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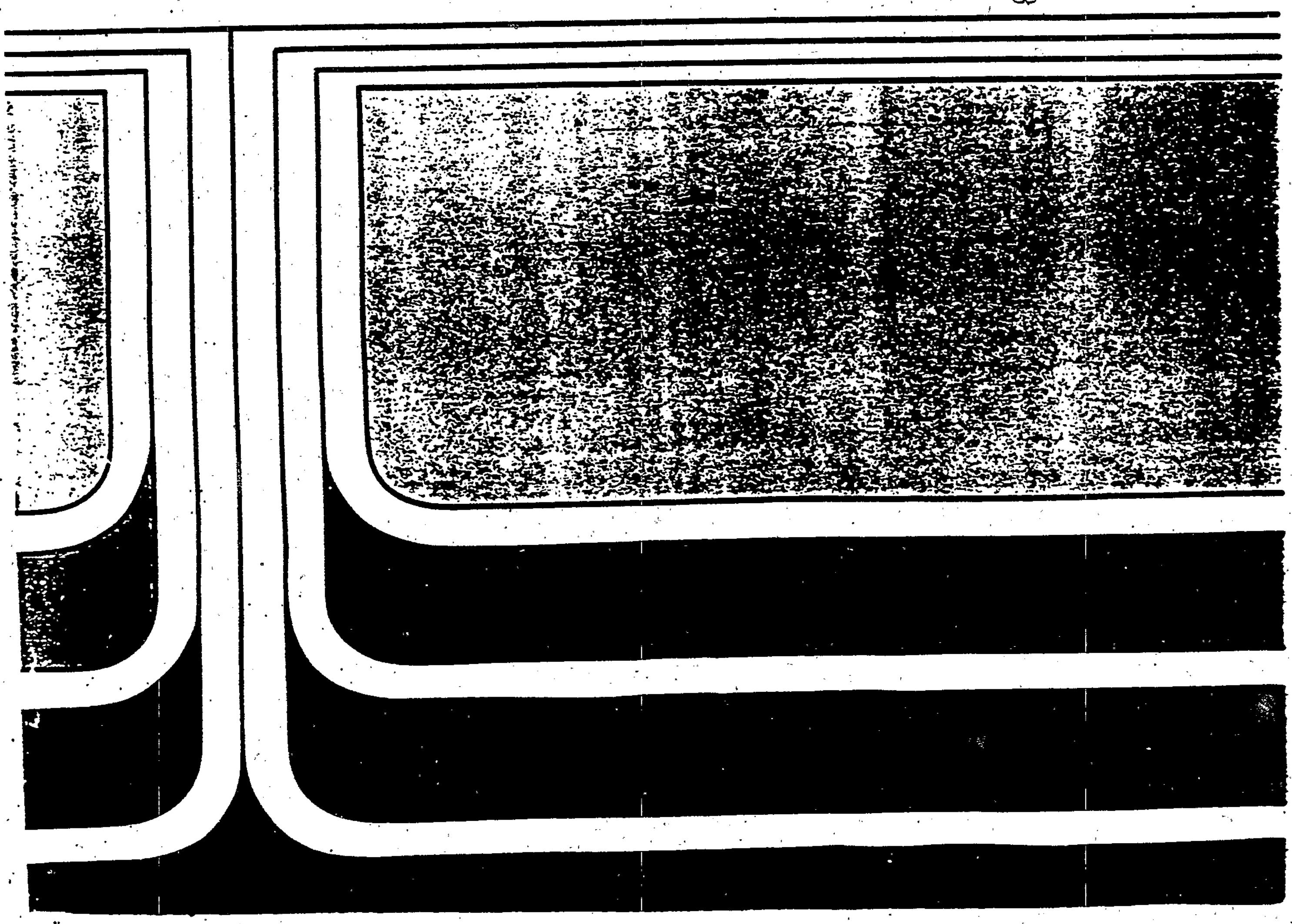
EPA

A Review Of Methods For Assessing Nonpoint Source Contaminated Ground-Water Discharge To Surface Water



A Review of Methods for Assessing Nonpoint Source Contaminated Ground-Water Discharge to Surface Water

April 1991

Office of Ground-Water Protection
U.S. Environmental Protection Agency

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Chapter I

Introduction

A. Purpose of this report

This report presents a summary of methods that have been applied to measure or estimate nonpoint source contaminated ground-water discharge to surface water. The U.S. Environmental Protection Agency (EPA) Office of Ground-Water Protection (OGWP) developed this analysis as part of an effort to broaden the understanding of the manner in which human activities can affect water quality in all phases of the hydrologic cycle within a watershed. EPA undertook this project in response to the growing awareness that contaminated ground-water discharge is a significant source of nonpoint source contaminant loading to surface waters in many parts of the country. In particular, this report is intended to stimulate understanding of the methods that may be applied to better account for nonpoint sources of contaminant loading. Improved characterization of nonpoint source loads to surface water may, in turn, lead to more comprehensive approaches for setting total maximum daily loads for surface waters and waste load allocation.[1] This report provides an overview of these methods, rather than a manual for employing the methods presented. Readers who intend to apply the methods summarized here should study the primary references cited.

While ground water and surface water are generally thought of as separate systems, they are highly interdependent components of the hydrologic cycle. The hydrologic cycle refers to the circulation of water among soil, ground water, surface water, and the atmosphere. Within a watershed, water may enter the basin through precipitation, upstream inflow, and ground-water discharge. Water leaves the watershed through downstream outflow, evaporation, and ground-water outflow (see Figure 1-1). Some rainwater never reaches surface water due to the evaporation of intercepted rainfall from vegetative surfaces and the soil matrix and transpiration of water by plants, returning water vapor back into the atmosphere.

Rainfall that reaches surface water may travel to the stream or lake as subsurface storm runoff, overland flow, or ground water. In most humid environments, about 80 percent of rainfall will infiltrate into the soil rather than travel by overland flow. Overland flow is more predominant in semi-arid rangelands, roadways, and cultivated fields in regions with high intensity rainfall. Rainfall that percolates into the soil matrix is held by capillary forces. As the soil moisture increases, older soil water is displaced and percolates laterally and/or vertically. Lateral percolation may eventually enter streams as subsurface storm runoff, while vertical percolation generally enters the saturated ground-water zone. Ground water moves more slowly than subsurface storm runoff and will eventually discharge and provide water to streams, wetlands, and lakes.

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[1] See Chapter 3 of this document for a discussion of total maximum daily load and the waste load allocation process.

Most streams and lakes are surrounded by bank storage zones that increase during storm events. During a rainstorm, precipitation entering areas of high soil moisture content which are closer to the stream displaces the water held in storage, thereby providing the stream with water from bank storage. In addition, when the topsoil is underlain by a less permeable horizon, water accumulates above that horizon and flows downhill through the soil. This represents a shorter route to the stream[2].

Saturated ground water enters surface waters in the form of springs or recharging zones to bank storage. If this ground water is contaminated by disperse nonpoint sources, the discharging ground water may affect surface-water quality over a wide area. This paper is concerned with the movement of nonpoint source contaminants through the ground water saturated zone to surface water and the methods that have been developed and applied for measuring this nonpoint source loading to surface water.

In preparing this report, EPA contacted over 100 individuals who are actively involved in developing or applying methods for measuring nonpoint source contaminated ground-water discharge to surface water. EPA also reviewed over 200 papers addressing this topic from the technical and professional literature. This report represents a synthesis of the information collected from these sources.

B. Organization of this report

This report is organized in three chapters. Following this introduction, Chapter II presents a summary of the analytical methods identified by EPA for measuring or estimating nonpoint source contaminated ground-water discharge to surface water. Chapter III presents an overview of the total maximum daily load assessment and waste load allocation processes and discusses the applicability of the methods described in Chapter II to support these analyses. In addition, an annotated bibliography of the papers that formed the basis for the analysis presented in Chapter II is provided in a companion volume to this document.[3]

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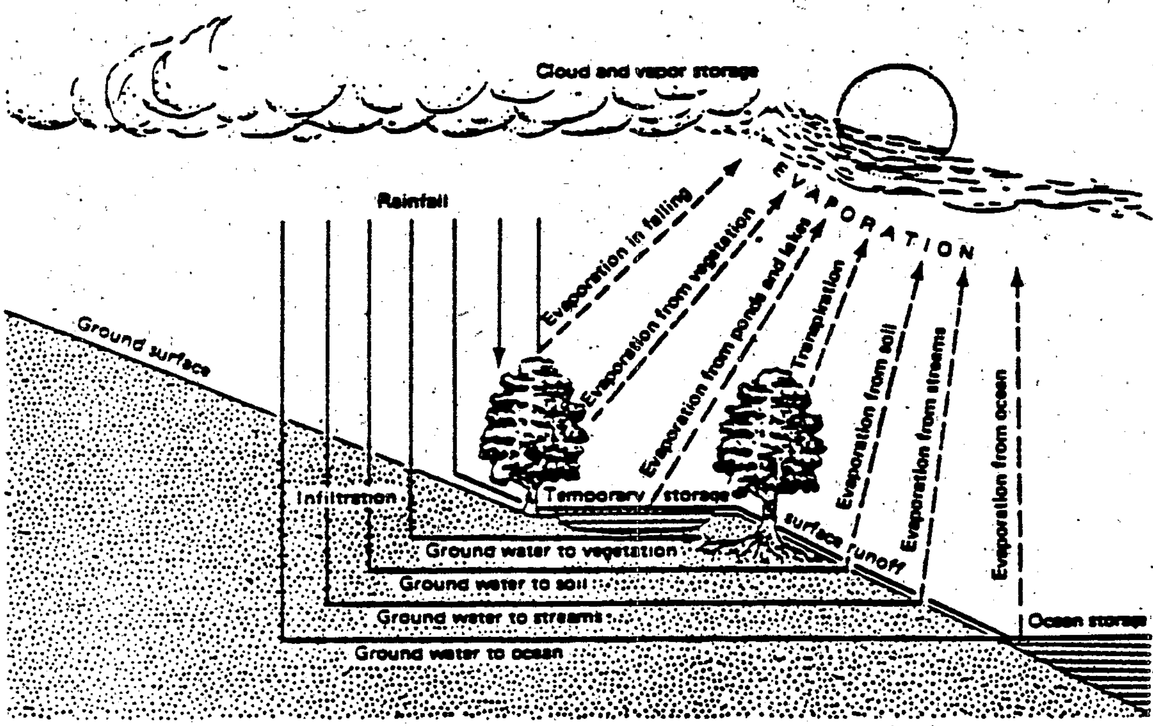
[2] Dunne, Thomas, and Luna B. Leopold: <u>Water in Environmental Planning</u>. W.H. Freeman and Company, 1978, pp. 255-277.

[3] See "An Annotated Bibliography to the Literature Addressing Nonpoint Source Contaminated Ground-Water Discharge to Surface Water," EPA 440/6-90-006.

Figure 1-1



Dunne, Thomas, and Luna B. Leopold: Water in Environmental Planning. W.H. Freeman and Company, 1978, p. 5.

Chapter II

Methods for Measuring or Estimating Nonpoint Source Contaminated Ground-Water Discharge to Surface Water

Introduction

Seven groups of methods have been identified in the literature for measuring or estimating contaminated ground-water discharge to surface water. Each section of this chapter presents a general description of the method, assumptions and limitations for the method, summarizes the data inputs and outputs for the method, describes the environmental settings and contaminant types that have been evaluated using the method, and presents a general evaluation of the suitability of the method for other applications. Each section also concludes with tables summarizing information presented in the companion volume to this document, "An Annotated Bibliography to the Literature Addressing Nonpoint Source Contaminated Ground-Water Discharge to Surface Water," EPA 440/6-90-006.

A limitation common to all of the methods discussed in this report is the high degree of uncertainty inherent in the study of ground water. The heterogeneity of geologic formations presents a major problem in ground-water study. For example, hydraulic conductivity values can range from 10^{-1} cm/s to less than 10^{-10} cm/s in different geologic settings. Furthermore, hydraulic conductivities and other hydrogeologic parameters can vary significantly over even small distances. Thus, errors inherent in ground-water parameter estimates can vary by 50 percent or more, whereas an acceptable error for surface-water work is about 10 percent. As a result, the reader should note that the methods described in this chapter may inherently encompass broad ranges of uncertainty in their estimates.

A. Studies involving use of seepage meters or mini-piezometers to measure ground-water discharge to surface water

The papers cited in this section are summarized in Section I of "An Annotated Bibliography of the Literature Addressing Nonpoint Source Contaminated Ground-Water Discharge to Surface Water," September 1990, EPA 440/6-90-006

i. General description of method

a. Description of method or procedure

Seepage meters and mini-piezometers may be used to measure the quantity and quality of ground water discharging to surface water. These methods measure a "point-location" ground-water discharge rate and allow for water-quality sampling over a very small area at the surface-water/sediment interface. In order to characterize larger areas, several measuring/sampling points must be selected. Areas with different sediment types may be mapped and several seepage meters/mini-piezometers installed in each sediment type.

The total discharge and loading rate to the surface-water body as a result of ground-water discharge can be estimated by applying average measurements per sediment type to the entire bottom area. Alternatively, reconnaissance over a large area may be used to identify areas where the greatest quantity of contaminants is entering the surface water body. Seepage meters and mini-piezometers may then be used to monitor and quantify discharge zones.

Seepage meters and mini-piezometers have been used to investigate ground-water discharge into lakes, streambeds, and marine environments. They are best suited for use in moderately permeable soils and relatively quiet waters, but adaptations allow successful use under more adverse conditions. They may be used in combination with one another or with piston corers to analyze soil permeability, ground-water quality, and ground-water discharge. An important function of these methods is to provide field verification for geophysical techniques that may be used to estimate ground-water discharge to a surface water body (see Section II.C).

Seepage Meter

In its simplest form, a seepage meter can be a 55-gallon drum with the bottom cut off and a vent hole placed in the closed end. The open end of the drum is pushed into the bottom sediments of the surface-water body until only the closed top of the drum is exposed (Lee, 1977). The vent hole remains unstoppered and the seepage meter equilibrates with the sediment environment. After several days, a collection system consisting of a tube and a deflated bag is attached to the vent hole (see Figure 2-1). One can use seepage meters to estimate discharge velocity of ground water to surface water. Dividing the collected volume of seepage by the duration of the collection period and by the area of the seepage meter produces an estimate of ground-water discharge velocity. Multiplying by the surface area of the stream or lake bottom estimates the total ground-water discharge rate through that area.

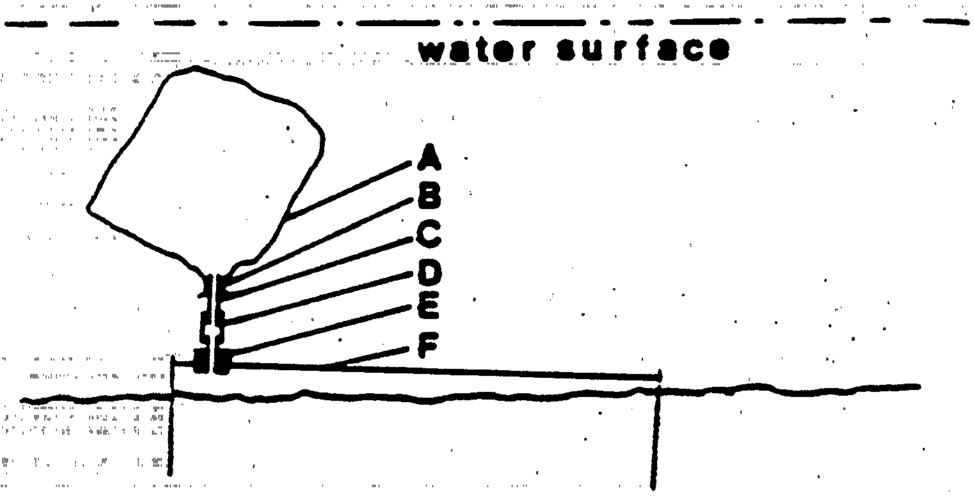
Provided that consideration is given to chemical alteration, seepage meters might be used to determine ground-water quality from collected seepage samples. Multiplying the measured chemical constituent concentration in the seepage by the calculated ground-water discharge rate to the surface-water body estimates the constituent's loading to the surface water (Goodman et al., 1989).

Mini-piezometer

Description and installation

Piezometers are devices consisting of pipes with slotted tips or well points on the end. They are used to measure hydraulic head in saturated geologic materials. Piezometers are usually installed in machine-drilled boreholes. Mini-piezometers are similar to piezometers, but are smaller in size and installed manually. A mini-piezometer consists of a small-diameter tube perforated over a short distance at one end. Nylon mesh covering the perforated tube keeps sediment from clogging the mini-piezometer. To place a mini-piezometer, a length of thick-walled pipe, with an inside diameter slightly larger than the tubing is hammered into the sediment. A temporary

Figure 2-1



Full section view of seepage meter showing proper placement in the sediment. A. 4 liter, 0.017 mm membrane plastic Baggies Alligator bag (open end was heat sealed): B. rubber-band wrap: C. 0.64 cm inside diameter, 6 cm long, polyethylene tube: D. 0.79 cm inside diameter, 4.5 cm long, amber-latex tube: F. 15 cm x 57 cm diameter epoxy-coated cylinder (end-section of a steel drum).

Lee, David R., and John A. Cherry: "A Field Exercise on Ground-Water Flow Using Seepage Meters and Mini-Piezometers," Journal of Geologic Education, 1978, Volume 27: p. 8.

plug attached to the end of the pipe keeps sediment from entering the pipe during placement. The plug is knocked free before the mini-piezometer is inserted to the bottom of the pipe, perforated tip first. The mini-piezometer tube is held in place as the length of pipe is removed. The pipe used in installation may be pulled back to expose the perforated section but left in place to provide added protection to the tubing. A collection system similar to the seepage meter tube and collapsible bag system can be used to collect seepage samples (see figure 2-2).

Mini-piezometer variations

One difficulty often associated with the installation of mini-piezometers is their tendency to move before the sediment collapses around the tube or if the tube is pulled later on. Lee and Welch (1989) have tested a harpoon piezometer which helps to alleviate this problem (see Figure 2-3). "Barbs" on the tip of the piezometer grip the sediment and help to keep the screen at the desired depth as the driving rod, or pipe, is withdrawn or if the screen in moved during use.

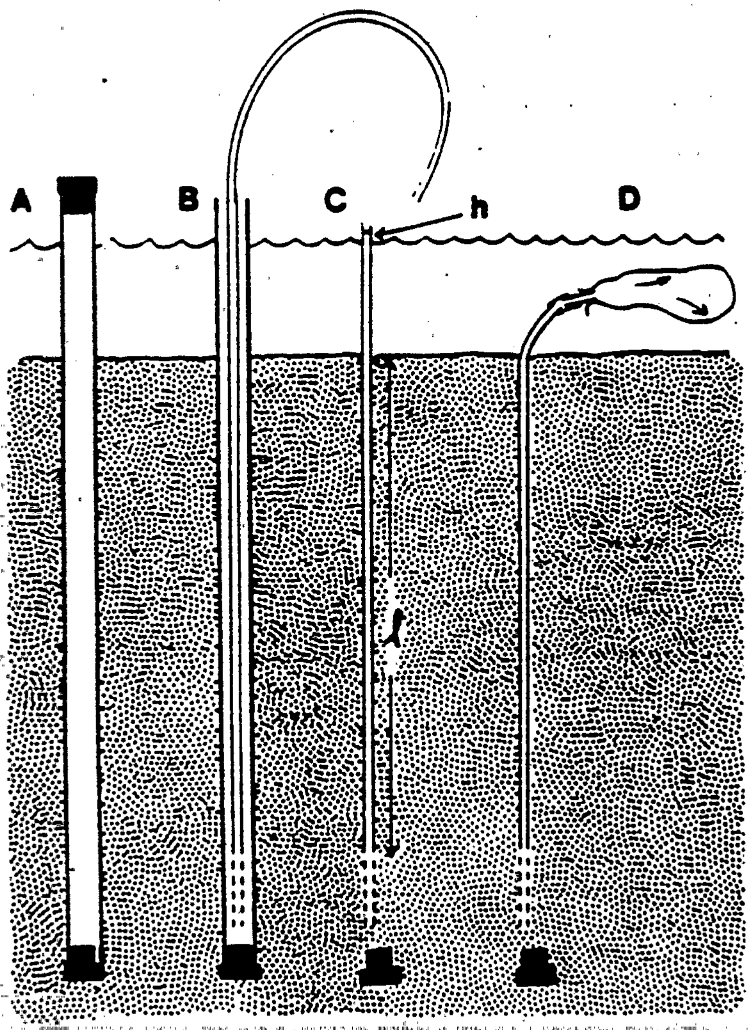
Another variation of the mini-piezometer is the bundle-type mini-piezometer, consisting of several small tubes placed within the pipe at one time. The tubes are placed at selected depths to allow detailed vertical resolution of head and pore-water chemistry at the selected mini-piezometer location. If bundle-type mini-piezometers are placed at selected points along a vertical plane, patterns of flow and geochemical processes in the subsurface are observable.

Mini-piezometer measurements

An alternative to direct measurement of ground-water discharge is to use hydraulic conductivity and hydraulic head data obtained from mini-piezometers to calculate the ground-water discharge rate to surface water using Darcy's Law. Comparing the hydraulic head in the mini-piezometer with the hydraulic head of the surface-water body determines the hydraulic gradient across bottom sediments. Hydraulic head differential may be measured using a manometer (see Figure 2-4) or a continuous water level recorder (see Figure 2-5). Head differential is divided by the depth of the piezometer screen below the sediment-water interface to obtain the vertical hydraulic gradient. Hydraulic conductivity of the bottom sediments may be estimated or be measured using either a constant head or falling head test. A constant head test has been developed using sections of sediment cut directly from a thin-walled piston core barrel (Munch and Killey, 1985). Once the hydraulic gradient and hydraulic conductivity have been determined, Darcy's Law may be used to determine the ground-water flux through the sediment (see Section A.i.d). Multiplying the calculated flux by the surface area of the surface-water bottom yields the ground-water discharge rate to the surface-water body.

The mini-piezometer yields seepage samples using a syringe or other sampling device. Multiplying the measured chemical constituent concentration in the ground water by the calculated ground-water discharge rate yields the loading rate to surface water.

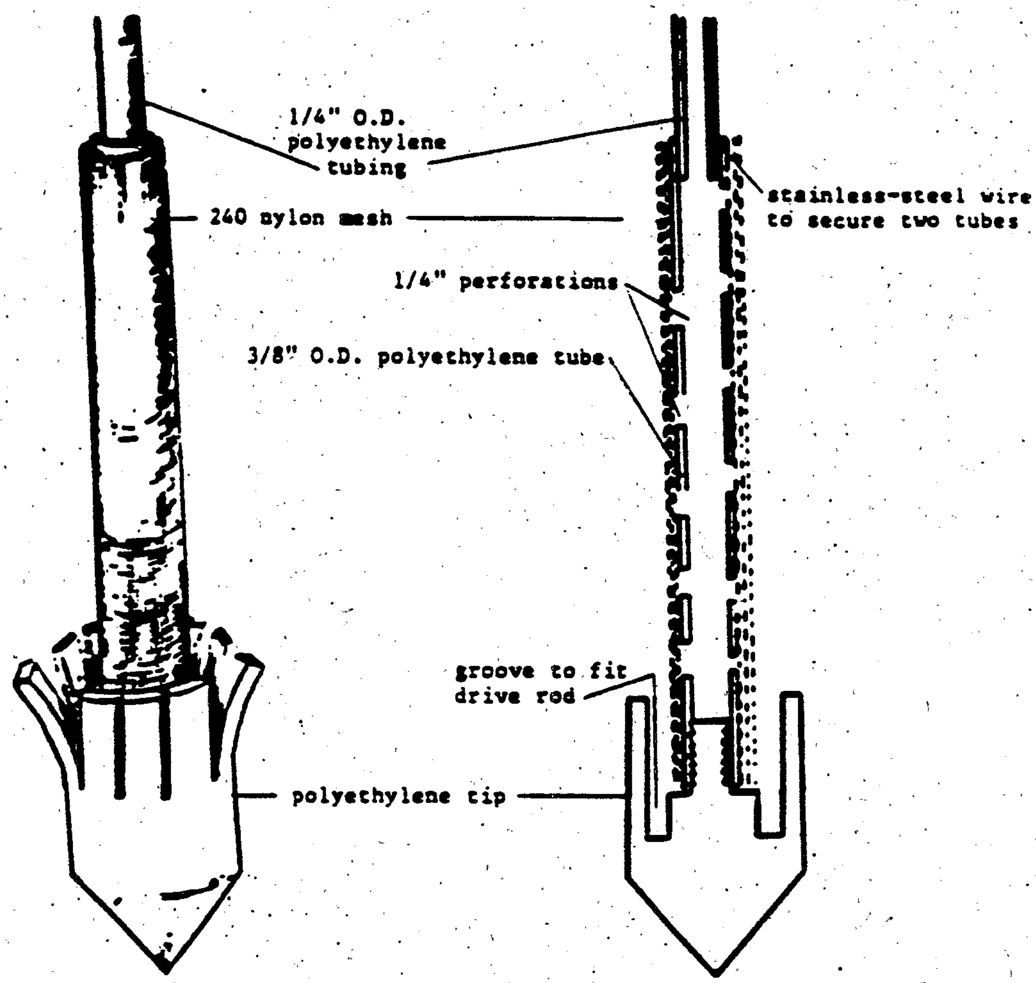
Figure 2-2



General features and method of installation of a mini-piezometer. A. casing driven into the sediment; B. plastic tube with screened tip inserted in the casing; C. plastic tube is a piezometer and indicates differential head (h) with respect to the surface water; D. plastic bag attached to the piezometer collects sediment-porewater.

Lee, David R., and John A. Cherry: "A Field Exercise on Ground-Water Flow Using Seepage Meters and Mini-Piezometers," Journal of Geologic Education, 1978, Volume 27: p. 7.

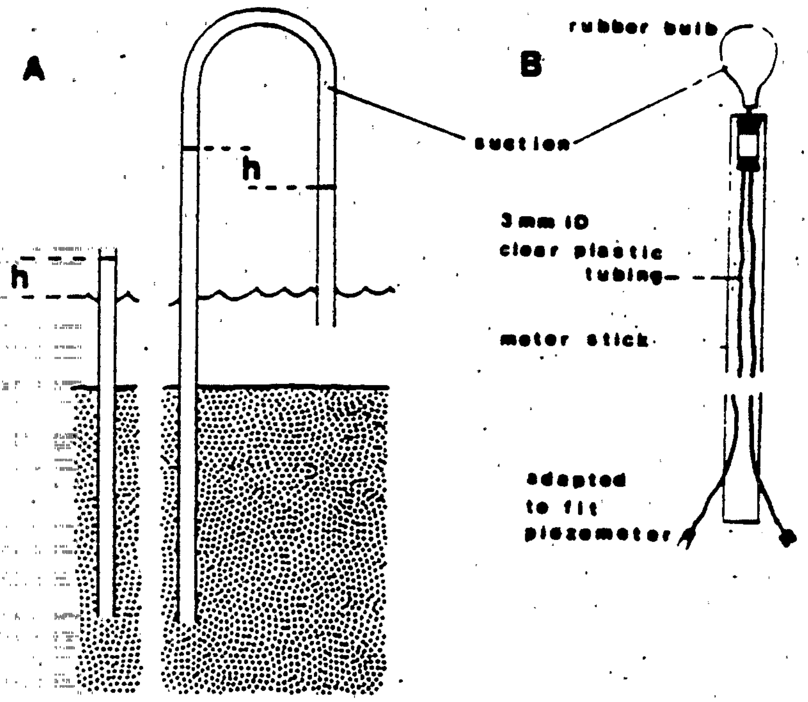
Figure 2-3



Harpoon piezometer tip, screen and tube. Dimensions for the small type are shown here. The screen is 10 cm long, has 8 1/4" diameter perforations and is covered with 3 layers of 240 μm mesh tightly rolled around the 3/8" O.D. polyethylene tube to prevent entry of sediment. The drive rod, not shown, fits loosely in groove. The "barbs" are folded back before driving in sediment to ensure that they grip in the sediment.

Lee, D.L. and S.J. Welch: "Methodology for Locating and Measuring Submerged Discharges: Targeting Tool, Harpoon Piezometer and More," FOCUS Conference on Eastern Regional Ground Water Issues," Kitchener, Ontario, Canada, October 17-19, 1989, p. 8.

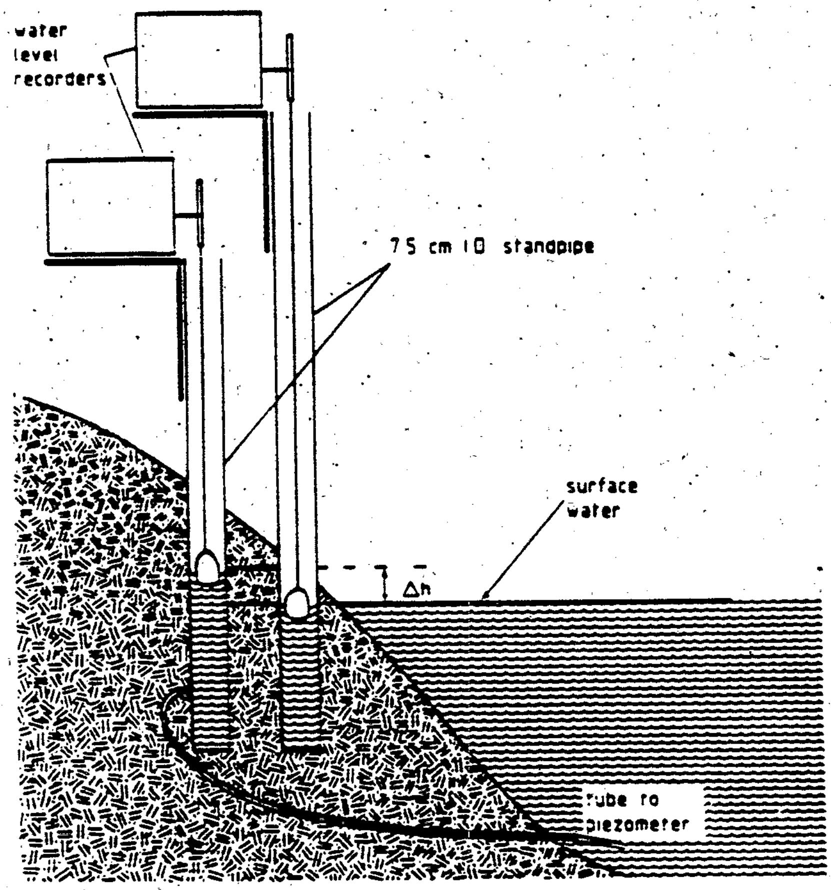
Figure 2-4



The manometer used to measure differential heads in minipiezometers. A. principle of operation: B. the field apparatus.

Lee, David R., and John A. Cherry: "A Field Exercise on Ground-Water Flow Using Seepage Meters and Mini-Piezometers," Journal of Geologic Education, 1978, Volume 27: p. 8.

Figure 2-5



Continuous measurement of head differences between piezometric level and river level using two water-level recorders.

Lee, David R., and Stephen J. Welch: "A Method for Installing and Monitoring Piezometers in Beds of Surface Waters," Ground Water, 1989, Volume 27(1): p. 89.

In geologic units having permeabilities too low to allow easy withdrawal of water from a piezometer tip, a piston corer may be used to obtain a continuous vertical profile of sediment (Lee, 1988). Munch and Killey (1985) have used a modified piston corer featuring a thin-wall core barrel and wireline recovery to sample both cohesive and cohesionless sediment from depths up to 30 m below the water table. The porewater can then be extracted from the piston core for chemical analysis.

b. Assumptions involved in using seepage meters and mini-piezometers

<u>Flow rate is uniform through the sampling interval</u>

Seepage meters and mini-piezometer infer that measured ground-water quantity and quality are representative of actual conditions throughout the sampling interval. The ground-water discharge rate recorded using a seepage meter or mini-piezometer represents the average ground-water discharge rate for the collection period. If a single discharge measurement or a series of discharge measurements recorded over a short time period are used to determine the ground-water discharge rate to a surface-water body, the calculated discharge rate may vary from actual rates.

<u>Sampled interval is representative temporally</u>

A series of discharge measurements taken over a short duration can vary substantially due to tidal cycles, storm events, and seasonal changes. Furthermore, seasonal variations within and between years may be substantial. It is possible that a series of discharge measurements recorded over a long duration may not be representative of actual conditions if the measurements were recorded in excessively wet or dry years.

<u>Sampling placement is representative spatially</u>

Seepage meters and mini-piezometers provide point measurements that determine the ground-water discharge rate and loading rate to a surface-water body through extrapolation (Goodman et al., 1989). The representativeness of the sampling locations and the number of locations influence the accuracy of the results. For example, Belanger and Connor (1980) not only found decreasing seepage rates with increasing distance from shore, but also that ground-water recharge occurred toward the center of East Lake Tohopekaliga. An overestimation of ground-water seepage would result if seepage meters used in the study were all located near shore. Conversely, if all the seepage meters were located toward the center of the lake, one would erroneously conclude that the entire lake was recharging ground water.

<u>Measured samples are representative of ground-water discharge quality</u>

Measured ground-water quality may not be representative of actual conditions because of interactions occurring at the sediment/surface-water interface. Belanger and Mikutel (1985) concluded that direct determination of water quality using seepage meters overestimated nutrient loading to lakes due to the enclosure of bottom sediments, which results in anaerobic conditions and increased release rates of ammonium, nitrogen, and phosphate. In addition, seepage meter or mini-piezometer samples from shallower, shoreline locations may be influenced by bank storage water.

c. Limitations of the methods

Placement on different surface-water bottoms

Not all bottom areas of a surface-water body are conducive to installation of seepage meters or mini-piezometers. Because seepage meters and mini-piezometers require insertion into bottom sediments, ideal installation locations are areas with relatively soft, fairly thick, moderately permeable sediments containing few cobbles or stones. However, German has successfully installed seepage meters in cobbles and rocks using bentonite placement.[4]

Deep surface waters require additional expertise and equipment

Seepage meters and mini-piezometers located in deep water require scuba abilities and equipment for installation, sampling, and maintenance. Depths that limit divers' safe performance control installation depths (Woessner and Sullivan, 1983). Additionally, some bottom locations are not suited to installation of seepage meters and mini-piezometers.

Strong currents and harsh seasons

Without modification, mini-piezometers and seepage meters should not be used with strong currents. Acceptable installation locations vary with seasons in areas due to wave and current action. Additionally, in colder climates, ice covering surface waters may limit seepage meters and mini-piezometers sampling and maintenance activities.

Sklash has overcome some of these problems.[5] In his investigation, handles placed on seepage meters aided divers in fast currents. Also, once the seepage meter is placed, bolts are used to clamp it down and ensure its stability. To protect seepage bags from the elements, Sklash used rapid disconnects for the bags and placed rigid containers around them.

Maintenance

Seepage meters and mini-piezometers equipped with sample collection devices require substantial maintenance. Without frequent changes, the increased pressure associated with a full catchment device reduces the amount

[4] German, Dave, personal communication, Nonpoint Source Contaminated Ground-water Discharge to Surface Water Workshop, Chicago, IL, November 30, 1989.

[5] Sklash, Mike, personal communication, Nonpoint Source Contaminated Ground-water Discharge to Surface Water Workshop, Chicago, IL, November 30, 1989.

of ground-water flow into the seepage meter. The sample collection tubing for seepage meters and mini-piezometers requires regular cleaning or replacement to prevent algae growth. There may be a need to periodically replace seepage meters and mini-piezometers due to wave and current action.

Anaerobic environment within equipment

A final limitation of seepage meters and mini-piezometers is that anaerobic conditions develop within the seepage meter, altering the chemistry of the discharging ground water (Belanger and Mikutel, 1985). As a result, calculated loading rates to surface water, determined using seepage meters, may not be representative of actual seepage from the ground water.

d. Representative equations

Darcy's Law is used to calculate the ground-water discharge rate to surface water when using mini-piezometers. The form of Darcy's Law used is:

$$Q = K(dh/dl)A$$

where

Q	=	Ground-water discharge rate $[L^3/T]$
K	=	Hydraulic conductivity $[L/T]$
dh/dl	=	Hydraulic gradient $[L/L]$
A	=	Surface area of the bottom of the surface-water body $[L^2]$.

The loading rate to surface water as a result of ground-water discharge is:

$$LR = QC$$

where

LR	=	Loading rate $[M/T]$
Q	=	Ground-water discharge rate $[L^3/T]$
C	=	Chemical constituent concentration in ground water $[M/L^3]$.

e. Description of field equipment

Equipment and materials often used for installation, sampling, and maintenance of seepage meters include:

- open-ended 55-gallon drum with vent hole,
- tubing,
- plastic seepage bag,
- boat, and
- scuba gear.

Typical equipment and materials required for installation, sampling, and maintenance of mini-piezometers include:

- metal pipe,

- end plugs,
- tubing,
- nylon mesh,
- hammer,
- sample bag, and
- boat.

For a detailed description of seepage meters installation and sampling, see Lee (1977). Installation and sampling of seepage meters in deeper, more turbulent water often requires additional equipment.

f. Expertise needed to apply method

Siting sampling locations requires a sufficient understanding of regional geology and hydrology. Extrapolating sampling results also requires expertise in geostatistical methods needed to delineate areas of similar sediment type for seepage meter or mini-piezometer location. In near shore areas where wading is possible, seepage meters and mini-piezometers are relatively easy to install and maintain. Farther from the shore, installation of mini-piezometers may be made from a surface platform as described by Welch and Lee (1989). The surface platform sits on top of two 14 foot boats, and consists of plywood flooring, 2 by 4's, ropes, bolts, and a central reinforced joist (see Figure 2-6). As water depth increases, seepage meter and mini-piezometer installation and maintenance may require scuba divers. The need to frequently change and check sample collection bags renders the method labor intensive, especially with dive team involvement.

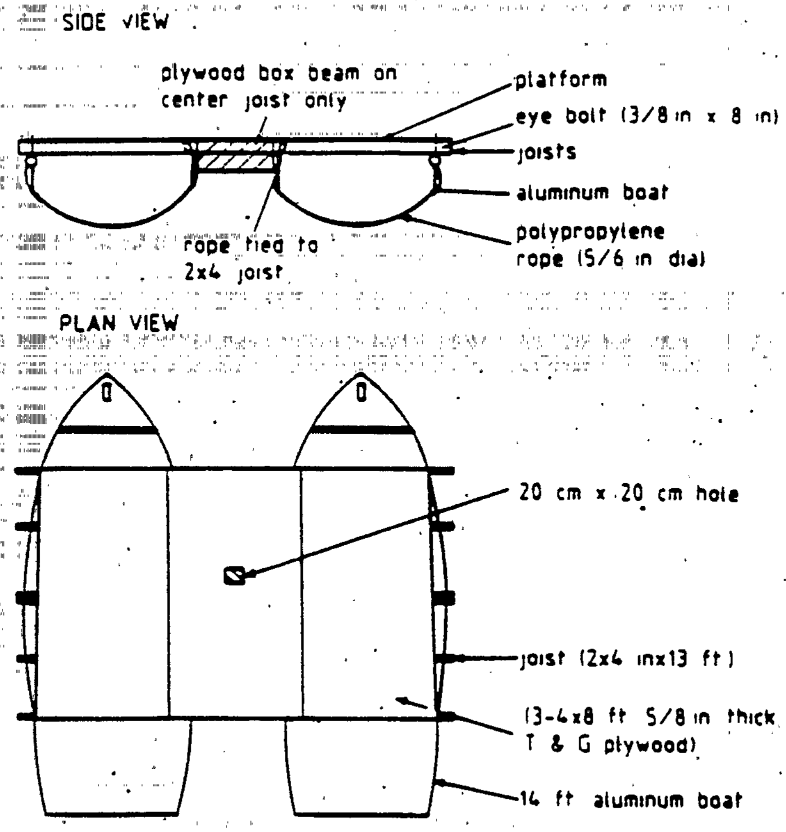
ii. Data inputs for the method

Extrapolation of sample results to areas with similar sediment characteristics requires knowledge of the spatial distribution of bottom sediments. If mini-piezometers are used, an estimate of the hydraulic conductivity of the bottom sediment is needed to determine the ground-water discharge rate.

iii. Outputs from the method

Seepage meters and mini-piezometers provide a direct measurement of ground-water discharge to surface water at a given location. In conjunction with data characterizing the areal distribution of sediment types, these data can provide estimates of the total loading from ground-water seepage.

Figure 2-6



Side and plan view. Working platform for installing piezometers and coring sediments from the water surface.

Lee, David R., and Stephen J. Welch: "A Method for Installing and Monitoring Piezometers in Beds of Surface Waters," Ground Water, 1989, Volume 27(1): p. 87.

a. Ground-water quantity discharge to surface water

A wide range of seepage rates can be measured using seepage meters and mini-piezometers. For example, in Minnesota and Nova Scotia, seepage velocities have been measured in the range of 0.1 to 2.58 $\mu m\ sec^{-1}$ using the simple seepage meter described above (Lee, 1977).

b. Ground-water quality discharge to surface water

Seepage meters allow for the collection of samples for water quality analysis. The quantification limit for the sample is a function of the detection limits for the constituent of concern. The anaerobic conditions within these sampling devices may affect sample integrity.

iv. Settings in which the method has been applied and contaminants that have been measured using this method

Summaries in Table A-1 describe some of the locations where seepage meters and mini-piezometers have been used successfully and the contaminants measured using the method.

v. Evaluation of the method

Seepage meters and mini-piezometers provide a simple, direct method to measure the quantity and quality of ground-water discharge to surface water. As with any ground-water method, the point measurements obtained represent moments in time and space for estimates of the quantity and quality of ground-water discharge to surface water. The inherent variability of most earth materials will require a large number of spatially distributed measurements in order to characterize discharge and loading rates accurately. Use of this method over large areas will require a substantial commitment of resources. However, the fact that sampling and flow measurements are made in or near the surface-water body can provide an accurate indication of the contaminant inflows without the necessity of installing monitoring wells on land and then extrapolating results to the points of discharge.

vi. References to annotated bibliography

References to the annotated bibliography presented in the accompanying volume to this document are provided in Table A-2.

Table A-1. Summary of Settings in Which the Method Has Been Applied and the Contaminants Measured.

Location	Aquifer	Contaminant	Author
Ontario	-	-	D. R. Lee, S. J. Welch
Sites near Leamington, Ontario and at Cape Cod Massachusetts	-	Nitrates (Ontario)	D. R. Lee
Chalk River, Ontario Douglas	-	-	J. B. Munch, R. W.
Key Largo National Marine Sancturay, Florida	- -	Pesticides Heavy Metals	G. M. Simmons Sr. F. G. Love
Key Largo National Marine Sanctuary, Florida	- -	Nitrates, selected cations, total phosphates	G. M. Simmons Jr. J. Netherton
Osceola County, Florida	Floridan Aquifer	Phosphorous, Nitrogen	T. V. Belanger D. F. Mikutel
Wisconsin	-	Nitrate, Phosphorous, Ammonia	T. D. Brock, D. R. Lee, D. Janes, D. Winek
Michigan and Wisconsin	-	-	D. A. Cherkauer, J. M. McBride
Brevard County, Florida Belanger	-	Chloride	J. N. Connor, T.V.
Upper Great Lakes Connecting Channels	Shallow glacial, glacial bedrock interface and bedrock units.	Phosphorous, Chromium, Lead, Barium, Zinc, Cobalt Nickel, Phenols	EPA Non Point Source Work Group
Orlando, Florida Brezonik	-	Nitrate	C. R. Fellows, P. L.
South Dakota	-	Nitrogen, Phosphorous, Pesticides	J. Goodman et al.
Colorado Winter	-	-	J. W. LaBaugh, T.C.
Eastern Ontario	-	-	D. R. Lee, J. A. Cherry, F. Pickens

Table A-1. Summary of Settings in Which the Method Has Been Applied and the Contaminants Measured. (Continued)

Location	Aquifer	Contaminant	Author
Eastern Ontario	-	Tritium	D. R. Lee, J. A. Cherry
Minnesota, Wisconsin North Carolina, Nova Scotia	-	Phosphates, Nitrates, Ammonia, Chloride	D. R. Lee
Southern Ontario	-	Nutrients	D. R. Lee, H.B.N. Hynes
Barbados, West Indies	Barbados Aquifer	Nitrogen, Phosphorous	J. B. Lewis
Southeastern Virginia	Shirley, Yorktown, and Tabb Formations	Inorganic Nitrogen	W. G. MacIntyre, G. H. Johnson, W. G. Reay, G. M. Simmons, Jr.
Minnesota	-	Phosphorous, Nitrogen	J. K. Neel, R. M. Brice
Holbrook, Massachusetts Ostyre,	-	Volatile Organics and Inorganics	W. R. Norman, D. P. J. S. Hobin
Mahantago Creek, Gburek, Pennsylvania	Manhantago Creek Basin	Nitrogen, Phosphorous	H. B. Poinke, N. J. N. J. Gburek et. al.
Chicago, Cook County, Demissie, Illinois	Cambrian and Ordovician Aquifers	E.P. Metals	M. S. Henebry, M. et al.
Virginia's Eastern Shore	-	Nitrate, Ammonia, Total Phosphorous	G. M. Simmons, Jr.
Sault Ste. Marie, Ontario	-	-	S. J. Welch, D. R. Lee
West Thorton, New Hampshire	-	-	T. C. Winter
Lake Mead, Nevada Sullivan	Tertiary-Cretaceous Gale-Hills Formation	TDS, calcium-sulfate	W. W. Woessner, K.
East Coast of Florida	-	Phosphate	C. F. Zimmerman, J. R. Montgomery, P. R. Carlson

Table A-2. References to Annotated Bibliography

Author	Citation	Reference to Annotated Bibliography
T. V. Belanger, D. F. Mikutel	"On the Use of Seepage Meters to Estimate Ground-Water Nutrient Loading to Lakes," Water Resources Bulletin, 1985, Volume 21(2): 265-272.	pp.2-3
R. Carr, T. C. Winter	"An Annotated Bibliography of Devices Developed for Direct Measurement of Seepage," U.S. Geological Survey Open File Report 80-344, 1980.	p.8
D. A. Cherkauer, J. M. McBride	"A Remotely Operated Seepage Meter for Use in Large Lakes and Rivers," Ground Water, 1988, 26(2): 165-171.	pp.9-10
J. N. Connor, T. V. Belanger	"Ground Water Seepage in Lake Washington and the Upper St. Johns River Basin, Florida," Water Resources Bulletin, 1981, 17(5): 799-805,	pp.11-12
EPA Non Point Source Work Group	"Upper Great Lakes Connecting Channel Study, Waste Disposal Disposal Sites and Potential Ground Water Contamination, St. Clair River," Non Point Source Work Group Report, April, 1988.	pp.13-17
C. R. Fellows, P. L. Brezonik	"Fertilizer Flux into Two Florida Lakes Via Seepage," Journal of Environmental Quality, 1980, Volume 10(2): 174-177.	pp.18-19
J. Goodman et al.	"Oakwood Lakes - Poinsett: Rural Clean Water Program Comprehensive Monitoring and Evaluation Technical Report, Project 20," Rural Clean Water Program Comprehensive Monitoring and Evaluation Technical Report, Project 20, May, 1989.	pp.20-21
J. W. LaBaugh, T. C. Winter	"In Impact of Uncertainties in Hydrologic Measurement on Phosphorous Budgets and Empirical Models for Two Colorado Reservoirs," Limnology and Oceanography, 1984, Volume 29(2): 322-339.	pp.22-23

Table A-2. References to Annotated Bibliography (Continued).

Author	Citation	Reference to Annotated Bibliography
D. R. Lee	"Six In-Situ Methods for Study of Groundwater Discharge," Proceedings of the International Symposium on Interaction Symposium on Interaction Between Groundwater and Surface Water, 30 May-3 June, 1988, Ystad, Sweden, edited by Peter Dahlblom and Gunner Lindh, Department of Water Resources Engineering, Lund University, Sweden.	pp.27-29
D. R. Lee, S. J. Welch	"Methodology for Locating and Measuring Submerged Discharges: Targeting Tool, Harpoon Piezometer and More," FOCUS Conference on Eastern Regional Ground Water Issues: October 17-19, 1989, Kitchener, Ontario, Canada, Co-sponsored by the Association of Ground Water Scientists and Engineers, Division of NWWA and Waterloo Center for Groundwater Research, University of Waterloo.	pp.37-39
T. D. Brock, D. R. Lee, David Janes, David Winek	"Ground-Water Seepage as a Nutrient Source to a Drainage Lake; Lake Mendota, Wisconsin,"	pp.6-7
D. R. Lee, J. A. Cherry, J. F. Pickens	"Ground-Water Transport of a Salt Tracer through a Limnology and Oceanography, 1980, Volume 25(1): 45-61.	pp.32-34
D. R. Lee, J. A. Cherry	"A Field Exercise on Ground-Water Flow Using Seepage Meters and Mini-piezometers," Journal of Geological Education, 1978, Volume 27: 6-10.	pp.30-31
D. R. Lee	"A Device for Measuring Seepage Flux in Lakes Lakes and Estuaries," Limnology and Oceanography, 1977, Volume 22(1): 140-147.	pp.24-26
D. R. Lee, H.B.N. Hynes	"Identification of Groundwater Discharge Zones in a Reach of Hillman Creek in Southern Ontario," Water Pollution Research Canada, 1978, 13: 121-133.	pp.35-36
J. B. Lewis	"Measurements of Ground-Water Seepage Flux onto a Coral Reef: Spatial and Temporal Variations," Limnology and Oceanography, 1987, 32(5): 1165-1169.	pp.40-41

Table A-2. References to Annotated Bibliography (Continued).

Author	Citation	Reference to Annotated Bibliography
W. G. MacIntyre, G. H. Johnson, W. G. Reay, G. M. Simmons, Jr.	"Ground-Water Non-Point Sources of Nutrients to the Southern Chesapeake Bay," Proceedings of <u>Ground Water Issues and Solutions in the Potomac River Basin/Chesapeake Bay Region</u>, Co-sponsored by the Association of Ground Water Scientists and Engineers, pp. 83-104.	pp.42-43
J. K. Neel, R. M. Brice	"Watershed and Point Source Enrichment and Lake State Index," US EPA, April 1979. EPA-600/3-79-046.	pp.44-46
W R. Norman, D. P. Ostrye, J. S. Hobin	"Use of Seepage Meters to Quantify Ground-Water Discharge and Contaminant Flux into Surface Water at the Baird and McGuire Site (NPL No. 14)," <u>Proceedings of Third Annual Eastern Regional Ground Water and Conference</u>, 1986, p. 472-491.	pp.47-49
H. B. Pionke, J. R. Hoover, R. R. Schnabel, W. J. Gburek, J. B. Urban, A. S. Rogowski	"Chemical-Hydrologic Interactions in the Near-Stream Zone," Zone," <u>Water Resources Research</u>, 1988, Volume 24(7): 1101-1110.	pp.50-52
P. E. Ross, M. S. Henebry, J. B. Risatti, T. J. Murphy, M. Demissie County, Illinois	"A Preliminary Environmental Assessment of the Contamination Associated with Lake Calumet Cook <u>Hazardous Waste Research and Information Center</u>, Illinois State Water Survey, 1988, HWRIC RR-019, 88/300.	pp.53-55
G. M. Simmons Jr., F. G. Love	"Water Quality of Newly Discovered Submarine Ground Water Discharge into a Deep (Coral Reef Habitat," <u>NOAA Symposium series for Undersea Research</u>, Volume 2(2): 155-163.	pp.60-61
G. M. Simmons Jr., J. Netherton	"Groundwater Discharge in a Deep Coral Reef Habitat: Evidence for a New Biogeochemical Cycle?" Diving for Science...86, Proceedings of the Sixth Annual Scientific Diving Symposium (1986), Tallahassee, Florida, Charles T. Mitchell, editor.	pp.62-64

Table A-2. References to Annotated Bibliography (Continued).

Author	Citation	Reference to Annotated Bibliography
G. M. Simmons, Jr.	"Understanding the Estuary Advances in Chesapeake Research," Proceedings of a Conference, March 29-31, 198, Baltimore, Maryland, Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.	pp.56-57
G. M. Simmons, Jr.	"The Chesapeake Bay's Hidden Tributary: Submarine Groundwater Discharge," Proceedings of Ground Water Issues and Solutions in the Potomac River Basin/Chesapeake Bay Region, Co-sponsored by the Association of Ground Water Scientists and Engineers, pp. 9-29.	pp.58-59
S. J. Welch, D. R. Lee	"A Method for Installing and Monitoring Piezometers in Beds of in Beds of Surface Waters," Ground Water, 1989 27(1): 87-90.	pp.65-66
T. C. Winter	"Geohydrologic Setting of Mirror Lake, West Thorton, New Hampshire," 1984, U.S. Geological Survey Water Resources Investigations Report, 84-4266, 61 pp.	pp.67-68
W. W. Woessner, K. Sullivan	"Use of Seepage Meters and Mini-piezometers for Identification of Reservoir - Groundwater Interactions in Lake Mead, Nevada," Desert Research Institute Water Resources Center, 1983, PB 83-226894.	pp.69-70
C. F. Zimmerman, J. R. Montgomery, P. R. Carlson	"Variability of Dissolved Reactive Phosphate Flux Rates in Nearshore Estuarine Sediments," Estuaries, 1985, 8(2B): 228-236.	pp.71-72

B. **Ground-water quality sampling and measurements of ground-water flow to estimate loading to surface water**

The papers cited in this section are summarized in Section VIII of "An Annotated Bibliography of the Literature Addressing Nonpoint Source Contaminated Ground-Water Discharge to Surface Water," September, 1990, EPA 440/6-90-006.

1. **General description of method**

a. **Description of procedures**

Water-level elevation measurements from piezometers and ground-water wells provide an indication of the quantity of ground water discharging to surface water in a watershed. This method uses water-level measurements obtained from wells located in the watershed to develop a water table contour map. Darcy's Law is then applied to calculate the discharge rate of ground water to surface water by incorporating estimates of hydraulic conductivity and the cross-sectional area of the aquifer. By assuming the aquifer underlying the watershed is homogenous and isotropic, the water-level contour map can be used to determine flow directions and horizontal gradients in the basin. In a homogeneous, isotropic aquifer, flow lines will be perpendicular to equipotentials. Hydraulic conductivity can be estimated for a particular rock or soil type or can be measured in-situ via aquifer tests. Aquifer geometry is estimated by examining lithologic logs of wells in the watershed.

This method is often used in conjunction with other methods, such as mini-piezometers, seepage meters, tracer studies, isotopic studies, or water and mass balances analyses to verify study results. This method has been practiced in both marine and fresh water environments and has been used on a large scale, such as in Long Island (Franke and McClymonds) and North Central Kansas (Spruill) and on a smaller scale such as in South Farmingdale, New York (Perlmutter and Lieber) and the Stockett-Sand Coulee coal field, Montana (Osborne, et.al.). This method has been used for glacial and dolomite aquifers. Contaminants studied include metals, nutrients, and some organic constituents.

Ground-water samples taken from wells within the basin can be used to characterize the spatial distribution of ground-water quality as a means of estimating nonpoint source contaminant load. To properly characterize ground-water quality in a drainage basin, potential nonpoint source loading areas should be identified and the underlying ground water sampled. Agricultural areas located on soils allowing rapid infiltration of precipitation are of particular concern. Such areas are identified from soil and land use maps (Hallberg et al., 1983). Evaluation of ground-water quality beneath nonpoint source loading areas over time will indicate qualitatively whether the loading rate to surface water, as a result of ground-water discharge, will increase or decrease in the future.

Ground-water quality in wells adjacent to the surface-water body are assumed to be representative of ground water discharging to surface water. By using the constituent concentrations and the calculated ground-water flux, the immediate loading rate to surface water from ground-water discharge can be calculated.

b. Assumptions involved in these methods

To use Darcy's Law to calculate ground-water discharge to surface water, it is assumed that the aquifer is homogeneous, isotropic, of constant thickness, and that flow is horizontal. The assumption that the aquifer is homogeneous and isotropic is necessary to ensure that flow lines are perpendicular to equipotentials (Perlmutter and Lieber, 1979). By assuming a constant aquifer thickness and horizontal flow, the one-dimensional version of Darcy's Law can be used. Additionally, water quality in sampled wells is assumed to be representative of the quality of the water discharging to the stream.

c. Limitations of the method

Aquifer characteristics

The limitations of this method reflect the natural variability of aquifers and the availability of information on aquifer characteristics. In nature, considerable heterogeneity exists and few, if any, aquifers are homogeneous, isotropic, and of constant thickness. In a heterogeneous, anisotropic aquifer, ground-water flow is not perpendicular to equipotentials, and the angle between flow direction and equipotentials is not constant. Because the predicted flow path length differs from the actual flow path length, the calculated hydraulic gradients will not be representative of the actual gradient. Additionally, horizontal gradients determined using wells screened at different depths below the water table or in different geologic formations may not represent the actual horizontal gradient. Temporal changes in the hydraulic conductivity due to changes in seepage face from precipitation events increase the difficulty of estimating an average ground-water discharge. Hydraulic conductivity also varies spatially and directionally in a heterogeneous, anisotropic aquifer. It would be difficult, if not impossible, to determine an accurate equivalent hydraulic conductivity and aquifer thickness for the basin. Because of the difficulties in determining horizontal hydraulic gradient, hydraulic conductivity, and aquifer thickness, precise determination of ground-water discharge to surface water is problematic (Koszalka, 1983).

Well installation

Installing the number of wells needed to properly characterize ground-water quality in a watershed is resource intensive. As an alternative to installing costly monitoring wells, existing production, domestic, and stock wells may be sampled. In many cases, however, these wells will not be in optimum locations or open to the geologic formation of interest. Additionally, water-quality results can be altered by well construction materials, faulty well construction, and sampling procedures. Consequently, ground-water quality in the basin may not be accurately characterized due to the construction and location of the well.

d. Representative equations

Darcy's Law is used in determining the quantity of flow entering surface water. The form of Darcy's Law used is:

$$Q = K(dh/dl)A$$

where

Q	=	Ground-water discharge rate (L^3/T)
K	=	Hydraulic conductivity (L/T)
dh/dL	=	Hydraulic gradient (L/L)
A	=	Cross sectional area of the aquifer (L^2).

The loading rate to surface water, as a result of ground-water discharge, is determined by multiplying the ground-water discharge rate by the concentration of the constituent in the ground water. The loading rate equation is:

$$LR = QC$$

where

LR	=	Loading rate (M/T)
Q	=	Ground-water discharge rate (L^3/T)
C	=	Concentration in ground-water (M/L^3).

e. Description of field equipment

After the wells have been installed, equipment is required to measure water levels and obtain ground-water samples. Suggested equipment includes:

- steel tape and chalk or electric well sounder,
- submersible pump,
- centrifugal pump,
- bailer,
- pH meter,
- conductivity meter,
- thermometer,
- appropriate sample containers, and
- coolers.

f. Expertise needed to apply method

This method involves an initial review of well log records for existing wells in the watershed to evaluate whether the existing well network can properly characterize the hydrogeology and water quality of the watershed (Hallberg et al., 1983). If more wells are needed, the number, locations, and depths are determined. After the well network for the watershed has been identified, water-level measurements and ground-water samples are obtained. Obtaining water-level measurements and ground-water samples is relatively easy, but often labor intensive. The water-level measurements are used to construct a water-table contour map from which the horizontal hydraulic gradient is determined. The greatest difficulty in applying the method is in determining aquifer characteristics and geometry. Aquifer tests are labor intensive and sometimes difficult to interpret. Additionally, aquifer characteristics determined for one portion of a watershed must be extrapolated to other portions of the watershed (Koszalka et al., 1985).

ii. Data inputs

To apply Darcy's Law and estimate the ground-water loading rate for a watershed, level elevations, hydraulic conductivity, aquifer geometry, and chemical constituent concentrations in ground water are needed. Well construction information is essential to determine the subsurface zones that the hydraulic head and water quality measurements represent.

iii. Outputs from the method

a. Quantity of ground water

A wide range of ground-water discharge to surface water rates can be estimated using this method. The factors controlling the quantity of ground-water discharge to surface water are hydraulic conductivity and gradient. If the hydraulic conductivity of the aquifer is low, and the hydraulic gradient across the aquifer is minimal, the ground-water discharge rate to surface water will also be low. Conversely, a high ground-water discharge rate to surface water will occur when the hydraulic conductivity and gradient for an aquifer are high.

b. Quality of ground water

A broad range of loading rates to surface water as a result of ground-water discharge can be predicted using this method. The ability to determine ground-water quality in sampled wells is limited only by the characteristics of the well materials and the quantitation limits for the individual constituents.

iv. Settings in which the method has been applied and contaminants that have been measured using this method

Some of the locations where this method has been used and the contaminants that have been measured using the method are summarized in Table B-1.

v. Evaluation of the method

Most watersheds contain observation or water supply wells that can be used to obtain water level elevations and water quality data, making this method applicable in many locations. The method can qualitatively determine the amount of ground-water discharge and the loading rate to surface water within a watershed. Increasing the number of sampling locations will improve the predictive capabilities of the method.

Because the method is often applied with limited knowledge of aquifer characteristics, a large number of sampling points will not necessarily result in accurate quantification of ground-water discharge or loading rates to surface water. A comparable qualitative indication of the loading rate to surface water as a result of ground-water discharge can be obtained by observing water-quality trends in the watershed. If, within a watershed, chemical constituent concentrations in ground water increase with time, the future loading rate to surface water as a result of ground-water discharge can also be expected to increase. In addition to using the method to quantitatively predict ground-water loading rates to surface water, this method can be used to qualitatively assess the impacts of various regulatory scenarios governing fertilizer and pesticide usage on ground-water quality.

vi. References to annotated bibliography

References to the accompanying annotated bibliography are located in Table B-2.

Table B-1. Summary of Settings in Which the Method Has Been Applied and the Contaminants Measured.

Location	Aquifer	Contaminant	Author
Patchogue, Long Island, New York	-	Nitrate Nitrogen	D. Capone, M. Bautista
Penfield, New York	Lockport dolomite	Sodium Chloride	L. R. Davis
Upper Great Lakes Connecting Channels	Shallow glacial, glacial bedrock interface, & bedrock units	Zinc, Phenols, Phosphorous	EPA Non-Point Source Group
Long Island, New York	Upper Glacial, Magothy, and Lloyd	TDS, Inorganic Metals	O. L. Franke, N. E. McClymonds
Clayton City, Iowa	Galena	Nitrates, Herbicide Pesticides, Bacteria & turbidity	G. R. Hallberg, B. E. Hoyer E. A. Bettis, III, R. D. Libra
Perth, Australia	-	Nitrate	R. E. Johannes
Niagara County, New York	-	Inorganic & Organic Constituents	E. J. Koszalka, J. E. Paschal T. S. Miller, P. B. Duran
Stockett and Sand Coulee, Montana	Kootenai Formation, Morrison Formation	Heavy Metals	T. J. Osborne, J. L. Sonnèregger, J. J. Donovan
Nassau County, New York	Magothy	Cadmium, Chromium	N. M. Perlmutter, M. Lieber
North Central Kansas	Almena, Kansas Bostwick, Cedar Bluff Units	Sulfate, Sodium Chloride, Calcium	T. B. Spruill
Butte, Mead and Lawrence, South Dakota	-	Arsenic, Selenium	R. L. Stach, R. N. Helgerson, R. F. Bretz, M. J. Tipton, D. R. Biessel, J. C. Harksen
Arkansas River Basin	-	TDS, Salinity, Chloride	J. D. Stoner
Schwatka Lake, Yukon Territory, Canada	-	Nitrogen, Phosphorous	P. H. Whitfield, B. McNaughton, W. G. Whitley

Table B 2. References to Annotated Bibliography

Author	Citation	Reference to Annotated Bibliography
D. Capone, M. Bautista	"A Ground-Water Source of Nitrate in Nearshore Marine Sediment," Nature, 1985, Volume 313: 214-216.	pp.198-199
L. R. Davis	"The Effects of Deicing Salt Usage on Surface and Ground Water Quality," 1982 International Symposium on Urban Hydrology, Hydraulics & Sediment Control, University of Kentucky, Lexington, Kentucky, July 27-29, 1982.	pp.200-201
EPA Non Point Source Group	"Upper Great Lakes Connecting Channel Study, Waste Disposal Sites and Potential Ground Water Contamination, St. Clair River" Non Point Source Work Group Report, April 1988."	pp.202-206
O. L. Franke, N. E. McClymonds	"Summary of the Hydrologic Situation on Long Island, New York, as a Guide to Water-Management Alternatives," U. S. Geological Survey Professional Paper 627-F, 59p.	pp.207-209
G. R. Hallberg, B. E. Hoyer, E.a. Bettis, III, R. D. Libra	"Hydrogeology, Water Quality, and Land Management in the Big Spring Basin, Clayton County, Iowa," Iowa Geological Survey, Open-File Report 83-3, 1983 Report on contract 82-5500-002.	pp.210-211
R. E. Johannes	"The Ecological Significance of the Submarine Discharge of Groundwater," Marine Ecology --- Progress Series, 1980, 3: 365-373.	pp.212-213
E. J. Koszalka, J. E. Paschal, T. S. Miller, P. B. Duran	"Preliminary Evaluation of Chemical Migration to Ground Water and the Niagara River from Selected Waste - Disposal Sites," USEPA, March 1985, EPA 905/4-85-001.	pp.214-216
T. J. Osborne, J. L. Sonneregger, J. J. Donovan	"Interaction between Groundwater and Surface Water Regimes and Mine-induced Acid - Mine Drainage in the Stockett-Sand Coulee Coal Field," Montana Joint Water Resources Research Center, 1983, Project No. A-129MONT, Bozeman, Montana.	pp.217-219
N. M. Perlmutter, M. Lieber	"Dispersal of Plating Wastes and Sewage Contaminants in Ground Water and Surface Water, South Farmingdale - Massapequa Area, Nassau County, New York," U.S. Geological Survey Water Supply Paper 1879-G.	pp.222-223

Table B 2. References to Annotated Bibliography (Continued)

Author	Citation	Reference to Annotated Bibliography
T. B. Spruill	"Statistical Evaluation of the Effects of Irrigation on Chemical Quality of Ground Water and Base Flow in Three River Valleys in North Central Kansas," U.S. Geological Survey Water Resource Investigation Report 85-4156, 1985.	pp.224-226
R. L. Stach, R. N. Helgerson, R. F. Bretz, M. J. Tipton, D. R. Biessel, J. C. Harksen	"Arsenic Levels in the Surface and Ground Waters along Whitewood Creek, Belle Fourche River, and a portion of the Cheyenne River, South Dakota," Completion Report, Project Number A-054-SDAK, Agreement Number 14-34-0101-6043, July, 1978.	pp.227-228
J. D. Stoner	"Dissolved Solids in the Arkansas River Basin," National National Water Summary 1984: Hydrologic Events, Selected Water Quality Trends, and Ground-Water Resources, U.S. Geological Survey Water Supply Paper 2275.	p.229
P. H. Whitfield, B. McNaughton	"Indications of Ground-Water Influences on Nutrient Transport Through Schwatka Lake, Yukon Territory," Water Resources Bulletin, 1982, 18(2): 197 - 203.	pp.230-232

C. Studies involving geophysical techniques to estimate ground-water discharge to surface water

The papers cited in this section are summarized in Section II of "An Annotated Bibliography of the Literature Addressing Nonpoint Source Contaminated Ground-Water Discharge to Surface Water," September 1990, EPA 440/6-90-006.

i. General description of method

a. Description of method or procedure

Ground water discharging to surface water is controlled by the hydraulic properties of the sediments of the surface-water body and the hydraulic gradient across those sediments. The sediment hydraulic properties of large water bodies, such as the Great Lakes or the Chesapeake Bay, are difficult to measure due to the depth of the sediments in open water. Standard methods of drilling and sediment sampling become slow and costly endeavors in deep aquatic environments. Less costly shipboard geophysical systems offer a method that continuously characterizes bottom sediments along the ship's track. Combining seismic and electrical geophysical measurements provides data to estimate sediment type, thickness, and sequence, as well as relative vertical hydraulic conductivity. Based on this information, one can calculate the volume of ground water discharging to surface water.

Geophysical methods have primarily been applied to lakebeds. Bradbury and Taylor (1984) collected geophysical data at an offshore site in Lake Michigan with sediment thicknesses ranging from 0.3 to 37 m and water depths from 2.5 to 27 m. Other investigators have used geophysical techniques in smaller lakes and in channels connecting the Great Lakes (see Bradbury and Taylor, 1984; Cherkauer and Taylor, 1987; Lee, 1989, and Taylor and Cherkauer, 1984). Zektser and Bergelson (1989) have used continuous measurements of temperature and electric conductivity and continuous seismoacoustic profiling to detect temperature and salinity anomalies in Lake Issyk-Kul in the southeastern USSR. One major difficulty associated with geophysical techniques is the need for field tests to verify the results. Field verification can be difficult to obtain in deep water.

<u>Seismic</u>

Seismic exploration involves generating seismic waves and measuring the time required for the waves to travel to a series of receiving devices called geophones. In seismic studies of large surface-water bodies, a shipboard seismic profiling system can generate and receive the seismic waves. Seismic waves generated on board the ship travel downward through the lake bottom sediment until they reflect off a hard surface and back up through the sediment to the ship's geophones. Information on sediment type, thickness, and sequence can be inferred through interpretation of the travel times of the seismic waves (Taylor and Cherkauer, 1984).

Induced Potential or Electrical Charge

This method involves charging the sediment with a current, shutting off the power source, and measuring the rate of current decay. An electrical array towed behind the boat charges bottom sediments and measures the rate of current decay. The relative clay content of bottom sediments can be determined using this method. These determinations are then used to estimate the vertical hydraulic conductivity of the sediment. Taylor and Cherkauer (1984) describe the equations characterizing the use of electrical conductivity and seismic readings used to estimate seepage (see Section C.i.d).

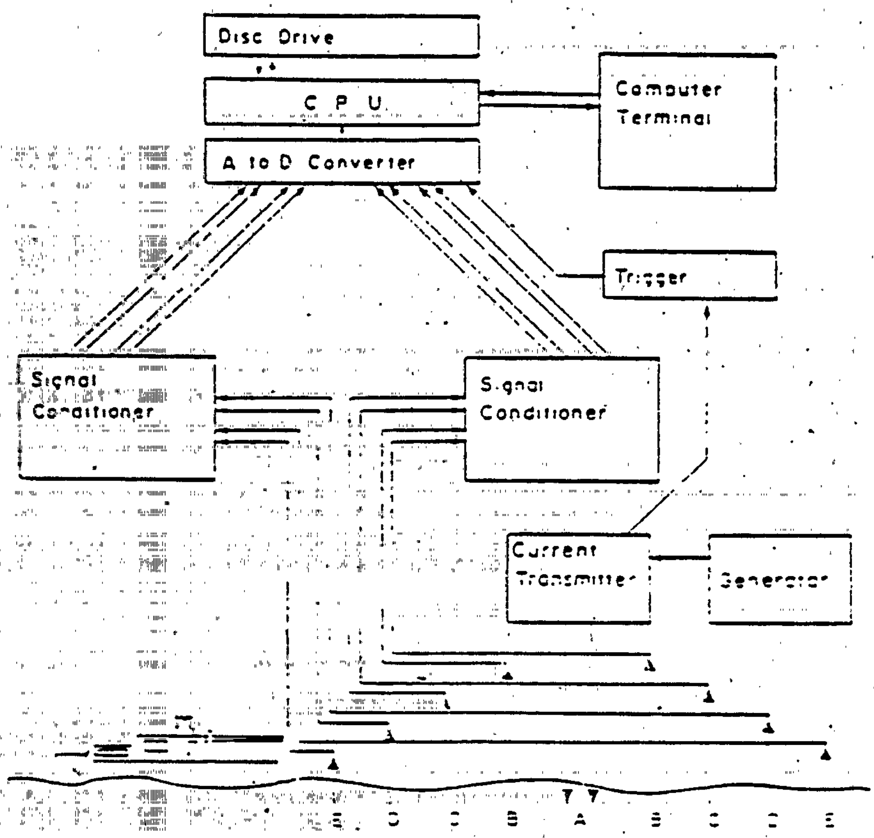
Resistivity

Resistivity methods also employ an artificial source of current which enters the subsurface through point-electrodes. Receiving electrodes measure the potentials of the electric flow field, which are influenced by the composition of the subsurface materials. An electrical array of source and receiving electrodes towed behind a boat (see Figure 2-7) measures the resistivity of an induced electrical field in the sediments. Sediment type, thickness, and sequence affect the configuration of the induced electrical field. Investigators infer the effective longitudinal conductance of the bottom sediments through interpretation of the resistivity of the induced electrical field. The effective longitudinal conductance, combined with sediment thickness information from seismic techniques and clay content estimates from electrical/resistivity techniques, provides data used to determine the effective vertical hydraulic conductivity of the sediment sequence. The effective vertical hydraulic conductivity, the hydraulic gradient over the sediment sequence (the change in hydraulic head over distance, measured at various points over a large area or assumed constant over the study area), and surface area of the water body bottom provide data to assess the likelihood and quantity of ground water discharging to surface water (Cherkauer and Taylor, 1984).

Temperature and Electrical Conductance

Another indirect method for locating ground-water discharge areas involves measuring temperature and bulk electrical conductance. A sediment probe with temperature and electrical conductance sensors is towed behind a boat along the bottom of a surface water body. From the continuous record of temperature and conductance, anomalies in temperature and bulk electrical conductance are located. These anomalies indicate the likelihood of ground-water inflow. Knowledge of the sediment type, water depth, and other geologic or hydrologic information concerning the nature of the possible discharge area may be needed for data interpretation. Investigators may correlate measured temperatures and conductivities with other techniques to better characterize the nature of sediment anomalies (Lee, 1989).

Figure 2-7



Block diagram of electrical instrumentation used in field survey

Taylor, Robert W., and Douglas S. Cherkauer, "The Application of Combined Seismic and Electrical Measurements to the Determination of the Hydraulic conductivity of a Lake Bed," Ground-Water Monitoring Review, 1984, Volume 4(4): p. 80.

b. Assumptions involved in using these models

Seismic

Reflective seismic methods assume an essentially horizontal reflective layer is located some distance beneath the sediment and will reflect the seismic waves. If the reflective layer is located a great distance beneath the sediment/surface-water interface, the seismic wave may be sufficiently damped, making reception and subsequent interpretation difficult. Seismic profilers have measured sediment thicknesses up to 37 m (Bradbury and Taylor, 1984).

Because the hydraulic gradient across the sediment sequence on the bottom of a surface-water body is difficult to measure, a uniform hydraulic gradient is often assumed for the entire body.

Electrical Charge and Resistivity

Electrical charge and resistivity studies require the assumption that a proportional relationship exists between clay content, hydraulic leakage (across the sediment/surface-water boundary) and the scaling coefficient. The scaling coefficient is required for the determination of hydraulic conductivity from electric and seismic readings (see Section C.i.d). For water bodies where clay content does not vary much, the scaling constant is assumed to be constant for the layered bottom sediments (Bradbury and Taylor, 1984).

Temperature and Electrical Conductance

Temperature and conductivity measurements require that temperature and conductivity differences in the sediment result from ground-water seepage. The temperature technique relies on the theory of ground-water flow distortion of thermal gradients, as indicated by ground-water discharge often producing relatively strong thermal gradients, while recharge gradients remain relatively weak. Other phenomena may be caused by sediment temperature anomalies, such as sediment storage of summer heat. The electrical conductance technique rests on the assumption that the distribution of solutes at the sediment/surface-water interface is expected to differ in areas where there is ground-water discharge as compared to areas where there is no ground-water discharge to the surface-water body (Lee, 1985).

c. Limitations of the methods

The uncertainties associated with geophysical methods limit their usefulness in determining ground-water discharge rates to surface water. Because these methods indirectly determine sediment thickness and effective vertical hydraulic conductivity for a sediment sequence, the uncertainty associated with the predicted values is greater than the uncertainty associated with values measured directly. If it is assumed that a single hydraulic gradient estimate represents the entire water body, further uncertainties may be introduced. No attempts were made in the studies reviewed for this report to quantify the effects that these assumptions had on the predicted rates of ground-water discharge to surface water. These uncertainties highlight the necessity of ground-truthing to verify the analytical results derived from these techniques.

The resistivity methods used to determined ground-water discharge rates are best suited to surface water environments where sediment overlies bedrock. The high resistivity of bedrock terminates the summation of the longitudinal electrical conductance equation and allows longitudinal conductance to be determined from a single electrical measurement.

d. Representative equations

The equations describing the longitudinal electrical conductance of a sequence of sediments are described by Taylor and Bradbury (1984):

$$S = \Sigma\, b_i/P_i$$

where:

S	=	Longitudinal electrical conductance [1/MLT]
b_i	=	Thickness of layer i [L]
P_i	=	Electrical resistivity of layer i [$1/ML^2T$].

The longitudinal electrical conductance, thickness, and clay content of a sediment sequence provide data for an equation describing the effective vertical hydraulic conductivity of the sediment sequence. To utilize the equation, it is necessary to assume clay content of a sedimentary sequence is accurately represented by an empirical scaling factor. The equation relating effective vertical hydraulic conductivity with longitudinal electrical conductance, sediment thickness, and a scaling factor is described by (Gherkauer and Taylor):

$$K_V = (C_o b_T)/S$$

where:

K_V	=	Effective vertical hydraulic conductivity [L/T]
b_T	=	Total thickness of sediment sequence [L]
S	=	Longitudinal electrical conductance [1/MLT]
C_o	=	Scaling factor [$1/MLT^2$].

After estimating the vertical hydraulic conductivity for a sediment sequence, one can apply Darcy's law to determine the quantity of ground water discharged to surface water by assuming a hydraulic gradient across the sediment sequence, and summing that gradient across the areal distribution of those sediment sequences with the same effective hydraulic conductivity. Darcy's Law is as follows:

$$Q = K_v dh/dlA$$

where:

Q	=	Ground-water discharge rate [L^3/T]
K_v	=	Effective vertical hydraulic conductivity [L/T]
dh/dl	=	Hydraulic gradient across the sediment sequence [L/L]

A - Area on bottom of surface-water body with similar effective vertical hydraulic conductivity $[L^2]$.

e. Description of field equipment

These methods usually require a boat, and sometimes a sizeable ship, to contain and deploy the geophysical instruments and support equipment used to characterize bottom sediments. Shipboard seismic instrumentation, used to determine sediment thickness, consists of a high resolution sounder and recorder. Electrical resistivity and chargeability equipment, used to determine the electrical longitudinal conductance and clay content of a sediment sequence consists of a long multiconductor cable equipped with source and receiving electrodes. The cable is towed behind the ship. A Loran navigation system determines the location of the ship's position for each measurement. A computer stores the position and measurement data and can assist in the interpretation of the data (Taylor and Cherkauer, 1984).

f. Expertise needed to apply the methods

The papers reviewed for this report suggest that geophysical methods require considerable expertise. Prior experience helps one to properly configure the instrumentation, conduct the tests, and interpret the results. Also, because a large boat must be used to house the geophysical system, these methods require navigational and piloting skills. For a more complete discussion of the expertise required to apply geophysical methods readers are referred to Taylor and Cherkauer (1984), Bradbury and Taylor (1989), Cherkauer and Taylor, and Lee (1985, 1989).

ii. Data inputs for the method

The geophysically determined effective hydraulic conductivity of bottom sediments is estimated for use with Darcy's Law to determine the quantity of ground water discharging to a surface-water body. Additional input data include a representative vertical hydraulic gradient for the entire surface-water body and the area of the bottom of the surface-water body.

When bottom sediment temperatures and conductivities are used to predict ground-water discharge areas to surface water, one known source of ground water seepage aids in calibrating the equipment. This technique is limited to general information about potential ground-water seepage zones.

iii. Outputs from the method

a. Ground-water quantity discharge to surface water

Geophysical methods estimate essentially any amount of ground-water discharge to surface water. The discharge rate relates to the effective hydraulic conductivity of the sediments and the hydraulic gradient across the sediments. Ground-water discharge rates to surface water of up to approximately 2.5 m^3/s were recorded using geophysical methods (Cherkauer and Taylor, 1984).

b. Ground-water quality discharge to surface water

Geophysical techniques do not determine ground-water quality, but conductivity measurements offer general water quality information. Conductivity values relate to the amount of dissolved chemical constituent in the water. A high conductivity value indicates the presence of dissolved chemical constituents in water, while a low conductivity value indicates the relative absence of chemical constituents in the water.

iv. Setting in which the method has been applied

Table C-1 summarizes some of the locations where this method has been used.

v. Contaminants that have been measured using the method

Geophysical methods do not directly measure the concentrations of chemical constituents in surface or ground water. Water quality can be qualitatively related to conductivity measurements. Generally the greater the chemical constituent concentration in water, the greater the conductivity measurement.

vi. General evaluation of the method

Geophysical methods characterize the spatial distribution of sediments on the bottom of surface-water bodies over large areas in a cost effective manner, as compared to other direct measurement techniques. As a result, there is an inherent trade off in the accuracy of the derived data. Geophysical techniques are of greatest value when applied to generally characterize the ground-water discharge characteristics of a large area of bottom sediment. Ground-truthing using conventional hydrogeologic measurements, such as mini-piezometers, piston corers, and seepage meters (described in Section A of this chapter), provides more precise estimates of discharge rates for specific areas. By identifying on-shore recharge zones for these areas and regulating land use in the recharge zones, surface-water quality can be better protected.

vii. References to annotated bibliography

References to the accompanying annotated bibliography are provided in Table C-2.

Table C-1. Summary of Settings in Which the Method Has Been Applied and the Contaminants Measured.

Location	Aquifer	Contaminant	Author
Ontario	-	-	D. R. Lee, S. J. Welch
Green Bay, Wi	-	-	K. R. Bradbury, R. W. Taylor
Detroit Metropolitan Area, MI	-	-	D. S. Cherkauer, R. W. Taylor
Mequon, WI	-	-	D. S. Cherkauer, B. Zvibleman
Great Lakes	Shallow glacial, glacial bedrock interface, bedrock units	organic solvents	EPA Non Point Source Group
Hardwick and New Braintree, MA Dover, NJ	-	-	W. W. Lapham
Chalk River Nuclear - Laboratories, Ontario	-	-	D. R. Lee
Southeastern U.S.S.R.	-	-	I. S. Zekster G. M. Bergelson

Table C 2. References to Annotated Bibliography

Author	Citation	Reference to Annotated Bibliography
K. R. Bradbury R. W. Taylor	"Determination of Hydrogeologic Properties of Lakebeds Using Offshore Geophysical Surveys," Ground Water, 1984 Volume 22(6): 690-695.	pp.74-75
D. S. Cherkauer R. W. Taylor	"Geophysically Determined Ground Water Flow into the Channels Connecting Lakes Huron and Erie," Proceedings of the Second National Outdoor Action Conference on Aquifer Restoration, Ground Water Monitoring and Geophysical Methods, Volume 2, Presented by the Association of Ground Water Scientists and Engineers and EPA/EMSL - Las Vegas, pp. 779-799.	pp.76-78
D. S. Cherkauer B. Zvibleman	"Hydraulic Connection between Lake Michigan and a Shallow Ground-Water Aquifer," Ground Water, 1981, Volume 19(4): 376-381.	pp.79-80
EPA Non Point Source Work Group	"Upper Great Lakes Connecting Channel Study, Waste Disposal Sites and Potential Ground Water Contamination St. Clair River," Non Point Source Work Group Report, April, 1988.	pp.81-85
W. W. Lapham	"Use of Temperature Profiles beneath Streams to Determine Rates of Vertical Ground-Water Flow and Vertical Hydraulic Conductivity," Draft Water Supply Paper No. 2337.	pp.86-87
D. R. Lee	"Method for Locating Sediment Anomalies in Lakebeds that that can be caused by Ground-Water Flow," Journal of Hydrology, 1985, 79: 187-193.	pp.88-89
R. W. Taylor D. S. Cherkauer	"The Application of Combined Seismic and Electrical Measurements to the Determination of the Hydraulic Conductivity of a Lake Bed," Ground-Water Monitoring Review, 1984, Volume 4(4): 78-85.	pp.92-93
I. S. Zekster G. M. Bergelson	"Effect of Ground Water on Lake Water Quality," Water Quality Bulletin, January, 1989, pp. 10-15.	pp.94-95

D. Studies involving hydrograph separation, regression analysis, or mass balance approaches to estimate the contribution of ground water to stream flow

The papers cited in this section are summarized in Section III of "An Annotated Bibliography of the Literature Addressing Nonpoint Source Contaminated Ground-Water Discharge to Surface Water," September, 1990, EPA 440/6-90-006.

i. General description of method

a. Description of method or procedures

The methods discussed in this section have been applied by investigators in areas throughout the U.S., in Ontario, Canada, and in the United Kingdom. Hydrograph separation has been used in conjunction with graphical techniques to estimate the distribution of ground-water flux to areas of the Great Lakes.[6] Other investigators have used analysis of conservative tracers along with hydrograph separation data to estimate ground-water flux and contaminant loading. The regression analysis and soil moisture balance methods rely on equations developed for specific regions. Arihood and Glatfelter (1986) have developed regression equations for northern Indiana, while Bevans's (1986) work was in eastern Kansas. Wilson and Ligon (1979) applied a water balance model to the Piedmont and Sandhill Regions of South Carolina.

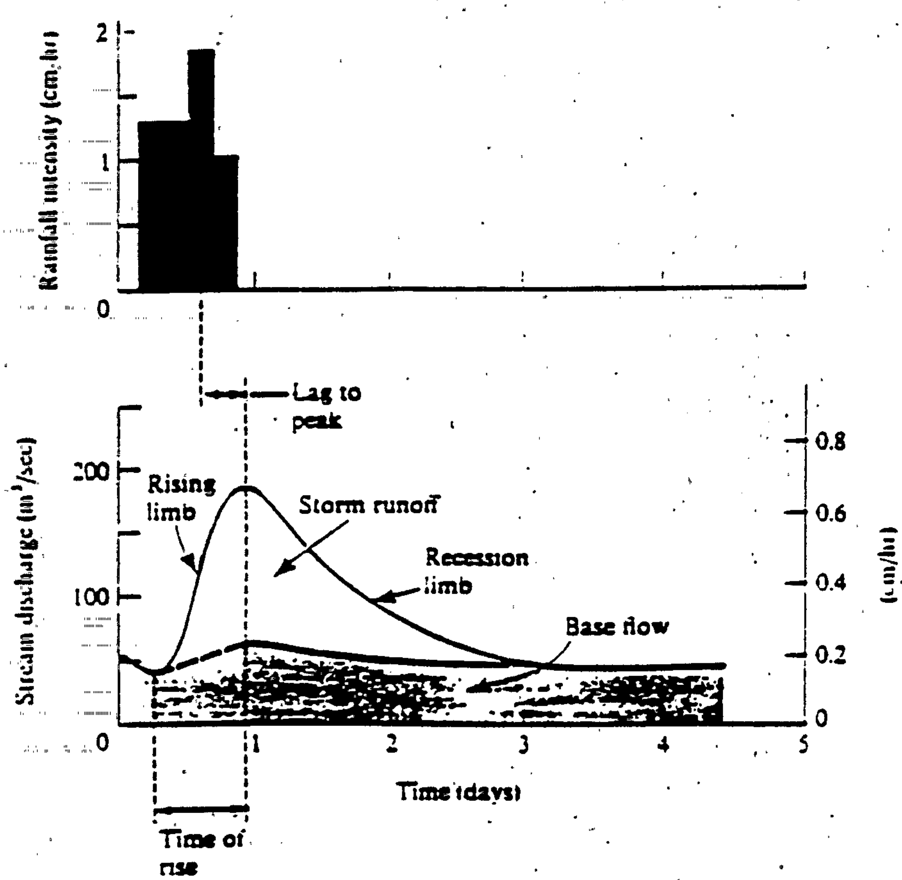
Hydrograph Separation

Precipitation entering a watershed travels to a stream by three main routes: surface runoff, interflow (or subsurface storm flow), and ground-water flow. The amount of water contributed to the stream by each of the three processes is reflected in the shape of the stream hydrograph, a graph of stream discharge at a particular point in the watershed versus time. The hydrograph for a single, short duration precipitation event, occurring over the entire watershed, follows a general pattern (see Figure 2-8). The hydrograph shows a period of increasing stage, or increasing discharge, known as the rising limb, that culminates in a peak or crest. Following the peak discharge, the hydrograph shows a period of decreasing discharge, referred to as the recession limb. Hydrograph separation techniques are applied to the recession limb to estimate contributions to stream flow from surface runoff, interflow, and ground-water flow.

When the hydrograph is plotted on semilogarithmic graph paper (discharge on the semilogarithmic y-axis), the recession limb has three identifiable line segments of different slopes, (see Figure 2-9). The slope of the line segment immediately after the peak discharge is the steepest and represents contribution to stream flow as a result of surface runoff and subsurface

[6] Pranckevicius, Pranas, personal communication, Nonpoint Source Contaminated Ground-water Discharge to Surface Water Workshop, Chicago, IL, November 30, 1989.

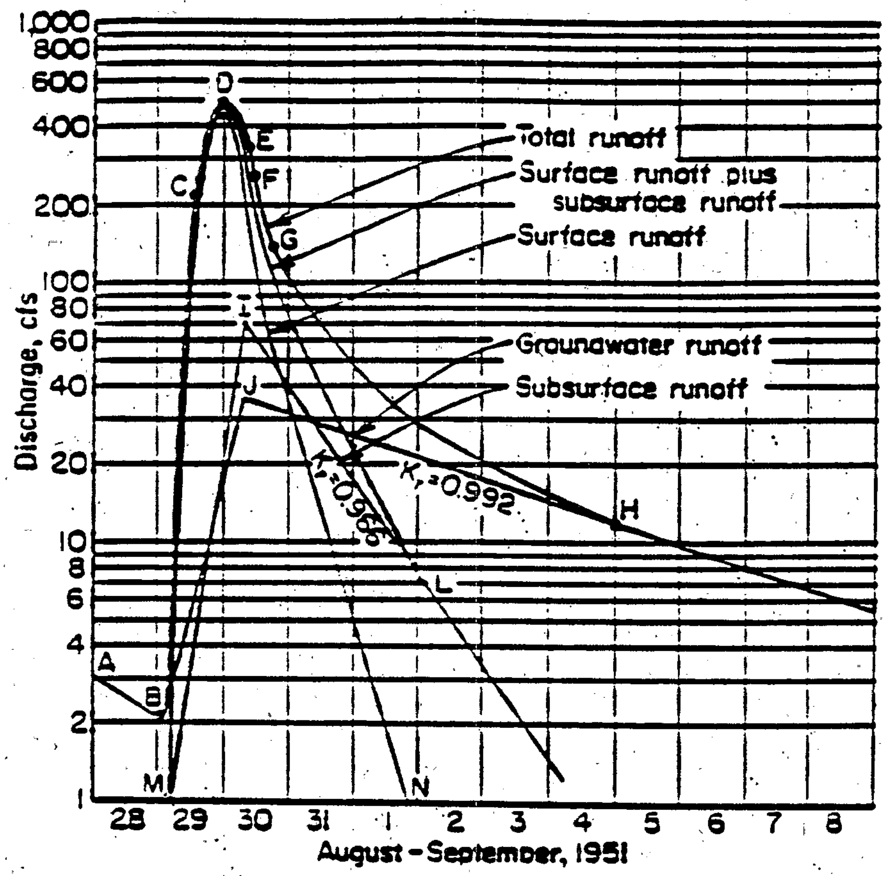
Figure 2-8



Hydrograph of streamflow in response to a rainstorm from a 100-square-kilometer basin.

Dunne, T. and L. Leopold. (1978) Water in Environmental Planning. San Francisco: W.H. Freeman and Co.

Figure 2-9



Semilogarithmic plotting of a hydrograph, showing separation of runoff components. (Panther Creek at El Paso, Illinois.)

Chow, V. (ed.) (1964) Handbook of Applied Hydrology. New York: McGraw-Hill.

runoff, which includes interflow and ground-water storage depletion. When surface runoff storage is depleted, the slope of the recession limb flattens. This portion of the recession limb represents contribution to stream flow as a result of interflow and ground-water storage depletion. The slope of the recession limb of the hydrograph changes again when interflow storage is depleted and contribution to stream flow is a result of ground-water storage depletion only. The ground-water contribution to stream flow is referred to as baseflow (see Figure 2-8). Surface runoff and interflow are often combined and referred to as direct runoff. The slope of the final segment of the recession limb is the ground-water recession constant, K_r, for the watershed. The line segment representing baseflow is extended back in time to a point under the hydrograph peak to determine maximum ground-water discharge to the stream as a result of the precipitation event. The ground-water recession constant for a watershed and the maximum ground-water discharge rate are used in an empirical formula to estimate ground-water discharge to surface water at any time after a precipitation event.

O'Brien (1980) has developed a "dynamic method" of hydrograph separation which matches the hydrograph of an index well with the stream hydrograph to determine the moment of maximum ground-water discharge for two small wetland-controlled basins in Massachusetts. The advantage of the method is that it is not rigidly tied to ground-water stage, and it accommodates variations in ground-water inflow and loss in channel storage in response to temperature, vegetation, stream stage, and change in seasons, causing shrinking and swelling of the peat and muck in the wetlands.

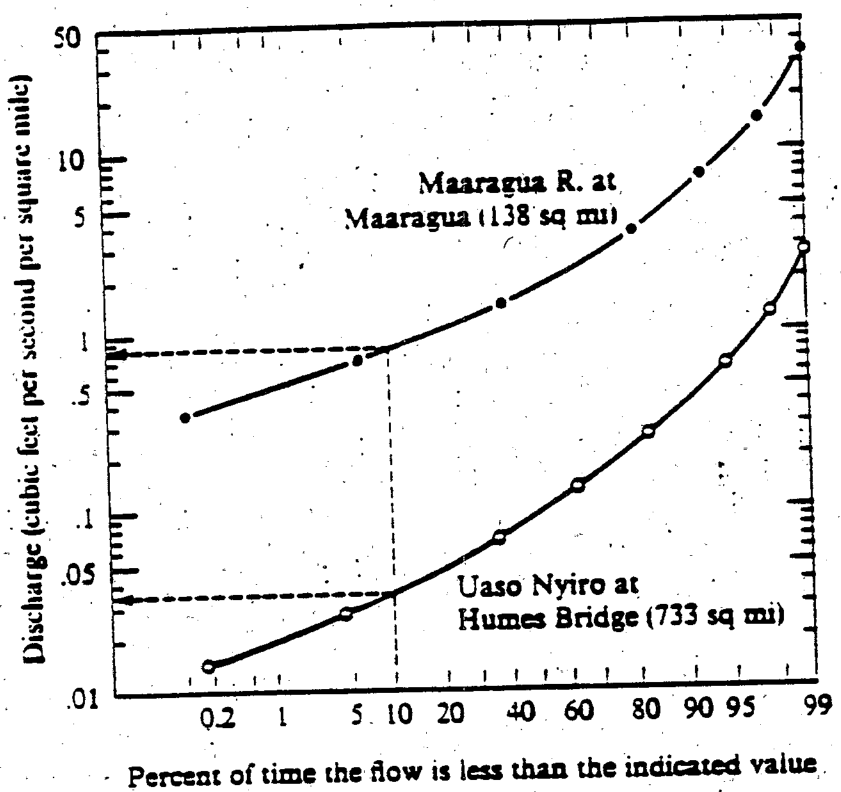
Regression Analysis

Equations developed with regression techniques that relate basin characteristics to baseflow characteristics in gaged streams can be used to estimate baseflow in ungaged streams. Examples of basin characteristics used in the regression analysis include drainage area of the watershed and flow duration ratio. The flow duration for a stream at a given point in the watershed is the proportion of time that discharge is less than a specific discharge value. Flow duration is commonly expressed as a curve representing the percent of time discharge is less than an indicated value versus discharge per area of the watershed, (see Figure 2-10). The flow duration ratio is the 20-percent flow duration divided by the 90-percent flow duration. The drainage areas of the watersheds and the flow duration ratios are transformed to logarithmic units and a regression equation is developed by backward elimination and maximum R^2 improvement procedures. For more information on regression analysis see Arihood and Glatfelter (1986) and Bevans (1986).

Soil Moisture Balance

The ground-water discharge component to a stream can also be estimated using a soil moisture mass balance approach, where inflow (precipitation) equals outflow (baseflow). Soil moisture water balance methods for a watershed assume that any excess soil moisture below the root zone ultimately will contribute to baseflow. The soil characteristics of the major soil types within the watershed are used to estimate the water-holding capacity of the different soil types. Excess soil moisture content below the root zone is

Figure 2-10



Flow duration curves for the River Maaragua in humid. central Kenya (mean annual rainfall 60 inches) and for the Uaso (River) Nyiro in semi-arid. north-central Kenya (mean annual rainfall 35 inches). The dashed lines indicate the flow values below which discharge declines for 10 percent of the time. The curves were constructed from records for the period 1956–1970.

Dunne, T. and L. Leopold. (1978) Water in Environmental Planning. San Francisco: W.H. Freeman and Co.

predicted using precipitation data, the evapotranspiration rate, and estimates of surface runoff for the watershed. The watershed is divided into two zones based on a predetermined depth to ground water from land surface. In the zone where depth to water from land surface is less than the predetermined depth, excess soil moisture below the root zone is assumed to discharge immediately to the stream. In the zone where depth to water is greater than the predetermined depth, excess soil moisture below the root zone is assumed to discharge to the stream in uniform increments, based on the time between precipitation events. For more information on soil moisture balance methods, see Wilson and Ligon (1979).

b. Assumptions involved in using these methods

Hydrograph Separation

Use of hydrograph separation techniques assumes that precipitation entering a watershed is evenly distributed and of the same intensity for the duration of the storm. Additionally, hydrograph separation techniques assume that the semilogarithmic plot of the recession limb of the stream hydrograph will have three identifiable segments of different slopes.

Regression Analysis

An important assumption when using regression equations to predict baseflow in unaged streams is that the basin characteristics used in the regression analysis are similar to basin characteristics of the unaged stream. Basin characteristics of concern are (a) the ground-water gradient, (b) the direction of the ground-water gradient, (c) the topography of the watershed, (d) the slope of the stream channel, and (e) the length of overland flow. Also, the geologic material underlying the basin will influence the shape of the stream hydrograph (Arihood and Glatfelter, 1986, and Bevans, 1986).

Soil Moisture Balance

The soil moisture water balance model assumes that any excess soil moisture below the root zone ultimately contributes to baseflow. Excess soil moisture below the root zone in the zone nearest the stream is assumed to enter the stream immediately following a precipitation event. Excess soil moisture below the root zone, in the zone farthest from the creek, is assumed to reach the stream in uniform increments, based on the time between precipitation events. Additionally, when the water table is below the root zone, it is assumed that no evapotranspiration occurs. Ground-water boundaries are assumed to correspond to surface-water boundaries, and there are no losses of ground water to other watersheds (Wilson and Ligon, 1979).

c. Limitations of the methods

Hydrograph Separation

In theory, it is straight forward to separate the recession limb of a stream hydrograph into three segments of different slopes from which the quantity of water contributed to the stream by surface runoff, interflow, and ground-water flow can be determined. In practice, separating the recession limb of a stream hydrograph into three segments of differing slope is an arbitrary process. Often no clear-cut change in slope exists. Given that precipitation events are not often of constant intensity or evenly distributed and considering the heterogeneity of a typical watershed, this is not surprising. Additionally, the effects of bank storage will make separating the recession limb of the storm hydrograph into three segments even more difficult. Because of the difficulties in determining the slope of the baseflow component of the recession limb of the stream hydrograph, predictions of ground-water contributions to stream flow may not be precise. Also, a continuous record of stream stage or discharge must be available to use this method.

In addition, several authors have questioned the ability of hydrograph separation techniques to determine the ground-water component of storm runoff accurately. Sklash and Farvolden (1979) report that ground water plays a much more active, responsive, and significant role in the generation of storm and snow-melt runoff in streams than hydrograph separations may predict. This increase in ground-water discharge may be caused by a rapid rise in hydraulic head along the perimeter of transient and perennial discharge areas.

Regression Analysis

Regression analysis equations developed using basin characteristics from one region should not be used to predict baseflow of streams located outside that region. Baseflow characteristics are dependent on the geology and geographic location of the watershed (Arihood and Glatfelter, 1986). Different geologic units will have different hydrologic properties, and different geographic locations will be exposed to different weather conditions and patterns. A given geologic formation may underlie one watershed and be absent in the neighboring watershed and weather conditions may be markedly different in adjoining watersheds as a result of orographic effects. Also, basin characteristics in a region may change gradually with distance from the study area. Because regression equations must be applied to basins with characteristics similar to those of the basins used in the regression analysis and because applicable basin characteristics are often difficult to define, the potential exists for regression analysis equations to be misapplied, resulting in inaccurate baseflow predictions.

Soil Moisture Balance

Like regression equations, mass balance soil moisture equations are based on basin characteristics common to a region. If the selection of constants in these questions is based on characteristics of basins that lie outside the region of concern, estimates of soil moisture balance and baseflow may be inaccurate.

d. Representative equations

If the break between direct runoff and baseflow on the semilogarithmic graph of the recession limb is difficult to define, an empirical equation can be used to estimate the number of days after the peak discharge at which direct runoff essentially ends. The empirical equation is as follows:

$$N = A^{0.2}$$

where:

N = Number of days after the peak when baseflow begins [dimensionless]
A = Watershed area in square miles [L^2].

The equation to determine the quantity of ground water discharging to surface water at any time after a precipitation event is:

$$Q(t) = Q_o e^{(\ln(K_r)t)}$$

where:

$Q(t)$ = Ground-water flow at time t after the peak discharge [L^3/T]
Q_o = Ground-water flow at time $t = 0$ [L^3/T]
K_r = Ground-water recession constant (derived from the hydrograph)
t = Time [T].

Equations for regression analysis and soil moisture balance methods are not presented here, as the equations are region specific and not universally applicable. Readers interested in these techniques are advised to read Wilson and Ligon (1979), Arihood and Glatfelter (1986), and Bevans (1986).

e. Description of equipment needs

Stream stage data are obtained using a continuous-chart recorder. Ideally the recorder should be located in a controlled section of the stream channel so a stream stage/discharge relationship can be developed.

For water balance methods, topographic maps are used to determine the area of a watershed. A rain gage can be used to measure precipitation; soil maps are used to determine soil types and to estimate soil properties.

f. Expertise required to apply this method

The greatest difficulty in using this method is in selecting the portion of the baseflow recession hydrograph to use in determining the ground-water recession constant (K_r) for the watershed. Estimation of K_r requires knowledge of basin characteristics and the temporal distribution of precipitation in the basin as well as considerable professional judgement. Once calculated, K_r can be used in the empirical ground-water discharge formula to determine the quantity of ground water discharging to surface water in a watershed as a result of a precipitation event.

Computer programs, as well as PC-based spreadsheets, are available that determine equations relating dependent and independent variables through regression analysis. The difficulty in using computer programs to determine the relationship of stream stage to basin characteristics is in determining which basin characteristics are appropriate input parameters. Additionally, it may be difficult to determine the area over which a regression equation is applicable. As with hydrograph separation, this method is best utilized by a hydrologist who is familiar with the subject basin.

A significant level of effort may be required to use soil moisture balance models to predict baseflow in a watershed. The upper and lower watershed zones must be delineated based on estimates of depth to ground water below land surface. Additionally, surface runoff and evapotranspiration rates must be estimated for the watershed. Because of a lack of representative precipitation measurements, the precipitation entering a watershed often must be estimated from precipitation measurements taken some distance away. Again, familiarity with the hydrologic and geologic characteristics and the temporal and spatial distribution of precipitation in the subject basin as well as surrounding basins is highly recommended for the successful use of these models.

ii. Data inputs for the method

Hydrograph separation techniques require a continuous record of stream stage or discharge to determine ground-water contribution to stream flow. Continuous stream stage/discharge data are available for many watersheds from the United States Geological Survey.

The drainage area of the watershed must be known to use regression equations to determine baseflow in an ungaged stream (Arihood and Glatfelter, 1986).

Soil moisture balance models require the following data: precipitation records, water-holding capacity of major soil classes in the watershed, area of the watershed, and drainable porosity measurements (Wilson and Ligon, 1979).

iii. Outputs for the method

a. Ground-water quantity discharge to surface water

Essentially any quantity of ground-water discharge to surface water can be predicted using hydrograph separation techniques. If the time between precipitation events is sufficiently long, the predicted ground-water discharge rate to surface water will decrease over time. The maximum ground-water discharge rate to surface water will be a function of the length and intensity of the precipitation event and the amount of ground water currently stored in the watershed.

As with hydrograph separation, any quantity of baseflow can be predicted using regression equations and soil moisture water balance models.

b. Ground-water quality discharge to surface water

Hydrograph separation, regression equations, and soil moisture water balance methods do not predict the quality of ground water discharging to surface water. Baseflow determined using these methods can be used in conjunction with ground-water quality measurements obtained in wells located adjacent to the surface-water body to estimate the ground-water loading rate to surface water.

iv. Settings in which the method has been applied

Some of the settings in which this method has been used are presented in Table D-1.

v. General evaluation of the method

Hydrograph separation techniques are an established method for estimating the ground-water discharge rate to surface water in a watershed. The method is well understood, simple to apply, and continuous stream stage/discharge data are readily available for many watersheds. The key to approximating actual ground-water discharge rates to surface water using this method involves correctly determining the slope of the baseflow portion of the recession limb of the hydrograph. Different slopes will produce markedly different predictions of ground-water contributions to stream flow.

Baseflow in ungaged streams can be estimated using regression equations. The equations are developed using regression analysis, relating drainage basin characteristics to baseflow, at gaging stations located in the same region. Because baseflow characteristics are dependent on the geology and geographic location of a drainage basin, a regression equation developed using drainage basin characteristics from one region should not be used to predict baseflow of streams located outside that region. Therefore, accurate estimation of baseflow in ungaged streams is dependent on the ability of the individual applying the method to identify regions having similar basin characteristics. Therein lies the problem-basin characteristics that influence baseflow are a result of a combination of components, some obvious, such as geographic location, and some not so obvious, such as geology. Additionally, the degree of interaction between the components affecting baseflow in a drainage basin is not well understood. Therefore, the potential exists that regression equations will be misapplied, resulting in inaccurate baseflow predictions.

Soil moisture balance techniques can be used to estimate baseflow in ungaged watersheds. Because so many of the input parameters for the model must be estimated, the error associated with baseflow predictions made using this method may be large.

vi. References to annotated bibliography

References to the accompanying bibliography are summarized in Table D-2.

Table D-1. Summary of Settings in Which the Method Has Been Applied and the Contaminants Measured.

Location	Aquifer	Contaminant	Author
Lincoln, Massachusetts	-	-	A. L. O'Brien
Northern and Central Indiana	-	-	L. D. Arihood, D. R. Glatfelter
Upper Coastal Plain of South Carolina; North Carolina; Georgia	Upper Coastal Plain Aquifer	-	W. R. Aucott, R. S. Meadows, G. G. Patterson
United Kingdom	-	-	M. D. Bako, Ayodele Owoade
Eastern Kansas	-	Sulfate; Coal mine drainage	Hugh E. Bevans
Elliot Lake, Ontario	-	Pyrite, Accessory metals, Radionuclides	D. W. Blowes, R. W. Gillham
Clayton County Iowa	Galena Aquifer	Herbicides, Pesticides, Nitrates and other agricultural inorganics	G. R. Hallberg, R. D. Libra, E. A. Bettis, III, B. E. Hoyer
Clayton County Iowa	Galena Aquifer	Nitrate nitrogen; Pesticides	R. D. Libra, G. R. Hallberg, B. E. Hoyer, L. G. Johnson
Illinois	-	-	Michael O'Hearn, James P. Gibb
Quebec and Ontario	-	-	M. G. Sklash, R. N. Farvolden
Cedar River Basin, Iowa-Minnesota	Cedar River Basin	Herbicides	P. J. Squillace, E. M. Thurman
Piedmont and Sandhill Regions, South Carolina	-	-	T. V. Wilson, J. T. Ligon

Table D 2. References to Annotated Bibliography

Author	Citation	Reference to Annotated Bibliography
L. D. Arihood, D. R. Glatfelter	"Method for Estimating Low-Flow Characteristics of Ungaged Streams in Indiana," U.S. Geological Survey, Open-File Report 86-323, 1986.	pp.97-98
W. R. Aucott, R. S. Meadows, G. G. Patterson	"Regional Ground-Water Discharge to Large Stream in the Upper Coasted Plain of South Carolina and Georgia," USGS Water Resource Investigations Report 86-4332, 1987.	pp.99-101
M.D. Bako, Ayodele Owoade	"Field Application of a Numerical Method for the Deviation of Baseflow Recession Constant," Hydrological Process, 1988, 2: 331-336.	pp.102-103
H. E. Bevans	"Estimating Stream-Aquifer Interactions In Coal Areas of Eastern Kansas by using Streamflow Records," USGS Water Supply Paper 2290 (January, 1986).	pp.104-105
D. W. Blowes, R. W. Gillham	"The Generation and Quality of Streamflow on Inactive Uranium Tailings Near Elliot Lake, Ontario," Journal of Hydrology, 1988, 97: 1-22.	pp.106-107
G. R. Hallberg, R. D. Libra, E. A. Bettis, III., B. E. Hoyer	"Hydrogeologic and Water Quality Investigations in the Big Spring Basin, Clayton County, Iowa," Iowa Geological Survey, 1984, Open-File Report 84-4.	pp.108-110
R. D. Libra, G. R. Hallberg, B. E. Hoyer, L. G. Johnson	"Agricultural Impacts on Ground-Water Quality," Proceedings of the Agricultural Impacts on Ground Water, 1986, National Water Well Association, Omaha, Nebraska, pp. 253-273.	pp.111-112
A. L. O'Brien	"The Role of Ground Water in Stream Discharges from Two Small Wetland Controlled Basins in Eastern Massachusetts," Ground Water, 1980, Volume 18(4):	pp.113-114
M. O'Hearn, J. P. Gibb	State Water Survey Report Number 246, 1980 Illinois Institute of Natural Resources.	pp.115-116
W. A. Pettyjon, R. J. Henning	"Preliminary Estimate of Regional Effective Ground Water Recharge Rates, Related Streamflow and Water Quality in Ohio," Water Resources Center, Preliminary Estimate of Regional Effective Ground Water Recharge Rates in Ohio, Project Completion Report, 323 pp., 1979.	pp.117-119

Table D-2. References to Annotated Bibliography (Continued)

Author	Citation	Reference to Annotated Bibliography
P. J. Squillace, E. M. Thurman	"Surface-Water Quality of the Cedar River Basin, Iowa-Minnesota, With Emphasis on the Occurrence and Transport of Herbicides, May 1984 through November 1985," U.S.G.S. Toxic Substances Hydrology Program, Abstracts of Technical Meeting, Phoenix, Arizona, September 26-30, 1988.	pp. 120-122
T. V. Wilson, J. T. Ligon	"Prediction of Baseflow for Piedmont Watersheds," Office of Water Research and Technology, Water Resources Research Institute, Report Number 80, 1979, 47 pp.	pp. 123-125

E. Numerical models of surface-water/ground-water interactions

The papers cited in this section are summarized in Sections IV, V, and VII of "An Annotated Bibliography of the Literature Addressing Nonpoint Source Contaminated Ground-Water Discharge to Surface Water," September, 1990, EPA 440/6-90-006.

i. General description of methods

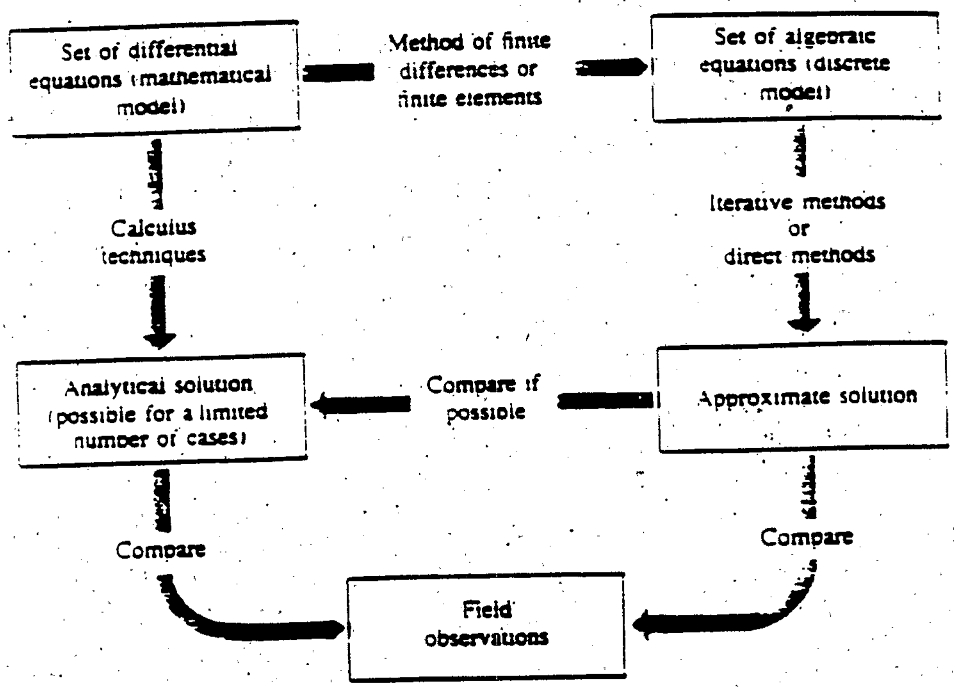
a. Description of method or procedure

Mathematical ground-water modelling simulates an aquifer or watershed system using a series of equations governing flow and/or mass balance properties. When developing a model, transport properties should be constructed using a framework of measured variables. Modelling represents a cumulative process of data gathering and model verification to ensure an accurate depiction of real world phenomena in the computer simulation. Models should not be used without field data and ground truthing, and the transient conditions of the study locations should be understood and incorporated into the analysis.

Mathematical ground-water models consist of sets of differential equations that describe or "govern" ground-water flow and/or contaminant transport. These equations can be solved to develop an analytical solution; however, field situations may be complex and difficult to solve exactly, and the assumptions that must be made to obtain the analytic solution are often unrealistic and are not representative of the flow or transport problem under consideration. In these situations, numerical methods can be used to solve the differential equations and obtain an approximate solution that can be used to simulate relatively complex ground-water flow and contaminant transport. This process is presented in Figure 2-11. Two popular numerical methods used to convert differential equations into algebraic equations are the finite difference method and the finite element method.

To utilize a numerical flow model, a flow system is defined and discretized into a finite number of rectangular blocks, in the case of finite-difference models, or triangles or quadrilaterals, in the case of finite-element models. Figures 2-12 and 2-13 show finite difference and finite element representations of an aquifer bounded on three sides by an impermeable boundary (i.e., no flow into or out of the aquifer) and on the fourth side by a river into which discharge from the aquifer occurs. Each cell in the flow region is assigned its own hydrologic properties based on measurements or observations from the flow region being modeled. Boundary conditions are then incorporated into the numerical model. Typical boundary conditions are ground-water divides (no flow), surface-water bodies (fixed head), and specified flow. The numerical model is run on a computer, and typically, the calculated head-field distribution at nodal points (the intersections of the lines delineating the region or centers of the blocks) is compared to the actual head-field distribution (obtained through measurement of water levels in wells) in the flow region and, if available, the results of an analytical solution (see Figure 2-11). If the predicted and actual head fields are not in close agreement, the model is adjusted by manipulating boundary conditions

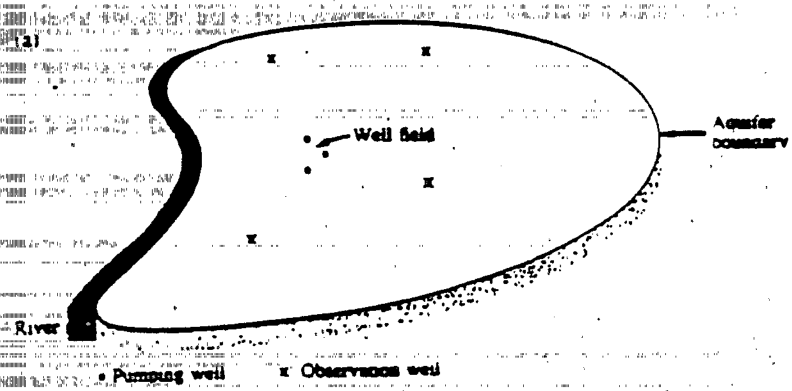
Figure 2-11

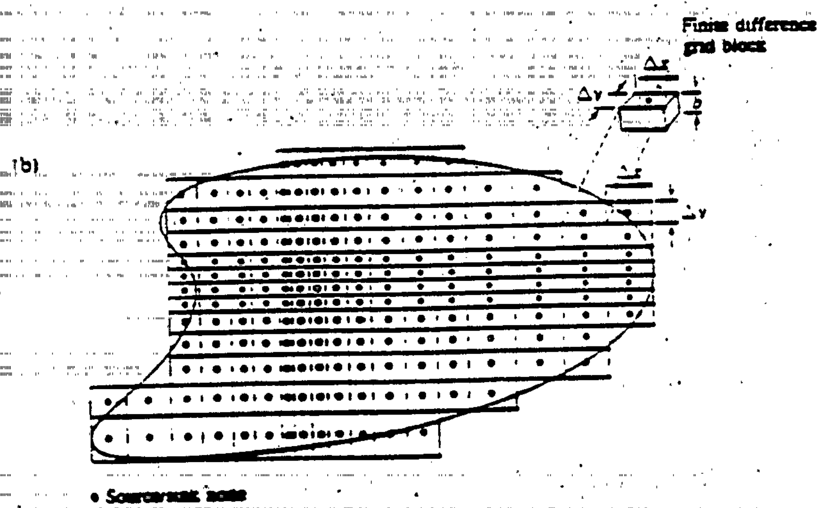


Relationships between mathematical model, discrete algebraic model, analytical solution, approximate solution, and field observations.

Wang, H. and M. Anderson. (1982) Introduction to Ground water Modeling: Finite Difference and Finite Element Methods. San Francisco: W.H. Freeman and Co., 237 p.

Figure 2-12

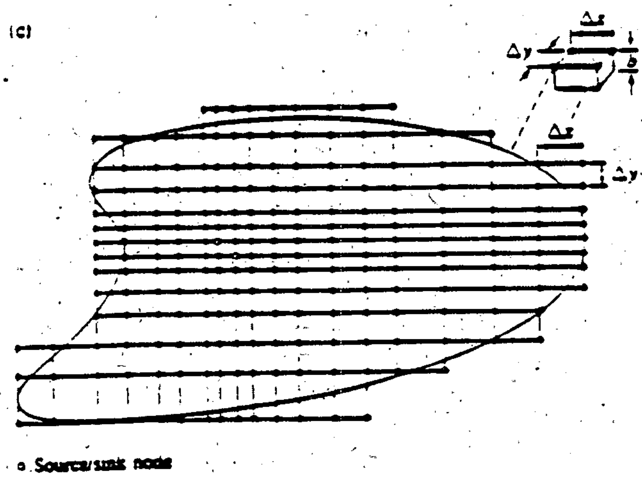


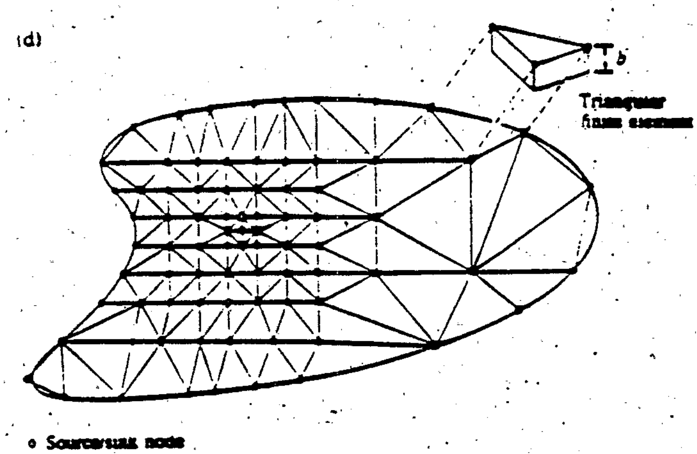


Finite difference and finite element representations of an aquifer region.
(a) Map view of aquifer showing well field, observation wells, and boundaries.
(b) Finite difference grid with block-centered nodes, where Δx is the spacing in the x direction, Δy is the spacing in the y direction, and b is the aquifer thickness.

Wang, H. and M. Anderson. (1982) Introduction to Ground water Modeling: Finite Difference and Finite Element Methods. San Francisco: W.H. Freeman and Co., 237 p.

Figure 2-13





(c) Finite difference grid with mesh-centered nodes.
(d) Finite element mesh with triangular elements where *b* is the aquifer thickness.
(Adapted from Mercer and Faust, 1980a.)

Wang, H. and M. Anderson. (1982) Introduction to Ground water Modeling: Finite Difference and Finite Element Methods. San Francisco: W.H. Freeman and Co., 237 p.

and/or the hydrologic properties of the individual cells. When close agreement is reached between predicted and actual head-field distribution, the model is considered calibrated.

After the numerical flow model is calibrated, the velocity field for the flow region calculated by the flow model is used as input to a solute transport model, which simulates the movement of dissolved chemical constituents through the aquifer by advection and dispersion. In many models, the advection-dispersion equation is solved numerically. In the alternative "random walk" method, a random process, based on a normal probability density function, is used to simulate dispersion (Prickett et al, 1981). Dissolved chemical constituents are represented by a finite number of discrete particles. Each particle is assigned a mass which represents a fraction of the total mass of the chemical constituent involved. Chemical constituents are introduced to the subsurface flow system in the recharge input for the model. The mass of chemical constituent input to the system is a function of factors such as land use and crop type. During each time step, the particles are moved advectively with the ground-water velocity field. Between time steps, the particles' positions are adjusted by moving individual particles a random distance in any direction. The magnitude and orientation of the random displacement is predicted by the normal probability density function.

Individual cell concentration for each time step is calculated by determining the number of particles located in the cell. Because each particle has a constant mass and the cell has a known volume, chemical constituent concentration in the cell can be determined. Predicted concentrations are compared to known concentrations in the flow region. To calibrate the model, input concentration, volume, and location are adjusted until predicted concentrations agree with observed concentrations. When both the predicted head and concentration fields reasonably agree with known values in the flow region, the model is considered calibrated and can be used for predictive purposes.

b. Assumptions involved in using these models

The primary assumption when using numerical models is that the predicted flow field is a close approximation of the actual flow field. Loading rates to surface water as a result of ground-water discharge are largely controlled by the ground-water velocity field. Even after calibration, using the assumptions that predicted and actual hydraulic head distributions are similar, the predicted velocity field may not be correct. This phenomenon results because a given hydraulic head distribution is not unique for a given combination of aquifer properties. When characterizing a watershed, a priority should be placed on delineating the spatial distribution of aquifer properties rather than on obtaining additional hydraulic head measurements. If input data are scarce, either in time or space, the predictive capabilities of the numerical model may be compromised.

An assumption of most finite-difference models is that there is only one set of principal anisotropic directions in the flow region and the principal directions are aligned with the grid system. Care should be taken when designing the model grid to align grid axes with principal directions of isotropy.

Chemical inputs to the flow region as a result of agricultural or other practices are assumed from generalized land use (Gburek et al., 1989). The input is assumed constant with time for a given land use for all contaminant sources within a study region. Instantaneous mixing is assumed to occur in each cell for each time step. Additionally, in the random walk process of transport simulation, it is assumed that dispersion in porous media can be considered a random process having a normal distribution.

c. Limitations of the models

The major limitation of numerical models is the large amount of data required to accurately calibrate them. To accurately calibrate a numerical model, information on the spatial and temporal distribution of land use, recharge, chemical input, hydraulic head, ground-water quality, and surface-water quality is needed. Also of prime importance is the spatial distribution of aquifer characteristics. Often, ground-water models do not take into account the variable effects of near shore phenomena. Generally, models will not simulate ground-water quality changes associated with seasons, or reflect the hydraulic conductivity changes associated with seepage face growth and capillary response to precipitation.

Prior to using a model, the scale and the geographic conditions of the study area must be incorporated into the model. For instance, fracture flow, macropore flow, karst terrain, and anthropomorphic effects on the study area's ground water may require that adjustments be made to the model's structure. Few watersheds have been monitored sufficiently to provide the data needed to calibrate a numerical model. Knowledge about one watershed in a region will assist in characterizing another watershed in the same region, but additional data probably will be required before the model is considered calibrated and can be used for predictive purposes. Without being able to reproduce flow-field conditions or chemical concentrations, little confidence can be placed in a model's predictive capabilities.

An additional limitation of numerical models results from the uncertainty associated with them. This uncertainty is a result of numerical models being based on mathematical expressions that are a simplification of the real world and the measurement error associated with input data. This uncertainty could result in predicted values that deviate significantly from the actual flow in the region being modeled. However, proper field data-collection techniques and the use of well-tested models by experienced personnel combine to produce reliable predictions in most cases.

d. Representative equations

The partial-differential equation that describes time-variable flow in a heterogeneous, anisotropic two-dimensional aquifer (one in which hydraulic conductivity varies both in direction and space throughout the aquifer) is:

$$\delta/\delta x(K_x \cdot \delta h/\delta x) + \delta/\delta y(K_y \cdot \delta h/\delta y) = S_s \cdot \delta h/\delta t$$

where:

K_x = Hydraulic conductivity in x direction [L/T]

K_y	=	Hydraulic conductivity in y direction [L/T]
S_s	=	Specific storage [L^{-1}]
h	=	Hydraulic head [L]
x	=	x direction [L]
y	=	y direction [L]
t	=	Time [T].

A finite-difference numerical model approximates the above differential equation using a series of finite-difference equations. The two-dimensional finite-difference equation for a homogeneous, isotropic medium, where the grid spacings in the x- and y- directions are the same and hydraulic conductivity is constant and isotropic throughout the aquifer ($K_x = K_y$), is:

$$h^k_{i,j} + S\Delta x^2/\Delta t \cdot (h^k_{i,j} - h^{k-1}_{i,j}) = \frac{1}{4}(h^k_{i-1,j} + h^k_{i+1,j} + h^k_{i,j-1} + h^k_{i,j+1})$$

where:

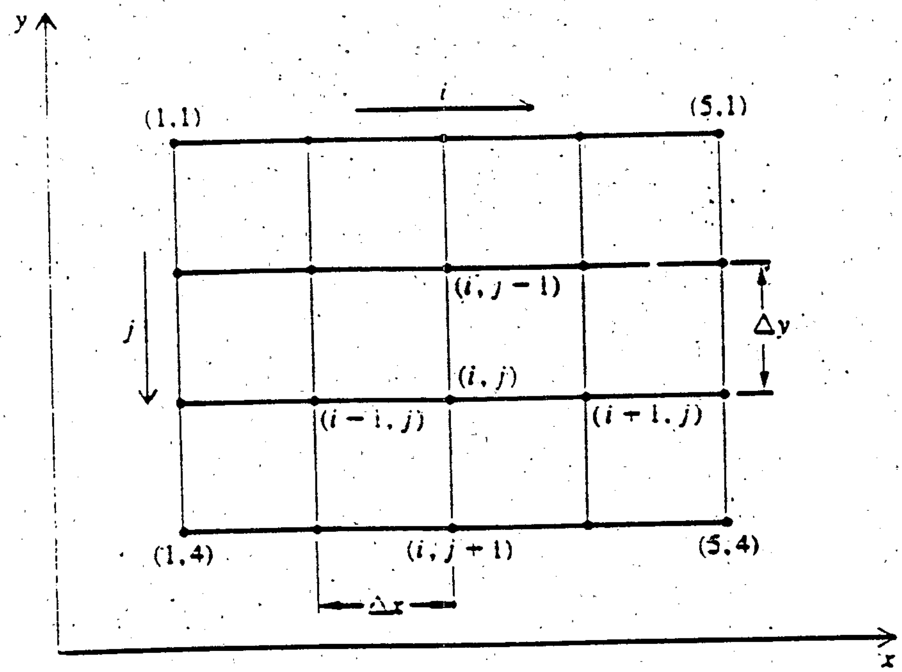
h	=	Hydraulic head [L]
T	=	Transmissivity [L^2/T]
S	=	Storativity [dimensionless]
t	=	Time increment [T]
x	=	Width of the grid spacing, where $\Delta x = \Delta y$ [L]
i	=	Column number [dimensionless]
j	=	Row number [dimensionless]
k	=	Time step or iteration index [dimensionless].

Column and row numbers in this equation correspond to those in the finite difference grid presented in Figure 2-14. Nodes (intersections of grids) are spaced horizontally by Δx and vertically by Δy. For the first iteration, or solution of the equation, the modeler estimates the value for the hydraulic head at each node. The head values of the first iteration (k=1) are used to calculate the head values for the second iteration (k=2). The equation is solved several times in this manner until the difference between the head values of the final iteration and the previous iteration is less than a value specified by the modeler, called a convergence criterion.

The partial-differential equation that describes solute transport in a two-dimensional, homogeneous aquifer through dispersion and advection is:

$$\underbrace{D_L \cdot \delta^2 c/\delta x^2 + D_T \cdot \delta^2 c/\delta y^2}_{\text{dispersion}} - \underbrace{V_x \cdot \delta c/\delta x}_{\text{advection}} = \delta c/\delta t$$

Figure 2-14



Finite difference grid showing index numbering convention.

Wang, H. and M. Anderson. (1982) Introduction to Ground water Modeling: Finite Difference and Finite Element Methods. San Francisco: W.H. Freeman and Co., 237 p.

where:

D_L = Longitudinal dispersion coefficient $[L^2/T]$
D_T = Transverse dispersion coefficient $[L^2/T]$
C = Concentration $[M/L^3]$
V_x = Average pore velocity in the x direction [L/T]
x = x direction (direction of flow) [L]
y = y direction [L]
t = Time [T].

In the random walk approach, solute transport in a porous medium is represented by a series of equations. Dissolved chemical constituents are represented by a finite number of discrete particles each having a mass representing a fraction of the total mass of the chemical constituent involved (Pricket et al., 1981). The total distance a particle travels between time steps is:

$$d_T = d + d^*$$

where:

d_T = Total distance traveled per time step [L]
d = Distance traveled as a result of advection per time step [L]
d^* = Distance traveled as a result of dispersion per time step [L].

The equation representing the distance a particle is transported by advection:

$$d = vt$$

where:

d = Distance particle travels for each time step [L]
v = Ground-water flow velocity [L/T]
t = Time step duration [T].

After the particles have been moved advectively, the position of each particle is adjusted a random amount in any direction to account for dispersion. The one-dimensional equation representing the influence of dispersion on a particle's position is:

$$d^* = N \cdot (2 \cdot d_L \cdot V \cdot t)^{1/2}$$

where:

d^* = Distance traveled as a result of dispersion per time step [L]
d_L = Longitudinal dispersivity [L]
V = Ground-water velocity [L/T]
t = Time step duration [T]
N = A number between -6 and 6, drawn from a normal distribution of numbers having a standard deviation of 1 and a mean of zero [dimensionless].

The equation determining individual cell concentration for each time step in a two-dimensional model is:

$$C_{i,j} = n \cdot m_p \cdot \Delta x \cdot \Delta y$$

where:

C	=	Concentration per unit width of chemical constituent $[M/L^3]$
n	=	Number of particles in a cell [dimensionless]
m_p	=	Mass per particle [M]
x	=	Cell length in x direction [L]
y	=	Cell length in y direction [L]
i	=	Column number [dimensionless]
j	=	Row number [dimensionless].

e. Description of computer hardware or software needs

Essentially any of the commercially-available numerical models can be run on a PC or mainframe computer. The requirements for a PC computer are a large memory capacity (640K is usually sufficient) and a math coprocessor chip. Without a math coprocessor chip, numerical models with a large number of cells can take hours to converge on a solution or even longer for large time-variable problems. Small memory capacity will limit the number of nodes used in simulating a flow region. Generally, the greater the number of nodes used, particularly near boundaries between property types or in areas of steep hydraulic gradients, the greater the model's accuracy. Limiting the number of nodes might compromise the accuracy of the simulation. Suggested computer hardware includes:

- a PC computer with math coprocessor chip and graphics card,
- a high resolution monitor (for plotting results on the screen),
- a printer, and
- a plotter.

Numerous software packages, both in the private and public domain, are available for simulating ground-water flow and contaminant transport. Some of the more popular models are codes by Konikow and Bredehoeft[7], Prickett and Lonnquist[8], Trescott et al.[9], and the International Ground-Water Modeling Center at Butler University in Indiana.

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[7] Konikow, L. and J. Bredehoeft. (1978) Computer model of two-dimensional solute transport and dispersion in ground water. U.S. Geological Survey, Techniques of Water Resources Investigations Book 7, Chapter C2, 90 p.

[8] Prickett, T. and C. Lonnquist. (1971) Selected digital computer techniques for groundwater resource evaluation. Illinois State Water Survey Bulletin 55, 62 p.

[9] Trescott, P., G. Pinder, and S. Larson. (1976) Finite-difference model for aquifer simulation in two dimensions with results of numerical experiments. U.S. Geological Survey Techniques of Water Resources Investigations, Book 7, Chapter C1, 116 p.

f. Expertise needed to run the models

To adequately simulate flow situations using numerical modeling techniques, knowledge of hydrology, numerical methods, computer language, and computers is needed. In addition to these skills, an intuitive sense, derived from experience, is needed to determine the proper grid spacing, boundary conditions, and geometry of aquifer properties. Also, once the numerical model has been calibrated, expertise is needed to determine if predicted values are as expected and are correct.

ii. Data inputs for the models

Basic input requirements for numerical models are the spatial distribution of hydraulic conductivity, anisotropy ratios, and boundary conditions. Examples of common boundary conditions are ground-water divides, surface-water bodies, and known flux boundaries. The spatial and temporal distribution of the hydraulic head field for the flow region is needed for calibration purposes. Also, for chemical transport modeling, the spatial and temporal distributions of chemical inputs and concentrations in ground water and surface water are required. For nonpoint studies, chemical loading may be a function of crop distribution and type; thus, agricultural records are required.

iii. Outputs from the models

a. Ground-water quantity discharge to surface water

Numerical models can simulate essentially any quantity of ground-water discharge to surface water. Discharges less than 1 cubic foot per second (cfs) have been simulated (Eddy and Doesburg, 1985).

b. Ground-water quality discharge to surface water

As with quantity of ground-water discharge to surface water, essentially any concentration of chemicals in both ground water and surface water can be simulated.

iv. Settings in which the methods have been applied and contaminant discharge that has been modeled

With enough information, a numerical model can be calibrated to simulate any watershed in any geologic region of the country. Given enough input data, the transport of any chemical in ground water can be simulated. Contaminant transport models are capable of simulating the effects of decay, retardation, and sorption on chemical constituent distribution and concentration. Table E-1 summarizes the settings in which numerical models have been used and the contaminants that have been transported.

v. General evaluation of the method

Numerical models can be used to simulate various nonpoint source loading scenarios for complex aquifer conditions. Before a numerical model can be used for predictive purposes, however, a large amount of input data is often required to properly calibrate the model. The amount of data required will depend on the type of model used, the objectives of the study, and the level of accuracy required. Acquisition of the needed data can require considerable time, expertise, and expense. Because of these constraints, the numerical model may have limited usefulness in cases where data are scarce and funding is limited.

The numerical model's strength is its usefulness as a screening tool. Numerical models calibrated to simulate watersheds in different regions of the country could be used to assess the general effect of various regulatory scenarios on ground-water quality in those watersheds located in those regions.

vi. References to annotated bibliography

References to the accompanying annotated bibliography are summarized in Table E-2.

Table E.1. Summary of Settings In Which the Method Has Been Applied and the Contaminants Measured.

Location	Aquifer	Contaminant	Author
Wisconsin	-	-	D. S. Cherkauer, B. R. Hensel
Kent, Washington	-	Organics, chloroform and tri-chloroethylene; zinc	C. H. Eddy, J. M. Doesburg
Pennsylvania	Trimmers Rock and Catskill Formations	Nitrates	W. J. Gburek, R. R. Schnabel S. T. Potter
Juneau, Alaska	Mendenhall Basin	-	D. I. Siegel
Northwestern Indiana	Calumet Aquifer	-	L. R. Watson, J. M. Fenelson
Central Sand Plain, Wisconsin	-	Agricultural chemicals	C. Zheng, K. R. Bradbury M. P. Anderson

Table E-2. References to Annotated Bibliography

Author	Citation	Reference to Annotated Bibliography
V. K. Barwell, D. R. Lee	"Determination of Horizontal-to-Vertical Hydraulic Conductivity Ratios from Seepage Measurements on Lake Beds," Water Resources Research, 1981, 17: 565-570.	pp.173-174
D. S. Cherkauer, B. R. Hensel	"Ground-Water Flow into Lake Michigan from Wisconsin," Journal of Hydrology, 1986, Volume 84: 261-271.	pp.175-177
V. T. Dubinchuk	"Radon and Radium Discharge to Surface Streams," Water Resources, 1981, 8(1): 102-116, translated from Vodnye Resursy.	pp.178-179
C. M. Eddy, J. M. Doesburg	"Remedial Action Modeling Assessment Western Processing Site, Kent, Washington," Report prepared for U.S. Environmental Protection Agency, Region X, Seattle, Washington 98101, July 1985.	pp.180-181
W. J. Gburek, R. R. Schnabel S. T. Potter	"Modeling the Effect of the Shallow Weathered Fracture Layer on Nitrate Transport," Unpublished Draft Report.	pp.182-184
T. A. Prickett, T. G. Naymik C. G. Lonnquist	"A 'Random Walk' Solute Transport Model for Selected Ground Quality Evaluations," Illinois State Water Survey, Champaign, IL, 1981, ISWS/BUL-65-81.	pp.185-187
D. I. Siegel	"The Recharge-Discharge Function of Wetlands near Juneau, Alaska: Part I. Hydrogeological Investigations," Ground Water, 1988, 26(4): 427-434.	pp.188-189
L. R. Watson, J. M. Fenelson	"Geohydrology of a Thin Water-Table Aquifer Adjacent to Lake Michigan, Northwestern Indiana," (in press).	pp.190-191
C. Zheng, K. R. Bradbury M. P. Anderson	"Role of Interceptor Ditches in Limiting the Spread of Contaminants in Ground Water," Ground Water, 1988, Volume 26(6): 734-742.	pp.192-193
C. Zheng, H. F. Wang M. P. Anderson, K. R. Bradbury	"Analysis of Interceptor Ditches for Control of Ground Water Pollution," Journal of Hydrology, 1988, 98: 67-81.	pp.194-196

F. Studies involving the application of functions estimating nonpoint source loading to surface water for various land use types

The papers cited in this section are summarized in Section VI of "An Annotated Bibliography of the Literature Addressing Nonpoint Source Contaminated Ground-Water Discharge to Surface Water," September, 1990, EPA 440/6-90-006.

i. General description of method

a. Description of method or procedure

Nonpoint source loading models combine surface runoff, sediment yield, and ground-water discharge with empirical loading rates to obtain estimates of nitrogen and phosphorous chemical concentrations in surface water. Runoff in the watershed is calculated from daily weather data using the U.S. Soil Conservation Service's Curve Number Equation. Sediment yield is calculated using the Universal Soil Loss Equation in conjunction with the Richardson daily rainfall erosivity index. Ground-water discharge is calculated from daily water balances for the unsaturated and saturated zones in a watershed or by using hydrograph separation techniques. Loading rates for runoff, sediment yield, and ground-water discharge are assigned based on land use. Land use is divided into residential, commercial, industrial, and agricultural categories. Agricultural land is further subdivided based on land use, crop type, and land management practices. The land use loading rates for runoff, sediment yield, and ground-water discharge are summed and multiplied by the total area of similar land use in the watershed to obtain the empirical loading rate as a result of that land use category. The total nonpoint source loading rate for the drainage basin rate is obtained by summing the calculated loading rates for each land use category (Haith and Shoemaker, 1987, and Ritter, 1986).

The estimation of ground-water discharge from functions is best used in conjunction with verification methods such as mass and water balance estimates, ground-water monitoring, piezometer sampling, and seepage meter monitoring. Functions have been used to estimate discharges in inland watersheds in Pennsylvania (Gburek, et. al.) and Wisconsin (Uttormark, et. al.), and in Inland Bays in Delaware (Ritter) and the Chesapeake Bay (Schnabel and Gburek). Most of the studies utilizing this method have examined nutrient loadings into surface waters.

b. Assumptions involved in using these models

The major assumption of nonpoint loading models is that the empirical loading rates assigned a land use category for runoff, sediment load, and ground-water discharge are representative of actual loading conditions. The assigned runoff and ground-water discharge loading rates for a land use category are assumed independent of topography, soil type, or tillage methods (Schnabel and Gburek, 1983).

Another assumption is that the transport process is not scale dependent; that is, the empirical loading rates for land use categories are equally applicable for both large and small drainage basins (Schnabel and Gburek, 1983).

c. Limitations of the methods

The ultimate purpose of loading models is to predict the impact various land management schemes will have on surface-water quality in a watershed through use of empirical loading factors. Ideally, loading factors for various land types should only be representative of on-field processes, such as tillage and fertilization practices. In reality, loading factors are a combination of on-field and off-field processes. Off-field processes such as non-crop plant nutrient uptake, deposition of sediment in buffer strips near streams in the watershed, and mixing of interflow and baseflow components of different chemical composition, are included in loading factors. Additionally, loading factors make no distinction between flowpaths, effectively masking the processes which contribute to sediment and chemical loss from a watershed. As a result, loading factors mask the interaction of on- and off-field processes and cannot be adjusted to account for individual changes in either on- and off-field management practices. Thus, as a predictive tool, loading factors may have limited use.

There was significant uncertainty associated with the input parameters for the model applications referenced in this review. Precipitation and temperature data were collected at one or two locations in a watershed and were assumed to be representative for the entire watershed (Haith and Shoemaker, 1987). The shallow ground-water storage value and recession index were assumed to represent the entire watershed even though several aquifers may discharge ground water to surface water. Because of these uncertainties associated with input values to the model, predicted loading rates to surface water may not be representative.

d. Representative equations

Ground-water discharge to a surface water is determined using a lumped parameter water balance model based on daily water balances from the unsaturated and shallow saturated zones (Haith and Shoemaker, 1987). The equation describing ground-water discharge is as follows:

$$G_t = S_t \cdot r$$

where:

G_t = Ground-water discharge $[L^3/T]$
S_t = Shallow saturated zone moisture content $[L^3]$
r = Ground-water recession constant $[1/T]$.

The loading rate to surface water as a result of ground-water discharge is:

$$LR = G_t \cdot C$$

where:

LR = Loading rate to surface water $[M/T]$
G_t = Ground-water discharge rate to surface water $[L^3/T]$
C = Concentration of chemical constituent in ground water $[M/L^3]$.

e. Description of equipment needs

Equipment needed for the method includes

- rain gages,
- continuous-chart recorder,
- soil maps,
- crop distribution maps, and
- land-use distribution maps.

A PC computer-based spreadsheet would be very useful to an application of the method.

f. Expertise needed to use the method

A significant amount of effort is required to use this method. The greatest level of effort is required to classify land into the various categories, which involves correlating soil type distribution with land use and crop distribution in the watershed. Knowledge of relationships between soil type, land use, and crop distribution within the watershed is useful. After the watershed has been sectioned into representative land-use categories and the recession constant has been determined, the method becomes a bookkeeping exercise. A computer spreadsheet can be utilized to multiply and add the calculated values to estimate the loading rate to surface water.

ii. Data inputs for the model

The model requires data describing land use and soil type distribution and daily precipitation. The ground-water recession constant can be estimated using standard hydrograph separation techniques and stream gage data.

iii. Outputs from the model

The method estimates the loading rate to surface water as a result of ground-water discharge, sediment load, and surface runoff for various land uses.

a. Ground-water quantity discharge to surface water

The amount of ground-water discharge to surface water predicted by this method is a function of the recession constant and the storage capacity of the shallow saturated zone. If both the recession index and the storage value are small, the predicted discharge rate will be small; conversely, if both are large, the predicted discharge rate will be large.

b. Ground-water quality discharge to surface water

The chemical constituents commonly modeled using this method include nitrogen and phosphorous (Haith and Shoemaker, 1987). The concentrations predicted in surface water using these methods are a function of the loading rates assigned to the various land types and land use distribution.

iv. Settings in which the models have been applied and contaminants that have been modeled

The settings and the contaminants that have been modeled using this method are summarized in Table F-1.

v. General evaluation of the method

The method has obvious appeal for many applications because information concerning land use, soil type distribution, precipitation and temperature data, and stream stage are readily available for most watersheds. Additionally, the method is relatively easy to use. Once a computer spread sheet containing the required inputs has been established for a given watershed, by inputting weather data, the loading rate to surface water can be estimated. However, ultimately any model used for management decisions must be able to predict future loading rates as a result of changes in management practices. Because loading models rely on a multicomponent loading factor, the effect of changing one component of the loading factor on surface-water quality may be difficult to determine, making the models less suited for management applications.

vi. References to annotated bibliography

References to the accompanying annotated bibliography are summarized in Table F-2.

Table F-1. Summary of Settings in Which the Method Has Been Applied and the Contaminants Measured.

Location	Aquifer	Contaminant	Author
Pennsylvania	Trimmers Rock Catskill formations	Nitrate	W. J. Gburek, J. B. Urban, R. R. Schnabel
Walton, New York	-	Nitrogen, phosphorous	D. A. Haith, L. L. Shoemaker
Inland Bays, Delaware	-	Nitrogen, phosphorous, BOD	W. F. Ritter
Chesapeake Bay, Virginia	-	Nitrogen, phosphorous	R. R. Schnabel, W. J. Gburek
Wisconsin	-	Nitrogen, phosphorous K. M. Green	P. D. Uttormark, J. D. Chapin

Table F 2. References to Annotated Bibliography

Author	Citation	Reference to Annotated Bibliography
W. J. Gburek, J. B. Urban R. R. Schnabel	"Nitrate Contamination of Ground Water in an Upland Pennsylvania Watershed," Proceedings of the Agricultural Impacts on Ground Water, A Conference, Omaha, Nebraska, August 11-13, 1986, pp. 352-380.	pp.160-162
D. A. Haith, L. L. Shoemaker	"Generalized Watershed Loading Functions for Stream Flow Flow Nutrients," <u>Water Resources Bulletin</u>, 1987, 23(3): 471-478.	pp.163-165
W. F. Ritter	"Nutrient Budgets for the Inland Bays," Report to Delaware Department of Natural Resources and Environmental Control, August, 1986.	pp.166-167
R. R. Schnabel, W. J. Gburek	"Calibration of NPS Model Loading Factors," <u>Journal of Environmental Engineering</u>, 1983.	pp.168-169
P. D. Uttormark, J. D. Chapin	"Estimating Nutrient Loading of Lakes from Non Point Sources," Office of Research and Monitoring, 1974, Environmental Protection Agency report number PA 660/3-74-020.	pp.170-171

G. Studies using environmental isotope methods to estimate the contribution of ground water to stream flow

The papers cited in this section are summarized in Section XI of "An Annotated Bibliography of the Literature Addressing Nonpoint Source Contaminated Ground-Water Discharge to Surface Water," September, 1990, EPA 440/6-90-006.

1. General description of method

a. Description of method or procedure

An assessment of naturally-occurring (or, in some cases, anthropogenically-produced) environmental isotope distributions can assist in the differentiating among sources of water supplying streams. Three environmental isotopes are commonly used in studies of runoff generation in catchments: oxygen-18 (^{18}O), deuterium (D or ^{2}H), and tritium (T or ^{3}H). These isotopes are almost ideal tracers for runoff generation studies, due to two principal characteristics: (1) since ^{18}O, D, and T are constituent parts of natural water molecules (e.g., $H_2{}^{18}O$, $HD^{16}O$, and $HT^{16}O$), they travel at the same rate through the catchment as "average" water ($H_2{}^{16}O$); and (2) ^{18}O, D, and T are chemically conservative at the low temperatures associated with most small watershed systems (i.e., their concentrations do not change by reactions with the catchment materials). The isotope concentrations in the flow system are altered only by physical processes such as: mixing, diffusion, dispersion, and radioactive decay.

Both ^{18}O and D are stable isotopes which occur naturally, accounting for about 0.2% of all oxygen atoms and about 0.015% of all hydrogen atoms, respectively. Their average concentrations in water, expressed as $H_2{}^{18}O$ and $HD^{16}O$ are about 2000 and 320 ppm, respectively. T is a radiogenic isotope of hydrogen whose half-life is in the order of 12.4 years. T atoms represent an extremely small proportion of terrestrial hydrogen (about 10^{-14} to 10^{-16}%) of all hydrogen atoms. Concentrations of T are expressed in tritium units (TU) in which 1TU = $1T/10^{18}$ atoms of hydrogen. Although very few T measurements were made on precipitation prior to the introduction of anthropogenically-produced T into the atmosphere, indications are that precipitation contains from 4-25 TU of naturally-produced T. Since the advent of atmospheric testing of thermonuclear devices in 1952, T produced as a by-product of this testing has been the dominant source of T in precipitation.

The term "hydrograph separation," discussed in section D of this report is normally associated with graphical hydrograph separation techniques which have been used for decades in predicting runoff volume and residence times. Another type of hydrograph separation, based on natural chemical or isotopic tracers in water, has been developed as a more "physically-based" runoff separation technique than the graphical technique described previously. This hydrograph separation technique apportions storm and snowmelt hydrographs into contributing components based on the distinctive chemical or isotopic signatures carried by each of the contributing components. The distinctive signature of each component is developed as the water molecules passing through the catchment take different flow paths and have different residence times.

The tracer-based hydrograph separation technique normally involves a two-component mixing model for the stream. The model assumes that water in the stream at any time during a storm or snowmelt runoff event is a mixture of two components: "new water," which is water from the current rain or snowmelt event, and "old water," which is the subsurface water which existed in the catchment prior to the current rain.

During baseflow conditions (the low flow conditions which occur between storm and snowmelt runoff events) in humid, headwater catchments, all the water in a stream is discharged ground water. The chemical and isotopic character of stream water at a given location during baseflow represents an integration of the upstream "old water" discharges. During storm and snowmelt runoff events, however, "new water" (rain or snowmelt) is added to the stream. If the "old" and "new" water components are chemically or isotopically different, the stream water becomes "diluted" by the addition of the "new" water. The extent of this dilution is a function of the relative contributions from the "old" and "new" water components.

The precursor to using natural isotopes as tracers in the simple two-component mixing model involved the use of various chemical parameter tracers such as total dissolved solids, chloride, and electrical conductivity. These mixing models are based on the general observation that "old water" has higher concentrations of most chemical parameters than "new water".

The major problem associated with separating hydrographs on the basis of chemical tracers is that most chemical tracers are not conservative. The chemistry of the "new water" may vary both areally and temporally as the rain or snowmelt water interacts with the catchment materials on the way to the stream. The chemical tracer technique could lead to an overestimate of the "old water" contribution since the "new water" progressively gains more solutes on its way to the stream and its chemistry becomes more like that of the "old water". One can, however, derive some valuable information characterizing runoff processes by using natural chemical tracers. Some parameters, such as silica are fairly conservative and can give reliable "old" and "new" water components. Other parameters such as chloride in coastal areas, can give some indication of the amount of "washoff" of atmospherically-deposited particulates by overland flow.

During the 1960's, hydrologists began to use the separation equations with anthropogenically-produced radioisotopes.

b. Assumptions involved in using the method

The environmental isotope hydrograph separation method uses an environmental isotope as the tracer in the separation equations (equations [2] to [4] in Section G.i.d). If the isotopic signatures of the "old" and "new" water are different, the contributions of "old" and "new" water in a storm or snowmelt hydrograph can be determined.

The isotopic signature of the "old" water in a catchment results from the mixing of rain and snowmelt which falls over a long period of time. For the stable isotopes, the "old" water del values (obtained from solving equation [1]) remain essentially constant year after year or they develop an annual sinusoidal cycle with the most depleted values in the winter and the least depleted values in the summer. Whether the "old" water del values are constant or cyclical is controlled by the residence time of the "old" water and the degree of dispersive flow in the watershed. The "old" water tritium concentrations in small watersheds generally show a gradual decrease with time, reflecting the progressively lower tritium concentrations in recent (post-1963) precipitation and tritium decay.

The isotopic signature of the "new" water component provides the contrast needed for the isotopic separation. Although the general seasonal variations in the ^{18}O, D, and T concentrations in precipitation and the general long-term decline in tritium levels are documented, it must be emphasized that there is no guarantee that the "new" water in an individual rain or snowmelt event will be isotopically different from the "old" water.

Four assumptions govern the reliability of hydrograph separations using environmental isotopes:

- The "new" and "old" water component can be characterized by a single isotopic value for each component or variations in each component's isotopic content can be documented.
- The isotopic content of the "old water" component is significantly different from that of the "new water" component.
- Vadose zone water contributions to the stream are negligible during the event or they must be accounted for (use an additional tracer if isotopically different from ground water).
- Surface water storage (channel storage, ponds, swamps, etc.) contributions to the stream are negligible during the runoff event.

c. Limitations of the method

One major assumption in using environmental isotopes is that the baseflow represents the "old" water component and the source of ground-water flow to the stream during storm and snowmelt events is the same as the source during baseflow conditions. However, ephemeral springs remote from the stream or deeper ground-water flow systems may contribute differently during events and if their isotopic signatures differ from that of baseflow, the assumed "old water" isotopic value may be incorrect. Although the occurrence of such situations could be tested by hydrometric monitoring and isotopic analyses of these features, qualification could be difficult.

Catchments with significant surface storage cannot be accurately characterized using isotope hydrograph separation methods. Isotopic enrichment of surface water in lakes, ponds, and swamps by evaporation may introduce complications in the simple two component model.

During some events, the "old" and "new" water isotopic contents may be too similar for meaningful hydrograph separations. Considering that substantial time may be spent waiting for and then sampling an event and considerable costs may be incurred for isotopic analyses, it is prudent to monitor an event using chemical tracers as well as isotope analysis. These other methods may include: (1) testing for other independent isotopes such as T rather than ^{18}O if ^{18}O is unsuitable and (2) testing for a conservative chemical constituent, such as silica, or other less conservative parameters such as electrical conductivity.

Estimating the actual isotopic composition of the rainfall reaching the ground surface is complicated in forested catchments because of the interception loss (by evaporation) from the forest canopy during rainfall. Evaporation from water stored on the forest canopy typically occurs at rates of 0.1-0.5 mm/hr and can account for the loss of about 20% of the gross rainfall. Depending on the ambient relative humidity at the canopy level, evaporation of 20% of the rainfall could substantially enrich the D or ^{18}O composition of throughfall and net rainfall compared with that of the gross rainfall usually measured and sampled. This is a potentially serious problem only when the gross rainfall is isotopically lighter (more negative in delta notation; equation [1] below) than the prestorm stream water.

d. Representative equations

Since D and ^{18}O concentrations in natural waters are much smaller than their common light isotopes (^{1}H and ^{16}O), D and ^{18}O concentrations are generally expressed in the conventional delta (δ) notation as per mil (0/00) differences relative to the international standard, SMOW:

$$\delta D \text{ or } \delta 180 = \frac{(Rsample - RSMOW)}{R_{SMOW}} \times 1000 \quad [1]$$

Analytical precision for δD and $\delta 180$ by mass spectrometry is better than 2 and 0.2%, respectively, with a confidence level of 95%.

Between storm events, stream base flow reflects the isotopic composition of the "old" (stored) water. During storm runoff events, however, the isotopic character of the stream may be altered by the addition of "new" water from rainfall. The "old" and "new" water contributions at any specified time can be calculated by solving the mass balance equations for the water and isotopic fluxes in the stream. These equations are expressed as:

$$Q_s = Q_o + Q_n \quad [2]$$

$$C_sQ_s = C_oQ_o + C_nQ_n \quad [3]$$

$$Qo = \frac{(Cs - Cn)}{(Co - Cn)} \times Q_s \quad [4]$$

where Q is discharge, C expresses tracer concentration, and the subscripts s, o, and n refer to the stream, "old water," and "new water," respectively. The utility of the mass balance equations for any particular storm event is controlled mainly by the magnitude of (C_o - C_n) relative to the analytical error and the recognition of areal and temporal variations in C_o and C_n. The equations can also provide estimates of "old" and "new" water percentage contributions to throughflow and overland flow.

Environmental isotope data can also used in estimating the areal extent of overland flow contributing areas and in calculating mean residence time of the "old" water in the catchment. Assuming that overland flow is generated entirely as saturated overland flow, the overland flow contributing area is estimated by the following equation:

$$Y = \frac{V_n}{V_{tn}} \qquad [5]$$

where Y is the discharge area expressed as a fraction of the total basin area, V_n is the total volume of "new water" which leaves the catchment and V_{tn} is the total volume of "new water" which falls on the catchment. The mean residence time of the "old water" in a catchment can be estimated by comparing the seasonal variations in del values for the precipitation and the streamflow or by analyzing the tritium input (precipitation) and output (streamflow) functions.

e. Description of equipment needs

Stage or precipitation activated time discrete automatic water samplers are needed to ensure sampling at the start of an event, especially for night-time storms or for remote catchments. Snowmelt lysimeters are needed for sampling snowmelt. To sample soil water, lysimeters are required. Ground-water samples are obtained by installing piezometers. Measurements of streamflow from catchments require weirs with stage recorders.

The concentrations of ^{18}O and D in a water sample are normally measured using a double-collecting mass spectrometer which compares the concentration of ^{18}O or D in the water sample to a standard water. Water samples for T analysis are measured using a liquid scintillation counter.

f. Expertise required to apply this method

The method requires knowledge of basin characteristics and the temporal distribution of precipitation and runoff in the basin as well as considerable professional judgement. Site water sampling requires a sufficient understanding of regional geology and hydrology.

ii. Data inputs for the method

To determine the "old" water isotopic value, the samples to be obtained include the following: ground water at various sites (shallow or deep) within the catchment, soil moisture at several sites (shallow or deep), and baseflow in a first order stream in the catchment or baseflow in a larger order stream. Most isotopic studies have used either ground water or baseflow to characterize the isotopic content of the "old" water. The isotopic value of stream baseflow is a good approximation of the isotopic value of ground-water discharging into the stream. Soil moisture is also appropriate for the "old" water component in certain hydrologic environments.

Many samples are needed when conducting environmental isotope studies of storm and snowmelt runoff; i.e., one must take as many time discrete samples of the precipitation (and snowmelt) and the stream as possible. Depending on the nature of the study, sampling frequency may vary from minutes to day or longer depending on the size of the catchment, type of sampling equipment, type of event, and detail desired. Through flow, overland and macropore flow sources can also be separated using isotopic methods.

iii. Outputs for the method

a. Ground-water quantity discharge to surface water

Essentially any quantity of ground-water discharge to surface water can be estimated using isotopic tracer-based hydrograph separation techniques. Isotopes leave signatures of stored water (in the unsaturated and saturated zones) that can be detected at the discharge points. Environmental isotope results can be used to test whether integrated water quality models represent catchment processes appropriately.

b. Ground-water quality discharge to surface water

Observed contaminant concentrations in surface water can be correlated with the runoff components indicated by the isotopic data. For example, if stream flow is found to be dominated by "old" water during a precipitation event and observed contaminant concentrations rise above baseflow concentrations, the increased contaminant levels may presumably arise from subsurface discharge.

iv. Settings in which the method has been applied

Some of the settings assessed and the isotopes analyzed in representative studies are presented in Table G-1. The predominant conclusion from these studies is that "old" water components normally dominate storm and snowmelt runoff in humid, headwater catchments. These studies demonstrate how isotope tracer studies can improve the characterization of runoff processes beyond those findings based upon hydrometric and/or hydrochemical data.

v. General evaluation of the method

Because isotopic tracers are constituent parts of natural water molecules, they can be used as excellent tracers of water origin and movement. The long term and widespread application of these tracers analyses, will allow researchers to study runoff generation on scales ranging from macropores to portions of catchment slopes to first and higher order streams (Sklash 1990).

Several disadvantages may arise in the use of isotopic tracers, however. Conditions for their use are not met in every event, and sample analysis is expensive. In some catchments, the isotopic content of "old" and "new" waters is not distinguishable in the snowmelt, and variability in the "new water" isotopic component may decrease the precision of the separation.

vi. References to annotated bibliography

References to the accompanying annotated bibliography are provided in Table G-2.

G-1. **Summary of Settings in Which the Method Has Been Applied and the Isotopes Measured**

#	Location	Catchment Size (km^2)	Isotope			Author(s)
			T	^{18}O	D	
1	France	24	*			E. Crouzet, P. Hubert, P. Olive, E. Siwertz, A. Marce (1970)
2	United Kingdom	1.0	*			D.S. Biggin (1971)
3	Netherlands	650 ha		*		W.G. Mook, D.J. Groenveld A.E. Bouwn, A.J. Van Ganswyk (1974)
4	Canada	22, 1.8		*		P. Fritz, J.A. Cherry, K.U. Weyer, M.G. Sklash (1976)
5	Canada	73 to 700		*		M.G. Sklash, R.N. Farvolden, P. Fritz (1976)
6	Canada	1, 1.2, 3.9		*		M.G. Sklash, R.N. Farvolden, (1976)
7	Canada	10.5, 1.24, 1.76		*		D.J. Bottomley, D. Craig, L.M. Johnston (1984)
8	USA	620	*		*	V.C. Kennedy, C. Kendall, G.W. Zellweger, T.A. Wyerman, R.J. Avangion (1986)

G-1. Summary of Settings In Which the Method Has Been Applied and the Isotopes Measured

#	Location	Catchment Size (km^2)	Isotope T	^{18}O	D	Author(s)
9	New Zealand	3 8 ha		*		A.J. Pearce, M.K. Stewart, M.G. Sklash (1986)
10	New Zealand	3.8 ha, 1.6 ha, 0.3 ha, 2.8		*		M.G. Sklash, M.K. Stewart, A.J. Pearce (1986)
11	Sweden	33.5		*		G. Jacks, E. Olofsson, G. Werne (1986)
12	Australia	82 ha		*	*	J.V. Turner, D.K. Macpherson, R.A. Stokes (1987)
13	Czechoslovakia	2.65	*	*		T. Dincer, B.R. Payne, T. Florkowski, J. Martinec,
14	Switzerland	43.4	*			J. Martinec, H. Siegenthaler, H. Oescheger, E. Tongiorgi (1974)
15	Canada	11.4, 111		*		H.R. Krouse, G. Holecek, H. Steppuhun (1978)
16	Canada	2.8		*		P.M. Wallis, H.B.N. Hynes, P. Fritz (1979)
17	West Germany	18.7			*	A. Herrmann, W. Stichler (1980) E. Tongiorgi (1974)

G-1. Summary of Settings in Which the Method Has Been Applied and the Isotopes Measured

#	Location	Catchment Size (km^2)	Isotope T	^{18}O	D	Author(s)
18	Canada	368	*	*		F.W. Schwartz (1980)
19	Canada	2.8	*	*		M.G. Sklash, R.N. Farvolden (1980)
20	Sweden	4.0, 6.8		*		A. Rodhe (1981)
21	Sweden	several		*		A. Rodhe (1984)
22	Canada	10.5		*		D.J. Bottomley, D. Craig, L.M. Johnston (1984)
23	Norway	41 ha		*		N. Christophersen, S. Kjaernsrod, A. Rodhe (1985)
24	USA	42.2		*		R.P. Hooper, C.A. Shoemaker (1986)
25	Canada	60		*		M.M. Obradovic, M.G. Sklash (1986)
26	Canada	10.5		*		A.J. Bottomley, D. Craig, L.M. Johnston (1986)
27	USA				*	J.R. Lawrence (1987)

Table G 2. References to Annotated Bibliography

Author	Citation	Reference to Annotated Bibliography
M. G. Sklash	"Environmental Isotope Studies of Storm and Snowmelt and Runoff Generation," In Surface and Subsurface Processes in Hydrogeology, M. G. Anderson and T. P. Burt, (ed.), John Wiley and Sons Ltd., Sussex, England, 73 p., 1990, In print.	p. 289
M. G. Sklash, I. D. Moore, G. J. Burch	"Environmental Isotope Tracer Studies of Catchment Processes: Tools for Verifying Integrated Water Quality Models," In: Proceedings of the USDA, AIRS-81, pp. 459-478. International Symposium on Water Quality Modeling of Agricultural Non-point Sources.	pp.293-295
R. P. Hooper, C. A. Shoemaker	"A Comparison of Chemical and Isotopic Hydrograph Separation," Separation," <u>Water Resources Research</u>, 1986, pp. 1444-1454.	pp.285-286
P. Maloszewski, W. Rauert, W. Stichler, W. Herrmann	"Application of Flow Models in an Alpine Catchment Area Using Tritium and Deuterium Data," <u>Journal of Hydrology</u>, 1983, 66: 319-330.	pp.287-288
M. G. Sklash, R. N. Farvolden	"The Role of Groundwater in Storm Runoff," <u>Journal of Hydrology</u>, 1979, 43: 45-65.	pp.291-292

Chapter III

The Impact of Nonpoint Source Contaminated Ground-Water Discharge to Surface Water in Water Quality-Limited Water Bodies:

Determining Total Maximum Daily Load and Waste Load Allocations

Introduction

This chapter provides a general overview of the process for determining the Total Maximum Daily Load (TMDL) for water quality-limited water bodies and the allocation of point source waste loads and nonpoint source loads to achieve the TMDL.

As used in this chapter, TMDLs are defined as the assimilative capacity of a waterbody, which is the sum of the individual Waste Load Allocations (WLAs) for point sources and Load Allocations (LAs) for nonpoint sources and natural background (see 40 CFR 130.2(h)), plus a safety factor. A Waste Load Allocation is the portion of a receiving water's loading capacity that is allocated to one of its existing or future point sources of pollution (see 40 CFR 130.2(g)). Similarly, Load Allocations (LAs) are the portions of a receiving water's loading capacity that is attributed either to one of its existing or future nonpoint sources of pollution or to natural background sources (see 40 CFR 130.2(f)). In sum, the TMDL should encompass the contaminant waste loads from point sources and nonpoint sources. However, the nonpoint source load allocation may be accounted for simply as a component of background contaminant concentrations. This chapter provides a preliminary discussion of the rationale for applying the methods described under 2 above to better measure or estimate the nonpoint source component of the load allocation under a TMDL.

Under Section 319 of the Clean Water Act, by August 4, 1988 the States were required to identify those water bodies that were not expected to attain or maintain their respective water quality standards due to point or nonpoint source loads. In addition, the States were directed to develop a program to alleviate these problems by describing how they will utilize the TMDL process to control nonpoint source pollution in accordance with Section 319 (b) (2) (B) of the Clean Water Act. This Section calls for "an identification of programs to achieve implementation of the best management practices (BMPs) by the categories, subcategories, and particular nonpoint sources designated under subparagraph (A)." Subparagraph (A) requires an identification of the BMPs and measures which will be undertaken to reduce pollutant loadings resulting from each category, subcategory, or particular nonpoint source designated under Section 319 (a) (1) (B). Presently, this requirement is the only regulatory tool available under the Clean Water Act to promote nonpoint source controls. To date, the Agency has prepared a variety of guidance documents and models to assist in determining TMDLs as part of the water quality-based permitting process.[1] This discussion specifically addresses the manner in which nonpoint source loads may be accounted for in this process.

The chapter is organized in four sections. Following this introduction, Section 2 introduces the regulatory concepts and statutory authorities mandating the TMDL and load allocation processes. Section 2 also provides a summary of the status of WLA applications and water quality-based permitting. Section 3 discusses the theory behind the TMDL process and the application of WLAs with regard to estimating water quality impacts from biochemical oxygen demand, nutrients, and toxic substances and provides a limited overview of WLA modeling approaches. Finally, Section 4 reviews the applicability of the methods described in 2 to supply the data needed to assess nonpoint source loads as part of a TMDL analysis.

A. Statutory and regulatory mandate for determining WLAs and LAs under the TMDL process

1. Rationale behind waste load allocation and water-quality based permitting

Under Section 303 of the Clean Water Act, the States are required to set water quality standards that protect the public health or welfare, enhance the quality of water, and serve the purposes of the Act for all waters in the State.[2] These standards are based on water quality criteria developed by U.S. EPA[3] and are to guarantee the achievement of a designated use for the water body. The State may not set a water body's designated use at less than fishable/swimmable without performing a use attainability analysis.[4]

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[1] EPA is in the process of preparing a series of nine Waste Load Allocation guidance documents. Several of those documents that are currently available from the Monitoring and Data Support Division/U.S. EPA are cited in this chapter.

[2] See also 48 FR 51400 for the regulations implementing the water quality standard process.

[3] These water quality criteria are developed under Section 304(a) of the Clean Water Act. Quality Criteria for Water 1986, published May 1987, is the most recent EPA summary of water quality criteria.

[4] Use attainability analyses involve a determination of the level of aquatic protection that can be achieved for a water body. The analysis includes an assessment of (1) what are the aquatic uses(s) currently being achieved in the water body, (2) what are the potential uses that can be attained based on the physical, chemical, and biological characteristics of the water body, and (3) what are the causes of any impairment of the uses? See Technical Support Manual: Water Body Surveys and Assessments for Conducting Use Attainability Analyses, November 1983. USEPA/OW.

Sections 302 and 304(l) of the Clean Water Act require the States to identify those waters for which technology-based effluent limitations are not sufficiently stringent to attain the water quality standards. The technology-based limits are mandated under Sections 301 and 307 of the Act and are implemented by the Agency through the promulgation of industry-specific effluent guidelines.[5] The States must rank their water quality-limited stretches for planning purposes and set total maximum daily loads of pollutants in the stretches that will achieve the applicable standard. Finally, the TMDLs are to be converted to wasteload allocations through modeling and ultimately to water quality-based effluent limitations on individual point source dischargers in the limited stretch.

ii. Implementing waste load allocations and load allocation in water quality-limited water bodies

Water quality-based controls are implemented for any stream segment in which it is know that water quality does not meet applicable water quality standards, and/or is not expected to meet applicable water quality standards, even after the application of the technology-based effluent limitations required by Sections 301 (B) and 306 of the Clean Water Act (see 40 CFR 130.2(i)). For these segments, water quality-based effluent limits may be

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[5] Under Sections 301 and 307 of the Clean Water Act, as amended in 1972, EPA is responsible for promulgating technology-based effluent guidelines, and applying these guidelines in permits to industrial point source dischargers. EPA is to review standards annually and to revise them every three years. Equivalent technology based standards also apply to municipal discharges; these have been defined by EPA as secondary treatment of municipal wastewater.

Under the 1972 amendments, industrial point sources were required to apply the best practicable control technology currently available (BPT) to their processes by July 1, 1977. BPT was interpreted as involving mainly "end-of-pipe" controls that imposed control costs and economic impacts that were not "wholly out of proportion" with water quality benefits. In the second phase of pollution control, the Act mandated that industries were to adopt best available technology economically achievable (BAT), or, if feasible, zero discharge, by July 1, 1983. In contrast to BPT, BAT was thought of primarily as in-plant process changes that had been or were capable of being achieved. Compliance costs were considered in setting BAT, but no cost-benefit analysis was necessary as with BPT. Finally, new sources were expected to immediately comply with strict standards of performance based on best available demonstrated control technology (BACT), a standard comparable to BAT for existing sources.

Under the 1977 amendments to the Act, Congress modified the original technology-forcing approach somewhat to include a new category of control, best conventional pollutant control technology (BCT), to be achieved by July 1, 1984. However, EPA found the BCT cost tests difficult to apply and, as a result, for most industry categories the BCT effluent limitations are virtually equivalent to BPT requirements.

imposed on point source dischargers under the authority of Sections 302 and 402 of the Clean Water Act. Water quality-based effluent limits are derived to protect water quality in the receiving water regardless of cost or waste stream treatability. In addition, however, Section 302 of the Act makes provision for permitees to apply for variances from the water quality-based effluent limits based upon the "relationship between the economic and social costs and the benefits to be obtained from achieving such limitation."

Control programs for nonpoint sources are developed as part of the planning process described under Section 319 of the Clean Water Act. In many cases, States do not have certified or established techniques or procedures for completing load allocations. In those cases, the State should present its overall schedule form implementing the TMDL process and a schedule with milestones for establishing the appropriate load allocations. Until the load allocation is approved by EPA, the State should pursue a technology-based approach. Technology-based controls are to be based upon water quality considerations and not just resource protection. For example, while agricultural management activities are directed at minimizing soil erosion for productivity purposes, the technology-based approach requires the landowner/operator to include not only productivity based controls but offsite measures such a filter strips and sediment and water control structures, as well. If BMP implementation is not adequate, the State should develop an action plan to develop additional BMPS, including a schedule to assess the water quality conditions and determine if standards are being met within an appropriate timeframe.

B. Determining the Total Maximum Daily Load

This section presents a brief description of the scientific understanding of the processes underlying estimates of total maximum daily load. Because the approach for assessing water quality impacts from different categories of contaminants varies, each of three classes of substances, biochemical oxygen demand, nutrients, and toxics is discussed separately.

1. Theory behind Waste Load Allocation

<u>Basic Principles</u>

Total Maximum Daily Load assessments provide information to assist in making effective decisions on levels of treatment required for a source or sources of pollutant load. WLAs are water quality oriented and are directed at establishing a qualitative relationship between a particular waste load and its impact on water quality. These relationships make it possible to compare incremental changes in concentrations of specific constituents in the receiving water system. One is then able to identify the maximum waste load that can be discharged without violating a water quality standard.[6]

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[6] <u>Technical Guidance Manual for Performing Waste Load Allocations, General Guidance</u>.

Because of the array of variable elements (e.g., temperature, stream flow, load level, reaction rates) that must be considered in WLAs, computerized mathematical models are generally employed to make the necessary calculations. Furthermore, the factors and model formats also differ depending on the water contaminant under investigation. The approaches taken for each of three major classes of water contaminants assessed in determining TMDLs are discussed below.

Biochemical Oxygen Demand/Dissolved Oxygen

Biochemical oxygen demand (BOD) is a measure of the amount of oxygen used in stabilizing a biodegradable material. Both carbonaceous organic compounds (CBOD) and nitrogenous forms (NBOD), such as ammonia and organic nitrogen, are subject to bio-oxidation. In most WLA applications, the amount of oxygen consumed for biodegradation over a five-day period (BOD_5) is used as the standard measurement. However, full oxidation of organic compounds generally requires in excess of twenty to thirty days for completion.[7]

When an organic waste is loaded into a water body, it is subjected to two processes that influence the transport of the waste: (1) advection, which represents the downstream transport of the waste load in stream flow, and (2) dispersion, which encompasses the turbulent and eddying processes that tend to mix the waste load with upstream and downstream waters. Under steady-state conditions (i.e., constant waste load and stream flow), the advection and dispersion processes can be assumed to be constant. In many WLA applications, one may assume steady-state conditions because critical low-flow conditions are modeled (e.g., the 7-day, 10-year low flow period). However, if conditions vary, the transport processes will also vary.

The biochemical oxygen demand and the resulting dissolved oxygen levels in the water body are a function of the ability of naturally occurring bacteria to decompose the organic waste materials, thereby utilizing the oxygen resources of the water body. Replenishment of the oxygen resources in the water body occurs either through transfer of atmospheric oxygen into the water column or, to a lesser extent, through oxygen production by aquatic plants.

The interaction of these processes produces the reduction in dissolved oxygen levels which is the focus of WLA modeling. The critical factor in the protection of water quality is an understanding of the rate at which reaeration takes place and the magnitude of this rate in relation to the rate of oxygen consumption. This relationship is generally expressed in terms of an oxygen deficit, which is defined as the difference in concentration between the saturation value and the actual dissolved oxygen concentration.[8]

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[7] Technical guidance Manual for Performing Waste Load Allocation. Book 2: Streams and Rivers. Chapter 1: Biochemical Oxygen Demand/Dissolved Oxygen. PB86-178936. September 1983. p. 2-13.

[8] Ibid, p. 2-19.

Nutrients and Eutrophication

The major water body impact associated with nutrient loading is eutrophication, or enrichment of the biological productivity of the water body. Waste load allocations for control of eutrophication are generally designed to reduce nutrient inputs. This strategy presumes that the nutrient to be controlled limits the rate of growth and subsequent population of phytoplankton. It further presumes that reducing the population level of phytoplankton will provide the desired control of the complex process of eutrophication and eliminate undesirable water quality situations such as algal blooms. Therefore, it should be noted that WLAs to control eutrophication in water bodies focus directly on nutrient reductions and indirectly on phytoplankton and dissolved oxygen conditions that result from overstimulation by nutrients.[9]

Nutrient levels in water bodies are controlled by external and internal sources. External sources of nutrients include municipal and industrial point sources, stream inputs, atmospheric deposition, urban runoff, ground-water discharge, agricultural drainage, and other nonpoint sources. Internal sources include sediment release, biological recycling, and nitrogen fixation.

Chemicals and Toxic Substances

The procedure for developing realistic mathematical models for chemical fate is similar to the mass-balance approach used for other measures of water quality, such as biochemical oxygen demand. The main differences involve the modeling of processes affecting the chemical constituent. These processes include chemical partitioning between the soluble phase and adsorption onto particulate matter, chemical transfers and kinetics involved in the decay or volatilization of the constituent, and sedimentation processes. In conducting WLA for toxic substances, all of these processes are accounted for in a mass balance equation. The result is a prediction of chemical concentrations in the water column, sediment, and, in some cases, in the biota present in the water body.

The fundamental transfer and kinetic characteristics are known for a wide variety of chemicals based upon laboratory analyses. These characteristics can be combined with other relationships, such as advection and dispersion predictions, to account for the manner in which any material is transported in a water body.[10]

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9. Technical Guidance Manual for Performing Waste Load Allocations. Book 4: Lakes and Impoundments. Chapter 2: Nutrient/Eutrophication Impacts. PB86-178928. August 1983. p. 1-4.

10. Technical Guidance Manual for Performing Waste Load Allocations. Book 4: Lakes, Reservoirs and Impoundments. Chapter 3: Toxic Substances Impact. EPA 440/4-87-002. December 1986. p. 6.

ii. **Waste Load Allocation models: Steady-state conditions**

General Approach

Conservation of mass is the fundamental principle which is used as the basis of all mathematical WLA models of real world processes. All material must be accounted for whether transported, transferred, or transformed. A rate equation which conforms to the requirements of mass balance is

$$V \cdot dc/dt = J + \Delta T + \Delta R + \Delta W$$

where

- c = concentration of the chemical
- J = transport through the system
- T = transfers within the system
- R = transformation reactions within the system
- W = chemical inputs
- V = volume of water body
- t = time.

This fundamental model forms the basis for assessing pollutant load to a water system. Most WLA applications also assume steady-state conditions, thereby eliminating the need to measure changes in parameters over time. The simplified steady-state framework for chemical WLA modeling also assumes complete mixing throughout the water body.

A steady-state model requires single, constant inputs for effluent flow, effluent concentration, background receiving water concentration, receiving water flow, and meteorologic conditions. As a result, the effects of variability in nonpoint source and point source contaminant discharge on receiving water quality cannot be predicted accurately using these steady-state techniques. Nonetheless, steady-state models provide a relatively simple and conservative tool for estimating water quality impacts from contaminant discharges. The specific analytical approaches for steady-state WLA modeling for BOD, nutrients, and toxic substances are described below.

Biochemical Oxygen Demand and Dissolved Oxygen Profile

A dissolved oxygen profile for a stream reach is based upon a simple mass balance which accounts for the mass of BOD entering a stream reach, the mass leaving the reach, and the biodegradation and reaeration processes that occur within the reach that result in the oxygen sag. At steady-state, the following mass balance applies:[11]

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[11] Technical Guidance Manual for Performing Waste Load Allocation. Book 2: Streams and Rivers. Chapter 1: Biochemical Oxygen Demand/Dissolved Oxygen. September 1983. PB86-178936, p. 2-40.

MASS IN - MASS OUT + SOURCES - SINKS = 0

$$QC - Q\cdot(C + dC/dx\cdot\Delta x) + K_a\cdot(C_s - C)\cdot V - K_d\cdot L\cdot V = 0$$

- if $U = Q/A$ and $V = A\cdot\Delta x$, then:

$$Q\cdot dC/dx\cdot\Delta x = (Q\cdot\Delta x)\cdot dC/dx = (U\cdot A\cdot\Delta x)\cdot dC/dx = U\cdot V\cdot dC/dx, \text{ and}$$

$$-U\cdot dC/dx + K_a\cdot(C_s - C) - K_d\cdot L = 0$$

- if the oxygen concentration (C) is expressed in terms of oxygen deficit (D) and the saturated oxygen concentration (C_s), then $C = C_s - D$, and

$$-U\cdot d(C_s - D)/dx + K_a\cdot D - K_d\cdot L = 0$$

- if C_s is constant over all x, then

$$U\cdot dD/dx + K_a\cdot D - K_d\cdot L = 0$$

- the rate of change in biochemical oxygen demand concentration (L) is expressed as:

$$L = L_0\cdot e^{(-Kr\cdot x/U)}$$

- therefore,

$$U\cdot dD/dx + K_a\cdot D = K_d\cdot L_0\cdot e^{(-Kr\cdot x/U)}$$

- integrating and solving the equation for the condition that $D = D_0$ at $x = 0$ yields the following:

$$D = D_0\cdot e^{(-Ka\cdot x/U)} + K_d/(K_a - K_r)\cdot L_0\cdot [e^{(-Kr\cdot x/U)} - e^{(-Ka\cdot x/U)}]$$

where

Q =	river flow rate
C =	concentration of dissolved oxygen entering the segment
C_s =	saturation concentration of dissolved oxygen
QC =	mass of oxygen entering the segment
dC/dx =	rate of change of oxygen (C) with distance (x); equivalent to rate of change with time (t) when converted by velocity (U)
$dC/dx\cdot\Delta x$ =	change in oxygen concentration during time of passage through segment of length Δx
Ka =	atmospheric reaeration rate coefficient
K_r =	BOD removal rate coefficient ($=K_d$).

This is a steady-state solution for the oxygen deficit in a stream segment of length Δx. The source term for biochemical oxygen demand (L) represents the concentration of BOD within the mixing zone below a point source outfall. However, the source may also include nonpoint loading from ground-water discharge.

Nutrients

In lakes and estuaries, nutrient inputs may promote increased biological productivity or eutrophication. Such eutrophication processes depend upon continual input of nutrients, as a net sedimentation of nutrients occurs over time. This process can be expressed in a general steady-state, mass balance equation which assumes a completely mixed water body. The removal rate of the nutrients is assumed to be proportional to their water concentration, which is expressed as follows[12]:

$$V \cdot dP/dt = \Sigma Q_i \cdot Pi - K_s \cdot p \cdot V - Qp$$

- where

$\Sigma Q_i \cdot Pi$	=	the sum of all the mass rates of total nutrients discharged to the lake from all sources (point source and nonpoint source) [M/T]; Q_i = flow [L^3/T]; and P_i = the initial nutrient concentration [M/L^3]
p =		lake nutrient concentration [M/L^3]
V =		lake volume [L^3]
K_s=		the net sedimentation rate of the nutrient [1/T]
Q =		lake outflow [L^3/T]

- assuming steady-state (dp/dt = 0), and letting $W = \Sigma Q_i \cdot Pi$:

$$p = W/(Q + K_s \cdot V)$$

- if $V = A \cdot z$ (where A= lake surface area and z = mean depth), then:

$$p = W/[A \cdot z((Q/V) + K_s)]$$

This expression provides a simple estimate of the ambient lake water nutrient concentration given a loading rate of W. However, the equation does not provide any indication of the water quality impacts resulting from the nutrient loading. Such an estimate could be made based upon the ambient nutrient concentration.

Toxic Substances

As mentioned above, the modeling approach for toxic substances is similar to that used for BOD and dissolved oxygen depletion WLAs. One of the principal differences between the approaches arises in the modeling of the

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12 Technical Guidance Manual for Performing Waste Load Allocations. Book 4: Lakes and Impoundments. Chapter 2: Nutrient/Eutrophication Impacts. PB86-178928. August 1983. p.3-18.

various fate processes and reaction rates affecting organic and heavy metal toxic constituents. Several references list reaction rates for toxic constituents.[13] If a first-order decay rate estimate for a particular constituent is available, the following equation for estimating the downstream concentration of the contaminant may be used:

$$C = C_0 \cdot e^{-K(x/u)}$$

where

C = downstream concentration
C_0 = concentration at the mixing zone
x = distance downstream of mixing zone
u = river velocity
K = measured decay rate.

However, the fate of many toxic constituents currently is not well understood because of the confounding effects of varying temperature, pH, and other environmental conditions in a water body.

EPA's TMDL guidance for modeling individual toxicants in streams and rivers recommends the following steady-state models: Simplified Lake/Stream Analysis (SLSA), Michigan River Model (MICHRIV), Chemical Transport and Analyses Program (CTAP), Exposure Analysis Modeling System (EXAMS), and Metals Exposure Analysis Modeling System (MEXAMS). All of these models except MEXAMS can simulate both organic chemical and heavy metal fate and transport in rivers. EPA also recommends these steady-state models for modeling individual toxicants in lakes and reservoirs.

In addition to steady-state models, research has continued on the development of dynamic or continuous simulation models. These modeling approaches are discussed below. EPA recommends the following continuous simulation models for rivers, streams, and lakes: Estuary and Stream Quality Model (WASTOX), Chemical Transport and Fate Model (TOXIWASP), Channel Transport Model (CHNTRN), Finite Element Transport Model (FETRA), Sediment Contaminant Transport Model (SERATRA), Transient One-dimensional Degradation and Migration Model (TODAM), and Hydrologic Simulation Program-Fortran (HSPF). All of these continuous models are designed for multiple reach, multiple source analyses of both organics and heavy metals.[14]

Detailed descriptions of these steady-state and dynamic models are provided in several of the EPA guidance documents cited in Chapter III of this document.

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13 For example, see Water Related Fate of 129 Priority Pollutants, Volumes I and II. EPA-440/4-81-014.

14 Technical Support Document for Water Quality-based Toxics Control. EPA-440/4-85-032.

iii. Dynamic Wasteload Allocation Modeling

At the present time, most States and EPA Regions use steady-state models, which assume the wastewater is completely mixed with the receiving water and the contaminant source loads are constant, to calculate WLAs for pollutants. This assumption may be adequate for conventional pollutants because the greatest environmental impact in the receiving water, such as severe oxygen depletion, is found downstream of the pollutant outfall. However, for toxic pollutants the highest concentration in the receiving water (i.e., near the pollutant outfall) may serve as the critical level for determining the waste load for the contaminant. As a result, dynamic modeling approaches are increasingly being applied to better account for variations in point source loads and variations in ambient water conditions resulting from changes in nonpoint source loads and other factors.

Dynamic TMDL models calculate an entire probability distribution for receiving water concentrations rather than a single worst case based on critical conditions. The prediction of complete probability distributions allows the risks inherent in alternative treatment strategies to be quantified. The dynamic modeling techniques have an additional advantage over steady-state modeling in that they determine the entire effluent concentration distribution required to produce the desired frequency of criteria compliance.

Continuous or dynamic simulation models use each day's effluent flow (Q_e) and concentration data (C_e) with each day's receiving water flow (Q_s) and background concentration (C_s) to calculate downstream receiving water concentrations. The model predicts these concentrations in chronological order with the same time sequence as the input variables. The daily receiving water concentrations can then be ranked from the lowest to the highest without regard to time sequence. A probability plot can be constructed from these ranked values, and the occurrence frequency of any one-day concentration of interest can be obtained.[15]

Several methods are available to compute the probability that downstream toxicants (or effluent toxicity) will exceed criteria. These approaches include an approximate method of moments and numerical integration.[16] The method of moments is based on the following equation:

$$C_t = C_e \cdot [Q_e/(Q_e+Q_s)] + C_s \cdot [1-(Q_e/(Q_e+Q_s))]$$

where C_t = downstream concentration of the contaminant at time t. Estimates of the mean and variance of the effluent concentration, effluent flow, and upstream concentration can be made by regressing the natural log of each of these variables against a standard lognormal random variable. More specific information concerning the variation in each of these terms may also be applied.

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[15] Technical Support Document for Water Quality-based Toxics Control. September 1985. EPA-440/4-85-032. p. 40.

[16] Ibid, p. 41.

An additional dynamic modeling approach involves the use of Monte Carlo simulation. Monte Carlo combines probabilistic and deterministic analyses since it uses a fate and transport mathematical model with statistically described inputs. While Monte Carlo simulations require more input data and calibration data than steady-state modeling approaches, they can account for interactions of time-varying water quality, flow, temperature, and point and nonpoint source loading terms.

The above discussion presents a brief overview of modeling approaches for WLAs. More detailed descriptions of the approaches outlined above are available in the documents summarized in the annotated bibliography which accompanies this report. The following section discusses how information characterizing nonpoint contaminated ground-water discharge to surface water may be incorporated in the WLA process.

C. Assessment of nonpoint source contaminated ground-water discharge to surface-water analysis methods as components of waste load allocation

i. NPS loading in current TMDL models

In steady-state TMDL models, all source terms, including NPS loads, are assumed to be constant. Therefore, one may conclude that NPS loading is accounted for as a component of ambient water quality conditions. However, because the application of steady-state models typically focuses only on point source loads, the contribution of nonpoint source inputs within the water body segment of concern may not be adequately assessed. Furthermore, if the WLA is determined for other than low flow conditions, the NPS load may vary greatly and a significant contaminant source term may be overlooked.

The need for a more thorough understanding of the change in ambient water body conditions brought about by contaminated ground-water discharge increases for dynamic WLA modeling applications. This understanding should include an assessment of the magnitude of NPS loading throughout the water body segment of concern and the spatial and temporal variations in that loading.

The accuracy of the waste load allocation process is highly dependent on the quality of the data used for simulation modeling. This data includes information concerning the ambient conditions in the water body, spatial and temporal variations in the source loading terms, and a detailed understanding of the kinetics of contaminant fate and transport. WLA models should also be calibrated and verified prior to allocating waste loads. Sufficient historical data to accomplish these objectives are often lacking, however, or of the wrong type. Therefore, improved data collection is often needed to better quantify ambient conditions and anticipated loadings.[17]

In addition to characterizing ambient conditions, a firm understanding of the contaminant source terms should be obtained. For systems that are not

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[17] See Stream Sampling for Waste Load Allocation Models. EPA/625/6-86/013.

in low flow (i.e., near steady-state) conditions, this understanding includes a quantitative measure of source constituent and concentration levels over time. For point sources, this information is readily available through analyses of permit conditions or past operating practices. For nonpoint sources, however, the amount of information available to characterize the source terms accurately typically is limited.

A sampling program to support a WLA assessment should, at a minimum, include the following sampling locations within the stream segment: upstream boundary, point source, upstream of point source, mouth of any tributaries entering the segment, upstream of the tributaries, upstream of any nonpoint sources, downstream of nonpoint sources, and downstream boundary of segment. In areas where significant nonpoint source loadings are known to exist, both the flow rate and constituent concentrations should be measured. If this area is not so large that other water quality changes are likely to occur during the travel time through the area, it is reasonable to assume that the changes in concentrations are due to the nonpoint sources and to use these differences as a basis for estimating the loads.[18] However, if the level of nonpoint source loading is significant, a more thorough characterization of the nonpoint source term may be needed.

The following section reviews the applicability of the methods described in 2 above to better characterize nonpoint source loading as part of the TMDL assessment process.

ii. Analysis of contaminated ground-water discharge to surface water assessment methods as sources of data for waste load allocation

As described above, there is no single analytical approach to waste load allocation. The TMDL analysis and the type and amount of data required for an assessment will differ depending upon the water body characteristics, the point and nonpoint source contaminant loads, and the level of water quality impairment. Furthermore, in many situations there may be no need to characterize the component of the ambient water contaminant concentrations contributed by nonpoint source loads. Such a circumstance may arise if the nonpoint source load is minimal and limited controls on the point sources within the watershed will achieve the applicable water quality standard. On the other hand, if nonpoint sources contribute a large portion of the ambient contaminant concentrations in a water body and stringent controls on point source discharges will not achieve the water quality standard, there may be a strong incentive to characterize the contaminant load provided by ground-water discharge to support development of a nonpoint source management strategy. This section discusses the applicability of the various contaminated ground-water discharge to surface water analysis methods for supporting such nonpoint source load assessments.

All of the methods described in Chapter II will provide an estimate of

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[18] Stream Sampling for Waste Load Allocation Applications. EPA-625/6-86-013. p. 2-7.

the loading of nonpoint source contaminants to a water body. However, the methods differ significantly in the level of effort required, the degree of specificity of the collected data, and the ability to accurately assess temporal and spatial variations in nonpoint source loads. As a result, different methods may be suitable for different levels of analysis. For example, in water bodies that are severely water quality-limited and that have high nonpoint source loads, the ability to accurately predict changes in contaminated ground-water discharge may be critical to support dynamic load allocation modeling. In contrast, for water bodies that are not as severely water quality-limited, a fundamental understanding of the component of the water contaminant concentration contributed by ground-water discharge at base flow or steady-state conditions may be sufficient for determining the TMDL.

The following table summarizes four attributes of the contaminated ground-water discharge to surface-water analysis methods. The characterizations are very general in nature and are intended to provide only a "first-cut" assessment of the various attributes of the methods. Because the categorization of the methods necessarily combines several different approaches under one general method heading, a more detailed review of each method application is needed to better assess the relevance of the approach to a particular situation. Nonetheless, this summary allows one to compare and contrast the suitability of the methods for specific applications.

The attributes are as follows: (1) resources needed to implement the method; (2) ability to assess spatial variations in ground-water discharge to a stream segment or water body; (3) ability to measure changes in ground-water discharge levels over time; and (4) the level of confidence in the method's ability to provide data that accurately reflects the "true" level of contaminated ground-water discharge to a water body. The attributes for each of the methods are ranked relative to one another. A more detailed analysis of specific method applications would be needed to provide absolute measures of the attributes for each method.

	Seepage Meter/ Mini-piezometer	Hydrograph Separation	Total Flux Measurement	Numerical Models	Loading Functions	Geophysical Methods	Isotopic Methods
Resources:	moderate	low	moderate	high	low	high	high
Spatial Variation:	possibly, with multiple sample points	no	no	yes	yes	yes	yes
Temporal Changes:	yes, with repeated samples	yes	yes	yes	no	yes	yes
Data Specificity:	high	moderate	moderate	moderate to high	low	moderate	high

This analysis indicates that no one method may be suitable for all applications. Nonetheless, one or more of the methods can provide sufficient data to support load allocations for many applications and environmental settings.

D. Summary

The preceding discussions outlined several methods that have been applied in a variety of environmental settings to assess nonpoint source contaminated ground-water discharge to surface water. Each of the methods is suitable for different applications and settings and the resources required to implement each of the methods also differ. An enhanced understanding of nonpoint source contaminated ground-water loadings to surface water may also improve the total maximum daily load assessment process in water quality-limited water bodies. These methods can support point and nonpoint source load allocations by better characterizing the component of ambient water quality contamination contributed by nonpoint sources under steady-state conditions and by improving the ability to characterize and predict changes in contaminated ground-water loading in dynamic simulation models. The manner in which several of the methods described above can be applied to better account for nonpoint source loading to a stream is the focus of a companion volume to this document.