOI assay (See *Soil Organic Matter: Loss on Ignition* above). Shallow compaction can be assessed by utilizing SET's data (See *Sedimentation-SETS* below) with accretion data collected from short-term accretion marker horizons as follows: compaction = sediment accretion − elevation change. For this approach, feldspar plots must be established before taking the SET baseline.

7.10.3 137Cs Dating

Marker integrity can be compromised by bioturbation, tidal action, resuspension, or erosion. Therefore, the reliability of the artificial marker horizon technique suffers in highly dynamic systems (DeLaune et al. 1989). An alternative approach to assessing long-term sediment accumulation is to use cesium-137 tracer methods (DeLaune et al. 1989; Milan et al. 1995). Production of 137Cs (half-life 30 year) resulted from aboveground thermonuclear weapons testing. Atmospheric deposition of 137Cs began in 1954, but peak fallout occurred in 1964 (Ritchie and McHenry 1990). Both dates are used to measure sediment accretion rates, but the 1954 sediment horizon is often difficult to discern (Ritchie and McHenry 1990), so the 1964 peak is most often used as the marker layer in wetland investigations (Reddy et al. 1993). In effect, the zone in the soil profile with the highest 137Cs level represents a "marker horizon", and all accretion above that zone occurred after 1964. The 137Cs signature may be compromised in areas with high erosion rates or where large amounts of sediment washed-in from other areas (Milan et al. 1995). In such situations, Milan et al. (1995) recommend that both 137Cs and 210Pb methods be used (see DeLaune et al. (1989)). The following 137Cs method was presented by DeLaune et al. (1989). Sampling sites are established along a

representative transect of the marsh. Sediment cores (15 cm in diameter, ≥50 cm in length) are taken along the transect with a thin wall cylinder, with care taken to prevent compaction of the cores. The retrieved core is sectioned, dried, and 137Cs activity is counted. Sediment accretion is measured by counting the 137Cs activity relative to distance down into the core. Bulk density, percent C, and percent organic matter should also be determined by depth in each core to allow for calculations relative to total mass, C, or organic matter.

7.10.4 SETS

Sediment erosion tables (SETS) measure net changes in sediment surface elevation relative to a benchmark (Cahoon et al. 2002a). A SET can be used to determine either the impact of a single meteorological event on sediment surface elevation or a long-term trend in elevation change. SETS consist of a supporting base pipe permanently placed at each site and a portable portion with four components: horizontal arm, vertical arm, flat table or plate, and pins. The base pipe, an aluminum pipe that is driven into the soil to the point of refusal, serves as a benchmark. The table provides a horizontal reference plane. The pins pass through the table at right angles until they touch the marsh surface. Marsh surface elevation relative to the benchmark is determined by measuring the length of pin above the table. Changes in the elevation of the marsh surface are indicated by changes in the distance between the marsh surface and the table. An integrated measure of elevation (accretion minus subsidence) can be calculated by combining data obtained from SETs with that provided by feldspar marker horizons.

SET stations should be located where the sediment surface is plainly visible and not obscured by vegetation or deep water. Data should be collected at times of lowest water level. Efforts should be made to minimize disturbance from foot traffic during SET installation and data collection. One option is to construct platforms with removable planks. The planks should be removed each time you exit the site. Data collection frequency should be based on expected sedimentation rates at the site. In areas with high sediment fluxes, a frequent collection schedule will produce a higher resolution of elevation changes over time. In areas with low sediment fluxes, a similar collection schedule may produce results within the range of methodological error so that actual changes in elevation will not be detected. The conservative approach is for each practitioner to construct a data collection schedule based on preliminary work onsite. For further details on SETS, including design specifics, the reader is referred to Cahoon et al. (2002a, b) and Callaway et al. (2001).

7.10.5 Sediment Collection Tiles

Sediment collection tiles provide a simple and inexpensive approach to assessing deposition in both tidal and non-tidal wetlands. Sediment is deposited on the upper

surface of the tile; the sediment is collected, dried, and weighed to produce a mass estimate. Ceramic tiles produced for home use (e.g., glazed kitchen tiles) are appropriate. Ceramic tiles placed on the soil surface will suffice for wetlands subject to low energy flooding. For high energy floodplains or tidal marshes, the sediment collection tile should be anchored to the soil. For sites that are subject to relatively deep flooding such as tidal marshes, tiles may be placed at several elevations relative to the soil surface to determine sediment availability within the water column. This is particularly important if the goal is to separate re-suspended sediments (defined as the amount of sediments lifted 6 cm or more above the sediment surface) from unsuspended, shifting surface material. Tiles may be easily positioned at specific heights relative to the soil surface by attaching PVC pipe to the base of the tile and inserting the pipe into the soil. Tiles are retrieved when the soil surface is exposed to air, usually at 2 week or 1 month intervals. Plant tissue is first removed from each tile. Sediment is scraped and rinsed off the tiles and into sample cups using a rubber scraper and deionized water. Samples are returned to the lab, filtered through pre-weighed paper filters, dried and weighed. Sediment deposition is commonly reported as g dry weight/m^2. Organic matter is an important constituent of sediment in many depositional settings; its content in a sediment sample can be determined by measuring weight loss in subsamples after burning at selected temperatures (see *Soil Organic Matter: Loss on Ignition* above).

The following approach was presented by Pasternack and Brush (1998) and at http://pasternack.ucdavis.edu/sedtiles.htm. The major advantages of this sturdier design is that it lends itself to high energy tidal marshes and can be used to assess erosion as well as accretion. It utilizes an aluminum pipe (1 m long $\times$ 2 cm dia.) 'anchor' sunk into the ground and capped with a detachable ceramic tile flush with the marsh surface. A plastic tube is attached to the bottom of each tile; the tube drops over the anchor pipe. This provides a stable tile surface that is not susceptible to motion unless subjected to extreme lift forces. Therefore, it is conducive to high energy tidal marshes. They constructed the tile and tube assembly by attaching (with plastic cement) a 7.5 cm long hard plastic pipe with an inner diameter of 2 cm to the center of a plastic square with plastic cement. Schedule 77 plastic is appropriate for the pipe and the square. The other side of the plastic square was attached to the bottom of a 20 cm $\times$ 20 cm glazed-ceramic tile with silicone glue.

Each sampling point should be marked by GPS. Even so, it may be difficult to find the anchor in subsequent visits. One approach is to mark each anchor with a second anchor as follows: Lay a meter stick on the ground with one end on the anchor and the other pointing to a pre-determined compass direction (e.g., due east) and hammer a second anchor ~0.5 m down into the ground at the east end of the meter stick. Flag the top of this second anchor (marker anchor) with a long strip of flagging attached with duct tape. Attach the tile assembly over the sunken auger by placing the plastic pipe over the metal pipe. The metal pipe may need to be filed to accommodate the plastic pipe.

The most appropriate sampling interval for sediment tiles depends on the hydrodynamics and sediment loading rate of the system in question. Extra care is needed when sampling very small or very large amounts of deposition. Pasternack

and Brush (1998) recommend the following sampling intervals: tidal freshwater wetland, 2–4 weeks; salt marsh, 4–8 weeks; floodplain, varies with flood regime. These should be taken as initial guidelines and adjusted as needed with experience.

Erosion monitoring requires that the height of each side of the tile above the marsh surface be determined at installation. Also during installation attempts should be made to keep the tile level with the bottom of the tile just touching the marsh surface. If not, any deviations should be recorded. If the tile was flat on the ground upon installation, erosion depth is determined by measuring the height of each tile side above the marsh surface and averaging the measurements. If the tile was not flat, measurements need to be adjusted for the known deviations. A mass estimate of erosion can be determined by multiplying erosion depth by sediment bulk density (see *Bulk Density* above).

7.11 Biogeochemical Indicators for Evaluating Wetland Condition

Many of the biogeochemical assays are expensive, require a high level of technical expertise, and require multiple site visits. Therefore, a number of wetland scientists have evaluated a surrogate approach in which readily measureable indicators of a biogeochemical process or pool are identified and used in lieu of the more complex assay. The surrogate approach may address an individual wetland service, the impact of a specific type of anthropogenic disturbance, or overall wetland condition. A major benefit of this approach is that since the required information can be collected in one visit, a greater number of wetlands can be assessed.

As an example of the surrogate approach to individual wetland services, researchers at the University of Delaware developed an assessment of denitrification capacity for slope wetlands based on soil morphology. In this procedure, wetlands are classified as having high, medium, or low denitrification capacity on the presence of specific hydric soil indicators. Field Indicators of Hydric Soils (USDA, NRCS 2010) are a series of individual soil morphology characteristics that were developed to identify hydric soils in the field. The majority of the indicators are visual and require only a ruler, a shovel, and a Munsell Color Chart. Differences between indicators reflect in part differences in hydroperiods which impact denitrification potential.

There is presently a national wetland monitoring effort based on the use of indicators to assess the impact of a specific type of anthropogenic disturbance or overall wetland condition. Natural and anthropogenic disturbances that negatively impact the functional capacity of a wetland are called “stressors”. Widespread anthropogenic disturbances include artificial drainage (e.g., ditches), timber harvesting, and nutrient loading. Commonly, wetland condition is assessed as the degree of deviation from the undisturbed condition. This requires the collection of

data from reference sites to identify baseline levels for comparison. Ideally, the reference wetland would be pristine (i.e., un-impacted by anthropogenic activities). However, in many regions such as the mid-Atlantic region of the U.S., it is difficult to find undisturbed wetlands for each class so that minimally impacted sites are used as reference sites. Reference sites should represent the same class of wetlands as the wetlands in question. For example, a mineral soil flat would be compared to a mineral soil flat reference site.

The U.S. EPA recognizes three levels of indicators for wetland monitoring – Levels I, II, and III (U.S. EPA 2008). Level I indicators are used for routine wetland monitoring for condition or impacts. Level II and III indicators are often used to test or support the validity of Level I indicators. The classes are based on the ease of measurement and sensitivity to respond to change as follows:

1. Level I: easily measureable, low cost, less sensitive to stress, long response time, low spatial variability
2. Level II: intermediate in complexity and sensitivity, intermediate response time, moderate spatial variability
3. Level III: highly complex and sensitive, short response time, high spatial variability

Because of the recognized importance of wetlands to water quality, one focus of these monitoring efforts has been to develop methods to evaluate nutrient enrichment of wetlands, which is one of the primary stressors to wetlands in many parts of the country. Eutrophication is commonly considered to be the enrichment of bodies of fresh water by inorganic plant nutrients (e.g., NO_3^-, PO_4^{2-}). This may result in increased primary productivity and a subsequent impact to water quality. Eutrophication of wetlands is usually attributed to nutrient loading, increased external inputs of nutrients from point and non-point sources. However, eutrophication can also result from an increase in nutrient cycling within the wetland itself without an increase in external inputs. For example, artificial drainage or increased development in the surrounding watershed will impact hydrologic characteristics of the wetland. These changes will impact SOM decomposition rates, soil mineralization rates, and denitrification rates. Destruction of vegetation in the wetland will diminish a primary sink for N and P resulting in an increase in those nutrients in the water column.

Not all of the monitoring indicators have a biogeochemical basis. However, because of the interdependence of biogeochemical cycles and the nutrient status of soil and the water column, biogeochemical characteristics are well suited to serve as indicators of wetland condition, especially with respect to nutrient enrichment. Table 7.2 presents some examples of potential biogeochemical indicators to assess nutrient impacts to wetlands. Additional information on using indicators to assess wetland condition can be found in the following references:

Reddy and DeLaune (2008), U.S. EPA (2008),
http://www.epa.gov/waterscience/criteria/wetlands/, and
http://www.epa.gov/owow/wetlands/bawwg/publicat.html.

Table 7.2 Examples of potential biogeochemical indicators to assess nutrient impacts to wetlands (Reddy and DeLaune 2008)

Level I indicators	Level II indicators	Level III indicators
Water column		
Dissolved O_2	Dissolved organic C	Microbial diversity
pH	Enzyme assays	Cellular fatty acids
Temperature	UV absorbance	
Turbidity		
Detritus and soil		
Soil bulk density	Soil O_2 demand	Microbial diversity
Soil Eh	Cation exchange capacity	Substrate-induced respiration
Total N	Soil respiration	Cellular fatty acids
Soil texture	Denitrification potential	Organic matter accretion rates

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Student Exercises

Classroom Exercises

Classroom Exercise 1: Understanding pH

The term pH is derived from the French term for hydrogen power, *pouvoir hydrog'ne*. In chemistry, pH is a measure of the acidity or basicity of an aqueous solution. Pure water is considered to be neutral, with a pH close to 7.0 at 25 °C. Solutions with a pH less than 7 are considered to be acidic; solutions with a pH greater than 7 are considered to be basic or alkaline. From a computational standpoint, pH is the negative logarithm (base 10) of the *active* hydrogen ion (H^+) concentration (in moles/L), or $-\log [H^+]$. pH does not precisely reflect H^+ concentration, but incorporates an activity factor to represent the tendency of hydrogen ions to interact with other components of the solution. For purposes of this exercise, consider the active hydrogen ion concentration to be equal to the hydrogen ion concentration. Because a pH value represents the negative log of a concentration, a pH of 4 corresponds to 10^{-4} mol/L, which corresponds to 0.0001 mol/L. What is the

Table 7.3 Ortho-P control chart

OP standards concentrations (mg/L)			
0.005	0.020	0.050	0.100
% transmittance			
99.1	96.8	91.8	84.8
99.2	96.7	91.6	84.6
99.2	96.8	92.6	84.3
99.3	96.2	91.0	84.0
99.8	96.8	92.6	86.9
98.9	96.2	92.8	85.4
98.8	96.6	91.8	84.8
Calculated OP concentrations (mg/L)			
0.005636	0.020227	0.053180	0.102463
0.005009	0.020869	0.054535	0.103930
0.005009	0.020227	0.047789	0.106138
0.004383	0.024090	0.058618	0.108353
0.001262	0.020227	0.047789	0.087264
0.006891	0.024090	0.046448	0.098082
0.007520	0.021512	0.053180	0.102463
Mean			
SD			
CV			
RSD			
MDL			

pH of a 0.1 M hydrochloric acid solution? What is the molar concentration of H^+ in a hydrochloric acid solution at pH 3? Given a 1 L container of pure water, how many hydrogen ions are present?

Classroom Exercise 2: Calculating Detection Limits

An analytical laboratory ran quality control charts to determine the method detection limit for their ortho phosphorus test method. They used the ascorbic acid method and read the color change on a spectrophotometer. The lab assistant made four standard solutions containing 0.005, 0.020, 0.050, and 0.100 mg/L PO_4^{-3}-P. The results are given in Table 7.3. The standard curve was determined before the control samples were run and the regression equation is as follows: $LN(Y) = -1.6094X + 4.6052$, where Y represents the % transmittance values and X represents the concentration. The curve was used to determine the associated concentration of the control sample results.

Calculate the mean (x), standard deviation (SD), coefficient of variation (CV), relative standard deviation (RSD) and the MDL for each of the control standard

Table 7.4 DBH data and allometric parameters for selected trees

	Dbh (cm) of individuals						Allometric parameters	
Species	1	2	3	4	5	6	a	b
Fagus grandifolia	43	28	33	27	26	31	0.0842	2.5715
Carya laciniosa	24	26	18				0.0792	2.6349
Acer rubrum	15	23	28	25	19		0.0910	2.5080
Quercus rubra	36	31	25	22			0.1130	2.4572
Quercus alba	19	36	34	29			0.0579	2.6887

concentrations. The standard deviation from a single standard concentration can be used to determine the MDL. However, the standard deviation is sometimes estimated from measurements made at a minimum of three levels (low, mid-range and high range) of standards. The value for the method standard deviation is calculated by plotting the standard deviation vs. concentration for the different concentrations. The method standard deviation is extrapolated from the curve, the value of the standard deviation as the concentration goes to zero. Plot the calculated standard deviations of the control standards versus concentration and determine the new MDL from the extrapolated SD. See *Quality Control and Detection Limits* for the required equations.

Classroom Exercise 3: Estimating Tree Biomass Using Allometric Equations

Tree biomass can be calculated using the allometric equation, $M = aD^b$, where M is the oven-dry weight of the biomass component of a tree (kg), D is diameter at breast height (DBH) (cm), and a and b are parameters unique to each species. The allometric parameters can be derived experimentally or selected from the literature.

A field investigation of a small wetland revealed 22 trees representing five species. Diameter breast height for each individual and allometric parameters for each species are presented in Table 7.4. The allometric parameters are for total aboveground biomass (Brenneman et al. 1978). Based on the data in Table 7.4, determine the total aboveground biomass for the tree stratum in the plot.

Reference

Brenneman BB, Frederick DJ, Gardner WE, Schoenhofen LH, Marsh PL (1978) Biomass of species and stands of West Virginia hardwoods. In: Pope PE (ed) Proceedings of Central Hardwood Forest Conference II. Purdue University, West LaFayette, pp 159–178

Laboratory Exercises

Laboratory Exercise 1: Measurement of Soil Respiration

Introductory Comments

Laboratory investigations of soil respiration generally monitor carbon dioxide production in either gas-tight, static microcosms or by using a dynamic, flowing-gas system (Zibilske 1994). The former approach is simpler and recommended except when experimental considerations dictate use of a dynamic system. Static systems make use of microcosms containing soil and possessing sufficient headspace to permit gas accumulation and simultaneously avoid the development of excessively low oxygen levels if maintenance of aerobic conditions is desired. If anaerobic conditions are to be investigated, the soil can be flooded with water and the headspace purged with N_2 or another suitable gas prior to incubation. Carbon dioxide produced by aerobic or anaerobic respiration, or by certain anaerobic fermentative processes, either is captured in alkali traps for measurement by acid-base titration or is measured by collecting gas samples for analysis by gas chromatography. The choice between the 2 CO_2 measurement methods is a matter of preference and instrument availability, as both provide good sensitivity. The reader is directed to Zibilske (1994) for details and a complete discussion of the various options.

Containers to be used as microcosms must be air-tight. Canning jars work well, and their lids can be equipped with septa for gas sampling if CO_2 is to be measured by gas chromatography. The nature of the soil placed in the microcosms may range from mixed, sieved soil to intact soil cores. In any case, the ratio of headspace volume-to-soil mass is an important consideration in aerobic studies; a ratio of 10:1 is recommended (e.g., 500 ml container for 50 g of soil), although lower ratios can be used, provided the jars are opened more frequently to allow for aeration. Regardless of the ratio used, it is important in aerobic studies that the oxygen levels of the microcosm be replenished on a regular basis. Presence of volatile fatty acids or reduced S-containing gasses in the headspace (e.g., a rotten egg smell) is an indication that the microcosms need to be opened more frequently. Mixing a bulking agent such as vermiculite or perlite with the soil can help prevent the development of anaerobic microsites (Thien and Graveel 2003). Additionally, for aerobic studies the moisture content of the soil should not exceed field capacity, and it may be desirable especially with fine-textured soils to use a lower moisture content that is still conducive to microbial activity. Regardless of the moisture content chosen, it should be standardized across treatments unless moisture content is being examined experimentally. Another variable affecting aeration status is the rate of any organic matter addition to the soil. A commonly used rate is 1 % by mass, although this will vary with the experimental objectives.

There are additional considerations when alkali traps are being used to collect CO_2. Care must be taken that excess alkali is used in the traps so that the CO_2-trapping ability is not exhausted between samplings. A reasonably conservative starting point when working with soil amended with organic matter at a 1 % rate is the equivalent of 15 ml of 1.5 M NaOH per 50 g of soil (Thien and Graveel 2003). It is absolutely essential that the alkali used in the traps and the acid used for titration be accurately standardized; the use of potassium hydrogen phthalate or a commercially available primary standard is strongly recommended. The concentration of NaOH used in the traps should be determined each time they are replenished. Additionally, it is critical that microcosm blanks (i.e., microcosms containing alkali traps only) be employed to account for any CO_2 in the headspace originally or added when the microcosms are opened for aeration; the CO_2 measured in the blanks should be subtracted from the values obtained from the microcosms containing soil.

Objective: To monitor soil respiration under laboratory conditions using acid-base titration to measure CO_2 evolution.

Materials and Equipment Needed

500-ml soil microcosms (16-oz canning jars with air-tight lids)
Balance
1.5 M NaOH, standardized
Vials for alkali traps (plastic, 50-ml centrifuge tube with air-tight cap)
1.0 M HCl, standardized
1.0 M $BaCl_2$
Phenolphthalein indicator solution
Burets
125-ml Erlenmeyer flasks

Procedures

Overview: The general approach outlined here is suggested for its simplicity and adaptability to a wide range of experimental objectives and is based on the procedure described by Thien and Graveel (2003) for use in instructional soil science laboratories.

1. Weigh 50 g of soil into each microcosm. Adjust moisture content and add a bulking agent and or organic amendment if desired. For anaerobic studies, the soil can be flooded with water at this time.
2. Dispense a precisely measured volume (~15 ml) of standardized NaOH into vials to be used as alkali traps. Place uncovered alkali traps into the microcosms

taking care not to spill the NaOH. Also, prepare microcosm blanks containing alkali traps only.

3. Securely attach the covers of the microcosms, first purging the headspace with N_2 gas for anaerobic treatments. Place the microcosms under the desired incubation conditions.
4. Collect samples for analysis at time intervals dictated by the experimental objectives, typically sampling more frequently early in the study in order to obtain an estimate of CO_2 evolution rate and thereby avoid exhausting the alkali traps by too infrequent sampling. Retrieve the alkali traps from the microcosms and attach air-tight covers to the vials to stop CO_2 collection.
5. After a brief period of time to allow for aeration, place fresh uncovered alkali traps in the microcosms. As noted previously, the NaOH used in the traps should be newly standardized. Replace microcosm lids.
6. For analysis of alkali traps, the solution in a trap is transferred quantitatively, using water rinses, to a titration flask; this procedure and the titration itself should be performed on one sample at a time to minimize spurious CO_2 collection. Add 25 ml of 1 M $BaCl_2$ to the flask, the contents of which should form a white precipitate of $BaCO_3$. A few drops of phenolphthalein indicator are added and should result in a pink coloration, indicating that the NaOH originally added to the trap was not exhausted during the soil incubation (i.e., excess NaOH remains). This excess NaOH is titrated from a pink suspension to a milky white endpoint using standardized 1.0 M HCl.
7. Carbon dioxide evolution is calculated as mg CO_2 = (meq base – meq acid) (22), where meq base is the milliequivalents of NaOH originally present in the traps, meq acid is the milliequivalents of HCl required to titrate the excess NaOH, and 22 is the milliequivalent weight of CO_2. Milliequivalents of acid or base are calculated as the product of concentration expressed as normality and volume expressed as ml [e.g., meq acid = (concentration of HCl as normality used in titration)] (ml of HCl used in titration). For NaOH and HCl, molarity concentration is equal to normality concentration (e.g., 1.5 M NaOH = 1.5 N NaOH).
8. Commonly calculated parameters in respiration studies are cumulative CO_2 evolution with time and mean rate of CO_2 evolution between sampling times. Depending on experimental objectives, it may be desirable to convert values from a CO_2 basis to an elemental carbon basis.

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Laboratory Exercise 2: Documenting Redox Conditions in Soil Mesocosms Using Pt Electrodes and Alpha-Alpha Dipyridyl Dye

Introduction and Background

When soil pores are filled with water and the soil becomes saturated, the movement of oxygen into the soil is greatly inhibited by the slow diffusion of gasses through liquids. If oxidizable carbon compounds are present and temperatures are sufficiently warm, aerobic heterotrophic microorganisms will begin to oxidize the organic matter using oxygen as the electron acceptor. Once the oxygen has been depleted, various anaerobic microbes will begin to use alternate compounds as electron acceptors. Some of the important and common compounds that serve as electron acceptors in saturated soils and thus can be reduced, include nitrate NO_3^- to nitrite NO_2^- (or eventually to dinitrogen N_2 gas), solid phase manganese oxides MnO_2 to soluble Mn^{2+}, solid phase iron oxyhydroxides FeOOH to soluble Fe^{2+}, and sulfate $SO4^=$ to sulfide $S^=$. When water is removed from the saturated soil through drainage or evapotranspiration, the reduced species can become oxidized. This reoxidation is often microbially mediated, but some reactions can occur chemically.

The redox potential, or Eh, can be measured using Pt electrodes with a reference electrode and the Eh, together with the pH, can be used to determine how reducing or oxidizing a soil is in order to predict whether particular compounds would be expected to be reduced or oxidized. This is done by comparing the measured Eh and pH values to a line calculated using the Ksp for particular compounds and phases based upon thermodynamic data. There are a number of variables which can affect the precise calculation of the line (such as the concentration of soluble components). Diagrams showing the redox stability lines (fields) for many compounds have been created and published.

Objectives

1. To make Eh measurements in a soil mesocosm
2. To interpret soil redox conditions by using Eh and pH measurements and redox stability diagrams
3. To use, and interpret the use of, alpha-alpha dipyridyl dye
4. To compare the Eh-pH data collected with the alpha-alpha dipyridyl dye reactions observed

Materials and Equipment Needed

Soil Mesocosm 7.5 cm diam by 40 cm high[1]
Six, 40 cm Pt wire electrodes

[1] A 7.5 cm schedule 40 PVC pipe 50 cm in length should be sharpened on one end (bevel out) so that the pipe can be driven into the soil to a depth of 40 cm and then excavated to collect the mesocosm.

One reference electrode (with salt bridge)
One, high impedance voltmeter
pH meter and buffers
Light's Solution
Bucket
Duct tape
alpha-alpha dipyridyl dye[2]

Procedures

1. *Selection and testing of electrodes*. Electrodes must be tested to ensure they are working properly. Each group of students will require 6 Pt electrodes. Electrodes can be made at fairly low expense if done according to the procedure described by Owens et al. (2005). To test the Pt electrodes, place a group of electrodes into a beaker containing Light's solution (Light 1972), which contains Fe(II) and Fe(III) in a sulfuric acid solution (be careful). This solution is poised and stable. Electrodes should be tested in groups using the same reference electrode, and all individual electrodes within the group should be within 1–2 mV the mean. Typically, they should read around Eh = 675 (raw voltage reading of about 430 if using a calomel reference). However, it is most important that the electrode readings group together and sometimes values for the Light's solution can drift. Groups of calomel reference electrodes can be similarly tested, although they typically vary a bit more.
2. *Selection of mesocosms*. Each group of (2–4) students will work with one mesocosm. See footnote #1 regarding collection of the mesocosm. A piece of geotextile fabric should be attached across the bottom of the mesocosm using duct tape, in order to help keep the soil in the mesocosm (Fig. 7.4). Care should be taken to give support to the soil within the mesocosm so that it does not accidentally slide out.

   The depth from the top of the mesocosm to the top of the soil should be measured and then marked on the outside of the core. After this, mark the core at the 2 depths at which the Pt electrodes will be later installed. These depths will be at 10 and 25 cm below the top of the soil (NOT below the top of the PVC cylinder) as shown in Fig. 7.4.
3. *Instrumentation of the mesocosms*. An overview of the instrumentation is shown in Fig. 7.5 which illustrates how the mesocosm will appear when all the electrodes have been installed. Probably, the best order in which to install the electrodes is the following: (1) the deep (25 cm) Pt electrodes, (2) the salt bridge and calomel electrode; and (3) the shallow (10 cm) Pt electrodes.

[2] This should be made according to the procedure of Childs (1981).

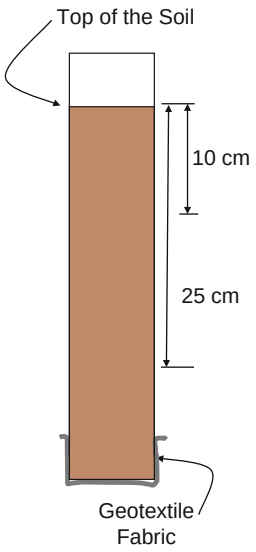

Fig. 7.4 Schematic of collected mesocosm with approximately 40 cm of soil within a 7.5 cm diameter, schedule 40 PVC pipe, 50 cm in length. A piece of geotextile fabric is taped to the bottom to prevent the soil from sliding out. Marks should be placed on the outside of the PVC at depths of 10 and 25 cm below the soil surface

4. *Installing the deep electrodes*. Note that the three deep Pt electrodes will be placed in an equally spaced arrangement around the mesocosm, optimizing the distance between the edge of the mesocosm and the future location of the salt bridge (see Fig. 7.5). Using colored tape, place a mark on each electrode that is 25 cm above the Pt point. Then, using a narrow sharpened stainless steel rod (slightly larger than the electrode), carefully make a VERTICAL pilot hole that is approximately ½ cm shallower than the depth where you will place the Pt electrode. Remove the rod and carefully slide a straight Pt electrode into the hole, and push the electrode to the exact depth where measurements are to be made (25 cm). Carefully install two additional deep electrodes in the same manner being sure to distribute them around the mesocosm (Fig. 7.5). Label these electrodes 25A, 25B, and 25C (for 25 cm below the soil surface).
5. *Installing the salt bridge*. The salt bridge is made from a piece of PVC pipe that has an OD of approximately 22 mm. It is filled with an agar which is saturated with KCl (similar to the calomel electrode itself) (Veneman and Pickering 1983). Prepare to use the salt bridge by inserting your previously selected (and tested) calomel reference electrode into the agar at the top of the salt bridge until the electrode is approximately 2–3 cm into the tube and the agar has encompassed the end of the electrode. Wipe any agar or salt from the electrode and PVC tube and carefully wrap with parafilm to produce a water tight seal that will hold the electrode in place. This will help to keep the electrode from drying out over time.

   A pilot hole approximately 5–10 cm deep should be made using a section of sharpened 2.5 cm dia. pipe, and the soil should be removed to make room for

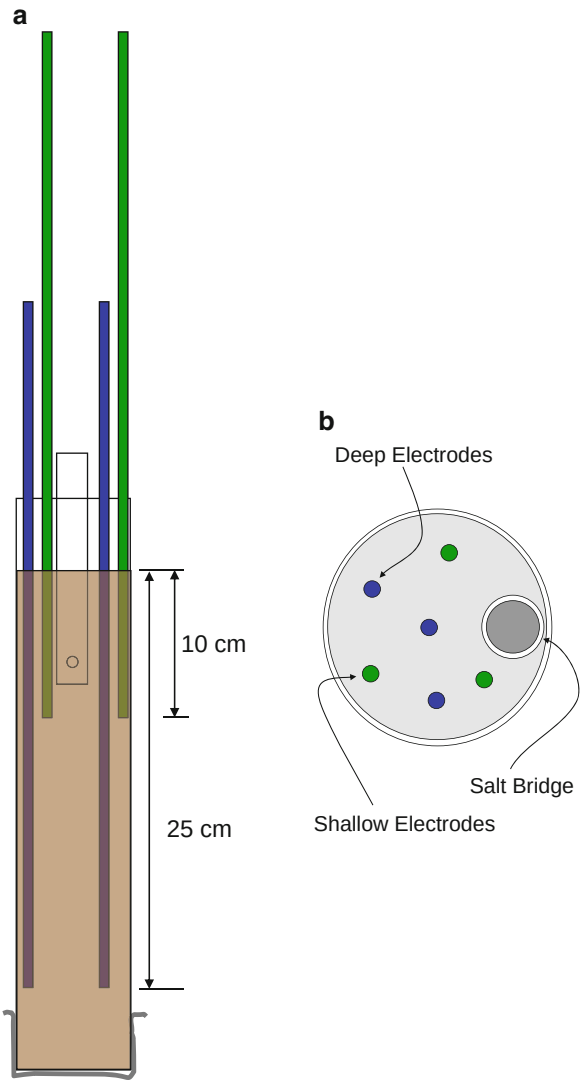

Fig. 7.5 Diagrams showing *side view* (**a**) and *top view* (**b**) of mesocosm indicating the spatial placement of the salt bridge and the Pt electrodes

the salt bridge. Then insert the salt bridge firmly into the pilot hole, being careful that the agar does not slide out of the tube. The salt bridge should extend about 5–10 cm above the top of the soil.

6. *Installing the shallow Pt electrodes*. Mark each of the other three electrodes at a depth of 10 cm using colored tape. The three shallow Pt electrodes should be placed in an equally spaced arrangement around the mesocosm, much like the deep electrodes and following a similar installation procedure, but being careful not to disturb the other Pt electrodes or the salt bridge and calomel

electrode (Fig. 7.5). Electrodes should be installed to a depth of 10 cm below the soil surface (which will be 15 cm above the deep electrodes) and should be labeled as electrodes 10A, 10B, and 10C.

7. *Taking initial Eh measurements (prior to saturation)*. Using either a lab grade Eh meter or a multimeter in conjunction with a device to create a high resistance circuit (Rabenhorst et al. 2009; Rabenhorst 2009), the voltage should be measured in the circuit created between the calomel reference electrode and each Pt electrode. The positive (red) wire should attach to the Pt electrode and the black wire should connect to the reference electrode. When the electrodes have been recently installed, there may be some slight drift during the measurement, but this drift should become less apparent on subsequent days. Typically, these measurements are recorded to the nearest 0.001 V (note there is too much variability to warrant recording with any greater precision than this so make sure you are NOT reading to tenths of a mV). Commonly, students will occasionally reverse the wires on the volt meter when making measurements. This will result in VERY LARGE ERRORS, because the voltage will have the opposite sign (−300 mV vs. 300 mV). Be very careful to ensure that the Pt electrode leads to the red (+) pole on the volt meter and the reference electrode is connected to the black (−) pole. Your initial readings will probably be somewhere in the range of 200–400 mV (before correction for the reference electrode). Over time, the voltages will likely become lower (especially for the deep electrodes installed below the water table.) Note that pH measurements must also be obtained at the same 2 depths where the electrodes will be placed. It is recommended that soil pH at 10 and 25 cm be collected from a replicate mesocosm so that the instrumented mesocosm does not need to damaged.
8. *Saturating the mesocosm*. Mesocosms will be saturated to a depth of 15 cm below the soil surface. In order to saturate each mesocosm, stand the mesocosm vertically in a container (bucket) where the water can be adjusted to the proper height, and secure it using duct tape as shown in Fig. 7.6. Distilled water should then be added slowly to the bucket which will result in filling the soil pore space in the mesocosm from below, which should help minimize the entrapment of air during saturation. The water level should be raised until it is at the appropriate height (Fig. 7.7). An alternate arrangement is also shown in Fig. 7.7 where the mesocosm is saturated to the soil surface. This can be used to illustrate differences in properties of contrasting soil horizons (such as OM content in A vs. B horizons) if different soil materials occur at 10 vs. 25 cm.
9. *Eh measurements after saturation*. Approximately 30–60 min after saturation, collect the second set of Eh measurements from the mesocosm as described previously. After this, Eh measurements should be made on the mesocosms daily for the first week, and then every other day through subsequent weeks. This should be continued for a minimum of 2 weeks and may produce better results if extended for 3 or 4 weeks. The level of the water in the bucket will need to be checked and maintained at the proper height.

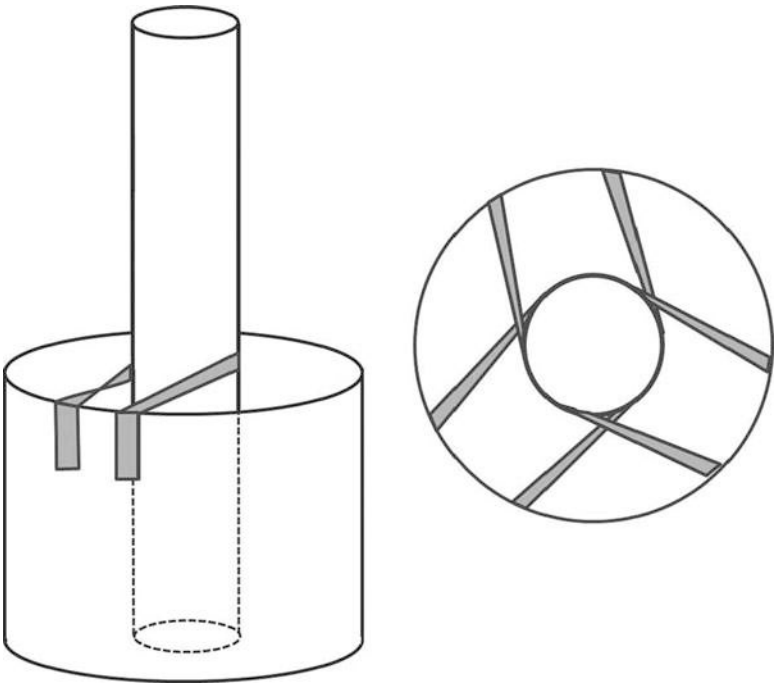

Fig. 7.6 Each mesocosm should be secured in a small bucket using three pieces of duct tape which wrap around the mesocosm and are affixed at approximately 120° from each other

10. *Disassembly of the mesocosms*. At the end of 2 (or more) weeks, remove the mesocosm from the water reservoir and place it on some absorbent material (newspaper) and allow it drain. Empty the water from the bucket into appropriate containers in the lab (do not empty soil down the sinks!)

    Carefully remove each of the Pt electrodes trying not to disturb the soil too much. Rinse off the Pt electrodes and set aside in a group being careful to keep them together and labeled with your mesocosm number. Check each electrode by placing them in the Light's solution and determining the voltage measured using a common calomel reference. This should be done to determine if any appear to be malfunctioning (in which case, data from those electrodes should not be included in the analysis).

    Remove the calomel reference electrode from the salt bridge, rinse off the electrode and make sure it is adequately filled with saturated KCl before storing. Using a rod, pole, or other device, extrude the soil from the PVC sleeve onto newspaper, trying to keep the core as intact as possible.
11. *Testing for Ferrous Fe using alpha-alpha dipyridyl (ααd) dye (or strips)*.[3] Using a knife or spatula, split open the extruded core lengthwise, trying mainly

[3] Alpha-alpha dipyridyl strips are available at http://www.ctlscientific.com/cgi/display.cgi?item_num=90725

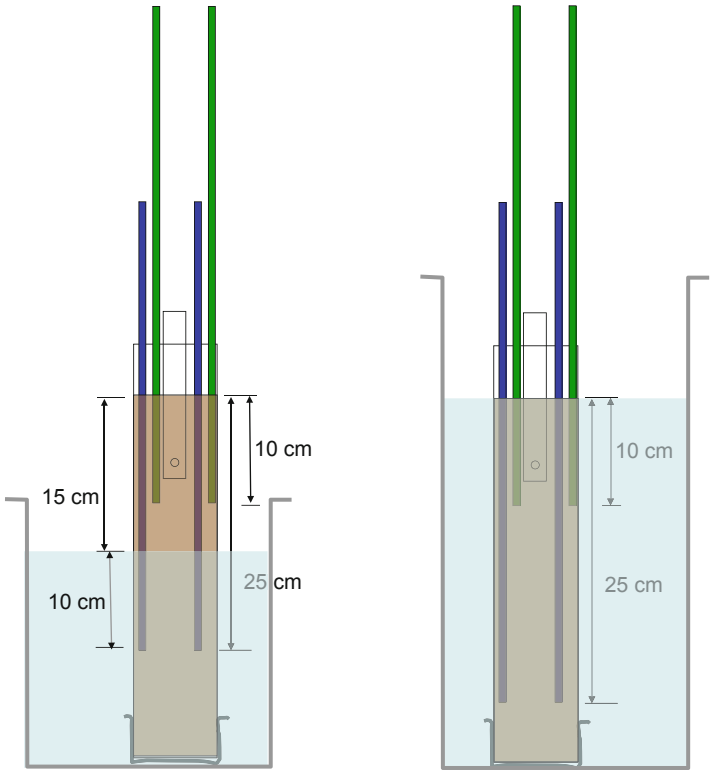

Fig. 7.7 On *left*, distilled water is added to the bucket until it reaches a level 15 cm below the soil surface. The tips of shallow and deep Pt electrodes should be located 5 cm above the water level and 10 cm below the water level, respectively. Water should be added periodically to preserve the desired level. An alternate arrangement (shown on *right*) is to use a taller bucket and to saturate the mesocosm to the soil surface. This can be especially informative if soil horizons with contrasting properties (such as OM content) occur at depths of 10 and 25 cm

to break open and expose the fresh soil surface (rather than a knife-cut soil surface). Note that you will need to test the freshly exposed and broken soil surface with the ααd (sometimes a false positive reaction to ααd is detected from where the low valence Fe from the steel in a knife blade or shovel has contacted the soil.)

Place a few drops of ααd (Childs 1981) on one-half of the core at various depths along the length of the mesocosm and note whether there is a reaction. Look for a pink color to develop. (If there is a lot of Fe^{2+} in the soil solution, this can sometimes occur quite rapidly. Other times, if there is very little Fe^{2+} in solution, it may take a few minutes for a subtle reaction to be observed.) In particular, pay attention to whether there is a reaction in the vicinity of where you had placed the electrodes (10 and 25 cm below the soil surface). Also keep in mind, the depth at which the water table was maintained in your mesocosm.

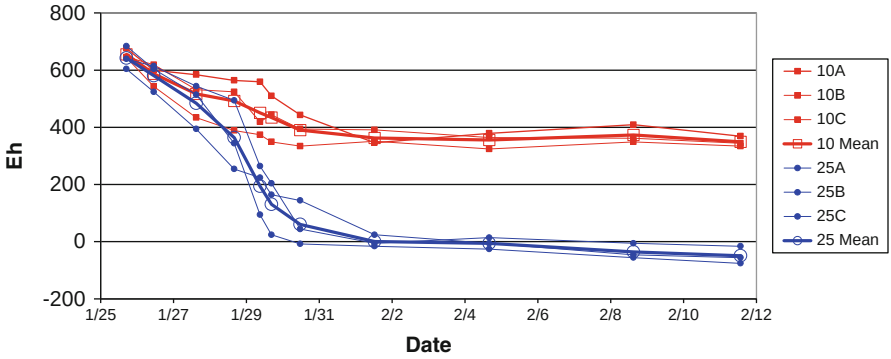

Fig. 7.8 Plotting of Eh measurements as a function of time. Note that replicate Eh measurements sometimes vary by more than 100 mV

Record your observations (especially at the depths of the electrodes), noting whether the reaction to αα*d* was negative, or positive, and if positive, indicate whether you think the reaction was weak, moderate or strong. Document any reaction to αα*d* at various other places up and down the core to try to obtain a better sense of where in the soil, Fe^{3+} has been reduced to Fe^{2+}. If you observe a positive reaction to αα*d* in the soil core, be careful to document where in the soil that reaction was observed, especially in relation to where the water table was located within the soil.

12. *Measuring soil pH*. In order to be able to plot data on an Eh-pH diagram, you will need to measure the pH of the soil at the same depths where you measured Eh. Use the OTHER HALF of the core to which you did NOT apply the αα*d* for measuring pH. Collect a few (10) ml of soil material from each of the two depths where Eh was measured and make a thick slurry by adding distilled water and stirring (the goal is the equivalent of 1:1 soil:water, but as you are not starting with dry soil, this will be an approximation). Allow the slurry to sit for 10–15 min and then mix again. Measure the pH by using a calibrated pH meter with a combination electrode and record to the nearest 0.1 units.
13. *Data Analysis*. Eh data should be reported in three ways. First, these data should be plotted as a function of time. This will allow you do evaluate whether any of the readings from any of the electrodes was spurious. Normally, a given electrode will show trends over time, and not provide erratic readings. If all the readings for a given electrode follow a trend and then one reading is way off, there is a good chance that the one reading is faulty. An example of Eh data plotted in this way is shown in Fig. 7.8.

    The second way in which the mesocosm data should be evaluated is with regard to an Eh-pH stability diagram. This is to determine whether the soil conditions were (theoretically) reducing with respect to Fe at any time during the experiment. You will only have pH measurements from your mesocosm at beginning and end of the experiment, and the pH is not be expected to change dramatically over the course of a couple of weeks (often one unit or less). For the sake of simplification, we will make the assumption that the pH changed

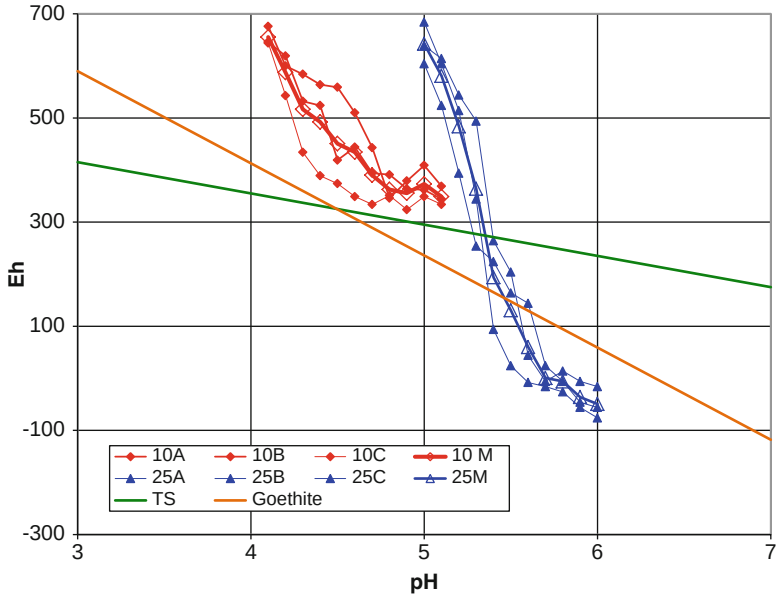

Fig. 7.9 Example data from mesocosms plotted on an Eh-pH diagram. Also shown is the "technical standard" line (TS) from the National Technical Committee for Hydric Soils which they use to define reducing soil conditions and also the stability line for goethite-Fe^{2+}

gradually (linearly) through the period of the experiment and thus use interpolated data for the dates in between. When plotted, these data may look something like those in Fig. 7.9. Also shown in Fig. 7.9 is the line for the equation Eh = 595 − (60 pH). This is the equation from the technical standard of the NTCHS (sometimes referred to as the "Technical Standard" line) (National Technical Committee for Hydric Soils 2007). This is an empirically derived line (shown in green), and data that plot above this line are considered to be oxidizing and those below the line indicate reducing conditions. The Eh-pH line showing the stability field for the crystalline iron oxide mineral goethite is also shown on the diagram (brown). Some people prefer to discuss iron reduction and Eh-pH data with respect to predictions using thermodynamically based equations such as this one. Above the line, goethite would be predicted to be stable and below the line, it would be predicted to unstable with Fe^{3+}being reduced to Fe^{2+}.

A third, and perhaps even more useful way to view the data is to plot the data over time, with the Eh values relative to the TS line (or this could also be done with respect to the a line like the goethite line). This is done by using pH data (for each date and depth) to calculate the corresponding Eh along the TS line, using the equation Eh = 595 − (60 pH). For example, if the pH ranged from 4.1 to 5.1 over a 6 day period, we would calculate the corresponding Eh values as shown in Table 7.5 (these change from 349 to 289 as the pH changes from 4.1 to 5.1).

Table 7.5 Calculation of Eh data with respect to the Technical Standard Line of the NTCHS

Day	pH	Eh of TS line	Measured Eh	Eh relative to the TS line
1	4.1	349	655	306
2	4.3	337	342	5
3	4.5	325	371	46
4	4.7	313	342	29
5	4.9	301	209	−92
6	5.1	289	−30	−319

These calculated values can then be subtracted from the measured Eh values (made using Pt electrodes). Positive values indicate that they are above the TS line (oxidizing) and if the values are negative, this means the data would plot below the TS line (reducing). If the data shown in Fig. 7.8 are plotted relative to the TS line, we obtain the graph shown in Fig. 7.9. On the graph in Fig. 7.9, the "zero" of the Y axis (Eh relative to the TS) represents the TS line itself. So any values that are greater than zero imply that the data plot above the TS line (in an Eh pH diagram) and are oxidizing with respect to Fe, and any negative values imply that the data plot below the TS line and are reducing with respect to Fe. This plot allows a quick visual evaluation of how these conditions change with time and at what point in time they become reducing.

Submission of Lab Report

Each student will write and submit a laboratory report based upon the data collected by the group (in the table below, data collected by students are shown by the shaded boxes). Each student should then calculate the Eh (considering the correction for the reference electrode). A graph similar to Fig. 7.8 should be constructed. The Eh of the TS line should be calculated for the pH values measured (and interpolated) for the soil. Then you should calculate, for each electrode, the Eh relative to the TS.

Using data measured and calculated in the student's version of Table 7.6, each of the following figures should be prepared:

- A figure showing Eh over time (similar to Fig. 7.8)
- A figure showing Eh and pH in relation to the TS line and to the goethite stability line (similar to Fig. 7.9)
- A figure showing Eh relative to the TS as a function of time (similar to Fig. 7.10)

Each student should provide a discussion of the data and figures that should include some reference to the following:

- Reproducibility and replication of the data measurements;
- Comparison of redox data collected at the two soil depths;
- The meaning of the Eh/pH data collected, implications of the data based on Eh/pH stability diagrams and where data plot relative to the TS line; and
- Comparisons between methods used – in particular the Eh/pH data and the reaction of the soil with alpha-alpha dipyridyl dye.

Table 7.6 Draft data sheet for recording measurements and calculations to be used in later analyses and figures

	Raw mV			Eh					Eh relative to the TS		
Date	Elect #	Elect #	Elect #	Elect #	Elect #	Elect #	pH	TS Eh	Elect #	Elect #	Elect #

A similar sheet should be used for collecting data both at 10 cm and at 25 cm

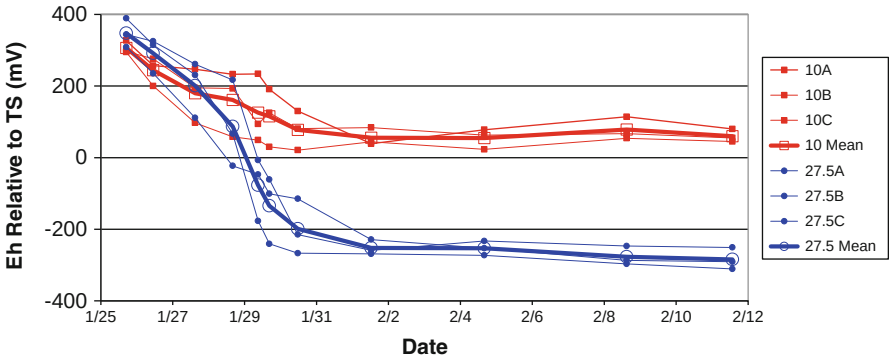

Fig. 7.10 Plot showing how Eh values are related to the Technical Standard line of the NTCHS, as a function of time. Positive values (above zero) indicate that the Eh is above the TS line (and thus is oxidizing). When values drop below zero, it indicates reducing soil conditions

References

Childs CW (1981) Field test for ferrous iron and ferric-organic complexes (on exchange sites on in water-soluble forms) in soils. Aust J Soil Res 19:175–180

Light TS (1972) Standard solution for redox potential measurements. Anal Chem 44:1038–1039

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Owens PR, Wilding LP, Lee LM, Herbert BE (2005) Evaluation of platinum electrodes and three electrode potential standards to determine electrode quality. Soil Sci Soc Am J 69:1541–1550

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Rabenhorst MC, Hively WD, James BR (2009) Measurements of soil redox potential. Soil Sci Soc Am J 73:668–674

Veneman PLM, Pickering EW (1983) Salt bridge for field redox potential measurements. Commun Soil Sci Plant 14:669–677

Field Exercises

Field Exercise 1: Use and Interpretation of IRIS Tubes

Objectives

1. To understand the principles behind the use of IRIS tubes.
2. To understand how to use IRIS tubes in the field.
3. To understand how to interpret the data from IRIS tubes.
4. To compare the functioning of IRIS tubes with other measures of reduction in soils.

Part I: IRIS Tube Installation

Materials and Equipment Needed

Five IRIS tubes
7/8″ push probe for making pilot holes
Spade or shovel
Tape measure
Transparent mylar grids
Equipment for making Eh measurements
Five, 40 cm Pt wire electrodes
One reference electrode (with salt bridge)
One, high impedance voltmeter
pH meter and buffers (This can be done in the field using a portable meter, or else samples can be returned to the lab for pH measurement.)
Alpha, alpha dipyridyl dye solution or test strips

Procedures

Overview: IRIS tubes will be installed following protocols spelled out in the Rabenhorst (2008) article. Tubes will remain in the soil for 4 weeks, after which they will be examined and paint removal will be quantified. On the dates on which the tubes are installed and extracted, water table levels will be documented. Also on these dates, soil reduction will be assessed using Eh (and pH) measurements and also by testing with alpha, alpha dipyridyl dye. Comparisons will be made among all three methods for assessing soil reduction.

1. Students should work in teams of 3–4 persons.
2. Each team will install a set of five IRIS tubes in the field (at a site provided by the instructor), following protocols spelled out in the Rabenhorst (2008) article.

3. At the location where the IRIS tubes were installed, an estimation of the level of the ground water table should be obtained by digging a small hole to a depth slightly greater than the depth to the water table and allowing it to equilibrate in the hole.
4. At the location where the IRIS tubes were installed, Eh should be measured using five replicate Pt electrodes at depths of 12.5, 25, and 40 cm. A stainless rod should be used to make pilot holes for the electrodes to the specified depths. First the five electrodes should be placed at 12.5 cm and measurements made. Then the pilot holes should be deepened and electrodes placed at 25 cm and measurements made a second time. A third set of measurements should be made after electrodes have been placed at 40 cm. Because electrodes will not remain in the field between the two measurement dates, the reference electrode does not need to be placed within a salt bridge. Rather, a small amount of water should be added to the soil surface (if not already saturated) and stirred to make a paste, into which the reference electrode should be placed. The reference electrode should be situated within 25–75 cm of the location of the Pt electrodes.
5. Using either the push probe, a spade or an auger, samples should be collected from depths of 12.5, 25 and 40 cm so that pH can be determined at the same depths where Eh was measured.
6. Using either soil on cores removed when making pilot holes for IRIS tubes or soil collected with a spade or auger, apply a few drops of alpha-alpha dipyridyl dye at various depths in the soil to see whether there is a positive reaction to the dye, indicating the presence of Fe^{2+}.
7. Make a brief soil description at the location where the IRIS tubes were installed making special note regarding whether or not the soil meets any of the approved field indicators of hydric soils (USDA-Natural Resources Conservation 2010).
8. After approximately 4 weeks have elapsed, on the date when the IRIS tubes are extracted and examined, you should again make measurements of Eh, pH and water table height, and evaluate whether the soil gives a positive reaction to the alpha-alpha dipyridyl dye.

Part 2: IRIS Data Collection and Analysis

1. Four weeks after the IRIS tubes were installed, they should be extracted from the soil. Adhering soil can be removed by rinsing with very gentle brushing (as needed) under a stream of water, and then placed aside to dry.
2. Quantification will be accomplished using the mylar grid method (Rabenhorst 2012)because this method is more accurate and more reproducible than visual estimations (Rabenhorst 2010). A 15 cm by 6.7 cm grid containing 390 50 mm by 51 mm sectors should be printed on transparent mylar sheet so that it can be wrapped around an IRIS tube and held in place using rubber bands (Fig. 7.11).
3. The Technical Standard (National Technical Committee for Hydric Soils 2007) requires that paint be removed from 30 % of the IRIS tube within a 15 cm zone

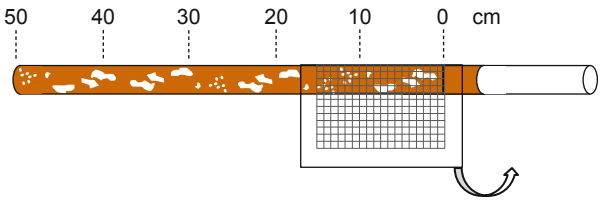

Fig. 7.11 Illustration showing placement of mylar grid around a 15 cm portion of an IRIS tube. By marking and counting sectors where paint has been removed, paint removal can be easily quantified (Modified from Rabenhorst 2012 with kind permission of © The Soil Science Society of America, Inc. 2012. All Rights Reserved)

that occurs somewhere within the upper 30 cm of the soil.[4] Therefore each group must determine which 15 cm zone within the upper 30 cm of each IRIS tube shows the maximum paint removal, and quantification should proceed for this zone.

4. A clear mylar grid should be wrapped around that portion of each IRIS tube showing the greatest degree of paint removal and should be secured using rubber bands. Using a fine point permanent marker, each sector with greater than 50 % paint removal should be marked. Once all the sectors are marked, the sectors should be counted and the percentage of the area can be calculated.

Part 3: Comparison of IRIS Data with Eh, pH and ααd Data

1. Using Eh and pH data collected and plotted on an Eh-pH stability diagram you should be able to evaluate whether the soil is reducing (or at what depths, the soil is reducing).
2. Based on the data you have collected, you now have 3 independent assessments of whether the soil is reducing: (1) Eh-pH; (2) ααd; (3) IRIS tubes.
3. Provide a thorough discussion of your data, being sure to include:

   - A summary of whether or not the soil was reducing according to each of the three assessment tools;
   - How results from each of these assessment methods were similar to or different from the others;
   - What factors might account for any differences that you observed; and
   - Your own conclusions (based upon your data) regarding whether or not the soil is reducing. Note that according to the Technical Standard of the NTCHS, in order for a soil to be reducing, it must meet the reducing specifications of one of the three methods, within the zone of 0–30 cm for a period of 14 days. For Eh and IRIS, a majority of the instruments (three-fifth) must fall within the specified range (not the mean).

[4] Other work has suggested that 20 % removal from a 10 cm zone somewhere within the upper 30 cm might be a better assessment of reducing conditions in soils (Castenson and Rabenhorst 2006; Rabenhorst 2008).

4. Your submitted lab report should include the following:

   - Data tables showing all data collected on each of the two dates (Eh, pH, ααd, temperature, water tables);
   - Your discussion of water tables and temperatures over the course of the month (using your data and the continuous class record);
   - Completed IRIS data forms with data from yourself and from others in your group;
   - Eh-pH diagrams showing your data plotted for each of the two dates;
   - Your soil description and assessment regarding whether the soil meets any of the Field Indicators; and
   - Your complete discussion of all the data (including morphological data) with your conclusions.

References

National Technical Committee for Hydric Soils (2007) Technical note 11: technical standards for hydric soils. USDA-NRCS, Washington DC. Available from: ftp://ftp-fc.sc.egov.usda.gov/NSSC/Hydric_Soils/note11.pdf

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Rabenhorst MC (2010) Visual assessment of IRIS tubes in field testing for soil reduction. Wetlands 30:847–852

Rabenhorst, MC (2012) Simple and reliable approach for quantifying IRIS tube data. Soil Sci Soc Am J 76:307–308

USDA, NRCS (2010) Field indicators of hydric soils in the United States. Ver. 7.0. In: Vasilas LM, Hurt GW, Noble CV (eds) USDA, NRCS in cooperation with the National Technical Committee for Hydric Soils

Field Exercise 2: Assessing Leaf Litter Decomposition with Litter Bags

Objectives

1. To understand the principles behind the use of litter bags.
2. To understand how to construct and deploy litter bags.
3. To understand the impact of soil moisture content on decomposition rates.

Materials and Equipment Needed

Leaf litter samples
Litter bags – fiberglass screening material, mesh size of 1–2 mm
Drying oven
Balance
Pin flags

Procedures

Overview: There are several options with this exercise with respect to the type of organic materials used and the location of bag deployment in the field. In the procedures below, we suggest collecting fallen leaves from one deciduous tree species. This is the simplest approach and promotes uniformity in the samples. However, the exercise can be modified to compare multiple species or even root samples. With respect to deployment location, we suggest two sites, a wetland and an adjacent upland. The main point to this exercise is that wet conditions slow down decomposition rates. To further illustrate this point, the class may choose to place some litter bags in inundated areas. We suggest avoiding permanently inundated areas as it may be difficult to retrieve the bags.

9. *Litter bag construction*. Ten litter bags will be needed for each team of students. For each litter bag cut a 20 cm × 40 cm section of screen. Double the screen section over and staple the sides together at 5 cm intervals. You should now have a screen pouch open at one end. The open end will be stapled shut after the bag is filled with leaf litter. Each bag should be numbered with a permanent marker for future identification. Weigh each bag with three additional staples. Record the weight. The weight of the additional staples is needed to account for the staples used to close the open end of the bag after the leaf sample is added.
10. *Leaf litter sample collection*. Collect fallen leaves from the chosen field site. To promote uniformity in the samples; collect all samples from the adjacent upland, sample only the top layers of the litter, and sample only one deciduous tree species.
11. *Sample preparation*. Chop the leaves into 2–5 cm lengths. Randomly collect five sub-samples of the chopped leaves. Determine fresh weight of the sub-samples, then dry them (70 °C, 48–72 h) and re-weigh to obtain their water content.
12. *Bag filling*. Fill the bags with the prepared samples. Try to place a similar amount of material in each bag (It does not need to be exactly the same). The material should be relatively loose in the bag to allow for air movement. Staple the open end closed with three staples. Weigh each bag and record. Sample fresh weight is calculated as the difference between the filled bag weight and the empty bag weight.
13. *Litter bag deployment*. Place five bags in the upland and five bags in the wetland. For placement in the wetland, choose deployment points with similar hydrologic conditions as soil moisture will impact decomposition rates. For example, if the site has pit and mound topography, do not place 1 bag on top of a mound and another bag in a pit. Avoid permanently inundated areas. Clear away any duff and place the bags directly on top of the mineral soil surface. Mark the location of each bag with a pin flag.
14. *Litter bag retrieval and analysis*. Retrieve the bags after 2–3 months. Dry the retrieved bags (70 °C, 48–72 h) and weigh. Calculate sample dry weight.

15. *Calculations*. Loss of biomass due to decomposition is calculated as the difference between initial biomass and remaining biomass. All values are expressed on a dry weight basis. Initial dry weight (pre-deployment) of the samples is calculated from the fresh weight of each sample and the average water content of the five sub-samples determined in Step 3. Average rate of decomposition (per day) is determined by dividing biomass loss by the incubation period. Compare the mean biomass loss for the upland samples to that of the wetland samples.

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