CHAPTER 6

Ecosystem Component Characterization

“Things don’t turn up in this world until somebody turns them up.”

James A. Garfield

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OVERVIEW

Ecosystem Structure and Integrity, Chaos and Disturbance

It is impossible to produce meaningful, representative, and reliable data to be used in decisions regarding the status of, or possible impacts to, the environment without first defining the environment, critical receptors, influencing factors, and natural dynamics. This requires the measurement of many aspects of the watershed, as previously described in this book. Simplistic and rapid approaches are fine for initial assessments, but may fall short in providing understanding of the causes of the observed problems. Therefore, later phases of watershed assessment projects generally need to examine more detailed aspects of a study area in order to obtain a better understanding of possible interactions. As an example, the majority of studies dealing with aquatic toxicity have used surrogate species (or a small number of species) and have not attempted to investigate ecosystem interactions a priori, such as ecosystem energetics or stress–productivity–predation relationships. For example, surrogate responses have simply been quantified based on sample toxicity, and then effects have been extrapolated to in situ conditions. While these exercises might satisfy the study objectives of defining sample toxicity to the test species, they do little to document or define ecosystem disturbance. Ecological processes can be ignored, to a degree, when acute toxicity scenarios are studied, such as in sediments that are severely degraded. However, “significant cases of acute toxic effects have been encountered infrequently” (Chapman 1986), and the more common situations in which effects and zones of contamination are “gray” (Chapman 1986) dictate that natural and anthropogenic effects be separated. This cannot be done accurately without an understanding of ecosystem dynamics such as spatial and temporal variance of chemical, physical, and biological systems and their interactive processes.

Community ecology in lotic and lentic systems has progressed substantially in recent years. “Biotic dynamics and interactions are intimately and inextricably linked to variation in abiotic factors” (Power et al. 1988), and lotic systems are not in equilibrium due to natural disturbances
which may occur frequently or infrequently (Resh et al. 1988). Disturbance can be defined as a discrete event that alters community structure and changes the physical environment and resource availability. These disturbances vary in type, frequency, and severity, both among and within ecoregions. The frequency and intensity of disturbances cannot be predicted (Resh et al. 1988). Intermediate levels of disturbance maximizes species richness (Resh et al. 1988). Equilibrium or steady-state conditions will tend to occur if disturbances are infrequent, thus excluding opportunistic species (Minshall 1988). In stream ecology, disturbance is the dominating organizing factor, having a “major impact on productivity, nutrient cycling and spiraling, and decomposition” (Resh et al. 1988). Disturbances such as storm events or the presence of toxicants can eliminate biota (Power et al. 1988). Recovery and succession of these systems between disturbances is typified by recurrent or divergent patterns (Pringle et al. 1988; Resh et al. 1988). Despite this inherent variability, benthic communities have been used effectively to classify community structure and functioning in aquatic ecosystems.

Ecotones are defined as zones of transition between adjacent ecosystems. Disturbance plays a major role in determining the structure and dynamics of ecotones, such as stream bank riparian zones. High relief areas are less stable due to more frequent and diverse disturbances combined with complex topographic effects. Both fluvial and geomorphic processes influence vegetation development along stream and lake embankments (Decamps et al. 1990).

The major role that natural and anthropogenic disturbances have on aquatic ecosystems increases the level of spatial and temporal variance. Spatial and temporal dimensions span 16 orders of magnitude in stream ecology (Minshall 1988; Pringle et al. 1988). Some suggest that spatial heterogeneity enhances the ability of an ecosystem to resist and recover after a disturbance (Fisher 1990). Significant spatial variance in sediments is common (Stemmer et al. 1990). Each level of the system has different dimensions, has different variances associated with it, and is interacting simultaneously with other ecosystem levels and their respective dimensions and variances. This complex reality is difficult, if not impossible, to define accurately but must be considered in all assessments of water quality or ecosystem health.

Orians (1980) stated that one of the greatest challenges in ecology (and ecotoxicology) is bridging the conceptual gap between micro- and macroecology. Aquatic systems can be considered as a mosaic of patches (Pringle et al. 1988). “A patch is a spatial unit that is determined by the organism and problems in question” (Pringle et al. 1988). The heterogeneous environment has highly clumped distributions (patches) of organisms whose spatial and temporal patterns and relationships change seasonally due to factors such as food (resource) patterns (Findlay 1981). These clumped distributions, therefore, pose severe sampling problems. The appropriate sampling scale will depend on the organism size, density, distribution, life cycle, and question being asked (Pringle et al. 1988), which, unfortunately, are often not considered. Aquatic ecosystems are open nonequilibrium systems (Carpenter et al. 1985; Pringle et al. 1988) where patches are in transitory steady state with other patches (Sheldon 1984). Many “ecosystems” are not independent units, and some processes (e.g., nutrient cycling) show no spatial threshold. That is, no one area bounds all processes, showing that ecosystems have both an open nature and are connected in many complex ways. Most aquatic organisms are aggregated at certain spatial scales and are random on other scales. In order to accurately determine total organism numbers and distribution patterns (patches) within and among sites, presampling should be conducted whereby the site is divided into quadrants, sampled, and coefficients of variation (standard deviation divided by the mean) determined. This will detect heterogeneity in density measurements (Westman 1985). Unfortunately, this level of accuracy is often beyond the resource capabilities of typical studies. Different life histories and variable interactions between species may prevent equilibrium (Carpenter et al. 1985).

Ecosystems tend to restore balance (homeostasis or resilience). Diversity does not equate to integrity. Biological integrity may be defined as the ability of species to interact and maintain their structure and function in some self-regulating, homeostatic fashion (Westman 1985). The rate, manner, and extent of recovery following a disturbance is a measure of resilience.
The influence of storm events and watershed characteristics on chemical element dynamics is poorly understood, particularly because some are lumped into operationally defined units such as dissolved or total organic carbon. Significant heterogeneity (62 to 100%) has been observed between adjacent sediment cores in concentrations of organic matter, water, and total phosphorus (Downing and Roth 1988). Some heterogeneity is likely due to small-scale variations in bottom profiles.

In stream benthic communities, hydraulics appear to be more important than substrate in determining distribution (Statzner et al. 1988). As in fish communities, populations will vary in type and number between pool and riffle areas. Most benthic macroinvertebrate testing occurs in riffle areas where continual flow exists and more types of organisms are present. Small-scale sampling is more likely to define benthic invertebrate patches than large-scale sampling, which homogenizes patchiness differences. The replicate number needed to obtain a given precision decreases with increased density and sample size, and the optimal sample size (considering cost and precision) depends on mean density (Morin 1985).

Other important considerations in valid hazard assessments are contaminant interactions and subsequent distribution in the aquatic system via solids. Sediment contaminant data should be evaluated based on grain size correction, which reduces the inert fractions (e.g., hydrates, sulfides, amorphous, and fine-grained organics). The most useful size fraction for contaminant assessments appears to be <63 µm (Håkanson 1984). This size fraction will tend to predominate in deposition areas and will play a major role in the transport, deposition, and resuspension of the fine-grained sediments. Particle diameters of suspended solids vary over two orders of magnitude and settling speeds in waters vary over four orders of magnitude (Gailani et al. 1991). Predicting transport is complicated by the lack of understanding of sizes and settling speeds, floc disaggregation due to shear, processes governing entrainment and deposition, and turbulence description (Gailani et al. 1991).

When resuspension events occur, predicting metal remobilization may be possible in site-specific studies; however, remobilization is dependent on particle residence time in the water column, which varies between sites, storms, and systems. In most systems, however, remobilization of metals from resuspended sediments is likely to be insignificant due to the slow reaction rates (Kersten and Forstner 1987).

Though resuspension effects appear limited if one considers the scavenging effects of solids, laboratory studies of bioturbation effects on contaminant movement and toxicity to planktonic species have shown otherwise. Bioturbation by benthic and epibenthic invertebrates occurs in many ways: by pumping pore water constituents out of the sediment into overlying waters; by injecting water into the sediment; by pumping particulates to the sediment-water interface; by depositing fecal pellets on the sediment surface; and by disrupting horizontal and vertical layering (Petr 1977).

Given the above discussion on the complexities of aquatic ecosystems, it is evident that it is no longer adequate to simply study separate components of the ecosystem, such as planktonic species in water-only systems or chemical dynamics in a water-only or sediment slurry system. This “reductionist” approach is essential for defining processes, but does not provide an accurate picture of the component–ecosystem interactions and, in fact, may produce misleading results. Examples of this disparity are becoming increasingly obvious, particularly in the field of aquatic toxicology, as more “holistic” types of studies are published (Chapman et al. 1992).

Sediments play a major role in ecosystem processes and ecosystem health (Chapman et al. 1992). Generally speaking, the surficial layer (upper few centimeters) is the active portion of the ecosystem, while deeper sediments are passive and more permanently “in-place.” These deeper layers are of interest as a historical record of ecosystem activity, but may also be reintroduced into the active portion of the ecosystem via dredging activities and severe storm or hydrogeological events. The usefulness of a sediment monitoring station as an indicator of contaminant presence is a function of the interactions between the change in contaminant net deposition rate, sediment accumulation rate, mixing zone depth and dynamics, sampling method and frequency, the type of laboratory method, and its precision and accuracy (Larsen and Jensen 1989). Sediments and soils typically exhibit more spatial variability from overlying waters but less temporal variation. This reality affects sampling design and statistical analyses.
This chapter describes a wide variety of tools that can be used for assessing the ecosystem and watershed physical characteristics because of the likely need to consider a broad range of assessment procedures. This chapter starts with discussions of rainfall and flow monitoring, as it is difficult to understand pollutant transport, fate, and effects without appreciating the physical movement of the water. The main sections of this chapter pertain to examinations of specific receiving water uses and associated ecosystem components: aesthetics and safety, habitat, water and sediment, microorganisms, benthos, zooplankton, fish, and toxicity and bioaccumulation.

**FLOW AND RAINFALL MONITORING**

It is essential that there be an accurate description of the system’s hydrodynamics when assessing the effects of stormwater runoff on receiving waters. Flow represents the pollutant loading mechanism, and its power and frequency of occurrence can degrade the physical habitat. One of the principal reasons there is a relatively poor understanding of stormwater runoff effects is because of the difficult logistics involved in measuring short-term, high-flow events quickly and accurately. Flow and rainfall monitoring are considered separately from other physical characteristics, which are discussed in the following section on habitat. The hydrology of the stream, reservoir, or lake which receives stormwater runoff is interrelated, directly and indirectly, with many other characteristics, such as substrate composition, temperature, suspended solids, channel morphology, and biological communities. Hydrology, as discussed here, is composed of flow, velocity, power, turbulence, mixing, sedimentation, and resuspension subcomponents. Each of these subcomponents is important to varying degrees depending on the site and study objectives, and each is relatively difficult to quantify accurately during storm events.

As with other major ecosystem components, the storm event hydrodynamics of the receiving water must be evaluated based on references for comparison. References may include an upstream station, present day baseflow conditions, predevelopment conditions, and/or a least disturbed watersheded of similar natural characteristics (e.g., soil, topography, drainage area, stream order, stream substrate, biological communities). The assessment should attempt to characterize the hydrology of the system by defining the loading dynamics (i.e., magnitude, duration, frequency) and the receiving system response (e.g., flow, spatial-temporal patterns). The physical characterization of the loading and system response will dictate the chemical sampling from which to determine pollutant (stressor) loading dynamics and optimal stormwater control programs and associated remediation measures.

The rate of stream discharge (flow) \((Q)\) is a function of the channel cross-sectional area \((A)\) and the mean velocity \((V)\), which is usually expressed as cubic feet per second (cfs). So, \(Q = AV\). Velocity is a function of runoff quantity, stream width, depth, and gradient, and channel roughness. Roughness is affected by channel sinuosity, substrate size, bottom topography, stream vegetation, debris, and other obstructions. Channelization increases velocity and also tends to make velocity more uniform (EPA 1983). Channelization practices, such as straightening, vegetation and debris removal, berming and leveeing, all increase drainage efficiency. These practices also produce sharper flow hydrographs, with much greater peak flow rates. The resulting higher flow rate and power increases the impact of storm events, including increased scour, erosion, bank cutting, sediment transport, flooding below the channelized section, reduced groundwater levels and stream dewatering, degraded habitat and water quality, promotion of land development, and lowered recreational values. Assessing channelization effects on habitat quality is discussed more fully in the following section on habitat.

Stream staff gauges, which measure stream depth, may be used to indirectly measure flow through the use of a rating curve which shows the relationship between stream depth and flow rate. The rating curve is developed by making velocity measurements in a cross-sectional area of the stream channel where the channel morphology and flow patterns are simple. This is done over a range of flows so that the curve can be constructed. This is discussed in a following subsection.
Stream power is the rate of potential energy expenditure per unit weight of water in a channel and is calculated as:

\[
SP = \frac{\Delta Y}{\Delta t} = \frac{\Delta X}{\Delta t} \frac{\Delta Y}{\Delta X} = VS_f
\]

where

- \(SP\) = stream power (ft-lbs/lb H\(_2\)O/s)
- \(t\) = time (s)
- \(V\) = velocity (ft/s)
- \(S_f\) = stream friction slope (ft/ft) (energy gradient)
- \(Y\) = energy grade line elevation above a point, equivalent to potential energy (ft-lb)/lb/H\(_2\)O = water surface elevation and velocity head (V\(^2\)/2g)
- \(X\) = longitudinal distance
- \(g\) = gravitational constant

Stream power can be used to estimate the energy available for sediment transport. This energy can be reduced by other habitat factors (e.g., bank and substrate stability, vegetation, or surface erosion).

“Time of passage” has been recommended as a parameter of pollutant movement through a stream more useful than the kinematic wave velocity that is typically used in hydrograph routing calculations (Velz 1970). The distinction is that the kinematic wave (hydrograph crest) moves faster than the waste in the body of water, particularly in large, deep water systems. Time of passage (as seconds or days) is based on the average flow rates that are measured when using current meters. It is determined by dividing the occupied channel volume (from cross-sectional area) (as cubic feet) by the runoff (from drainage area and yield) (as cfs).

**Flow Requirements for Aquatic Biota**

A popular evaluation tool for evaluating flow effects on aquatic communities was published by Tennant (1976). He found the following in 11 streams of three western states:

- Changes in habitat were similar among streams with similar flow regimes.
- A depth of at least 0.3 m and velocity of at least 0.75 ft/s were required for most fish.
- Thirty percent of the annual flow provided good habitat.
- Sixty percent of the annual flow provided outstanding habitat.

Stream velocity plays a major role in determining the composition of benthic communities (Cummins 1975): invertebrate drift increases as the velocity increases (Minshall and Winger 1968; Walton 1977; Zimmer 1977).

The U.S. Fish and Wildlife Service developed the Instream Flow Incremental Methodology (IFIM) computer program to evaluate changes in aquatic life from alteration of channel morphology, water quality, and hydraulic components. Each species has a range of habitat (including flow) conditions it can tolerate, which can be defined (or is defined) for the species, as can stream conditions. IFIM simulates hydraulic conditions — habitat availability for a species and size class, or usable waters for a particular recreational activity. This is done through use of the Physical Habitat Simulation Model (PHABSIM), which relates changes in flow and channel structure to changes in physical habitat availability.

The basic steps in the IFIM can be summarized by the following:

- Project scoping — Define objectives for the delineation of study area boundaries, determine the species, and define their life history, food types, water quality tolerances, and microhabitat.
- Study reach and site selection — Identify and delineate critical reaches to be sampled, delineate major changes and transition zones and the distribution of the evaluation species.
• Data collection — Transects are selected to adequately characterize the hydraulic and in-stream habitat conditions. Data gathering must be compatible with IFIH computer models.
• Computer simulation — Reduce field data and input into programs described above.
• Interpretation — The output is expressed as the Weighted Usable Area (WUA), a discrete value for each representative and critical study reach, for each life stage and species, and for each flow regime.


**Urban Hydrology**

Basic watershed characteristics need to be known in order to understand stream flow conditions. These include topography (watershed divide plus stream and land slopes), drainage efficiency (stream orders and types of artificial drainage systems), and, to a lesser extent in urban areas, soil characteristics (disturbed or compacted, age since development, type of ground cover, soil texture, etc.). It is important that characteristics throughout the watershed be evaluated when studying streams. Looking only at characteristics adjacent to the stream is very misleading, as urban drainage systems are very efficient transporting systems, capable of carrying water and pollutants to the stream from locations far away. These topics are beyond the scope of this book, but several good books are available that describe urban hydrology and associated drainage design (including McCuen 1989; WEF and ASCE 1992; Debo and Reese 1995; and Wanielista et al. 1997).

Urban hydrology can be used to divide rain into different major categories, each reflecting distinct portions of the long-term rainfall record (Pitt et al. 1999). When monitoring runoff, it is therefore important to include a sampling effort that represents each of these categories. All too often, the small rains are not sampled because of misunderstandings of their significance. It is easy to ignore these small events, considering the problems that occur when trying to program automatic sampling equipment. However, small events are extremely important when conducting a receiving water investigation. As an example, consider the following rainfall and runoff data for Milwaukee, WI, what were obtained during the Nationwide Urban Runoff Program (EPA 1983). Figure 6.1 shows measured rain and runoff distributions for Milwaukee during the 1981 NURP monitored rain year. Rains between 0.05 and 5 in were monitored during this period. Two very large events
(greater than 3 in) occurred during this monitoring period, which greatly bias this distribution, compared to typical rain years. The following observations are evident:

- The median rain depth was about 0.3 in.
- 66% of all Milwaukee rains are less than 0.5 in in depth.
- For the medium-density residential area, 50% of the runoff was associated with rains less than 0.75 in for Milwaukee.
- Observable runoff occurred with rain as small as 0.05 in in depth.

In addition, a 100-year, 24-hour rain of 5.6 in for Milwaukee could produce about 15% of the typical annual runoff volume, but it only contributes about 0.15% of the average annual runoff volume, when amortized over 100 years. Typical 25-year drainage design storms (4.4 in in Milwaukee) produce about 12.5% of typical annual runoff volume but only about 0.5% of the average runoff volume.

Figure 6.2 shows measured Milwaukee pollutant discharges associated with different rain depths for a monitored medium-density residential area. Suspended solids, COD, lead, and phosphate discharges are seen to closely follow the runoff distribution shown in Figure 6.1. Therefore, the concentrations of most runoff pollutants do not vary significantly for runoff events associated with different rain depths.

The monitored rains at this Milwaukee medium-density residential location can be divided into four categories:

- <0.5 inch. These rains account for most of the events, but little of the runoff volume. They produce much less pollutant mass discharge and probably have fewer receiving water effects than other rains. However, the runoff pollutant concentrations likely exceed regulatory standards for several categories of critical pollutants, especially bacteria and some total recoverable metals. They also cause large numbers of overflow events in uncontrolled combined sewers. These rains are very common, occurring once or twice a week (accounting for about 60% of the total rainfall events and about 45% of the total runoff events that occurred), but they only account for about 20% of the annual runoff and pollutant discharges. Rains less than about 0.05 in did not produce noticeable runoff.
- 0.5 to 1.5 inches. These rains account for the majority of the runoff volume (about 50% of the annual volume for this Milwaukee example) and produce moderate to high flows. They account for about 35% of the annual rain events, and about 20% of the annual runoff events. These rains
occur on the average about every 2 weeks from spring to fall and subject the receiving waters to frequent high pollutant loads and moderate to high flows.

- 1.5 to 3 inches. These rains produce the most damaging flows, from a habitat destruction standpoint, and occur every several months (at least once or twice a year). These recurring high flows, which were historically associated with much less frequent rains, establish the energy gradient of the stream and cause unstable stream banks. Only about 2% of the rains are in this category, and they are responsible for about 10% of the annual runoff and pollutant discharges. Typical storm drainage design events fall in the upper portion of this category.

- >3 inches. The smallest rains in this category are included in design storms used for drainage systems in Milwaukee. These rains occur only rarely (once every several years to once every several decades, or even less frequently) and produce extremely large flows. The monitoring period during the Milwaukee NURP program was unusual in that two of these events occurred. Less than 2% of the rains were in this category (typically <<1%), and they produced about 15% of the annual runoff quantity and pollutant discharges. During a “normal” period, these rains would produce only a very small fraction of the annual average discharges. However, when they do occur, great property and receiving water damage results. Receiving waters can conceivably recover naturally from this damage (mostly associated with habitat destruction, sediment scouring, and the flushing of organisms great distances downstream and out of the system) and return to before-storm conditions within a few years, depending on riparian vegetation growth rates and nearby “reservoir or refugia” areas for aquatic organisms.

The above specific rain values are given for Milwaukee, WI, selected because of the occurrence of two very rare rains during an actual monitoring period. Obviously, the critical values defining the design storm regions would be highly dependent on local rain and development conditions. Computer modeling analyses from 24 urban locations from throughout the United States were conducted by Pitt et al. (1999) to examine these patterns nationwide. These locations represent most of the major river basins and much of the rainfall variations in the country. These simulations were based on 5 to 10 years of rainfall records, usually containing about 500 individual rains each. The rainfall records were from certified NOAA weather stations and were obtained from CD-ROMs distributed by EarthInfo of Boulder, CO. Hourly rainfall depths for the indicated periods were downloaded from the CD-ROMs into an Excel spreadsheet. This file was then read by a utility program included in the Source Loading and Management Model (SLAMM) package (Pitt and Voorhees 1995). This rainfall file utility combined adjacent hourly rainfall values into individual rains, based on user selections (at least 6 hours of no rain was used as the criterion to separate adjacent rain events and all rain depths were used, with the exception of the “trace” values that were <0.01 in). These rain files for each city were then used in SLAMM for typical medium-density and strip commercial developments. SLAMM utilizes unique prediction methods that were especially developed by Pitt (1987) to accurately predict runoff during these small rains. Conventional runoff prediction methods are based on drainage design storms (of several inches in depth) and are not accurate when predicting runoff during small rains.

Table 6.1 summarizes these rain and runoff distributions for these different U.S. locations. Lower and upper runoff distribution breakpoints were identified on all of the individual distributions. The breakpoints separate the distributions into the following three general categories (similar to the regions identified for the Milwaukee rains):

- Less than lower breakpoint: small, but frequent rains. These generally account for 50 to 70% of all rain events (by number), but only produce about 10 to 20% of the runoff volume. The rain depth for this breakpoint ranges from about 0.10 in in the arid Southwest, to about 0.5 in in the wet Southeast. These events are most important because of their frequencies, not because of their mass discharges. They are therefore of great interest where water quality violations associated with urban stormwater occur. This would be most common for bacteria (especially fecal coliforms) and for total recoverable heavy metals, which typically exceed receiving water numeric criteria during practically every rain event in heavily urbanized drainages having separate stormwater drainage systems.
<table>
<thead>
<tr>
<th>Location</th>
<th>Median Rain Depth, by Count (in)</th>
<th>Corresponding Percentage for the Median Rain Depth</th>
<th>Rain Depth Associated with Median Runoff Depth (in)</th>
<th>Lower Breakpoint Rain Depth (in)</th>
<th>Percentage of Rain Events Less Than Lower Breakpoint</th>
<th>Percentage of Runoff Volume Less Than Lower Breakpoint</th>
<th>Upper Breakpoint Rain Depth (in)</th>
<th>Percentage of Rain Events Less Than Upper Breakpoint</th>
<th>Percentage of Runoff Volume Less Than Upper Breakpoint</th>
<th>Percentage of Runoff Volume Between Breakpoints</th>
<th>Percentage of Rain Events Between Breakpoints</th>
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</thead>
<tbody>
<tr>
<td>Columbia North Pacific</td>
<td>0.07</td>
<td>3–5</td>
<td>0.30–0.35</td>
<td>0.10</td>
<td>52</td>
<td>9–11</td>
<td>0.91</td>
<td>99</td>
<td>89–93</td>
<td>80–82</td>
<td>47</td>
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<tr>
<td>Seattle, WA</td>
<td>0.12</td>
<td>4–6</td>
<td>0.62–0.80</td>
<td>0.18</td>
<td>60</td>
<td>8–11</td>
<td>3.4</td>
<td>99</td>
<td>92–96</td>
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<td>California</td>
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<td>1.2–1.5</td>
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<td>0.55–0.68</td>
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<td>9–12</td>
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<td>0.55–0.60</td>
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<td>8–10</td>
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<td>89–93</td>
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<td>Denver, CO</td>
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<td>0.50–0.60</td>
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<td>71</td>
<td>13–17</td>
<td>1.8</td>
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<td>92–96</td>
<td>82–83</td>
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<td>Arkansas-White-Red</td>
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<td>65</td>
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<td>99</td>
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<td>78–80</td>
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<td>2–3</td>
<td>1.4–1.8</td>
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<td>72</td>
<td>8–12</td>
<td>6.0</td>
<td>99</td>
<td>88–94</td>
<td>80–82</td>
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<td>Upper Mississippi</td>
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<td>3–5</td>
<td>0.73–1.0</td>
<td>0.22</td>
<td>65</td>
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<td>80–83</td>
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<td>80–82</td>
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## Great Lakes

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<td>99</td>
<td>92–95</td>
<td>85–84</td>
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<td>7–11</td>
<td>0.61–0.72</td>
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<td>64</td>
<td>8–12</td>
<td>2.1</td>
<td>99</td>
<td>88–93</td>
<td>80–81</td>
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<td>Columbus, OH</td>
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<td>0.80–1.0</td>
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<td>63</td>
<td>8–12</td>
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<td>99</td>
<td>85–91</td>
<td>77–79</td>
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## North Atlantic

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<tbody>
<tr>
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<td>7–11</td>
<td>1.1–1.5</td>
<td>0.30</td>
<td>64</td>
<td>8–12</td>
<td>4.5</td>
<td>99</td>
<td>90–96</td>
<td>82–84</td>
<td>35</td>
<td></td>
</tr>
<tr>
<td>Newark, NJ</td>
<td>0.28</td>
<td>7–11</td>
<td>1.2–1.5</td>
<td>0.33</td>
<td>54</td>
<td>8–12</td>
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<td>99</td>
<td>89–94</td>
<td>81–82</td>
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</tbody>
</table>

## Lower Mississippi

<table>
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<tr>
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<th></th>
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<th></th>
<th></th>
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</tr>
</thead>
<tbody>
<tr>
<td>New Orleans, LA</td>
<td>0.25</td>
<td>7–11</td>
<td>1.7–2.2</td>
<td>0.45</td>
<td>62</td>
<td>7–11</td>
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<td>99</td>
<td>88–93</td>
<td>81–82</td>
<td>37</td>
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## South Atlantic Gulf

<table>
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<th></th>
<th></th>
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<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Atlanta, GA</td>
<td>0.22</td>
<td>7–11</td>
<td>1.2–1.7</td>
<td>0.32</td>
<td>58</td>
<td>5–9</td>
<td>4.0</td>
<td>99</td>
<td>91–95</td>
<td>86</td>
<td>41</td>
<td></td>
</tr>
<tr>
<td>Birmingham, AL</td>
<td>0.20</td>
<td>7–11</td>
<td>1.2–1.5</td>
<td>0.40</td>
<td>64</td>
<td>8–13</td>
<td>5.0</td>
<td>99</td>
<td>90–96</td>
<td>82–83</td>
<td>35</td>
<td></td>
</tr>
<tr>
<td>Raleigh, NC</td>
<td>0.18</td>
<td>7–11</td>
<td>1.0–1.2</td>
<td>0.26</td>
<td>60</td>
<td>7–11</td>
<td>2.5</td>
<td>99</td>
<td>87–93</td>
<td>80–82</td>
<td>39</td>
<td></td>
</tr>
<tr>
<td>Miami, FL</td>
<td>0.13</td>
<td>7–11</td>
<td>1.2–1.6</td>
<td>0.30</td>
<td>67</td>
<td>9–13</td>
<td>4.0</td>
<td>99</td>
<td>87–93</td>
<td>78–80</td>
<td>32</td>
<td></td>
</tr>
</tbody>
</table>

Between the lower and upper breakpoint: moderate rains. These rains generally account for 30 to 50% of all rain events (by number), but produce 75 to 90% of all the runoff volume. The rain depths associated with the upper breakpoint range from about 1 to 2 in in the arid parts of the United States and up to 5 or 6 in in wetter areas. These runoff volume distributions are approximately the same as the pollutant distributions. Therefore, these intermediate rains also account for most of the pollutant mass discharges and many of the actual receiving water problems associated with stormwater discharges.

Above the upper breakpoint: large but rare rains. These rains include the typical drainage design events and are therefore quite rare. During the period analyzed, many of the sites only had one or two, if any, events above this breakpoint. These rare events do account for about 5 to 10% of the runoff on an annual basis. Obviously, these events must be evaluated to ensure adequate drainage.

The fourth category, evident in the Milwaukee monitoring results and shown in Figures 6.1 and 6.2, was not obvious during these computer analyses. These extremely rare events, which exceed the drainage capacity of most areas, do not significantly affect these long-term probability distributions. During the isolated years when they occur, such as during the monitoring period in Milwaukee, they have significant effects, but when averaged over long periods, their contributions diminish rapidly.

The small rains, generally less than about 0.5 in, are very important in a wet-weather monitoring program. They represent the vast majority of rains that occur in an area, and may represent the majority of runoff events. Water quality violations associated with wet-weather flows are typically common for these events. Similarly, the medium-sized events (from the 0.5-in rains to rains of several inches in depth) contribute the majority of runoff volume and mass pollutant discharges and are therefore likely responsible for most of the biological effects (especially habitat destruction and sediment contamination) in receiving waters. The largest rains (greater than several inches) are the primary focus of drainage design. Therefore, efforts must be made to characterize runoff and receiving water conditions in each of these different categories in order to understand the varying receiving water problems that may be occurring.

Pollutant Transport

The routing of pollutants through a watershed is a complex issue and beyond the scope of this book. One of the most important goals of a monitoring effort is collecting representative samples. In many cases, pollutant routing can affect pollutant concentration distributions. At outfalls, or in receiving waters, stormwater pollutant concentrations are random, with little of the observed variations being explainable by normal parameters (such as time since the event started or rain depth). As noted by Roa-Espinosa and Bannerman (1994), obtaining many discrete subsamples over the event duration likely results in a composite sample that has pollutant concentrations very similar to a flow-weighted composite sample. However, if collecting samples from a relatively small homogeneous area (such as a paved parking area), high concentrations of practically all pollutants are commonly observed near the beginning of the rain.

This “first-flush” phenomenon is most prevalent for rains having relatively constant intensities and for small areas. As a drainage area size increases (or as the surfaces become more complex, such as in a residential area), multiple first-flush waves travel through the drainage system, arriving at a single downstream location at different times. This moderates obvious concentration trends with time during the event. Also, as the rain intensity varies throughout an event, the washoff of pollutants at the sources also varies. Peak washoff occurs during periods of peak rain energy (high rain intensity). Therefore, periods of high concentrations may also occur later in a rain, as high intensities occur. Generally, lighter (more soluble) hydrocarbons and the smallest particles will “always” show a first-flush of high concentrations from small paved areas, while larger particulates and heavier hydrocarbons will wash off more effectively with high rain energies, which may occur randomly during a rain.

Sampling strategies must therefore consider these possible scenarios. The most effective sampling (but most expensive) is flow-weighted composite sampling throughout the entire storm event.
However, compositing many discrete subsamples collected throughout the event is likely to result in similar concentration values. If sampling a small critical source area (such as a gas station or convenience store), it may be useful to obtain an initial sample during the first few minutes of the event, and a composite over the complete event. In all cases, it would be difficult to justify analyzing many individual discrete samples collected throughout an event.

**Flow Monitoring Methods**

There are a wide variety of methods (Table 6.2) to determine flow in open and closed (e.g., conduit) channels. For additional information, see EPA (1982 and 1987a). Most flow measurements to assess receiving water effects from stormwater are conducted in relatively small streams. Often, channel cross-sectional area is determined and the velocity measured at intervals across the channel using a current meter. In some situations, discharge from a pipe, notched weir, or small dam can be caught in a container of known volume and mean fill-up time used to calculate flow (e.g., liters per second). A variety of flumes and weirs have been used successfully in assessing flow and runoff.

Mechanical current meters are commonly used because they are simple, rugged, accurate at low velocities (0.03 m/s, 0.1 ft/s) and operate at shallow depth (0.1 m). A manufacturer’s calibration table converts the meter rotation number into meters or feet per second. Many modern meters are direct reading. The mean velocity at each cross-sectional interval is multiplied by the area of the subsection to calculate volumetric flow for each subsection. These are then summed to obtain the total stream flow.

Salt or fluorescent dyes have been used effectively to estimate velocities and time of passage when other methods are not practical, especially for highly irregular stream shapes or highly turbulent low flows. They depend on determining the amount of dilution that a known concentration of tracer receives as it mixes in the stream. The velocity between two stations is determined by knowing the travel time of the dye, or by comparing the dilute dye concentration to the injected dye concentration. The tracer may be added continuously or as a slug. A common tracer is Rhodamine WT dye which is measured with a fluorometer.

Flow monitoring in streams and other open channels is usually a necessary component of receiving water investigations. Flow estimates need to be made whenever any in-stream measurements are made, or samples collected, for example. In addition, equipment for continuous flow monitoring must be periodically calibrated using manual procedures. The following paragraphs briefly describe several common manual flow monitoring procedures.

**Drift Method**

The drift method is simply watching and timing debris floating down the stream. This velocity is then multiplied by the estimated or measured stream cross-sectional area to obtain the stream discharge rate. Of course, this method is usually the least accurate flow estimation method. The accuracy can be improved by choosing drift material that floats barely under the stream surface (such as an orange). Do not use material that floats high in the water (such as Styrofoam debris, for example), as it will be strongly influenced by winds. Drift measurements made in the center of a stream will tend to be the highest stream velocities, so the values should be reduced (by roughly 0.6, but highly variable) to better represent average stream flow rates.

**Current Meter Method**

The most traditional method of measuring flow is using a mechanical current meter. This method requires at least two people (one person should never be working alone near a stream anyway), a current meter, and simple surveying equipment. The stream discharge is measured at a cross section, usually selected along a relatively straight stretch (about 10 stream widths downstream from any major bends). If the stream discharge is being used to calibrate a stage recorder for continuous
### Table 6.2 Methods for Flow Measurement and Their Application to Various Types of Problems

<table>
<thead>
<tr>
<th>Device or Method</th>
<th>Flow Range Measurement</th>
<th>Applicable to Type of Water and Wastewater</th>
<th>Cost</th>
<th>Ease of Installation</th>
<th>Accuracy of Data</th>
<th>Pressure Loss thru the Device</th>
<th>Volumetric Flow Detector</th>
<th>Flow Rate Sensor</th>
<th>Transmitter Available</th>
</tr>
</thead>
<tbody>
<tr>
<td>Formula</td>
<td>Small to large</td>
<td>All</td>
<td>Low</td>
<td>NA</td>
<td>Fair</td>
<td>NA</td>
<td>NA</td>
<td>NA</td>
<td>NA</td>
</tr>
<tr>
<td>Bucket and stopwatch</td>
<td>Small</td>
<td>All</td>
<td>Low</td>
<td>Fair</td>
<td>Good</td>
<td>NA</td>
<td>NA</td>
<td>NA</td>
<td>NA</td>
</tr>
<tr>
<td>Floating objects</td>
<td>Small to medium</td>
<td>All</td>
<td>Low</td>
<td>NA</td>
<td>Good</td>
<td>NA</td>
<td>NA</td>
<td>NA</td>
<td>NA</td>
</tr>
<tr>
<td>Rotating elements</td>
<td>Small to medium</td>
<td>All</td>
<td>Low</td>
<td>NA</td>
<td>Good</td>
<td>NA</td>
<td>Yes</td>
<td>NA</td>
<td>Yes</td>
</tr>
<tr>
<td>current meters</td>
<td></td>
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<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Dyes</td>
<td>Small to medium</td>
<td>All</td>
<td>Low</td>
<td>NA</td>
<td>Fairly good</td>
<td>NA</td>
<td>NA</td>
<td>NA</td>
<td>NA</td>
</tr>
<tr>
<td>Salt dilution</td>
<td>Small to medium</td>
<td>All</td>
<td>Low</td>
<td>NA</td>
<td>Fair</td>
<td>NA</td>
<td>NA</td>
<td>NA</td>
<td>NA</td>
</tr>
<tr>
<td>Magnetic flowmeters</td>
<td>Small to large</td>
<td>All</td>
<td>High</td>
<td>Fair</td>
<td>Excellent</td>
<td>None</td>
<td>Yes</td>
<td>Yes</td>
<td>Yes</td>
</tr>
<tr>
<td>Weirs</td>
<td>Small to large</td>
<td>All</td>
<td>Medium</td>
<td>Difficult</td>
<td>Good to excellent 2–5%</td>
<td>Minimal</td>
<td>Yes</td>
<td>Yes</td>
<td>Yes</td>
</tr>
<tr>
<td>Flumes</td>
<td>Small to large</td>
<td>All</td>
<td>High</td>
<td>Difficult</td>
<td>Good to excellent 2–5%</td>
<td>Minimal</td>
<td>Yes</td>
<td>Yes</td>
<td>Yes</td>
</tr>
<tr>
<td>Acoustic flowmeters</td>
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<td>All</td>
<td>High</td>
<td>Fair</td>
<td>Excellent 1%</td>
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<td>Yes</td>
<td>Yes</td>
<td>Yes</td>
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</table>

*Assume proper installation and maintenance.

flow monitoring, the cross section being measured must not be affected by backwater conditions. If the selected cross section is in the vicinity of sampling and will not be used to calibrate a flow equation but will be used to determine the instantaneous current conditions at the time of sampling, then backwater influences and affects from meanders need to be included in the measurements. Instantaneous flows are determined using current meters to document flows occurring in a sampling period. However, this procedure can also be used to calibrate a state–discharge curve that can be used in conjunction with a conventional continuous stage recording device for long-term studies. Figures 6.3 and 6.4 illustrate common current meters used for stream studies.

In order to calibrate a flow or discharge model (especially the Manning’s equation), the stream is assumed to have normal flow where the water surface is parallel with the stream bottom. This is unusual under real stream conditions, where actual water surface profiles exist. In this case, Manning’s equation can still be used, but by substituting the friction slope for the water surface (or stream bed) slope. The friction slope is elevated above the water surface by the velocity head \( (v^2/2g) \). It is therefore easy to adjust the surveyed water surface slope to the friction slope by adding the velocity heads at the upstream and downstream locations. The calibration procedure usually involves calculating the Manning’s roughness factor \( (n) \) in the stream stretch. Manning’s equation is (in U.S. customary units):

\[
V = 1.49\left( R^{2/3} \right) S_f^{0.5}/n
\]

where
- \( V \) = velocity of the open channel flow (ft/s)
- \( R \) = hydraulic radius (area/wetted perimeter, ft)
- \( S_f \) = friction slope
- \( n \) = Manning’s roughness coefficient

Biological monitoring is normally conducted during relatively low flow periods, whereas Manning’s equation was developed for channel design for large, rare events. Manning’s equation is a
conservative design formula (when using the published roughness coefficients). It is not an analysis method and it must be used with care during low flow conditions. During low flows, the roughness coefficient is usually much greater than during high flows, for example, requiring equation calibration at different stream stages.

Current meter flow monitoring requires that the stream be divided into several sections. About 10 sections from 1 to several feet wide are usually adequate, depending on overall stream width. The depth of the stream is measured at each section edge, and the current velocity is measured in a vertical profile in the center of each section. The average velocities in each section are multiplied by the section areas to obtain the discharge rates for each section. These are then summed to obtain the total stream discharge. Table 6.3 is an example calculation for a section on Cahaba Valley Creek, in Shelby County, AL, that is generally used as a field demonstration site for UAB hydrology classes. Figure 6.5 is a cross-sectional diagram of this site, also showing the flow profile distributions. It is interesting to note that the peak water velocity for this stream section is seen to be near the bottom of the stream, close to the middle, but off-set, likely due to the slight meandering of the stream at this location. This is in contrast to the typically assumed velocity profile where the peak velocity is very near the top of the stream (and near the center). Figures 6.6 and 6.7 are photographs of a UAB hydrology class obtaining current measurements at this location.

Stream discharge monitoring is obviously a multiperson job, both from a safety standpoint and in order to take the actual measurements. A safety throw rope should always be ready, and great care should be exercised when working in a fast-moving or deep stream. If a stream has too great a velocity (especially greater than about 2.5 ft/s), or if it is too deep, then current measurements

Table 6.3 Example Calculation for Flow and Current Measurements

<table>
<thead>
<tr>
<th>Section Interval (ft)</th>
<th>Midpoint, Distance from Shore (ft)</th>
<th>Depth at Midpoint (ft)</th>
<th>Section Area (ft²)</th>
<th>Velocity at 0.2 Depth (ft/s)</th>
<th>Velocity at 0.8 Depth (ft/s)</th>
<th>Average Velocity (ft/s)</th>
<th>Discharge (ft³/s)</th>
</tr>
</thead>
<tbody>
<tr>
<td>0–1</td>
<td>0.5</td>
<td>0.21</td>
<td>0.2</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>1–3</td>
<td>2</td>
<td>0.74</td>
<td>1.5</td>
<td>0.3</td>
<td>0.4</td>
<td>0.4</td>
<td>0.6</td>
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<td>3–5</td>
<td>4</td>
<td>1.42</td>
<td>2.8</td>
<td>1.1</td>
<td>1.6</td>
<td>1.4</td>
<td>3.9</td>
</tr>
<tr>
<td>5–7</td>
<td>6</td>
<td>1.70</td>
<td>3.4</td>
<td>1.8</td>
<td>2.0</td>
<td>1.9</td>
<td>6.5</td>
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<tr>
<td>7–9</td>
<td>8</td>
<td>1.93</td>
<td>3.9</td>
<td>1.5</td>
<td>2.5</td>
<td>2.0</td>
<td>7.8</td>
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<tr>
<td>9–11</td>
<td>10</td>
<td>1.94</td>
<td>3.9</td>
<td>1.4</td>
<td>2.5</td>
<td>2.0</td>
<td>7.8</td>
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<td>11–13</td>
<td>12</td>
<td>1.79</td>
<td>3.6</td>
<td>2.0</td>
<td>3.0</td>
<td>2.5</td>
<td>9.0</td>
</tr>
<tr>
<td>13–15</td>
<td>14</td>
<td>1.54</td>
<td>3.1</td>
<td>1.5</td>
<td>2.2</td>
<td>1.9</td>
<td>5.9</td>
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<tr>
<td>15–17</td>
<td>16</td>
<td>1.19</td>
<td>2.4</td>
<td>0</td>
<td>0</td>
<td>0</td>
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<tr>
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<td></td>
<td>26</td>
<td></td>
<td>1.6</td>
<td></td>
<td>42</td>
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</tr>
</tbody>
</table>

Figure 6.5 Cross section of stream velocities (ft/s) at Cahaba Valley Creek, Shelby County, AL.

Figure 6.6 Obtaining elevation contours at Cahaba Valley Creek, Shelby County, AL.
should be conducted from a bridge, or cable system, and personnel should not be allowed to enter the water. Urban streams are also known for hidden debris and very soft bottoms. As in all work in urban streams, waders are necessary to minimize water contact and to prevent injuries from sharp objects. Riparian plants (such as poison oak and poison ivy) and slippery banks can also present additional hazards near streams. And do not step on any water moccasins.

A suitable current meter is obviously needed for a stream discharge study. Direct-reading digital meters (instead of older audible counter meters, where the operator must count clicks that are related to the water velocity) are now most commonly used. The current meter should be able to measure to 0.1 ft/s, have a threshold velocity of at least 0.2 ft/s, and preferably have an averaging mode in addition to an instantaneous mode. The meter should also be capable of measuring in very shallow water and next to the stream bottom (within a few inches of the stream bottom). The readout should also be selectable between metric and U.S. customary units. The meter must be recalibrated at least every year, preferably in the manufacturer’s tow tank or in an open channel test facility. Numerous hand-held current meters are available. Forestry Suppliers, Inc. (800-543-4203) has several different mechanical models, as listed below:

- Swoffer Model 2100-1514 (#94161) 0.1 to 25 ft/s 1% accuracy $2300
- Handheld Flowmeter (#94303) 0.5 to 25 ft/s ± 0.5 ft/s accuracy $700
- Gurley Model 625 Pigmy (#94993) 0.05 to 3 ft/s audible counter $1320
- Gurley Model 625 Pigmy (#94983) 0.05 to 3 ft/s digital indicator $2600
- Gurley Model D622F Meter (#94982) 0.2 to 32 ft/s digital indicator $2940

All of these current meters meet the desirable performance criteria, except for the much less expensive flowmeters. Newer portable meters are available that have no moving parts, typically using sonic pulses and Doppler measurements of reflected sound waves from moving particles in the water. These meters are costly (>$3000) and may have a more limited life span than the traditional current-driven meters.

An engineering level, rod, stakes, and tape are also needed to measure the water surface slope between adjacent cross sections when calibrating Manning’s equation. Fiberglass tapes are suitable for measuring the stream widths, and rigid (but thin) rules are useful for measuring water depth at the stream sections. When measuring water velocities with a current meter, operators must stand to the side and behind the meter and ensure that no turbulence from their legs (or from others) affects the measurements.

**Flow Monitoring Using Tracers**

The most precise method of stream current measurements is through the use of tracers. This method is especially important when measuring flows in areas having karst conditions where surface waters frequently lose and/or gain substantial flows to and from underground flows. A single upstream dye injection location and multiple downstream sampling stations through the study area are used in this situation. Tracers are also needed if there is an obviously large fraction of
of inter-bed flow or if the stream flow is very turbulent. The flow in very shallow streams, especially when the stream is cobble lined, is also very difficult to monitor with current meters, requiring the use of tracers. Another common use of tracers is when measuring the transport and diffusion of a discharge into a receiving water. Hydraulic detention times in small ponds and lakes can also be determined using tracers. Orand and Colon (1993) state that the use of tracers for water discharge measurements is not a new concept. They admit that the use of current meters is usually much simpler and therefore more common. However, current meters are not applicable in many situations, as noted above. As an example, they routinely use dye tracers and a field fluorometer with continuously recording output to measure the discharge of very turbulent mountain streams, which would not be possible with current meters.

Unfortunately, tracers are rarely useful for continuously monitoring flows, but they can be used for instantaneous flow determinations or for calibrating conventional continuous flow monitoring equipment in actual installations.

Brassington (1990) lists the desired traits for a tracer:

- An ideal tracer should be detectable in very small concentrations.
- It should not be naturally occurring.
- If an artificial tracer is being used, it should exhibit conservative behavior.
- It should be safe to use and produce no harmful environmental effects.
- It should be relatively inexpensive and readily available.

Three main classifications of tracers are generally used. Dyes give a specific and distinctive color to the water that can be detected easily. Chemicals, especially naturally occurring salts, can be used effectively if a discharge into a receiving water has a unique water chemistry and the tracer study objective is to determine the behavior of the discharge. Mechanical tracers can also be used to tag the water, much like the drift method described previously.

The most common mechanical tracer is a spore of Lycopodium, a club moss (Brassington 1990). The spores can be dyed and used to measure the surface and groundwater interactions in complex systems. Another approach in monitoring complex surface–groundwater interaction is to use bacteriophages to trace groundwater movement, including the role of septic tank discharges on local receiving waters. Paul et al. (1995) injected prepared bacteriophage cultures (φHSIC-1 and Salmonella phage PRD1) as viral tracers, along with 1-µm fluorescent spheres and fluorescein dye, into septic tanks and injection wells and identified their presence in local surface waters in Key Largo, FL. They found relatively rapid movement of the viral tracers (from 0.5 to 25 m/h) in the subsurface limestone environment into the surrounding marine waters. These rates were more than 500 times faster than had been previously measured. They concluded that the subsurface flows may not have reflected uniform diffusion through a homogeneous matrix, but were rather “channeling” through the limestone. Another possibility they suggested was that viruses travel like colloids through the subsurface, moving faster than the bulk water flow. They concluded that the bacteriophages were much more efficient than the fluorescent tracers due to their much better detection limits.

The most efficient tracer is a naturally occurring one. Johnson (1984) concluded that using naturally occurring materials (such as salinity, turbidity, temperature, or other suspended or dissolved materials) allows much more data to be collected and is usually relatively inexpensive (compared to using artificial tracers). In order to use a natural tracer, the material must be:

- Conservative
- Highly soluble under a variety of conditions
- Not amenable to sorption or precipitation or degradation
- Linear with mixing
- Present in greatly contrasting concentrations in the two water bodies that are mixing
The tracer must also be easily and cheaply analyzed. In many cases, specific conductivity can be used. Specific conductivity is especially useful when examining freshwater inflows into saline receiving waters. Field et al. (1994) described the use of specific conductivity to measure the effectiveness of a combined sewer overflow (CSO) capture and control device in Brooklyn, NY. In this example, the CSO (which had a specific conductivity of about 1000 \( \mu \text{S/cm} \) and a standard deviation of about 250 \( \mu \text{S/cm} \)) was contrasted with Fresh Creek water (which had a specific conductivity of about 20,600 \( \mu \text{S/cm} \) and a standard deviation of about 2600 \( \mu \text{S/cm} \)). Standard conductivity meters were used to trace the CSO water as it displaced the Fresh Creek water in the treatment facility during rains, and to measure the leakage of Fresh Creek water into the treatment facility between rains, as shown in Figure 6.8. The mass \((M)\) of the tracer is equal to the water volume \((V)\) times the concentration \((P)\). It does not matter that there is no adequate conversion for specific conductivity to be expressed as a mass, as specific conductivity concentrations were shown to be linearly related to dilution with the receiving water. A Monte Carlo mixing model was used to calculate the unknowns in this diagram, considering the variabilities of the concentrations in the two water bodies. Stable isotopes have been used successfully as tracers by some researchers with access to sensitive mass spectrophotometers, if the waters being distinguished have a sufficiently different source (Sangal et al. 1996). Ratios of major ions have also been used successfully to identify different waters, especially in groundwater studies.

In most cases, naturally occurring tracers cannot be effectively used because of their non-conservative behavior, insufficient concentration contrasts, or expense. A later section in this chapter discusses the use of natural tracers to identify sources of discharges. Commercially produced fluorescent dyes have been available for many years and have been extensively used for water tracer analyses. Fluorescein (a green fluorescent dye) has been used since the late 1800s, for example, but is not very stable in sunlight. However, it is still commonly used in visual leak detection tests and to visually trace discharge connections (such as determining if floor drains are connected to the sanitary wastewater lines or the storm drain system). Color Figures 6.1 and 6.2* show fluorescein being used to trace sanitary sewage connections to a storm drainage system in Boston.

Rhodamine B was used in the 1950s for water tracing in Chesapeake Bay because it was more stable in sunlight than fluorescein, but it readily adsorbed to sediments, making quantitative measurements difficult (Johnson 1984). Forestry Suppliers, Inc. (800-543-4203) sells liquid and compressed tablets and cakes of Rhodamine B and fluorescein for visual tracer work (but not for use

* Color Plates follow p. 370.
near water intakes). Bottles of 200 tablets of either dye, having a total weight of about 10 oz., or a 3” donut, also weighing 10 oz., of either dye costs about $35.

The most common artificial tracer currently used is Intracid Rhodamine WT dye, a 20% (by weight) stock of dye in water and other solvents having a specific gravity of 1.2. It is available from Crompton and Knowles (Reading, PA, 215-582-8765), at about $400 per 10 L. It is greatly diluted before use in the working stock solution for continuous dye injection studies. Chemical and laboratory suppliers also sell much more dilute mixtures (but at a much greater cost per unit of dye). Forestry Suppliers, Inc., sells a 1-gallon bottle of Rhodamine WT, unspecified dilution, (catalog #92969) for about $100, and bottles of 200 compressed Rhodamine WT tablets (catalog #92991) (weighing 11 oz.) for about $36.

Rhodamine WT was specifically developed in the 1960s for water tracing applications and is much superior for quantitative work compared to the earlier dyes. It is generally easier to detect in much lower concentrations, less toxic, has lower sorption to particles, and exhibits slower decay. Even though it is very expensive by volume, its very low detection limit (about 0.01 ppb of the 20% stock solution) and conservative behavior make it cost-effective.

Rhodamine WT is generally thought to have low toxicity; however, the USGS limits its concentrations at water supply intakes to 10 ppb (Johnson 1984). The biggest toxicity problem associated with Rhodamine WT is apparently associated with reactions with very high concentrations of nitrates. In all cases, it is important to contact local drinking water and state water regulatory agencies when planning a dye tracer study. The largest concern is probably associated with complaints of red water (which should not occur if proper dye concentrations are used).

The Corps of Engineers (Johnson 1984) has published a comprehensive description of the use of water tracing using fluorescent dyes. This report stresses monitoring inflows into reservoirs, with information applicable for a wide range of surface water conditions, including small streams, large rivers, and lakes. Johnson (1984) reports that no significant decay of Rhodamine WT is likely to occur due to chemical or photochemical decay for conditions found in natural waters. However, high chlorine levels (several mg/L, such as are found in many drinking waters) can cause significant decay during long exposure tests (tens of hours). As an example, Johnson reports that chlorine concentrations of 5 mg/L in tests run over 20 hours caused about a 5% decay of fluorescent activity. If operating in urban areas, where the chlorine concentrations may be periodically high or the turbidity variable, it is important to test decay and sorption of the dye. This is best done by using actual receiving water collected at the time of the tracer study as the dilution water when preparing the dye standards. These standards should be compared to standards using proper laboratory dilution water (preferably prepared using ion exchange, and/or reverse osmosis, as laboratory distilled water can contain very high chlorine levels).

Johnson (1984) states that total fluorescent decay of Rhodamine WT is probably about 0.04/day, from both sorption and photochemical decay. Almost all of this loss is likely associated with sorption. The sorption of Rhodamine WT onto particles, according to Orland and Colon (1993), had less than a 7% effect on the measured stream discharges (overestimated) in water having suspended solids concentrations ranging from 200 to 2000 mg/L (particle diameter <200 µm).

Johnson (1984) also reports the effects of pH, temperature, and salinity on the fluorescence of Rhodamine WT. The most serious problem with precise measurements is that the fluorescent intensity decreases with increasing temperature, requiring temperature corrections. This change is a decrease in fluorescence by about 5% for every 2°C increase in temperature. If collecting discrete samples that are brought back to the laboratory for analysis, the samples and the standards can be kept at the same temperature for analysis, eliminating this problem. In situ fluorescent measurements require temperature corrections (available as an option in the Turner Designs 10-AU, for example). It is recommended that discrete samples also be periodically collected, along with the continuous field measurements, for temperature-controlled laboratory analysis to confirm the automatic corrections.

The pH of the receiving water affects the sorption of the Rhodamine WT to organic material. Below a pH value of 5.5, the carboxyl acid group of the dye becomes protonated, increasing
adsorption. Johnson (1984) reviewed studies that showed that humic sediment solutions of 2.0 and 20 g/L and 100 ppb Rhodamine WT caused 18 and 89% decreases in fluorescence, respectively. The high humic concentrations lowered the pH values of the water and increased the organic content of the water. In similar solutions using a kaolinite clay, the fluorescent losses were only 11 and 23%. These clay concentrations are very high (2000 and 20,000 mg/L) and would be likely to occur only in construction site runoff in urban areas. The very high associated turbidity of these samples would also greatly complicate fluorescent measurements. The samples would likely have to be clarified (by centrifuge or filtering) before measurement (see also below).

The most commonly used fluorescent measurement instrumentation for fluorescent dye studies has been the older and obsolete Turner model 111 fluorometer that is still available in many laboratories, and the newer Turner Designs (408-749-0994) model 10-AU fluorometer (Figures 6.9 and 6.10). Both of these instruments are filter fluorometers and are very sensitive. The Turner Designs 10-AU is a much superior unit for field measurements, as it is designed to operate on 12-volt batteries, has newer and more stable electronics, a wider dynamic range, and has a water-resistant case. It is also suitable for laboratory measurements. The Turner Designs unit also has a flow-through cell, plus built-in temperature correction and data logging options, which are convenient for field use.

The downstream dye concentrations should be measured over a long period and at many locations across the stream to obtain the best flow estimate. In practice, an automatic water sampler is used to obtain samples, or manual grab samples are obtained, at the downstream location for laboratory analyses, or less commonly, a flow-through fluorometer is used to measure the dye concentrations on a real-time basis. If manual sampling is used, subsamples can be obtained from several locations across the stream for compositing. If a flow-through instrument is used, the intake can be moved to various locations across the stream to investigate mixing conditions. In all cases, the downstream location should be well beyond the predicted fully mixed area. Variations in dye concentrations observed are therefore assumed to be associated with flow variations in the stream.

Background fluorescence in the water must be determined before and during the test. During some tests, we have detected residual background fluorescence. In receiving waters affected by sanitary sewage (such as from raw overflows, inappropriate connections, leaks, septic tank influences, and treated effluent), background fluorescent can be very high due to detergents in the water. Almost all of this interference is eliminated by using specific Rhodamine WT filter sets in the fluorometer. The use of the actual water being tested as the injection water diluent during a continuous test reduces background problems, as do the highly selective optics available for Rhodamine WT analyses. However, background water samples need to be collected for analyses before any dye is added to the water. In addition, it is a good idea to collect upstream water samples before any dye is added to the water. In addition, it is a good idea to collect upstream water samples before any dye is added to the water.
periodically during the test to check for changing background conditions (especially important when conducting a tracer test in a sanitary sewer where background water quality can change dramatically over a relatively short period of time). If turbidity levels vary greatly during the test, Johnson (1984) recommends that the samples be filtered or centrifuged prior to analysis. Continuous dye analysis in the field does not allow a correction for turbidity (like the built-in temperature correction option available from Turner Designs), but periodic grab samples analyzed in the laboratory after turbidity reduction enables these effects to be determined.

An example of continuous background corrections was described by Dekker et al. (1998) using Rhodamine WT in Detroit to accurately calibrate flow-metering equipment. They found that abrupt changes in suspended solids in the sewage were very common and that this could radically change the fluorescent response. They therefore collected background (upstream) sewage samples every 15 min during the dye tests and prepared calibration curves with this water, changing the response factors for the measurements accordingly. They also monitored the light absorbance at the Rhodamine WT excitation wavelength (550 nm) simultaneously with the dye concentrations to screen out periods of abrupt changes in suspended solids that would affect the calibration curves.

The careful calibration of fluorometers is critical because of their great sensitivity. Calibration solutions from about 0.1 to 500 ppb should be used (these concentrations are in relation to the 20% stock solution). Two sets of calibration solutions need to be prepared. The initial laboratory series is prepared using laboratory-grade clean water, and another set must be prepared using the receiving water. As noted previously, if using distilled water, ensure that the chlorine concentrations are very low. Never use tap water. Deionized water (at 18 meg-ohm resistance) is probably the best. Preparing such low concentration standards requires a great deal of care, especially when withdrawing the stock and making the initial dilution. Needless to say, the largest hazard associated with working with Rhodamine WT is the mess that it can make if splashed or spilled. The stock solution is stratified in the shipping container, requiring stirring, but trying to stir or shake the stock container is a challenge, as it is heavy and minor spills or leaks are a great nuisance.

It is recommended that the amount of dye needed for the test be withdrawn from the stock shipping container, including the minor amount needed for preparing the standards. This will be only a very small amount, usually only a few hundred mLs for a slug dose test, or a few liters if conducting a continuous injection experiment in an urban stream. This aliquot doesn’t have to be perfectly representative of the stock solution. The goal is to withdraw the amount needed without spilling any, with minimal mixing. The initial dilution is usually made using 10 mL of the stock diluted in a liter of dilution water, using a volumetric flask. The 10 mL of stock is very dark and viscous, making it almost impossible to measure with a standard pipette. Many people weigh the initial amount, correcting for the 1.2 specific gravity, but unless the aliquot was from a well-mixed stock container, the specific gravity can be quite different. An automatic pipette (capable of handling viscous fluids) is probably better, as volume dilutions are being measured during the test. Serial dilutions are then usually made, making weaker and weaker standards. The strong concentrations foam if violently mixed, making it difficult to fill the volumetric flasks accurately to the calibration marks.

Analytical chemists do not approve of serial dilutions, as errors are easily compounded, but the nature of Rhodamine WT and the great dilutions needed would otherwise require measuring very small quantities of stock. Using a 1-µL pipette and a 1-L volumetric flask would only produce a 1 ppm (1000 ppb) solution, by volume. At least a second (serial) dilution would still have to be made to obtain a 1 ppb concentration, and a third dilution to obtain a fraction of a ppb standard. Inaccuracies associated with serial dilutions are probably less of a problem than trying to pipette such small amounts.

Fluorescent analyses can be conducted in the field or in the laboratory. In situ (flow-through) dye analyses (for which the Turner Designs 10-AU is specifically designed) can be much more efficient than collecting water samples and bringing them back to the laboratory for analyses. However, a combination approach is usually best, where periodic samples are collected and brought to the laboratory for temperature controlled analyses for comparison to the in situ values. The
situ analyses allow immediate evaluation of the sampling program, especially when the dye is being used at proper concentrations, making it nearly invisible to the eye, or if complex hydraulics (such as in an estuary with strong currents) prohibit easy prediction of the flow path. However, using a fluorometer in flow-through mode presents special problems. Johnson (1984) stresses the need to ensure that all water connections are air tight to prevent bubbles from entering the flow path. In addition, the pump should be located above the light cell to decrease bubbles from leaky pump seals. The intake of the water delivery system should also be screened to decrease the chance of sand and other debris from scratching the instrument optics.

The two main types of dye injection include instantaneous or continuous releases. Instantaneous dye releases are much more efficient in the use of dye. The amount of dye quickly added to the water usually results in a visible dye cloud that is easy to follow manually. In addition, no special dye injection equipment is required, as the dye is simply poured quickly into the water body. However, continuous releases of dye, especially in conjunction with in situ analyses, is necessary when simply tracking the dye is challenging. Continuous dye releases require substantially more dye and usually more field personnel, but changing conditions can be easily measured (Color Figures 6.3 and 6.4).

Thomann and Mueller (1987) present a USGS method used to estimate the amount of Rhodamine WT dye needed for an instantaneous release experiment. The amount is usually much less than needed for a continuous release experiment. They also present several methods to evaluate the observations and obtain estimates of flow, diffusion coefficients, and recovery of dye.

Continuous release rates of dye are dependent on the desired downstream concentration of dye, the concentration of the dye being released, the injection rate, and the estimated stream discharge. Figure 6.11 shows a basic mass balance for a discharge into a river or stream. This can be easily applied to a dye injection experiment, with the dye being considered as the effluent being discharged into the receiving water.

The mass balance for this situation is:

\[ Q_u s_u + Q_e s_e = Q_s \]

where

- \( Q_u \) = upstream flow rate
- \( s_u \) = upstream concentration
- \( Q_e \) = effluent discharge (or dye injection) rate
- \( s_e \) = effluent (or dye injection solution) concentration
- \( Q \) = resulting downstream discharge rate (equal to \( Q_u + Q_e \))
- \( s \) = resulting downstream concentration

\[ W = Q_e s_e \]

Solving for Q, the downstream discharge rate:

\[ Q = (Q_u s_u + Q_e s_e)/s \]

If the background concentration \( s_u \) is zero (as desired in a tracer experiment), this further reduces to:

\[ Q = Q_e (s_e/s) \]

where \( (s_e/s) \) is the dilution ratio of the dye.

Therefore, the stream discharge (Q) is the ratio of the concentration of the dye injection solution \( s_e \) to the measured downstream dye concentration \( s \), multiplied by the dye injection rate. As an example, assume the following conditions:

- \( Q_e = 10 \text{ cm}^3/\text{s} \)
- \( s_e = 1.0 \) (injection dye solution concentration, a given arbitrary concentration of 1.0)
- \( s = 12 \text{ ppm}_{\text{vol}} \) compared to injection concentration (average dye concentration from numerous samples collected).

The average value for \( s \) was determined to be 12 ppm (relative to the injection dye concentrations); therefore, the calculated stream discharge rate is:

\[ Q = Q_e (s_e/s) = 10 \text{ cm}^3/\text{s} \left(1.0/12 \times 10^{-6}\right) = 830,000 \text{ cm}^3/\text{s} \]

This is equal to 830 L/s, or about 29 ft³/s (cfs). As noted in this example, the absolute concentration of the injection solution does not need to be known, as long as calibration solutions are made using the injection solution and the receiving water.

The injection solution needs to be discharged at a constant rate. This is made much easier by using a special metering pump (as supplied by Turner Designs, for example, or a battery-operated peristaltic pump available from Cole-Parmer). In all cases, someone must be at the injection site for the duration of the experiment to ensure that the discharged dye is well mixed and that constant pumping of the injection solution is occurring. This is achieved by periodically measuring the time needed to fill an appropriate graduated cylinder (retain some of the solution from the filled cylinder for use in later calibration solutions, and dump the remainder of the material from the cylinder into the stream when finished timing). The injection solution samples should be analyzed to detect variations in injection dye concentration during the study period.

Fortunately, as is evident from the above equation, everything is relative to the injection concentration, or the mass of dye used, with tracer work. The stock solution concentrate is never directly used in dye studies because the intense color would make the injection plume visible for a large downstream distance; also, the high 1.2 specific gravity affects the plume buoyancy, and precisely pumping very small dye injection rates is difficult. The stock is therefore greatly diluted (by about 10 to 100 times) to create an injection solution to minimize these problems. When conducting a continuous injection experiment, one measures the ratio in concentrations between the injection dye stream and the resulting receiving water concentration. This initial dilution causes a loss of sensitivity, so more dye is required in a continuous injection experiment. In small urban streams, this loss of efficiency is not too serious. When conducting a large-scale injection experiment, specific gravity adjustments are usually made and close to full-strength dye is injected to minimize costs. In a slug discharge test, much less dye is usually needed, and the full amount of tracer dye is introduced into the water as rapidly as possible (within a few seconds).
instantaneous tests, the strength of the dye solution is not important. It is only necessary to know
the mass of the dye used. Therefore, the small amount of dye needed can be effectively diluted in
a several-gallon container that can be rapidly poured into the stream to initiate the test.

Experimental conditions needed for various estimated stream discharges can be predetermined
by knowing the injection pump rates available and the sensitivity of the fluorometer. A Cole-Parmer
Masterflex peristaltic pump can supply a wide range of dye injection rates, depending on the pump
rotational speed and the size of tubing. With #13 tubing, the pump can be set to deliver between
0.2 and 0.5 mL/s. Number 16 tubing has a useful range of between 2.0 and 8.0 mL/s, while #18
tubing can be used between 10 and 40 mL/s. A Turner model 111 fluorometer has a range of
sensitivity from less than 1 to more than 150 ppb Rhodamine WT, depending on the sensitivity
setting. The newer Turner Designs model 10-AU has a much wider dynamic range. The combination
of these settings allows measurement of a wide range of flow rates. Table 6.4 illustrates some of
the flow rates that can be measured using some of these combinations. The downstream concen-
trations shown on this table are in relation to the injection concentration, which should be diluted
by at least 10 times compared to the 20% stock solution. Therefore, the downstream concentration
of 10 ppb shown may actually be closer to 1 ppb of the 20% stock. Intermediate downstream
concentrations should be targeted to ensure that variations in stream flow can be accommodated.
If a needed injection rate is too low, it may be unstable. The concentration of the dye being injected
should then be decreased so a higher pumping rate can be used.

As an example, consider a stream having an estimated discharge rate of 25 cfs and the target
downstream concentration is 25 ppb (compared to the injection dye strength which is diluted 10
times from the 20% stock solution; the actual downstream dye concentration is therefore about
2.5 ppb, which would be about mid-scale on the most sensitive setting for a Turner model 111
fluorometer). An injection rate of about 20 mL/s will therefore be required. Therefore, 2 mL of
20% stock will be used per second, or 120 mL of stock per minute of the test, or 7.2 L of stock
per hour of the test — a large amount of dye. The injection duration depends on the duration of
the steady flow period to be monitored. This should be long in comparison to the flow duration
from the injection location to the monitoring location to minimize sampling problems. The sampling
location must be located far enough downstream to ensure complete mixing. This length (in feet)
can be estimated using the equation presented by Thomann and Mueller (1987):

\[ L_{nm} = \frac{(2.6 UB^2)}{H} \]

where
- U = the stream velocity in ft/s
- B = the average stream width in feet
- H = average stream depth in feet

As an example, the discharge rate is estimated to be 25 cfs, the stream velocity is estimated to
be about 1 ft/s, the stream width about 25 ft, and the depth about 1 ft. The “complete mixing”
length is therefore about 1600 ft. About half of this distance would be needed if the dye injection

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<th>Table 6.4 Stream Discharge Rates (cfs) That Can Be Measured for Different Experimental Conditions</th>
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<td><strong>Injection Rate (mL/s)</strong></td>
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<tr>
<td>0.3</td>
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<tr>
<td>0.5</td>
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<tr>
<td>2</td>
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point is located at the centerline of the stream. The travel time needed (if injected at midstream) is about 13 min, at least. Therefore, an hour-long injection period would not be unusually long, requiring about 7 L of 20% Rhodamine WT dye, for this example.

**The Use of a Multiparameter Probe to Indicate the Presence, Duration, Severity, and Frequency of Wet-Weather Flows**

Most receiving water problems are highly dependent on the duration, severity, and frequency of wet-weather events. Habitat effects, for example, are greatly dependent on the frequency of erosive flows that cause bank instability. Sediment scour and deposition is also dependent on the flow energy. Bacteria, turbidity, and other water quality standard violations are much more serious if they occur commonly. Toxicity effects on receiving water organisms are also greatly dependent on the frequency and duration of exposure to excessive concentrations. Knowing when an event occurred, plus knowing the duration and severity of the event, is critical when conducting a long-term exposure experiment using many of the techniques described in this book. Therefore, knowing these basic wet weather event parameters is very important and enables a more complete evaluation of wet-weather problems in receiving waters. The following discussion presents a simple way to automatically monitor these important hydraulic characteristics in a stream without installing a permanent flow monitoring station.

Continuous sondes for water quality monitoring have been available for some time, but current models are vastly improved compared to earlier ones. It is now possible to deploy a water quality sonde for up to several weeks, with little drift and other degradation in performance. This allows the units to be left unattended for extended periods to obtain diurnal variations of constituents (such as DO, temperature, conductivity, turbidity, and water depth) for varying environmental conditions. One application is to examine the duration of degraded receiving water conditions following rains.

The following example is based on work by Easton et al. (1998) as part of an investigation studying the effects of SSOs (sanitary sewer overflows) on small urban streams. This study used YSI 6000 UPG water quality sondes to indicate the duration, frequency, and magnitude of wet-weather events in both surface waters and surficial sediments. Short-term, or runoff-induced, pollution effects can be studied in detail using these instruments. Long deployment time and the continuous monitoring capability of the YSI 6000 enables acquisition of data for multiple events, i.e., as many as occur during the time of deployment. The YSI 6000 UPG sonde is a multiparameter water quality monitor manufactured by YSI Incorporated, Yellow Springs, OH. The 6000 UPG is capable of performing a subset of the following measurement parameters: dissolved oxygen, conductivity, specific conductance, salinity, total dissolved solids, resistivity, temperature, pH, ORP (oxidation reduction potential), depth, ammonium/ammonia, nitrate, and turbidity. The 6000 UPG can be left unattended in the field for approximately 45 days, depending on the frequency of data logging and parameters being recorded. The instrument is constructed of PVC and stainless steel and is 3.5 in in diameter and 19.5 in in length. It weighs approximately 6.5 lb, with batteries. The sonde is capable of interfacing with an IBM PC-compatible computer for downloading data, or a hand-held unit can be used for direct observations. In addition, a software package, Ecowatch for Windows, is available for sonde setup, data acquisition, and data presentation/analysis. The sondes used in these experiments were configured to acquire the following parameters: dissolved oxygen, conductivity, specific conductance, temperature, pH, ORP, turbidity, and depth.

Five-Mile Creek (which is actually about 50 miles long) is a typical medium-sized Alabama stream, originating in a rural area, then flowing through a suburban, and then a heavily urbanized area. The flow in the creek ranged from approximately 2 to 10 m$^3$/s, depending on recent rainfall conditions. At each test site, one sonde was located on the creek bottom and the second was buried under approximately 6 in of sediment. The buried sondes were protected by placing them inside 75-mm-aperture nylon mesh bags and were used to measure interstitial water characteristics in situ and continuously. The sondes were anchored to the bottom by a chain attached to cinder
blocks. The cinder blocks were then attached to a tree to prevent the sondes from being washed downstream during major events. One set of sondes was located in an area having coarse sediments (stones of about 1 in in diameter), while the other set was located in an area having finer sediments (sandy grained).

The duration, frequency, and magnitude of runoff events is apparent from an examination of plots constructed from the sonde data (Figures 6.12 and 6.13). These sonde data show a large fluctuation in depth, specific conductance, and turbidity in the water column at both sites on July 1 at 5:00 pm, roughly corresponding to the 0.6 in of rain observed at the Birmingham International Airport several miles away. No site-specific rain information was available, as may be typical for many small-scale studies.

The rise period for all of the parameters was very rapid, and the peaks occurred very early in the runoff event. They then returned to previous levels within 1 to 2 days, depending upon the parameter. The data acquired for water depth are obviously the parameters that best correlate to tracking runoff hydrographs as they pass. There is an obvious change in flood stage (approximately 0.5 m increase in depth), as indicated on these figures. There were two slightly separated, but very similar, runoff hydrographs that passed through the creek; the depth data show two obvious peaks spaced about 3 hours apart. The other two parameters do not distinguish between these two separate, but close events, as is evident in the time taken to return to baseline (Tables 6.5 and 6.6). The turbidity and specific conductance data also substantiate the presence of a runoff event, but with an additional perspective on the duration of the potential effects from elevated turbidity levels and possibly other pollutants. Notice the almost immediate increase in depth and turbidity, and corresponding decrease in specific conductance. These changes are easily explained by a sudden increase in runoff water within the creek. Furthermore, the depth sensors indicate the timing and severity of the runoff event from a hydrologic perspective, while the specific conductance and turbidity sensors indicate the extended duration of probable adverse water quality conditions due to contaminated baseflows entering the stream.
The data in Tables 6.5 and 6.6 show the differences in water exchange between the water column and the interstitial water occurring in the two different sediment types (coarse and fine). These experiments show that the interstitial water at the coarse sediment site changes with the water column, although at a slightly reduced magnitude, while the interstitial water at the fine sediment site shows no change. Most urban streams have sediments represented by the fine sediment site (sand sized) or finer. Therefore, very little direct water exchange occurs between the water column and the interstitial water. The interstitial water quality is much more affected by the quality of the deposited sediments (especially decomposable material and toxicants) than by the water column quality directly. This rapid fluctuation of interstitial water in coarse-grained sediments has important implications on evaluations of sediment quality. The benthic micro-, meio-, and macrofaunal exposures in these environments will be more dynamic than typically assumed. Interstitial water sampling and sediment sampling were discussed in Chapter 5.

<table>
<thead>
<tr>
<th>Sonde Location</th>
<th>Magnitude of Change (µS/cm)</th>
<th>Time to Return to Baseline (hr)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Water column</td>
<td>210</td>
<td>42</td>
</tr>
<tr>
<td>Fine sediment</td>
<td>Not obvious</td>
<td>Not obvious</td>
</tr>
<tr>
<td>Water column</td>
<td>260</td>
<td>44</td>
</tr>
<tr>
<td>Coarse sediment</td>
<td>230</td>
<td>46</td>
</tr>
</tbody>
</table>

The duration of the water column effects from the wet-weather events is seen to be much greater than the duration indicated by the high flows alone (30 to 45 hours vs. 12 hours). This has a major impact on evaluating biological effects of the receiving waters. As an example, rains only occur for about 4.5% of all hours in Birmingham. Periods of extended high flows in Five-Mile Creek may occur for about 15% of the time. However, periods of elevated turbidity (and likely other constituents of concern) may occur for about 40% of the time. This extended time has a significant effect on in-stream beneficial uses and risk assessments from wet-weather toxicants and pathogens.

### In-Stream and Outfall Flow Monitoring

Monitoring of flows in storm drainage systems is typically done to supplement stormwater sampling activities. In most cases, flow monitoring equipment available from the same vendor that supplied the automatic water samplers is selected. The flow sensors typically measure depth of flow in the sewerage and apply Manning’s equation to calculate the flow rate and discharge. Unfortunately, Manning’s equation was developed as a design equation and not as an analysis equation. It was not intended for accurate measurements for shallow flows and does not consider debris that accumulates in sewerage. A better approach is to use a control section in the sewerage and calibrate a stage-discharge relationship. The ultimate solution is to use a special prefabricated manhole that contains a flume. Plasti-Fab (503-692-5460) offers many options of manhole and flume sizes and types for a broad range of sites and conditions. A less expensive alternative (and more suitable for temporary installations) is a manhole flume insert. These are available from Plasti-Fab and from Badger Meter (918-836-8411). These are installed in the discharge sewer line from a manhole, causing a backwater in the manhole that provides an accurate stage-discharge relationship that can be measured. Acoustical flowmeters (measuring water surface distances from a reference location above the water using reflected sound) or bubbler flowmeters (measuring the depth of water above the sensor based on hydrostatic pressure) are usually used to measure the water depth. If the storm sewer line is debris and obstruction free, Manning’s equation can be used, but a site-specific stage–discharge relationship must be developed and calibrated over a wide range of depths. Flow calibration is most effectively conducted using Rhodamine WT dye as a tracer, as described previously.

It is critical that the flow monitoring sites be selected to provide accurate flow measurements, along with providing safe and easy access. Sites for flow monitoring must meet numerous criteria in order to obtain accurate results. The most critical criteria require the absence of backwater conditions at the monitoring location and a reasonably straight and homogeneous stream character upstream of the monitoring location for a length of at least 10 times the stream width. Since the stream depth measurements will need to be translated into flow values using a depth–discharge curve, the stream banks and stream bottom need to be reasonably stable at the monitoring locations. The best way to provide the stability and constant stage–discharge relationship at a flow monitoring

---

**Table 6.6 Values for Magnitude of Change and Time to Return to Baseline for Turbidity, Due to Period of High Flow**

<table>
<thead>
<tr>
<th>Sonde Location</th>
<th>Magnitude of Change (NTU)</th>
<th>Time to Return to Baseline (hr)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Water column &gt;1000</td>
<td>30</td>
<td></td>
</tr>
<tr>
<td>Fine sediment</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Water column &gt;1000</td>
<td>30</td>
<td></td>
</tr>
<tr>
<td>Coarse sediment</td>
<td>210</td>
<td>30</td>
</tr>
</tbody>
</table>

station is to construct a control section, usually a flume or a weir. If the stream to be monitored is moderate in size and in a natural setting, especially with important in-stream biological resources, constructing a flume or weir is usually not practical.

The electronic components of typical in-stream flowmeters need to be secured near the stream edge, but outside the zone of common flooding. It would be best to secure them within a heavy steel contractor’s box permanently mounted onto an oversized concrete slab. A heavy padlock normally provides adequate security. This enclosure can also contain the necessary deep-cycle batteries recommended for power. If an external data logger is needed, it can also be secured within the box. In many instances, a solar panel can be installed to provide a trickle charge to the battery (but the solar panel would be exposed to vandalism, and riparian locations might be heavily shaded). The bubble tube can be easily run inside a steel pipe (2 to 3 in in diameter) buried in the stream bank. The upper end can come through the concrete pad directly into the steel instrument shelter. The lower end must terminate below the lowest expected stream depth, coming up through a moderate-sized concrete pad to protect the pipe and bubbler tube. The bubbling end must lie on top of the in-stream concrete pad and needs a heavy, but shallow, wire cage covering. This covering needs to be relatively easy to remove (while submerged) in order to provide intermittent service to the end of the bubbling tube. This installation can be easily upgraded to include an automatic water sampler, with the sampler (and its deep cycle battery) also enclosed in the steel shelter and the sampler tube also running down the pipe. If a water sampler is also to be used, a galvanized steel pipe must not be used because of zinc contamination. A very heavy-duty plastic pipe, sufficiently buried and protected may be suitable, or a much more expensive stainless steel pipe could be used to encase the bubbling and sampler tubes.

Another option for a shelter is to use concrete pipe rings stacked to a sufficient height and a steel plate padlocked to the top. This is a more temporary (and cheaper) alternative that usually works well. The bubbling tube should also be protected, if possible, within a large-diameter heavy plastic pipe. Another alternative is to mount the flowmeter and ancillary components on a road crossing where a stilling well can be run down into the water, usually on the downstream side of a bridge pier. The equipment can be mounted inside a heavy plywood box on top of the stilling well and accessed from the bridge. In this case, the pier may cause water level interferences.

Many flow measurement equipment vendors now offer simultaneous stage and velocity sensors. The velocity sensors directly measure the flow rate of the water, reducing the need for a stage–discharge relationship. The two major types of velocity sensors are the time-of-transit flowmeter and the Doppler flowmeter. Time-of-transit flowmeters use acoustical signals directed diagonally across the water flow path to a receiver. The acoustical signal travel time can be very accurately predicted. Any difference between the predicted and measured travel time is associated with the water motion. Accusonic (508-548-5800) is one vendor of these devices, which have been reliably used in large conduits. A series of three Accusonic sensors is placed in each of three parallel 10 ft × 15 ft CSO outfalls in Brooklyn, NY, as part of the Fresh Creek CSO treatment study (Field et al. 1995). The three sensor and receiver pairs in each outfall are placed in three vertical zones in each outfall, representing three layers of flow that can measure the severe backwater conditions due to daily tides. As an example, the individual sensors can measure tidal flows entering the bottom of the outfall and any floating CSO discharging on top of the saline receiving water.

Rob Washbusch and Dave Owens of the USGS in Madison, WI, recently (1998) tested several different flow monitoring devices simultaneously in a single storm drain pipe for comparison (Figures 6.14 to 6.19). A unique aspect of these tests was the use of continuous dye injection and downstream water sampling that was automatically activated when rainfall started. The samples were then brought to the laboratory for fluorometric determinations and actual flow values. These actual flows were then compared with the flows indicated by the different flow monitoring equipment. The box plots show the observations from 60 events examined over a 6-month period of time. Flow measurement errors of ±25% were not uncommon. They emphasize that these results
are for only one site (an industrial area in Madison, WI) and are not likely directly indicative of conditions that might be found elsewhere. They recommend that all runoff flow monitoring equipment be carefully calibrated at the time of installation and periodically rechecked.

Doppler velocity sensors are more commonly used in small storm and sanitary sewer lines. These reflect acoustic signals from particles flowing toward the sensors. The signals reflect off the fastest moving particles, and signal processing then determines the average water velocity. Several vendors sell Doppler units that are constantly improving in accuracy and ease of use. ADS Environmental
Services, Inc. (800-633-7246) maintains many large-scale flow monitoring networks around the world using its Doppler velocity and ultrasonic level sensors. ISCO (800-228-4373) also sells a Doppler unit that can be used in conjunction with its automatic water samplers. Unidata America (503-697-3570) sells the Starflow ultrasonic/Doppler flowmeter that is very compact and can be used in small open channels and sewer and drainage lines.

**Summary of Flow Monitoring Methods**

Table 6.7 is a list of some of the advantages and disadvantages of the different flow monitoring/measurement techniques that are most commonly used in urban receiving water studies. The previous discussion presented both manual flow monitoring procedures and methods for flow monitoring that can be used in conjunction with automatic water samplers. In most cases, standard bubble depth sensors supplied by the sampler manufacturer are probably the best choice for an automated station. However, these should be placed in a control section where the stage–discharge curve is specifically known and has been calibrated. Time-of-travel (sonic) current meters can be extremely valuable in situations where stratified flow may occur, but custom interfaces with the sampling equipment may be needed. Basic velocity meters are best used for more casual flow measurements, especially when flow measurements are being taken simultaneously with biological sampling. Dye testing is usually reserved for absolute calibration of flow monitoring setups and to measure in difficult situations, especially during low flow conditions in rocky streams where much of the flow may be actually occurring within gravel deposits, and in streams in karst areas where the interactions between surface and subsurface flows can be dramatic.
### Table 6.7 Comparisons of Available Flow Measurement Instruments

<table>
<thead>
<tr>
<th>Flow Monitoring Instrument Type</th>
<th>Advantages</th>
<th>Disadvantages</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Manual Instruments</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Velocity meters</td>
<td>Simple and rapid results</td>
<td>Instantaneous results, not long-term</td>
</tr>
<tr>
<td>Tracers (fluorescent dye)</td>
<td>Direct readout of current velocity</td>
<td>Requires multiple measurements across stream to obtain average condition. Can be dangerous during high flows.</td>
</tr>
<tr>
<td>Tracers (naturally occurring salts)</td>
<td>Considered the standard flow calibration procedure</td>
<td>May be subject to interferences from changing water quality (solids and temperature) or pipe materials. May be difficult to design and to conduct measurements for large systems. Required fluorometer is expensive.</td>
</tr>
<tr>
<td><strong>Automated Instruments</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Bubble sensor depth indicators</td>
<td>Simple and easy to interface with automatic samplers. Most choice and experience from many vendors.</td>
<td>Long-term placement</td>
</tr>
<tr>
<td>Propeller velocity meters</td>
<td>Direct measurement of current velocity.</td>
<td>More expensive and needed for each monitoring location</td>
</tr>
<tr>
<td>Time-of-travel (sonic) velocity meters</td>
<td>Direct measurement of velocity. Can be used to measure velocity of specific layer of the water to indicate shear; especially useful in tidal conditions with stratified water moving in different directions.</td>
<td>Only measures depth; requires stage-discharge relationship. Should be used in conjunction with a control section (weir or flume) and be verified with frequent velocity meter studies (not commonly done).</td>
</tr>
<tr>
<td>Acoustic velocity meters</td>
<td>Direct measurement of current velocity. Usually measures the peak velocity, and the average velocity for the relatively large sensing zone is calculated as a fraction of the peak velocity.</td>
<td>Foul easily and only indicate velocity at location of propeller. Relatively expensive and several may be needed to accurately measure flow in different flow strata.</td>
</tr>
</tbody>
</table>

**Rainfall Monitoring**

Rainfall data are very important when monitoring receiving water quality and quantity. As an example, the rainfall history in a watershed is needed before interpretation of biological monitoring data can be used to identify possible sources of degraded conditions. The hydrology texts listed previously all contain excellent summaries of rainfall aspects of importance in runoff studies. An especially important reference on rainfall depth measurements and interpretation is the *National Engineering Handbook Series* (Part 630, Chapter 4, Storm Rainfall Depth) published by the USDA (Soil Conservation Service, SCS, now the Natural Resources Conservation Service, NRCS), commonly referred to as NEH-4. This is available from the Consolidated Forms and Distribution Center, 3222 Hubbard Road, Landover, MD 20785. This handbook is supplied in a three-ring binder and sections are periodically updated.
Placement and selection of rain gauges are described in these references, along with calculating and interpreting watershed-wide rainfall. This section briefly summarizes several important aspects of rainfall monitoring not usually discussed in available reference texts, especially selecting the proper rain gauge network density and the need for calibration.

Rain gauges suitable for stormwater monitoring are available from many sources. A new small and self-contained weather station is available from Hazco (800-332-0435) that contains sensors for wind speed, wind direction, temperature, relative humidity, dew point, barometric pressure, and rainfall. It has a built-in data logger for up to 6 months of recording and is even available with a modem for connecting to a cellular telephone for telemetry. The cost is about $8500 (catalog #B-W010010M) with a modem and $6600 (catalog #B-W010010) without a modem. Tipping bucket recording rain gauges and data loggers, standard 8" rain gauges, and wind screens are available without the other sensors from several sources, including Qualimetries, Inc. (800-824-5873) and Global Water (916-638-3429) (Figures 6.20 and 6.21).

The other extreme in rainfall monitoring is the “Clear View” rain gauge from Cole-Parmer (800-323-4340) that is only about $35 (catalog #H-03319-10). This is a nonrecording rain gauge (having a 4” funnel diameter) requiring manual readings of the rain depth. Many other types of “garden store” accumulative rainfall gauges (Figure 6.22) are also available for as little as $5 each, including simple ones that can be made using 3-L plastic soft drink bottles (requiring the collected rain to be poured out and measured). As noted below, relatively few recording rain gauges (for accurate rainfall intensity measurements and start and end rain times) are needed for most urban catchment.
studies. However, numerous nonrecording gauges should be placed throughout the study area to indicate rainfall variations, especially for small rains.

**Determining Watershed Averaged Rainfall Depths**

Three methods are most commonly used to determine representative watershed-wide rainfall amounts from several point observations. These include the station-average method, the Thiessen polygon method, and the isohyetal method. These methods are briefly described in the following paragraphs.

**Station-Average Method**

The simplest and easiest method of estimating watershed-wide rainfall amounts is simply to compute the numerical average of all observed values in the watershed. Only those rain gauges physically located in the watershed of interest are usually considered. This method yields good estimates if most of the following conditions are present: the watershed has little topographical relief, a sufficient number of rain gauges are present, the rain gauges are reasonably uniformly distributed throughout the area, and the individual rain depths observed for the different rain gauges do not vary widely from the overall mean. The most important criterion is the need for a large number of rain gauges uniformly distributed throughout the area.

**Thiessen Polygon Method**

The Thiessen method uses a weighted average for the rain gauge network, based on the area assumed to be represented by each rain gauge. Closely spaced rain gauges have smaller weightings than do rain gauges spaced farther apart. The area weightings generally do not consider topography, or other watershed characteristics, although the polygons can be manually adjusted to account for these potential effects, with experience. The area represented by each station is assumed to be the area that is closer to it than to any other station. These areas are determined by drawing connecting lines between all adjacent rain gauges. These connecting lines are then bisected. The perpendicular bisectors then describe a polygon surrounding each rain gauge. Figure 6.23 is a simple illustration of the construction of the polygons surrounding each rain gauge. Figure 6.24 is an example of a Thiessen polygon system for the Toronto, Ontario, metropolitan area which has 35 rain gauges over an area of about 4000 km$^2$. These polygons were prepared using the SYSTAT computer program.

Results from the Thiessen polygon method are usually assumed to be more accurate than those obtained by the simple station-average method because the Thiessen method accounts for non-uniform distributions of stations. Rain gauge measurements from surrounding areas are also used in the analysis. The polygons also do not change for different rains, unless data are missing from one or more rain gauges. The weightings therefore are relatively constant, making the calculations reasonably simple for multiple rains, after the polygons are initially determined and measured.

**Isohyetal Method**

This is the most complex method for determining rainfall depths over a watershed and is usually considered the most accurate. It was rarely used before the common availability of computers that
simplified the necessary calculations. In contrast to the Thiessen polygon method, the isohyetal method requires extensive calculations for each individual rain event. In this method, contours of equal precipitation depth are constructed over the watershed. The construction of the contours can consider the presence of topographic or lake effects. The precipitation averaged over the entire area is computed by multiplying the area enclosed between adjacent isohyetal lines by the average rain depth values of the two adjacent isohyetal lines. Figure 6.25 is an isohyetal map (rain depths in mm) for a single rainfall over the Toronto area, using data from many individual rain gauges. This map was also prepared using SYSTAT.

The Toronto rain gauge network density resulted in small differences between the three averaging methods because of the large number of rain gauges available. The use of the 35 rain gauges was a lot compared to available rain gauge networks in most urban areas. The resulting errors in using the simple averaging method or the Thiessen polygon method, compared to the
isohyetal method, were all less than 1 mm in rain depth for rains of just a few mm in depth to over 25 mm in depth.

**Rain Monitoring Errors**

There are several common aspects of rainfall monitoring that can cause measurement errors. Most of these errors result in decreased rainfall values compared to true conditions. These include too few rain gauges for the area, poor placement of the rain gauges, wind effects, splashing of rain out of the gauge during high-intensity rains, tipping rate of tipping bucket rain gauge not keeping up with high-intensity rains, and calibration errors. These problems can usually be identified when reviewing the data. The errors can be corrected during the monitoring period, one hopes; otherwise the rain data might not be usable.

The easiest way to identify questionable rainfall data is to compare the site data with data collected from nearby and independent rain gauge locations. Residual analyses (differences between the site data and surrounding data) may indicate a consistent bias. This may be expected if there is a good reason for the bias (such as topographic differences or nearby large water bodies). The residuals also need to be examined for changes with time. This pattern should also be random, with no obvious trends or abrupt changes. In all cases, a recording rain gauge (especially a tipping bucket rain gauge) must have a standard rain gauge located in close proximity. The total rainfall recorded between observation times of the tipping bucket rain gauge is adjusted based on the standard gauge readings. These adjustment factors should be reasonably consistent. Another way to check rain gauge data is by comparing the watershed rainfall quantity with the stream flow quantity. This relationship should follow a reasonable rainfall–runoff pattern, with no abrupt deviations. Finally, recording rain gauges need to be periodically calibrated against different artificial rain intensities. The measured rainfall causing a tip of the bucket in a tipping bucket rain gauge should remain constant for a wide range of rain intensities. This quantity should also not change abruptly with time.

**Needed Rain Gauge Density**

One of the most common problems with rainfall monitoring is simply not having enough rain gauges in the watershed. Typical guidance for appropriate rain gauge densities does not consider the likely errors associated with too few gauges located in relatively small urban watersheds. The absolute number of rain gauges is probably more important than the simple rain gauge density. In all cases, multiple rain gauges are needed, even in the smallest study area. The number of rain gauges required depends on local conditions (Curtis 1993). Areas of higher rainfall variability require a greater number of rain gauges to adequately estimate rainfall over a watershed. As an example, mountainous areas will require more gauges than flat lands, and areas subject to convective storms will require more gauges than areas subject to frontal-type storms.

The spatial variability and intended use of the data should be used in determining the needed number of rain gauges. Typical guidance for flat terrain indicates rain gauge spacing of about 25 to 30 km, while this spacing is reduced to 10 to 15 km for mountainous areas. Most monitored urban watershed areas are quite small: almost all are less than 100 km², and typically less than 10 ha in area. These small areas seem to justify only a single rain gauge. Wullshleger et al. (1976) made one of the earliest recommendations for the number of rain gauges needed in small urban runoff catchments. They found about one gauge was needed in 0.5- to 1-km² watersheds, and about 12 gauges for larger (25-km²) watersheds. However, multiple rain gauges are needed in all monitored watersheds. This should include a tipping bucket rain gauge and a single standard rain gauge, at least, for the smallest area, if rain intensities are to be monitored. When the study area increases, and if smaller rains are of interest, the number of rain gauges must be increased to compensate for the increased variation in the rain depth throughout the area. These additional rain gauges can be
additional pairs of tipping bucket and standard rain gauges, or simple accumulative (garden-store type) rain gauges, if intensities are not needed.

The National Engineering Handbook Series contains a simple chart, shown here as Figure 6.26, that can be used to estimate the 90% confidence limits of a rainfall located a specific distance from a rain gauge (NEH undated). As an example, if the measured rainfall at a rain gauge is 2 in, the 90% confidence limit in rain depth for a location 0.5 miles away can be estimated as:

- The “plus error” is about 0.8 in, or 2.8 in for the upper limit.
- The “minus error” is assumed to be about one half this amount, or 0.4 in, with a lower limit of 1.6 in.

The NEH also contains a nomograph (Figure 6.27) that can be used to estimate the error in measurement of watershed average rainfall depth, based on the size of the watershed, the number of rain gauges, the annual average precipitation depth, and the storm rainfall depth of concern. The example shown in this figure is for a watershed of 200 acres, having two rain gauges. In the example shown, the annual rainfall is about 33 in, and the rain of interest is 5 in. The average error is estimated to be about ±12%, or ±0.6 in.

Lei and Schilling (1993) studied the rainfall distribution in two urban watersheds located in Essen, Germany. The catchment had an area of 34 km² and was represented by 17 rain gauges. Rainfall data for five summers (1980–1984) were analyzed. They only examined rains that had all stations represented and that had at least 0.5 mm of rain. They compared catchment-wide averaged rain depth using subsets of the complete rain gauge network against the data from all 17 rain gauges as a reference. Figure 6.28 shows the basin-wide runoff volume errors that would result if only one rain gauge was used in rainfall–runoff modeling. It shows that relative errors of computed runoff volume decreased with increasing rain depth. Rains greater than about 8 mm had about ±20% errors in modeled runoff volume with a single rain gauge over the 34 km².
Figure 6.27- Errors in watershed rain depth. (From NEH (National Engineering Handbook). Part 630, Chapter 4, *Storm Rainfall Depth* (NEH-4). USDA (Natural Resources Conservation Service), Consolidated Forms and Distribution Center, 3222 Hubbard Road, Landover, MD 20785. Periodically updated.)

drainage area. However, smaller rains could have rain depth errors of up to 250% with only a
single rain gauge.

Ciaponi et al. (1993) studied rainfall variability in the 11.4-ha Cascina Scala experimental urban
catchment watershed in Pavia, Italy, for a 3-year period. Two rain gauges separated by 310 m were
used in this study. During this period, 233 storm events were selected for analysis, all greater than
1 mm in depth. The following list shows the percentage differences between the rain depths
measured at the two monitoring locations for three rain depth categories:

- For $1 \text{ mm} < h < 5 \text{ mm}$ (135 storms), the average error was 31%.
- For $5 \text{ mm} < h < 20 \text{ mm}$ (75 storms), the average error was 10%.
- For $h > 20 \text{ mm}$ (23 storms), the average error was 8%.

These results show that the rainfall monitoring variations over even a very small watershed and
with two closely spaced rain gauges can be quite large for small rain depths (<5 mm), with the
differences decreasing for larger rains.

The National Weather Service guideline (Curtis 1993) used to determine the minimum number
of gauges required in a local flood warning system is:

$$N = A^{0.33}$$

where $A$ is the basin area in square miles. As an example, a 10-mi$^2$ watershed would require at
least two rain gauges, while a 100-mi$^2$ watershed would require at least five.

Figure 6.29 shows the expected coefficients of variation for different rain gauge numbers and
watershed sizes (Curtis 1993). For a fast-responding watershed, a coefficient of variation (the
standard deviation divided by the mean) goal of 0.10 would require about six rain gauges for a
50-mi$^2$ watershed, while a 500 mi$^2$ watershed would require about 13 rain gauges for the same
COV of observed rain depths in the watershed. Average and slow-responding watersheds would
require slightly fewer rain gauges for the same watershed areas.

Rodda (1976) presented recommendations (Tables 6.8 and 6.9) for the minimum number of
rain gauges required for small and moderate-sized watersheds and for larger watersheds. Table 6.8
shows the number of rain gauges needed for observations of daily rain depth totals and for monthly
rain depth totals.

According to Chow (1964), one rain gauge per 625 mi$^2$ is the minimum for general climatological
purposes, while for hydrologic purposes, each study basin should have at least one rain
gauge per 100 mi$^2$. However, one rain gauge per 1 mi$^2$ was recommended for the analysis of
thunderstorms.

![Figure 6.29 - Areal rainfall accuracies for fast-responding watersheds.](image)
Pitt and McLean (1986) investigated rainfall distributions in the Toronto area as part of the Humber River pilot watershed study. Rainfall data were available for 35 rain gauges over an area of about 4000 km². This high number of gauges allowed sensitivity calculations to be made to determine the appropriate number of rain gauges that may be needed. Numerous random subsets of these rain gauge data were used to analyze potential errors associated with using fewer gauges for 46 different rains greater than 1 mm in depth. Figure 6.30 shows the likely errors for different numbers of rain gauges over this area. The largest rains (>20 mm) had the smallest rainfall variations over the area and therefore had the smallest errors for a specific number of rain gauges. The smallest rains (<5 mm in depth) had much greater errors because their variations were much larger throughout the area. This plot shows that the errors would be very large (several hundred percent in error) for all rains with only one rain gauge for the complete area. The errors somewhat leveled off after about 12 rain gauges were used. However, the rain depth errors for the largest rain category would remain greater than 10% even for 25 rain gauges, and the smallest rains may still have about 50% errors associated with this large number of gauges.

The small catchment monitoring effort by Pitt and McLean (1986) in Toronto illustrated the need to include multiple rain gauges even in very small areas. The two urban watersheds monitored were 39 and 154 ha in area and were located about 3 km apart. Rainfall was monitored at one of the areas only, and the rainfall at the airport several kilometers away was used for comparison. Partway through the monitoring program, a large deviation was noted between the local and airport monitored rain depths. The local rain gauge was then recalibrated, with a 40% increase in the

<table>
<thead>
<tr>
<th>Area (mi²)</th>
<th>Daily</th>
<th>Monthly</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>1</td>
<td>2</td>
<td>3</td>
</tr>
<tr>
<td>2</td>
<td>2</td>
<td>4</td>
<td>6</td>
</tr>
<tr>
<td>8</td>
<td>3</td>
<td>7</td>
<td>10</td>
</tr>
<tr>
<td>16</td>
<td>4</td>
<td>11</td>
<td>15</td>
</tr>
<tr>
<td>31</td>
<td>5</td>
<td>15</td>
<td>20</td>
</tr>
<tr>
<td>47</td>
<td>6</td>
<td>19</td>
<td>25</td>
</tr>
<tr>
<td>63</td>
<td>8</td>
<td>22</td>
<td>30</td>
</tr>
</tbody>
</table>

volume needed for a single bucket tip compared to the initial calibration value. This of course had a significant effect on the rainfall quantity monitored, and much time was spent in identifying why and when the rain gauge had changed so much since its initial calibration. After much analysis using surrounding rainfall data and investigating the history of the specific rain gauge, it was determined that the rain gauge used had a historical problem with its bearings and several repairs had been made in an attempt to correct it. Unfortunately, the gauge calibration was found to be highly variable, and all the locally monitored data were therefore questionable and not used. Thankfully, the Toronto rain gauge network had six other rain gauges surrounding the two study areas within a few km. These data were extensively evaluated, including examining the storm tracks across the city during all monitored rains, to derive suitable rain depth and intensity values for the storms of interest. This analysis required much time, but was possible because of the additional rain gauge data. This problem could have been prevented with the use of a standard rain gauge located next to the tipping bucket rain gauge (as required in professional rain monitoring installations) for more frequent checks on the calibration factor. Nonrecording rain gauges could also have been located in several locations in the small test watersheds to indicate variations throughout the drainage. Both of these options would have cost a fraction of the amount associated with the additional detailed rainfall analysis required during this project and would have alerted the field personnel to the rainfall monitoring problem much sooner.

Proper Placement of Rain Gauges

Precipitation measurements are greatly influenced by wind. Careful placement and shielding of rain gauges are both necessary to reduce wind-induced errors. The upward movement of air over a rain gauge reduces the amount of precipitation captured in a rain gauge. Proper placement is needed to minimize wind-induced turbulence (and to minimize rain shadow effects) from nearby obstructions. Linsley et al. (1982) concluded that reliable measurements of wind-induced errors are difficult because of problems involved in determining the actual amounts of precipitation reaching the ground. They reported that wind-induced errors during rainfall monitoring exceed about 10% for winds greater than about 8 mph, for both shielded and unshielded rain gauges. This error increases to about 20% during 20 mph winds. Shielded rain gauges perform slightly better, with a wind-induced error about 3% less than for an unshielded rain gauge during 10 mph winds, and about 5% less during 20 mph winds. The effects of winds on snowfall is much greater, with shielded gauges having about half the magnitude of errors as unshielded gauges when monitoring snowfalls. Snowfall errors (all underreported) for unshielded gauges may be about 50% for 10 mph winds and increase to about 70% for 20 mph winds. Various types of wind shields have been used, but the Alter shield (loose-hanging vanes in a circle around the rain gauge) has been adopted as a standard in the United States. Its open and flexible construction provides less opportunity than solid shields for snow buildup, and the flexible design allows wind movement to help keep the shield free of accumulated snow and ice.

Rain gauge exposure and placement are very important to reduce rainfall measurement errors. The higher the rain gauge is located above the ground, the greater the wind error. It is therefore best to locate the rain gauge on level ground, definitely avoiding roof installations and steep hillsides. Linsley et al. (1982) and Shaw (1983) both recommend a partially sheltered site. Brassington (1990) stated that the rain gauge should be located at a distance that is at least twice the height of surrounding obstructions: the vertical angle from the rain gauge to the top of the surrounding trees and buildings should be no greater than 30°. Also, Shaw (1983) recommended that a turf wall be used in overexposed locations where natural shelter is rare. A surrounding small grassed embankment decreases wind turbulence around the rain gauge which can inhibit raindrops from falling into an unprotected gauge. The turf wall should form a circle having an inside diameter of about 3 m, and be built up to the top of the rain gauge. The inside wall should have vertical walls, while the outside should have a slope of about 1 to 4. The inner area must be drained to the outside to prevent flooding.
Rain gauges must also be placed level. If a rain gauge is inclined 10° from the vertical, it will catch 1.5% less than it should due to a decreased open area exposed to the rain. In addition, if a rain gauge is inclined slightly toward the wind, it will catch more rain than the true amount.

**Proper Calibration of Rain Gauges**

The standard U.S. Weather Bureau rain gauge is a nonrecording, but accumulating rain gauge that has an 8-in-diameter funnel opening. The opening directs the water into a measuring tube that has 1/10 the cross-sectional area of the gauge opening. The depth of accumulated rain in the measuring tube is therefore 10 times the depth of rain that fell since the gauge was last checked. This gauge is usually used to measure the 24-hour total rain depths, usually read at 8:00 am each day. This standard gauge should be located adjacent to any recording rain gauge to check the total amount of rain that has fallen during the observation period.

A tipping bucket rain gauge is the most common type that measures rainfall intensity. This gauge has an internal tipping mechanism that fills with water from the funnel connected to the standard 8-in-diameter opening (see Figure 6.21). The tipping mechanism is balanced to dump its contents after a specific amount of water has accumulated (usually 0.01 in). Upon dumping, another small bucket rises to collect the next increment of rainfall. Each tipping motion is recorded on an event recorder, along with its time. Rainfall intensity is therefore related to the number of tips per time period.

Tipping bucket rain gauges must be periodically calibrated by measuring the number of tips associated with a specific amount of water slowly introduced into the gauge. The calibration water must be introduced at a rate comparable to that of the rainfall of interest. Several rainfall rates should be checked over the range of interest. This calibration should be conducted in the field, with the gauge installed, at least every 6 months. As noted previously, tipping bucket rain gauges are most accurate for small to moderate rain intensities. Significant rain can be missed during the time that the tipping action is moving and before the other bucket is in place. Heavy rains also tend to hold the buckets in intermediate positions for long periods, preventing the rain from accumulating in the buckets. The use of a standard accumulating rain gauge adjacent to any recording rain gauge is therefore highly recommended.

Table 6.10 shows the water delivery rate to a tipping bucket rain gauge needed for calibration for different equivalent rainfall intensities, assuming a standard 8-in opening. The rates needed to calibrate a tipping bucket rain gauge for the smallest rainfall intensities shown on this table are very low and would require special low flow pumps. As an example, a Masterflex® portable pump can pump from 0.06 to 1100 mL/min, depending on pump head, tubing size, and pump speed (available from Forestry Suppliers, catalog #76899, model 7570-10 variable speed pump with rechargeable battery, and #76888 pump head with #16 tubing, for 0.80 to 320 mL/min, at a total cost of about $900). This pump can therefore be used for all the rainfall intensity calibrations listed in Table 6.10. Of course, other available peristaltic pumps can also be used for this calibration.

<table>
<thead>
<tr>
<th>Rainfall Intensity (mm/hour)</th>
<th>Rainfall Intensity (in/hour)</th>
<th>Water Delivery Rate for Calibration (mL/min)</th>
</tr>
</thead>
<tbody>
<tr>
<td>2</td>
<td>0.078</td>
<td>1.1</td>
</tr>
<tr>
<td>5</td>
<td>0.20</td>
<td>2.7</td>
</tr>
<tr>
<td>10</td>
<td>0.39</td>
<td>5.4</td>
</tr>
<tr>
<td>25</td>
<td>0.98</td>
<td>14</td>
</tr>
<tr>
<td>50</td>
<td>2.0</td>
<td>27</td>
</tr>
<tr>
<td>100</td>
<td>3.9</td>
<td>54</td>
</tr>
<tr>
<td>200</td>
<td>7.9</td>
<td>110</td>
</tr>
</tbody>
</table>
When the rainfall intensity becomes great, the tipping bucket mechanism cannot keep up, resulting in a decreased amount of rain recorded. As an example, Ciaponi et al. (1993) used a peristaltic pump to calibrate two gauges in an urban test watershed in Pavia, Italy. The calibrations showed that the rain gauges could accurately measure rainfall intensities at 44 mm/hour (the lowest rate calibrated with the pump) with errors less than 1%. However, at rain intensities of about 250 mm/hour, the errors were about 10%, and at 400 mm/hour, the errors increased to about 15%. The measured rain intensities were all less than the actual intensities due to missing rain during the tipping time of the individual buckets. Of course, very few rains would be expected to have prolonged large intensities that would cause errors greater than about 10%. However, short-duration, very high rain intensities are much more likely, and accurate rates in these high-intensity ranges may be needed. Therefore, care must be taken when calibrating rain gauges to use appropriate water delivery rates that correspond to a wide range of expected rainfall intensities.

Summary of Rainfall Monitoring Methods

Table 6.11 lists the main advantages and disadvantages of the different basic types of rainfall monitoring methods. In all cases, a tipping bucket rain gauge is needed in an urban study area, with a standard gauge located nearby for proper calibration. In addition, at least several rain gauges (need not be recording, but that would obviously be most helpful) must be placed throughout the study area. For large areas, many gauge installations are needed. In areas of snowfall, special modifications are also required. Proper placement and shielding of the rain gauges are also needed but frequently overlooked. Radar rainfall information can be valuable, but only as a supplement to standard rain gauges in a study area. Proper use of radar rainfall data generally requires an expert and specialized software, and it is only useful relatively close to the radar installation.

SOIL EVALUATIONS

Knowing local soil properties is critical for many aspects of watershed evaluations. Soil properties are extremely important for less-developed areas, because they control many of the hydrologic and sediment aspects of the stormwater. As a watershed becomes developed, however, soil characteristics may become less important than other aspects (especially the nature and extent of the paved and roofed surfaces). Nonetheless, it is important to acknowledge that soils become dramat-

<table>
<thead>
<tr>
<th>Rainfall Monitoring Method</th>
<th>Advantages</th>
<th>Disadvantages</th>
</tr>
</thead>
<tbody>
<tr>
<td>Tipping bucket rain gauges</td>
<td>Most commonly used and available gauge. Obtains high resolution rainfall intensity data. Relatively inexpensive for current versions of recording models.</td>
<td>Must be frequently calibrated and located adjacent to a standard rain gauge (not usually done). Usually insufficient numbers of recording gauges in most local networks.</td>
</tr>
<tr>
<td>Standard rain gauges</td>
<td>Standard rain gauge and most accurate. Can be heated and used for monitoring snowfall.</td>
<td>Does not obtain rain intensity information. Must be manually read at least once a day.</td>
</tr>
<tr>
<td>“Garden store” rain gauges</td>
<td>Inexpensive and can be placed throughout a study area. Best use to supplement standard and tipping bucket rain gauges.</td>
<td>Does not obtain rain intensity information. Must be manually read.</td>
</tr>
<tr>
<td>Radar rainfall measurements (such as NEXRAD)</td>
<td>High resolution data over a large area. Real-time measurements.</td>
<td>Most indicative of severe weather conditions. Can be very inaccurate and requires substantial calibration from standard rain gauges. Only suitable for areas relatively close to a radar installation.</td>
</tr>
</tbody>
</table>
ically altered with typical urban development and to understand how these changes affect local stormwater. The following paragraphs describe the unusual soil conditions found during some studies of urban soils and the methodologies that were used.

Local USDA Soil Conservation Service (now NRCS, Natural Resources Conservation Service) offices have a wealth of information pertaining to soils in all areas of the nation. The county soil surveys should be carefully reviewed for important information. However, urbanization typically alters many “natural,” or mapped, soil characteristics beyond recognition through removing vegetation and topsoil, large-scale cut-and-fill operations, compaction, and artificial landscaping. Unfortunately, these changes usually all adversely affect the soils’ abilities to infiltrate runoff and to retain soil during storms. It may therefore be important to directly measure some of these critical soil characteristics in watersheds undergoing study. This section briefly describes the experimental design and numerous test procedures and some results for a recent EPA-sponsored research project (Pitt et al. 1999a) that investigated adverse soil changes with urbanization (mostly compaction) and possible mitigation methods (amending soil with compost).

Numerous methods have been used to measure infiltration in urban areas. Figure 6.31 is a double-ring infiltration apparatus used to measure infiltration through disturbed urban soils in Oconomowac, WI. Figures 6.32 and 6.33 are photographs of an infiltration test apparatus developed by Dr. Wolfgang Geiger at the University of Essen, Germany, and Figures 6.34 through 6.35 are photographs of the Pac Forest soil infiltration test site developed by Dr. Rob Harrison of the Ecosystem Science and Conservation Division, College of Forest Resources at the University of Washington, Seattle.

Case Study to Measure Infiltration Rates in Disturbed Urban Soils

The soil characteristics of most interest for a receiving water investigation include the soil texture, the soil erosion factors (NRCS K and T factors), and the soil infiltration rates. Because soils in urban areas are greatly disturbed during construction activities, the information contained in the county soil surveys will not be directly applicable, requiring site investigations. Soil infiltration may be related to the time since the soil was disturbed by construction or grading operations (turf age). In most new developments, compact soils are expected to be dominant, with reduced infiltration compared to preconstruction conditions. In older areas, the soil may have recovered.
some of its infiltration capacity due to root structure development and from soil insects and other digging animals.

The following discussion presents a case study that was conducted by Pitt et al. (1999) that investigated infiltration rates in disturbed urban soils. These types of data can be used to more accurately predict watershed hydrology and associated receiving water problems, compared to using published information for natural soil conditions. The results presented in the following example
show how site measurements can be significantly different from published and traditional data. This case study is presented as an example of how this type of study can be conducted to obtain this critical, site-specific information.

**Experimental Design**

A series of 153 double-ring infiltrometer tests were conducted in disturbed urban soils in the Birmingham and Mobile, AL, areas. The tests were organized in a complete $2^3$ factorial design (Box et al. 1978) to examine the effects of soil moisture, soil texture, and soil compactness on water infiltration through historically disturbed urban soils. Moisture and soil texture conditions were determined by standard laboratory soil analyses. Compaction was measured in the field using a cone penetrometer (Dickey-John Corp. 1987) and confirmed by the site history. Moisture levels were increased using long-duration surface irrigation before most of the saturated soil tests. From 12 to 27 replicate tests were conducted in each of the eight experimental categories in order to measure the variations within each category for comparison to the variation between the categories.

Table 6.12 shows the analytical measurement methods used for measuring the infiltration rates, and supporting measurements, during the tests of infiltration at disturbed urban sites. Table 6.13 defines the different levels for the experimental factors used during these tests.

**Infiltration Rate Measurements**

The infiltration test procedure included several measurements. Before a test was performed, the compaction of the soil was measured with the DICKEY-john Soil Compaction Tester and a sample was obtained to analyze moisture content. TURF-TEC Infiltrometers (1989) were used to measure the soil infiltration rates. These small devices have an inner ring about 64 mm (2.5 in) in diameter and an outer ring about 110 mm (4.25 in) in diameter. The water depth in the inner compartment starts at 125 mm (5 in) at the beginning of the test, and the device is pushed 50 mm (2 in) into the ground. The rings are secured in a frame with a float in the inner chamber and a pointer next to a stopwatch. These units are smaller than standard double-ring infiltrometers, but their ease of use allowed many tests to be conducted under a wide variety of conditions. The use of three infiltrometers placed close together also enabled better site variability to be determined than if larger, standard-

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### Table 6.12 QA Objectives for Detection Limits, Precision, and Accuracy for Critical Infiltration Rate Measurements in Disturbed Urban Soils

<table>
<thead>
<tr>
<th>Measurement</th>
<th>Method</th>
<th>Reporting Units</th>
<th>MDL</th>
<th>Precision</th>
</tr>
</thead>
<tbody>
<tr>
<td>Double-ring infiltration rate measurements</td>
<td>ASTM D3385-94</td>
<td>in/hr</td>
<td>0.05</td>
<td>10%</td>
</tr>
<tr>
<td>Soil texture</td>
<td>ASTM D 422-63, D 2488-93, and 421</td>
<td>plots</td>
<td>na</td>
<td>10%</td>
</tr>
<tr>
<td>Soil moisture (analytical balance)</td>
<td>ASTM D 2974-87</td>
<td>Percentage of moisture in soil (mg)</td>
<td>5% (0.1 mg)</td>
<td>10% (1%)</td>
</tr>
<tr>
<td>Soil compaction</td>
<td>History of site activities and cone penetrometer</td>
<td>psi</td>
<td>5</td>
<td>10%</td>
</tr>
<tr>
<td>Soil age</td>
<td>Age of development</td>
<td>years</td>
<td>na</td>
<td>na</td>
</tr>
</tbody>
</table>


### Table 6.13 Experimental Test Levels during Infiltration Rate Tests

<table>
<thead>
<tr>
<th>Moisture</th>
<th>Disturbance</th>
<th>Soil Texture</th>
</tr>
</thead>
<tbody>
<tr>
<td>Enhanced infiltration</td>
<td>Dry (&lt;20% moisture)</td>
<td>Uncompacted (&lt;300 psi)</td>
</tr>
<tr>
<td>Decreased infiltration</td>
<td>Wet (&gt;20% moisture)</td>
<td>Compact (&gt;300 psi)</td>
</tr>
</tbody>
</table>

sized units were used. These small units are available from Forestry Suppliers, Inc., while the standard-sized units are available from Gilson, or other soil engineering equipment suppliers.

Three infiltrometers were inserted into the turf within a meter of each other to indicate the infiltration rate variability of soils in close proximity. Both the inner and outer compartments were filled with clean water by first filling the inner compartment and allowing it to overflow into the outer compartment. As soon as the measuring pointer reached the beginning of the scale, the timer was started. Readings were taken every 5 min for 2 hours. The instantaneous infiltration rates were calculated by noting the decline in the water level in the inner compartment over the 5-min period.

Tests were recorded on a field observation sheet as shown in Figure 6.37. Each document contained information such as relative site information, testing date and time, compaction data, moisture data, and water level drops over time, with the corresponding calculated infiltration rate for the 5-min intervals.

All measurements were taken in soils in the field (leaving the surface sod in place), with no manipulation besides possibly increasing the moisture content before “wet” soil tests are conducted (if needed).

**Soil Moisture Measurements**

Moisture values relating to dry or wet conditions are highly dependent on soil texture and are mostly determined by the length of antecedent dry period before the test. Soil moisture was determined in the laboratory using the ASTM D 2974-87 (1994) method (basically weighing a soil before and after oven drying). For typical sandy and clayey soil conditions at the candidate test areas, the dry soils had moisture contents ranging from 5 to 20% (averaging 13%) water, while wet soils had moisture contents ranging from 20 to 40% (averaging 27%) water.

The moisture condition at each test site was an important test factor. The weather occurring during the testing enabled most site locations to produce a paired set of dry and wet tests. The dry tests were taken during periods of little rain, which typically extended for as long as 2 weeks with no rain and with sunny, hot days. The saturated tests were conducted after thorough artificial soaking of the ground, or after prolonged rain. The soil moisture was measured in the field using a portable moisture meter (for some tests) and in the laboratory using standard soil moisture methods (for all
The moisture content, as defined by Das (1994), is the ratio of the weight of water to the weight of solids in a given volume of soil. This was obtained using ASTM method D 2974-87 (1994), by weighing the soil sample with its natural moisture content and recording the mass. The sample was then oven-dried and its dry weight recorded.

**Soil Texture Measurements**

At each site location, a soil sample was obtained for a texture classification. The texture of the samples was determined by ASTM standard sieve analyses (1994) to verify the soil conditions estimated in the field and for comparison to the NRCS soil maps. The sieve analysis used was the ASTM D 422-63 *Standard Test Method for Particle Size Analysis of Soils* for the particles larger than the No. 200 sieve, along with ASTM D 2488-93 *Standard Practice for Description and Identification of Soils (Visual — Manual Procedure)*. The sample was prepared based on ASTM 421 *Practice for Dry Preparation of Soil Samples for Particle Size Analysis and Determination of Soil Constants*.

The texture analyses required a representative dry sample of the soil to be tested. After the material was dried and weighed, it was crumbled to allow a precise sieve analysis. The sample was then treated with a dispersing agent (sodium hexametaphosphate) and water at the specified quantities. The mixture was then washed over a No. 200 sieve to remove all soil particles smaller than the 0.075 mm openings. The sample was then dried again and a dry weight obtained. At that point, the remaining sample was placed in a sieve stack containing No. 4, No. 8, No. 16, No. 30, No. 50, No. 100, No. 200 sieves, and the pan. The sieves were then placed in a mechanical shaker and allowed to separate onto their respective sieve sizes. The cumulative weight retained on each sieve was then recorded.

The designation for the sand or clay categories follows the *Unified Soil Classification System*, ASTM D 2487. Sandy soils required that more than half of the material be larger than the No. 200 sieve, and more than half of that fraction be smaller than the No. 4 sieve. Similarly, for clayey soils, more than half of the material is required to be smaller than the No. 200 sieve. Figure 6.38 is the standard soil texture triangle defining the different soil texture categories.

**Soil Compaction Measurements**

The extent of compaction at each site was also measured before testing using a cone penetrometer. The compaction of the test areas was obtained by pushing a Dickey-john Soil Compaction Tester (available from Forestry Suppliers, Inc.) into the ground and recording the readings from

![Figure 6.38 Standard soil triangle.](image-url)
the gauge. For these tests, compact soils were defined as a reading of greater than 300 psi at a depth of 3 in, while uncompacted soils had readings of less than 300 psi.

Compaction was confirmed based on historical use of the test site location, as moisture levels affected the cone penetrometer readings. Soils, especially clay soils, are obviously more spongy and soft when wet compared to when they are extremely dry. Therefore, the penetrometer measurements were not made for saturated conditions, and the degree of soil compaction was also determined based on the history of the specific site (especially the presence of parked vehicles, unpaved lanes, well-used walkways, etc.). Other factors that were beyond the control of the experiments, but also affect infiltration rates, include bioturbation by ants, gophers and other small burrowing animals, worms, and plant roots.

**Bulk Density**

Bulk density was estimated using a coring device of known volume (bulk density soil sampler). The core was removed, oven dried, and weighed. Bulk density was calculated as the oven-dry weight divided by the core volume. Particle density was determined by using a gravimetric displacement. A known weight of soil or soil/compost mixture was placed in a volumetric flask containing water. The volume of displacement was measured and particle density was calculated by dividing the oven-dry weight by displaced volume.

Gravimetric water-holding capacity was determined using a soil column extraction method that approximates field capacity by drawing air downward through a soil column. Soil or soil/compost mixture was placed into 50 mL syringe tubes and tapped down (not compressed directly) to achieve the same bulk density as the field bulk density measured with coring devices. The column was saturated by drawing 50 mL of water through the soil column, then brought to approximate field capacity by drawing 50 mL of air through the soil or soil/compost column.

Volumetric water-holding capacity was calculated by multiplying gravimetric field capacity by the bulk density. Total porosity was calculated by using the following function:

$$\text{total porosity} = 1 - \frac{\text{bulk density}}{\text{particle density}} \times 100\%$$  \hspace{1cm} (6.1)

Particle size distribution was determined both by sieve analysis and sedimentation analysis for particles less than 0.5 mm in size. Due to the light nature of the organic matter amendment, particle size analysis was sometimes difficult, and possibly slightly inaccurate. Soil structure was determined using the feel method and comparing soil and soil/compost mixture samples to known structures.

**Subsurface Flow Measurements**

Subsurface flows and surface runoff during rains were measured and sampled using special tipping bucket flow monitors collecting the samples from the tubing shown in Figure 6.39 (Harrison et al. 1997). The flow amounts and rates were measured by tipping-bucket-type devices attached to an electronic recorder, as shown in Figure 6.40 (a close-up of the tipping bucket flowmeters shown previously in Figure 6.35), taken at the University of Washington installation. Each tip of the bucket was calibrated for each site and checked on a regular basis to give rates of surface and subsurface runoff from all plots.

**Observations — Infiltration Rates in Disturbed Urban Soils**

The initial exploratory analyses of the data showed that sand was most affected by compaction, with little change due to moisture levels. However, the clay sites were affected by a strong interaction