Preface

The table of contents of this document are linked to the referencing text through *hypertext links*. The *hypertext links* are indicated by blue text and are activated with a single click of the left mouse button near the highlighted word. The large left facing arrow (go to previous view icon) on the toolbar will return the display to the document.

THE WES HANDBOOK ON WATER QUALITY ENHANCEMENT TECHNIQUES FOR RESERVOIRS AND TAILWATERS

REFERENCES

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Simple Random Sampling

APPENDIX 2.1.A. SIMPLE RANDOM SAMPLING

A procedure for selecting n units out of the N possible units so that all possible samples have an equal chance of being selected

Process:

1. Number the units of the target population from 1 to N

2. Randomly select n units of the target population

These n units comprise the simple random sample

Example

Objective: Determine the average dissolved oxygen concentration (over an annual period) in the releases from a flood control project.

Target population: 365 days

Sample size: 52

Simple random sample: Select 52 random numbers between 1 and 365

Random sampling assures the independence of the observations

STRATIFIED RANDOM SAMPLING

Consists of dividing the target population into a distinct set of subpopulations, followed by taking a simple random sample from each subpopulation

The subpopulations are referred to as strata

Advantages over simple random sampling:

1. Useful to have data on subsets of the population

2. May produce an increase in precision

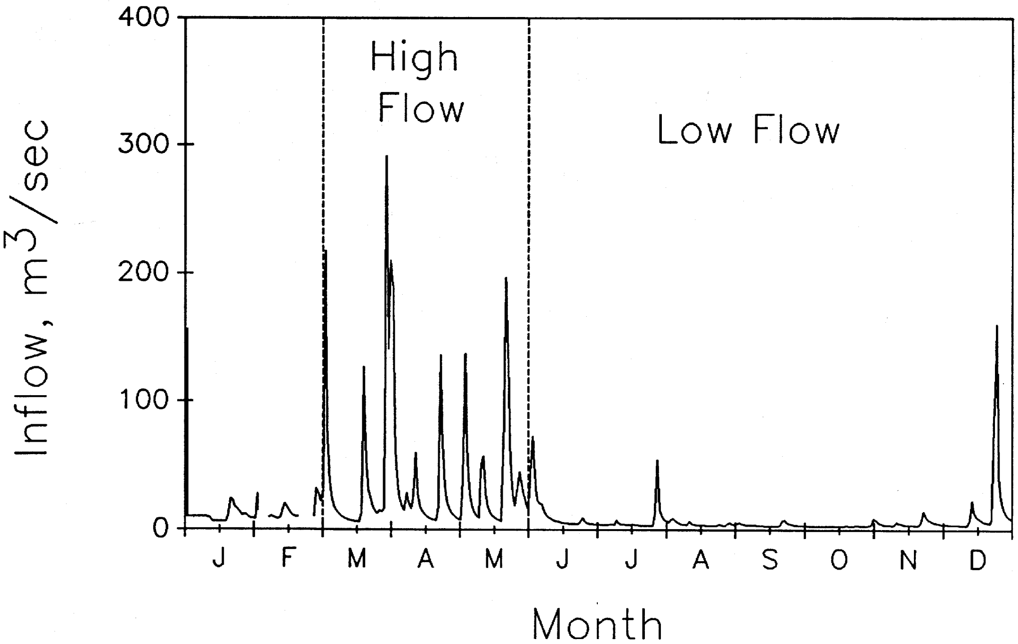
Three steps to stratified sampling

1. Identify and weight the strata
2. Determine the total sample size, n_{st}
3. Allocate the samples to the strata

Stratified Sampling Example

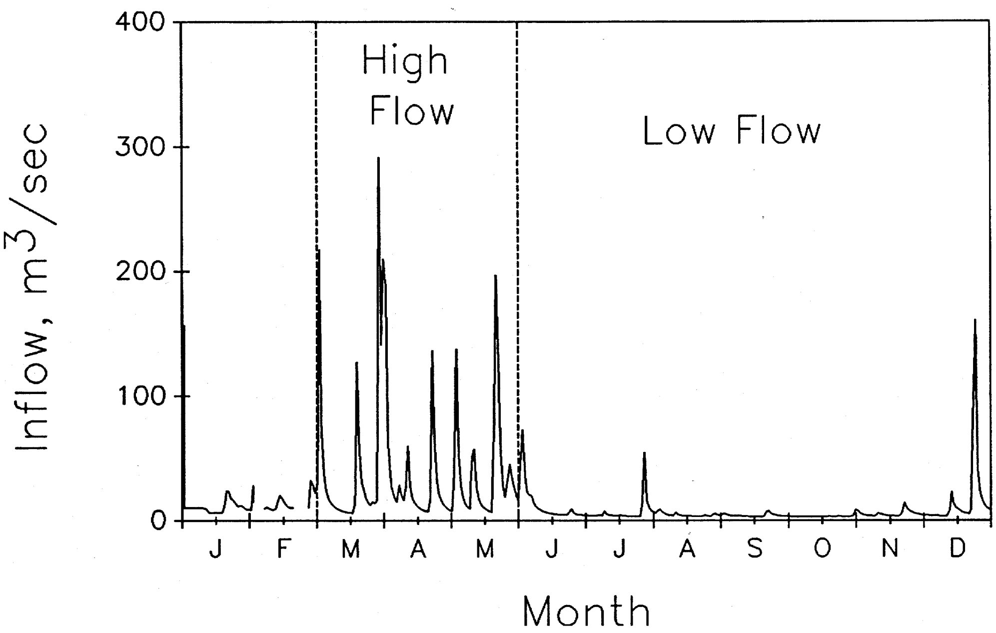
Objective: Calculate nutrient load contributed by the major tributary to a flood control project.

Sampling Design: Divide the target population into two strata based on flow conditions.



High flow period: Mar - May --- 92 days, 70% of the water load

Low flow period: Jan - Feb, Jun - Dec --- 273 days, 30% of the water load



Strata weights

1. Based on time

   High flow period 92/365 = .252

   Low flow period 273/365 = .748

2. Based on water load

   High flow period .70

   Low flow period .30

Sample Size

Assume a sample size of 52 was calculated

Allocation of samples

1. Equal

    high flow - 26

    low flow - 26

2. Proportional

    a. based on time

        high flow (.252)(52) = 13.1 => 13

        low flow (.748)(52) = 38.9 => 39

    b. based on water load

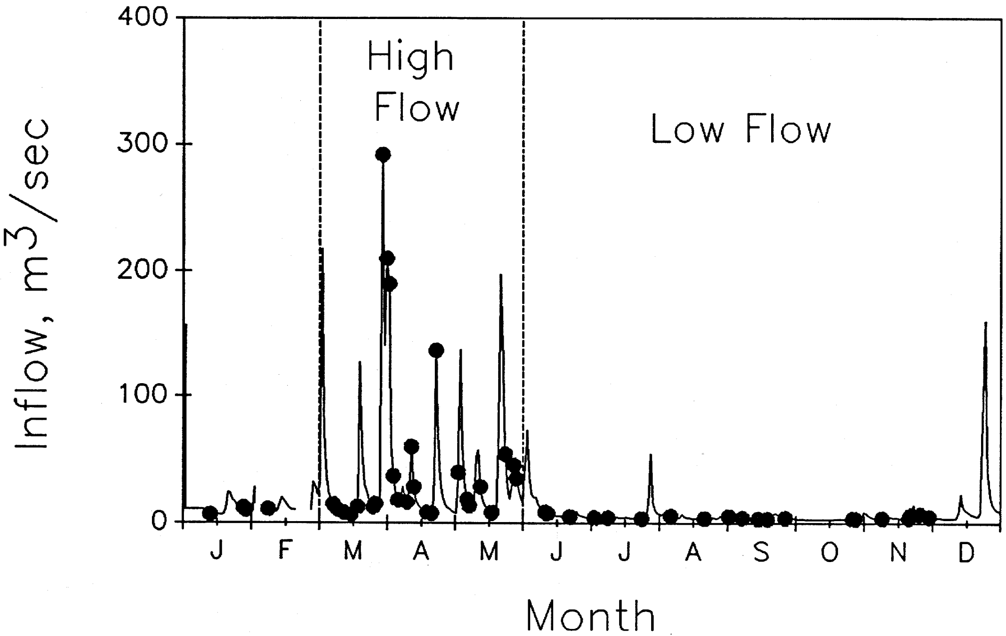
        high flow (.70)(52) = 36.4 => 36

        low flow (.30)(52) = 15.6 => 16

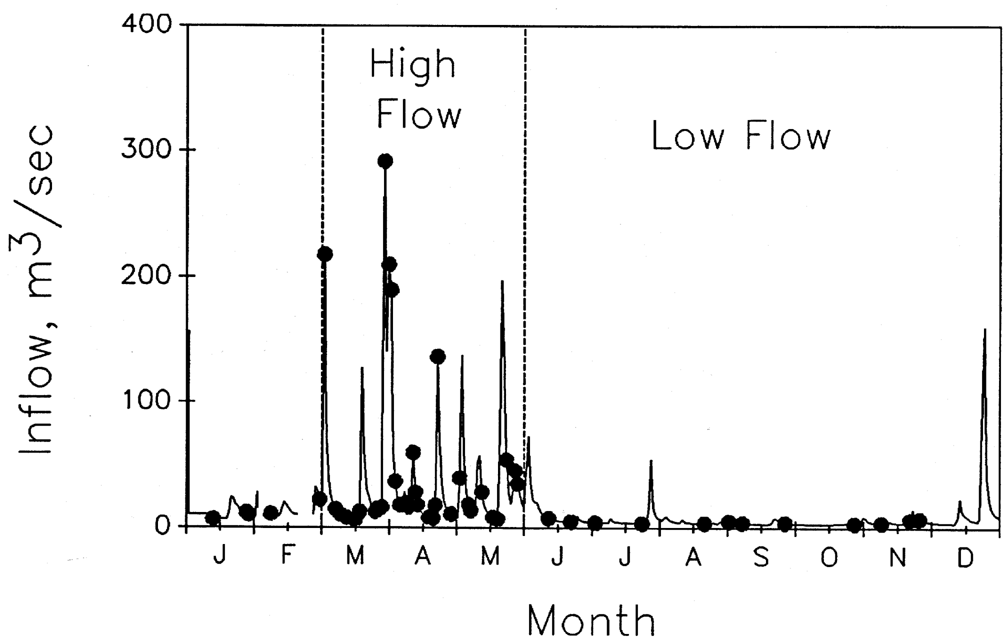
3. Optimal (using strata based on water load and assuming the high flow period is twice as variable as the low flow period)

high flow	43
low flow	9

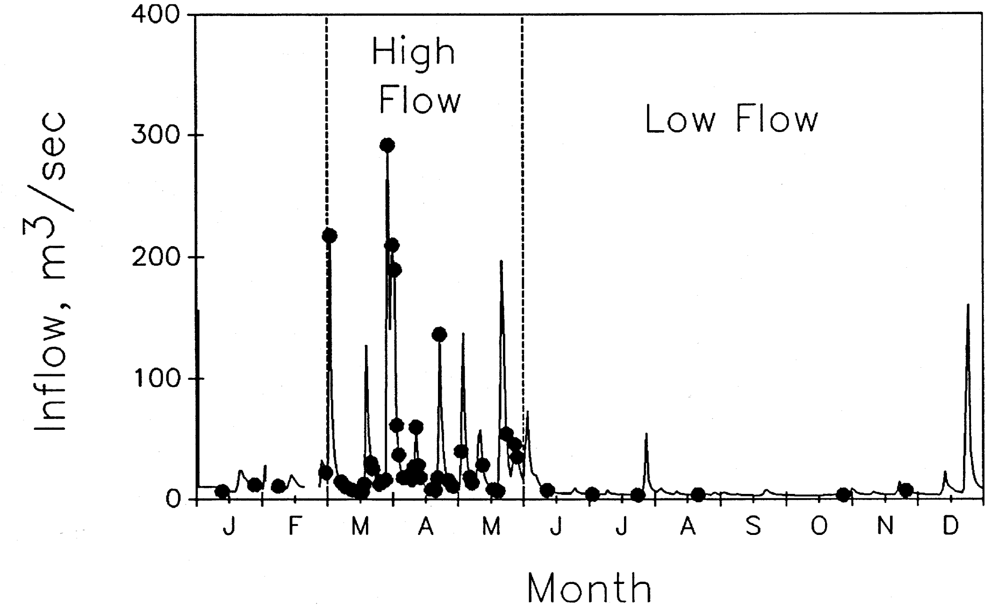
EQUAL ALLOCATION



PROPORTIONAL ALLOCATION



OPTIMAL ALLOCATION



SYSTEMATIC SAMPLING

Differs considerably from simple random and stratified random sampling

Sampling population is selected by taking n units at some predefined interval

Examples:

1. Sampling with respect to depth
2. Sampling with respect to time

Apparent advantages over simple random sampling

1. Development of the sampling program is simple and easy to execute

2. Probably more precise - view the systematic sample as a stratified design

Major disadvantage to systematic sampling

The observations that make up the systematic sample will, in most cases, lack independence

The lack of independence makes it impossible to calculate the sample variance and error variance without bias

APPENDIX 2.1 B

Sampling Design Software User's Manual

by: Robert F. Gaugush

Instruction Report W-93-I
February 1993

US Army Corps
of Engineers
Waterways Experiment
Station

Water Operations Technical Support Program

Sampling Design Software User's Manual

by Robert F. Gaugush
Environmental Laboratory

Prepared for Headquarters, U.S. Army Corps of Engineers

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Water Operations Technical
Support Program

Instruction Report W-93-l
February 1993

Sampling Design Software User's Manual

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Environmental Laboratory

U.S. Army Corps of Engineers
Waterways Experiment Station
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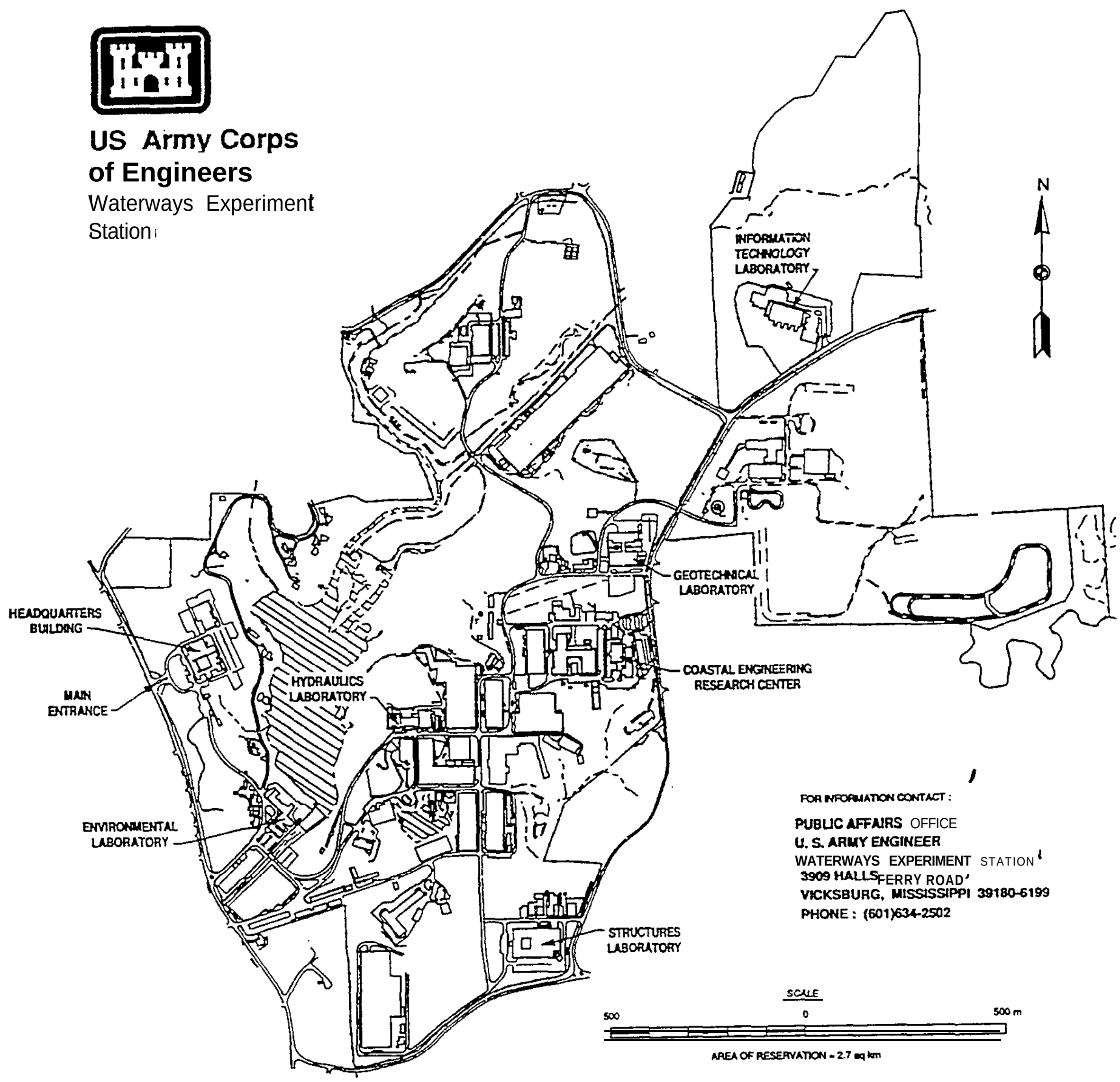
Final report

Approved for public release; distribution is unlimited

Prepared for U.S. Army Corps of Engineers
Washington, DC 20314-l 000

US Army Corps of Engineers
Waterways Experiment Station



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Contents

Preface

This report was prepared by the Environmental Laboratory (EL) of the U.S. Army Engineer Waterways Experiment Station (WES), as part of the Water Quality Management for Reservoirs and Tailwaters Demonstration of the Water Operations Technical Support (WOTS) Program, sponsored by the U.S. Army Corps of Engineers (HQUSACE). Mr. Pete Juhle, HQUSACE, is Technical Monitor. The WOTS is managed under the Environmental Resources Research and Assistance Programs (ERRAP), Mr. J. Lewis Decell, WES, Manager. Dr. A. J. Anderson was Assistant Manager, ERRAP, for the WOTS program.

This report was prepared by Dr. Robert F. Gaugush of the Aquatic Processes and Effects Group (APEG), EL, under the direct supervision of Dr. Robert H. Kennedy, APEG, and under the general supervision of Mr. Donald L. Robey, Chief, Ecosystem Research and Simulation Division, EL, and Dr. John Harrison, Chief, EL.

At the time of publication of this report, Director of WES was Dr. Robert W. Whalin. Commander was COL Leonard G. Hassell, EN.

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1 Introduction

Background

The Sampling Design Software (SDS, Version 2.0) was developed as a companion to the Instruction Report "Sampling Design for Reservoir Water Quality Investigations" (Gaugush 1987). Four programs were developed to assist the user with problems with sampling design and its evaluation. The programs aid the decision-making process in sampling design through the use of decision matrices (the DECMATRX program). Sampling design evaluation is performed using variance component analysis (the VARCOM program), error analysis (the ERROR program), and cluster analysis (the CLUSTER program).

The purpose of this user's manual and the SDS disk provided with it is to assist the user in the implementation of these programs and is not intended to provide instruction on the assumptions and calculation methods of the statistical techniques used by these programs. The Bibliography presents a number of sources for basic statistics, sampling design, and more advanced statistical topics. The instruction report mentioned previously represents an introduction to the topic of sampling design. An introduction to statistics from a reservoir water quality perspective can be found in "Statistical Methods for Reservoir Water Quality Investigations" (Gaugush 1986).

Contents of the SDS Disk

A total of 39 files are provided on the SDS disk. The .EXE files are the compiled program files for DECMATRX, VARCOM; ERROR, and CLUSTER. These programs were developed and compiled using Turbo Pascal 5.5 (Borland International, Copyright 1984, 1989). The program files also have associated help files (files with an extension of .Hxx). Three example data sets are provided for the programs VARCOM, ERROR, and CLUSTER. These data sets are EG.VAR, EG.ERR, and EG.CLS, respectively.

Some files are required for all of the programs. The files with an extension of .BGI are graphics device drivers. Only one of these files will be used for any particular application, but all are provided for maximum compatibility with the numerous graphics cards to be found in personal computers (PC's). The files with an extension of .CHR are graphics character sets that are used in the introductory screens for each program. These files are supplied with the Turbo Pascal 5.5 compiler (Borland International, Copyright 1984, 1989).

The COLORS.DAT file is a short ASCII-format text file that is read by all of the programs to set the screen colors. If, after running the programs, you would like to change the screen colors, then simply edit this file. Notes on color selection are included in the file.

A complete listing of the files on the SDS disk is provided below:

Decision Matrices files:

DECMATRX.EXE - program file

DECMATRX.H01 - help files
DECMATRX.H02
DECMATRX.H03
DECMATRX.H04
DECMATRX.H05

Variance Component Analysis files:

VARCOM.EXE - program file

VARCOM.HOl - help files
VARCOM.H02
VARCOM.H03

EG.VAR - example data file

Error Analysis files:

ERROR-EXE - program file

ERROR.H01 - help files
ERROR.H02
ERROR.H03
ERROR.H04
ERROR.H05

EG.ERR - example data file

Cluster Analysis files:

CLUSTER-EXE - program file

CLUSTER.HOO - help files
CLUSTER-HO 1
CLUSTER.H02
CLUSTER.H03
CLUSTER.H04
CLUSTER.H05
CLUSTER.H06
CLUSTER.H07
CLUSTER.H08
CLUSTER.H09

EG.CLS - example data file

Files used for all programs:

ATT.BGI - graphics drivers
CGA.BGI
EGAVGA.BGI
HERC.BGI
IBM85 14.BGI
PC3270.BGI

LITT.CHR - character sets
TRIP.CHR
COLORS.DAT - data file for setting screen colors

installation

The SDS software will run from a single 360K 5.25-in. floppy disk (the software is supplied in this format), but performance will be improved considerably by installing the software on a hard disk drive.

To install the software on a hard disk:

a. Create a subdirectory for the software

MD C:\SAMPLING

b. Copy all files from the SDS disk to the new directory

CD \SAMPLING
COPY A:*.*

(The above examples assume that your C: drive is a hard disk and that the SDS disk is in drive A:)

Hardware Requirements

The SDS software has been tested on a number of different PC configurations. Testing has included 8088 (basic PC's), 80286 (AT types), and 80386 machines. Numeric co-processors are not required, but will be used if present. The CGA, EGA, VGA, and Hercules graphics drivers are supported.

User Assistance

Please contact:

Robert H. Kennedy, CEWES-ES-A
U.S. Army Engineer Waterways Experiment Station
3909 Halls Ferry Road
Vicksburg, MS 39 180-6 199

Telephone: (601) 634-3659

if you need assistance with the operation of the SDS software.

2 Decision Matrices

A decision matrix is an aid to the 'determination of sample size for multi-variable sampling programs and can be used for either simple random or stratified random sampling designs. The decision matrix is simply a tabular presentation that incorporates the factors necessary to determine sample size: (a) an estimate of the mean, (b) an estimate of the variability, (c) desired precision, (d) the acceptable probability of error, and (e) the costs associated with sampling. See Gaugush (1987) for a more complete discussion of determining sample size and the use of decision matrices.

Program Execution

To run the Decision Matrices program, simply type "decmatrx" at the DOS prompt. Be sure your default directory (i.e., the directory that you are in when you enter the above command) contains all of the files on the Sampling Design Software disk.

After the above command is entered, the program will prompt you for all of the necessary inputs. Program flow is as follows:

a. Introductory screen.

b. Prompt for output route - output may be routed to either the screen only or to a disk file as well as the screen (if disk file output is chosen, the program will prompt for a file name).

c. Data entry.

d. View output.

e. Repeat analysis with new data.

f. Exit program.

A documented session presented below provides a more complete view of the program flow.

Data Entry

DECMATRX is an interactive program and allows you to enter data during the execution of the program. Two data entry windows are used to (a) specify the parameters to be used by the program, and (b) enter estimates of the central tendency (i.e., the mean) and dispersion (Le., the variance) of the variables to be sampled.

In the first data entry window, six fields are highlighted for input. (In the representations of the data entry windows shown below, highlighted fields are indicated by underlining the field.) In the first field enter the value (from 1 to 6) of the number of variables to be used in the decision matrix. The remaining fields are for the error probabilities and the levels of precision to be used in the analysis. Default values are provided for these fields, but they can be changed by entering the desired value in the respective field. Five possible values for the error probability are supported and are restricted to these values because of the method used to calculate the t statistic in the program. Values for precision can fall anywhere within the specified range of possible values. Generally, you will only need to specify the number of variables because the default values for error probability and precision provide a wide range of sample sizes.

```
                    DECISION  MATRIX
       Number of variables (maximum of 6) : _

       Error Probabilities :   .05  .10  .20
            Default to .05  .10  .20
            Possible values: .01  .05  .10  .20  .50

       Levels of Precision     .10  .20
            Default to .10  .20
            Range of possible values  .01 TO  .50

F1 - Help    F2 - Continue
```

The arrow keys allow movement between the fields. The right and down arrows move the cursor to the next field while the left and up arrows move the cursor to the previous field. Typographical errors within a field can be corrected by using the backspace key to delete the error and then retyping the field. Errors can also be corrected after leaving the field that contains the error, but in this case the entire field must be retyped.

The second data entry window consists of four fields for each of the n variables specified in the first window. The example shown below assumes that the analysis is to be performed on three variables. As shown, a name, mean, coefficient of variation (C.V.), and cost must be specified for each variable. As before, the arrow keys allow for movement between the fields. Variable names can contain any characters (uppercase or lowercase, numbers may also be used), but blank spaces are not allowed in variable names. Decimal points are not required in the remaining fields but should be used for clarity. Values for the C.V.'s are expressed as a decimal fraction and not as a percentage. For example, the C.V. would be expressed as 0.50, not as 50.0 percent, for a variable with a mean of 50.0 and a standard deviation of 25.0.

```
                    DECISION  MATRIX

VARIABLE      NAME          MEAN        C.V.      UNIT  COST

   1       ________________________________      _______
   2       ________________________________      _______
   3       ________________________________      _______

F1 - Help    F2 - Continue
```

Error Messages

As the data are entered into the program, DECMATRX checks for errors. The program checks the fields for number of variables, error probability, and precision 'for nonnumeric characters. If any are found, DECMATRX will issue one of the following error messages:

INPUT ERROR: NUMBER OF VARIABLES INCORRECTLY ENTERED
INPUT ERROR: ERROR PROBABILITY INCORRECTLY ENTERED
INPUT ERROR: PRECISION INCORRECTLY ENTERED

The program also checks these same fields to determine if the values entered are within the range of values supported by the program. If any fall outside of the range of supported values, the program will issue one of the following messages:

INPUT ERROR: NUMBER OF VARIABLES IS OUT OF RANGE
INPUT ERROR: ERROR PROBABILITY IS OUT OF RANGE
INPUT ERROR: LEVEL OF PRECISION IS OUT OF RANGE

The second data entry window is also checked for errors. If a C.V. is less than or equal to zero, DECMATRX reports:

```
INPUT ERROR: C.V. <= 0
```

If a sampling cost is entered as a negative number, then the program issues the following error message:

```
INPUT ERROR: COST < 0
```

If any nonnumeric characters are entered for any of the means, C.V.'s, or costs, then one of the following messages will be displayed:

```
INPUT ERROR: MEAN INCORRECTLY ENTERED
INPUT ERROR: C.V. INCORRECTLY ENTERED
INPUT ERROR: COST INCORRECTLY ENTERED
```

Pressing any key after. an error message has been reported will return the program to the data entry screen with the error. Correct the error and continue.

Documented Session

This example session with DECMATRX uses the following data:

Variable	Mean	C.V.	Cost
TP	95.	0.56	25.0
TN	1614.	0.28	25.0
CHLA	35.	0.52	25.0

The object of the analysis is to determine sample sizes and costs associated **with** sampling these three variables over an annual period. Sample sizes and costs for each variable are presented with respect to error probability and precision. The results of the analysis can be used to develop a sampling design within both **statistical** and financial constraints.

Entering the command "DECMATRX" at the DOS prompt begins the program.

Decision Matrices

Sampling Design Software – Version 2.0

Developed by
Dr. Robert F. Gaugush
Environmental Laboratory
USAE Waterways Experiment Station

(Press any key to continue.. .)

Created using Turbo Pascal, Copyright Borland International 1984, 1989

After pressing any key, the program prompts for the output route.

```
Select output route

          1) Screen only
          2) Disk file

                    Enter value to continue...
F1 - Help
```

Press F1 for help.

```
Select output route

                    1) screen only
                    2) Disk file


                              Enter value to continue...
F1 - Help

Help - Output routing

Output from the Decision Matrices program can be routed to a disk
file as well as to the screen.  If you select to output to a disk
file, you will be prompted for a file name (paths can be included).
                                                  F2 - Continue
```

Press F2 to continue and clear the help window.

```
Select output route                                          I

                    1) Screen only
                    2) Disk file


                              Enter value to continue...
F1 - Help
```

Select 2 (disk file output). DECMATRX then prompts for the output file name. Use **MATRIX.OUT** for this session.

```
Input disk file name: matrix.out
```

DECMATRX then displays the first data entry window. (Underlined fields represent fields that will be highlighted on the PC screen).

```
                DECISION  MATRIX
Number of variables  (maximum of 6): _

Error  Probabilities  :  ___ ___ ___ 05 10 20
     Default  to   .05  .10  .20
     Possible  values: .01  .05  .10  .20  .50

Levels  of  Precision    .10  .20
     Default  to   .10  .20
     Range of possible values  .01 TO  .50

F1 - Help   F2 - Continue
```

Press **F1** for help.

```
                         DECISION  MATRIX
                                                   6) : _

                 Default  to   .05  .10  .20
 ==== Help - Data input ==========================================
 Enter data in each of the high-lighted fields. Default values exist
 for the error probabilities and the levels of precision. If these
 values are satisfactory then you only need to enter a value for the
 number of variables.

 To move between fields:    left or up arrow    - previous field
                            right or down arrow - next field
 ================================================ F2 - Continue ===
 ====F1 - Help====F2 - Continue=====================================
```

Press F2 to continue and clear the help window. Enter a “3” in the field for the number of variables.

Press F2 to continue and the program displays the second data entry window.

```
                         DECISION  MATRIX

   VARIABLE      NAME          MEAN        C.V.      UNIT  COST
      1       __________    ________    ________     _____
      2       __________    ________    ________     _____
      3       __________    ________    ________     _____



 ====F1 - Help====F2 - Continue=====================================
```

Press Fl for help.

```
                         DECISION  MATRIX

 VARIABLE        NAME            MEAN          C.V.       UNIT  COST
     1       ____________   _______________________      ______
     2       ____________   _______________________      ______
 ╔═══ Help - Data input ══════════════════════════════════════════╗
 ║ Enter data in each of the high-lighted fields. Provide a name, mean,
 ║ coefficient of variation, and sampling cost for each variable.  The
 ║ sampling costs are usually analytical costs per sample. If costs are
 ║ nor an issue, simply enter a 1 for the cost for each variable.
 ║
 ║ To move between fields:    left or up arrow    - previous field
 ║                            right or down arrow - next field
 ╚═════════════════════════════════════════ E2 - Continue ════════╝
 ═════F1 - Help═════F2 - Continue══════════════════════════════════
```

Press F2 to continue and clear the help window. DECMATRX returns to the data entry window. Enter data to produce the screen shown below.

```
                         DECISION  MATRIX

 VARIABLE        NAME            MEAN          C.V.       UNIT  COST
     1       TP                  95.           0.56          25.
     2       TN                  1614.         0.28          25.
     3       CHLA                35.           0.52          25.



 ═════F1 - Help═════F2 - Continue══════════════════════════════════
```

When data entry is completed, press F2 to continue. The program displays sample sizes with respect to variable, error probability, and precision.

SAMPLE SIZE

PRECISION:		0.10			0.20	
ERROR:	0.05	0.10	0.20	0.05	0.10	0.20
VARIABLE						
TP	123	87	53	33	23	14
TN	33	23	14	10	7	4
CHLA	106	75	46	28	20	12

F1 - Help F2 - Exit F3 - Costs

Press **F1** for help.

SAMPLE SIZE

PRECISION:		0.10			0.20	
ERROR:	0.05	0.10	0.20	0.05	0.10	0.20
VARIABLE						
T?	123	87	53	33	23	14
TN	33	23	14	10	7	4
CHLA	106	75	46	28	20	12

Help - Sample sizes

Sample sizes are provided for each combination of variable, error probability, and precision.

F2 - Continue

F1 - Help F2 - Exit F3 - Costs

Press F2 to continue and clear the help window. Press F3 to see the costs window.

COST

PRECISION:		0.10			0.20	
ERROR:	0.05	0.10	0.20	0.05	0.10	0.20
VARIABLE						
TP	3075	2175	1325	825	575	350
TN	825	575	350	250	175	100
CHLA	2650	1875	1150	700	500	300

F1 - Help F2 - Exit F3 - Sample size

Press F1 for help.

COST

PRECISION:		0.10			0.20	
ERROR:	0.05	0.10	0.20	0.05	0.10	0.20
VARIABLE						
TP	3075	2175	1325	825	575	350
TN	825	575	350	250	175	100
CHLA	2650	1875	1150	700	500	300

Help - Sampling costs

Sampling costs are provided for each combination of variable, error probability, and precision.

F2 - Continue

F1 - Help F2 - Exit F3 - Sample size

Press F2 to continue and clear the help window. 'Any time after data entry, F3 allows switching between the sample size and cost windows. Press F3 to return to the sample size window.

SAMPLE SIZE

PRECISION:		0.10			0.20	
ERROR:	0.05	0.10	0.20	0.05	0.10	0.20
VARIABLE						
TP	123	87	53	33	23	14
TN	33	23	14	10	7	4
CHLA	106	75	46	28	20	12

F1 - Help F2 - Exit F3 - Costs

Press F2 to **exit.**

Repeat program with new data? (Y or N)

At this point, you can either repeat the program with new data or exit the program. Respond with **"N"** to end the documented session.

Example Output File

DECISION MATRIX

INPUT DATA

ERROR PROBABILITIES : 0.10 0.20

LEVELS OF PRECISION : 0.05 0.10 0.20

VARIABLE	MEAN	C.V.	UNIT COST
TP	9.500E+01	5.600E-01	2.500E+01
TN	1.614E+03	2.800E-01	2.500E+01
CHLA	3.500E+01	5.200E-01	2.500E+01

SAMPLE SIZE

PRECISION:		0.10			0.20	
ERROR:	0.05	0.10	0.20	0.05	0.10	0.20
VARIABLE						
TP	123	87	53	33	23	14
TN	33	23	14	10	1	4
CHLA	'106	75	46	28	20	12

COST

PRECISION:		0.10			0.20	
ERROR:	0.05	0.10	0.20	0.05	0.10	0.20
VARIABLE						
TP	3075	2175	1325	825	575	350
TN	825	575	350	250	175	100
CHLA	2650	1875	1150	700	500	300

3 Variance Component Analysis

Variance component analysis is a technique for quantifying the sources of variability in the data resulting from a given sampling design. The analysis results in the determination of each design component's contribution to the overall variance. Based on these results, sampling effort allocated to a given component of the design could be reduced or eliminated. See Winer (1971) for a comprehensive treatment of variance component analysis.

Data Set Preparation

The VARCOM program requires that input data sets be prepared prior to its use (i.e., data input during the program is not available). Data sets can be prepared with most text editors and word processing software. The data sets may contain only ASCII characters and none of the special characters used by most word processors for formatting. If you use a word processor to generate your data sets, be sure to save the files in DOS or ASCII format.

Data in VARCOM input files are organized into four groups:

Group 1 - title
Group 2 - problem size identifiers
Group 3 - factor and level information
Group 4 - data records

An example data set, EG.VAR, is provided on the SDS distribution diskette and is shown below:

```
EAU GALLE - CHLOROPHYLL - MAY 1981               Data Group 1
3 24                                             Data Group 2
STATION                3
          10
          20
          30
DAY                    2
           5                                     Data Group 3
          19
DEPTH                  4
           0
           1
           2
           3
          10           5         0    66.92
          10           5         1    69.89
          10           5         2    69.97
          10           5         3    68.51
          10          19         0     4.77
          10          19         1     6.23
          10          19         2     4.38
          10          19         3     3.88
          20           5         0    46.13
          20           5         1    39.85
          20           5         2    44.17
          20           5         3    46.45       Data Group 4
          20          19         0     3.37
          20          19         1     3.38
          20          19         2     6.11
          20          19         3     4.71
          30           5         0    57.28
          30           5         1    48.00
          30           5         2    59.71
          30           5         3    58.39
          30          19         0     3.50
          30          19         1     3.70
          30          19         2     8.64
          30          19         3     6.47
```

Data Group 1 consists of a single line specifying a title for the data set (maximum of 60 characters). Data Group 2 is a single line with two items. The first is the number of factors in the data set (VARCOM allows a maximum of three factors), and the second indicates the number of observations in Data Group 4. Data Group 3 names the factors, specifies the number of levels for each factor, and provides the name for each of the levels. A maximum of 100 levels is supported by VARCOM. In the example data set, three factors are specified in Data Group 2. The three factors used in the example data set are STATION, DAY, and DEPTH. STATION has three levels (10, 20, and 30) which means that three stations were sampled. DAY has two levels (samples were taken on the 5th and the 19th of May). Depth has four levels (samples were taken at 1-m intervals from the surface to 3 m). Data Group 4 lists the value of the variable to be analyzed (chlorophyll *a* in the example data set) for each combination of the factors. For example, at station 10 on the 5th of May at a depth of 1 m, the chlorophyll *a* concentration was 69.89 µg/l (second line of Data Group 4).

VARCOM requires that the data in Data Groups 3 and 4 be placed in specific columns. A portion of Data Group 3 with column identifiers is shown below.

```
         1         2         3
123456789012345678901234567890   ——— Column numbers

STATION                     3 ——— Number of levels
            10
            20   ├— Level names
            30

  └— Factor name
```

A factor name can have a maximum of 20 characters and must begin in column 1 (i.e., factor names must be left-justified). Separate the factor name and the number of its levels by one blank space. Therefore, the value for the number of levels should begin in column 22 or greater. A level name (in the following row) can have a maximum of 15 characters and must end in column 15 (i.e., all level names must be right-justified). A portion of Data Group 4 with column identifiers is shown below.

```
         1         2         3         4         5         6
123456789012345678901234567890123456789012345678901234567890  ——— Column numbers

            10             5              0     66.92
            10             5              1     69.89
            10             5              2     69.97   ├— Variable data (chlorophyll a)
            10             5              3     68.51

                                          └— Level 3 names (depths)

                           └— Level 2 names (dates)

             └— Level 1 names (stations)
```

Level 1 names must end in column 15, level 2 names end in column 30, and level 3 names end in column 45. At least one blank column must separate the last level name from the variable data.

A data set for a two factor variance component analysis would appear as follows:

```
EAU GALLE - CHLOROPHYLL - MAY 1981
2 24
STATION               3
            10
            20
            30
DAY                   2
             5
            19
            10               5      66.92
            10               5      69.89
            10               5      69.97
            10               5      68.51
            10              19       4.77
            10              19       6.23
            10              19       4.38
            10              19       3.88
            20               5      46.13
            20               5      39.85
            20               5      44.17
            20               5      46.45
            20              19       3.37
            20              19       3.38
            20              19       6.11
            20              19       4.71
            30               5      57.28
            30               5      48.00
            30               5      59.71
            30               5      58.39
            30              19       3.50
            30              19       3.70
            30              19       8.64
            30              19       6.47
```

Note that multiple observations for combinations of levels are allowed. In the above data set, there are four observations for each combination of station and day. It also important to note that the order of lines in Data Group 4 is not important. The above data set could be just as correctly specified as:

```
EAU GALLE - CHLOROPHYLL - MAY 1981
2 24
STATION               3
            10
            20
            30
DAY                   2
             5
            19
            10               5   66.92
            10               5   69.89
            10               5   69.97
            10               5   68.51
            20               5   46.13
            20               5   39.85
            20               5   44.17
            20               5   46.45
            30               5   57.28
            30               5   48.00
            30               5   59.71
            30               5   58.39
            10              19    4.77
            10              19    6.23
            10              19    4.38
            10              19    3.88
            20              19    3.37
            20              19    3.38
            20              19    6.11
            20              19    4.71
            30              19    3.50
            30              19    3.70
            30              19    8.64
            30              19    6.47
```

As long as the level names and the variable data on each line are placed in the proper position, then the lines of Data Group 4 can be arranged in any convenient order. A one factor data set would appear as follows:

```
EAU GALLE - CHLOROPHYLL - MAY 1981
1 24
DAY                 2
               5
              19
               5      66.92
               5      69.89
               5      69.97
               5      68.51
              19       4.77
              19       6.23
              19       4.38
              19       3.88
               5      46.13
               5      39.85
               5      44.17
               5      46.45
              19       3.37
              19       3.38
              19       6.11
              19       4.71
               5      57.28
               5      48.00
               5      59.71
               5      58.39
              19       3.50
              19       3.70
              19       8.64
              19       6.47
```

Suggestion: use an extension of .VAR for VARCOM data files. This will distinguish them from other data files.

Program Execution

To run the Variance Component Analysis program, simply type "varcom" at the DOS prompt. Be sure your default directory (i.e., the directory that you are in when you enter the above command) contains all of the files on the Sampling Design Software disk.

After the above command is entered, the program will prompt you for all of the necessary inputs. Program flow is as follows:

a. Introductory screen.

b. Prompt for output route - output may be routed to either the screen only or to a disk file as well as the screen (if disk file output is chosen, the program will prompt for a file name).

c. Prompt for input file name.

d. View output.

e. Repeat analysis with new data.

f. Exit program.

A documented session presented below provides a more complete view of the program flow.

Error Messages

After prompting for the input and output file names, VARCOM performs an error check on the input data set. If the data set specifies more than three factors for the analysis, the program reports:

```
ERROR: NUMBER OF FACTORS EXCEEDS MAX.  FACTORS
```

If the number of levels for any of the factors exceeds 100, the following error message is reported:

```
ERROR: NUMBER OF LEVELS FOR FACTOR i
       EXCEEDS THE MAX. NUMBER OF LEVELS
```

If the number of observations is greater than 3,500, VARCOM reports:

```
ERROR: NUMBER OF OBSERVATIONS EXCEEDS MAXIMUM
```

If, for any factor, the number of level names does not agree with the names listed, the program provides the following error message:

```
ERROR: LEVEL ID NOT FOUND
```

VARCOM terminates after reporting any of the above error messages. Edit the input data file and run the program again.

Documented Session

This example session with VARCOM uses the EG.VAR data set provided on the SDS distribution diskette. These data were derived from studies conducted on Eau Galle Reservoir in west-central Wisconsin. The data set has three factors: STATION, DAY, and DEPTH. STATION has three levels (stations 10, 20, and 30), DAY has two levels (the 5th and 19th of May), and DEPTH has four levels (depths of 0, 1, 2, and 3 m).

The object of the analysis is to determine the distribution of the variance in chlorophyll *a* among the three factors. If all of the factors account for a significant fraction of the variance in chlorophyll *a*, then the sampling design is efficient. If, on the other hand, one or two of the factors account for most of the variance, then the sampling effort could be reduced. The sampling design could be modified to include only those factors that explain the majority of the variance.

Entering the command "VARCOM" at the DOS prompt begins the program.

Variance Component Analysis

Sampling Design Software – Version 2.0

Developed by
Dr. Robert F. Gaugush
Environmental Laboratory
USAE Waterways Experiment Station

(Press any key to continue...)

Created using Turbo Pascal, Copyright Borland International 1984, 1989

After pressing any key, the program prompts for the output route.

```
Select output route

            1) Screen only
            2) Disk file

                    Enter value to continue...
F1 - Help
```

Press F1 for help.

```
Select output route

                  1) Screen only
                  2) Disk file


                        Enter value to continue...
F1 - Help

Help - Output routing

Output from the Variance Component Analysis program can be routed
to a disk file as well as to the screen.  If you select to output
to a disk file, you will be prompted for a file name (paths can be
included).
                                             F2 - Continue
```

Press F2 to continue and clear the help window.

```
Select output route

                  1) Screen only
                  2) Disk file


                        Enter value to continue...
F1 - Help
```

Select 2 (disk file output). VARCOM then prompts for the output file name. Use EG.OUT for this session.

```
Disk file name for output: eg.out
```

The program then prompts for the input file name. Use EG.VAR for this session.

```
Input data file name?  eg.var

Help - Input data file
Provide the name of your input data file (previously prepared).
Paths can be included in the file specification.
```

VARCOM then displays the results of the variance component analysis.

```
VARIANCE COMPONENT ANALYSIS
        EAU GALLE - CHLOROPHYLL - MAY 1981

        SOURCE                  DF        SS           MS

        STATION                  2     6.30E+02     3.15E+02
        DAY                      1     1.58E+04     1.58E+04
        DEPTH                    3     4.52E+01     1.51E+01
        ERROR                   17     6.91E+02     4.07E+01
        CORRECTED TOTAL         23     1.72E+04

        VARIANCE COMPONENT            ESTIMATE     PERCENT TOTAL

        VAR(STATION             )      3.43E+01         2.47
        VAR(DAY                 )      1.31E+03        94.90
        VAR(DEPTH               )     -4.27E+00       < .01
        VAR(ERROR)                     4.07E+01         2.94


    F1 - Help      F2 - Exit
```

Press F1 for help.

```
VARIANCE COMPONENT ANALYSIS
        EAU GALLE - CHLOROPHYLL - MAY 1981

        SOURCE                  DF        SS           MS

        STATION                  2     6.30E+02     3.15E+02
    ==== Help - Variance component analysis ==========================

    Output is divided into two sections. The upper section of the
    window provides the output of an n-way analysis of variance.
    The "Source" column lists the sources of variability within the data
    set. The "DF" column provides the degrees of freedom for each of the
    sources. The sum of squares and the mean square error are given in
    the "SS" and "MS" columns, respectively. The lower section of the
    output lists the variance component estimates and the relative
    contribution of each source to the overall variance.
    ======================================== F2 - Continue ===========


    F1 - Help      F2 - Exit
```

Press F2 to continue and clear the help window.

```
VARIANCE COMPONENT ANALYSIS
        EAU GALLE - CHLOROPHYLL - MAY 1981

        SOURCE                   DF        SS          MS

        STATION                   2     6.30E+02    3.15E+02
        DAY                       1     1.58E+04    1.58E+04
        DEPTH                     3     4.52E+01    1.51E+01
        ERROR                    17     6.91E+02    4.07E+01
        CORRECTED TOTAL          23     1.72E+04

        VARIANCE COMPONENT              ESTIMATE    PERCENT TOTAL

        VAR(STATION              )      3.43E+01        2.47
        VAR(DAY                  )      1.31E+03       94.90
        VAR(DEPTH                )     -4.27E+00      < .01
        VAR(ERROR)                      4.07E+01        2.94

  F1 - Help       F2 - Exit
```

The variance component analysis indicates that most of the variance (almost 95 percent) is explained by sampling date (the DAY factor). For this data set, sampling stations and dates account for less than 3 percent of the total variance. Press F2 to exit.

```
  Repeat program with new data? (Y or N) N
```

After finishing the analysis, you can repeat the program with a new data set or exit the program.

Example Output File

```
VARIANCE COMPONENT ANALYSIS
        EAU GALLE - CHLOROPHYLL - MAY 1981                    Title

        SOURCE                 DF       SS          MS
        STATION                 2    6.30E+02    3.15E+02
        DAY                     1    1.58E+04    1.58E+04     N-way analysis of variance
        DEPTH                   3    4.52E+01    1.51E+01
        ERROR                  17    6.91E+02    4.07E+01
        CORRECTED TOTAL        23    1.72E+04

        VARIANCE COMPONENT          ESTIMATE    PERCENT TOTAL
        VAR(STATION           )     3.43E+01        2.47
        VAR(DAY               )     1.31E+03       94.90      Variance component estimates
        VAR(DEPTH             )    -4.27E+00       < .01
        VAR(ERROR)                  4.07E+01        2.94
```

4 Error Analysis

Error analysis is a statistical technique that can be used to improve an existing sampling design that uses the observed distribution of variance to redefine the sampling design. The results of the error analysis are used to redistribute samples to the existing strata to produce the minimum variance about the mean. The technique can be applied to the data of a stratified sampling design or to the data from a simple random or a systematic sample that has been subjected to poststratification (i.e., defining strata a posteriori). See Gaugush (1987) for a more detailed description of stratified sampling and the use of error analysis.

Data Set Preparation

The ERROR program requires that input data sets be prepared prior to its use (i.e., data input during the program is not available). Data sets can be prepared with most text editors and word processing software. The data sets may contain only ASCII characters and none of the special characters used by most word processors for formatting. If you use a word processor to generate your data sets, be sure to save the files in DOS or ASCII format.

Data in ERROR input files are organized into four groups:

Group 1 - title
Group 2 - problem size identifier
Group 3 - strata weights
Group 4 - data records

An example data set, EG.ERR, is provided on the SDS distribution diskette and is shown below:

```
EAU GALLE - 1981 - STATION 20        Data Group 1
4                                    Data Group 2
1 .167
2 .334                               Data Group 3
3 .167
4 .332
1 7.8748E+01
1 1.6735E+02
1 4.2722E+01
1 4.3925E+00
2 5.8933E+00
2 2.7610E+01
2 2.9570E+01
2 5.7273E+01
2 4.2378E+01
2 4.0602E+01
2 5.5306E+01
2 6.5534E+01                         Data Group 4
2 5.2158E+01
3 3.1465E+01
3 2.4320E+01
3 4.0684E+01
3 3.1248E+01
4 1.3363E+01
4 1.8966E+01
4 1.0322E+01
4 2.8420E+00
4 3.5075E+00
4 8.2300E+00
4 2.8575E+01
4 2.5618E+01
```

Data Group 1 consists of a single line for the title of the data set (maximum of 60 characters). Data Group 2 also is a single line that specifies the number of strata in the data set. The ERROR program supports a maximum of 25 strata. Data Group 3 specifies the strata numbers and weights. The strata numbers must be in numerical order and start with 1. The strata weights must sum to 1.00. At least one blank space must separate the stratum number and stratum weight in Data Group 3. Data Group 4 lists the observations of the sample data set consisting of the stratum number and the value of the variable (separated by at least one blank space). (Note: Although the example data set uses the computer representation of scientific notation (i.e., 2.5618E+01 is the computer form of 2.5618×10^1) for the data values, this is not required. These numbers could have been entered in a more typical decimal notation.)

Suggestion: use an extension of .ERR for ERROR data files. This will distinguish them from other data files.

Program Execution

To run the Error Analysis program, simply type "error" at the DOS prompt. Be sure your default directory (i.e., the directory that you are in when you enter the above command) contains all of the files provided on the Sampling Design Software disk.

After the above command is entered, the program will prompt you for all of the necessary inputs. Program flow is as follows:

a. Introductory screen.

b. Prompt for output route - output may be routed to either the screen only or to a disk file as well as the screen (if disk file output is selected, the program will prompt for a disk file name).

c. Prompt for input file name.

d. View output.

e. Repeat analysis with new data.

f. Exit program.

A documented session presented below provides a more complete view of program flow.

Error Messages

After prompting for the input and output file names, ERROR performs an error check on the input data set. If the data set specifies more than 25 strata for the analysis, the program reports:

```
ERROR : NUMBER OF STRATA EXCEEDS MAXIMUM
```

If the strata weights do not sum to 1.00, the following error message is reported:

```
ERROR : WEIGHTS DO NOT SUM TO 1.00
```

ERROR reports the following message if any of the strata have less than three observations:

```
ERROR : LESS THAN 3 SAMPLES IN STRATUM i
```

Documented Session

This example execution of ERROR uses the EG.ERR data set provided on the SDS distribution diskette. These data were derived from studies conducted on Eau Galle Reservoir in west-central Wisconsin. Composite epilimnetic samples for chlorophyll *a* were taken at approximately 2-week intervals at Station 20 (a station located at the deepest part of the lake).

The data were stratified a posteriori into four strata: spring, summer, fall, and winter. The strata were defined as follows: "1" for spring - April and May (61 days), "2" for summer - June, July, August, and September (122 days), "3" for fall - October and November (61 days), and "4" for winter - December, January, and February (121 days). Strata weights were calculated by dividing the number of days in the stratum by 365.

The object of the analysis is to determine if the sampling design can be improved through the use of a stratified design using an optimal allocation of samples to the strata. Error analysis calculates the error variance associated with existing distribution of samples and determines an optimal distribution based on the observed variance among strata. If the existing and the optimal distribution of samples are considerably different, the sampling design can be improved by adopting the optimal distribution.

Entering the command "ERROR" at the DOS prompt begins the program.

Error Analysis

Sampling Design Software – Version 2.0

Developed by
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Environmental Laboratory
USAE Waterways Experiment Station

(Press any key to continue...)

Created using Turbo Pascal, Copyright Borland International 1984, 1989

After pressing any key, the program prompts for the output route.

```
Select output route

                    1) Screen only
                    2) Disk file


                            Enter value to continue...
F1 - Help
```

Press F1 for help.

```
Select output route

                    1) Screen only
                    2) Disk file


                            Enter value to continue...
F1 - Help

Help - Output routing

Output from the Error Analysis program can be routed to a disk
file as well as to the screen.  If you select to output to a disk
file, you will be prompted for a file name (paths can be included).
                                              F2 - Continue
```

Press F2 to continue and clear the help window.

```
Select output route

                1) Screen only
                2) Disk file


                      Enter value to continue...
F1 - Help
```

Select 2 (disk file output). ERROR then prompts for the output file name. Use EG.OUT for this session.

```
Disk file name for output: eg.out
```

The program then prompts for the input file name. Use EG.ERR for this session.

```
Input data file name? eg.err

=== Help - Input data file ===
Provide the name of your input data file (previously prepared).
Paths can be included in the file specification.
```

ERROR then displays the statistics for the stratified sample.

```
EAU GALLE - 1981 - STATION 20

 STRATIFIED SAMPLE STATISTICS

           MEAN       3.62E+01
       VARIANCE       1.85E+02
 ERROR VARIANCE       3.97E+01

F1-Help - F2-Sample Stat   F3-Strata Stat  F4-Analysis    F5-Exit
```

Press F1 for help.

```
EAU GALLE - 1981 - STATION 20

 STRATIFIED SAMPLE STATISTICS

          MEAN      3.62E+01
      VARIANCE      1.85E+02
ERROR VARIANCE      3.97E+01

 Help - Stratified sample statistics
 Statistics (mean, variance, and error variance) for the stratified
 sample.
                                          F2 - Continue

F1-Help  F2-Sample Stat  F3-Strata Stat  F4-Analysis  F5-Exit
```

Press F2 to continue and clear the help window.

```
EAU GALLE - 1981 - STATION 20

 STRATIFIED SAMPLE STATISTICS

          MEAN      3.62E+01
      VARIANCE      1.85E+02
ERROR VARIANCE      3.97E+01

F1-Help  F2-Sample Stat  F3-Strata Stat  F4-Analysis  F5-Exit
```

Press F3 to see the strata statistics.

```
EAU GALLE - 1981 - STATION 20

STRATA STATISTICS
     STRATUM      N      MEAN        VARIANCE        ERROR VARIANCE

        1         4    7.33E+01     4.85E+03          1.21E+03
        2         9    4.18E+01     3.42E+02          3.80E+01
        3         4    3.19E+01     4.51E+01          1.13E+01
        4         8    1.39E+01     9.34E+01          1.17E+01

F1-Help   F2-Sample Stat   F3-Strata Stat  F4-Analysis   F5-Exit
```

Press F1 for help.

```
EAU GALLE - 1981 - STATION 20

STRATA STATISTICS
     STRATUM      N      MEAN        VARIANCE        ERROR VARIANCE

        1         4    7.33E+01     4.85E+03          1.21E+03
        2         9    4.18E+01     3.42E+02          3.80E+01
        3         4    3.19E+01     4.51E+01          1.13E+01
        4         8    1.39E+01     9.34E+01          1.17E+01
 ==== Help - Strata statistics ====================================
 Statistics (number of samples, mean, variance, and error variance)
 for each of the sampled strata.

 ==================================================== F2 - Continue ==

F1-Help   F2-Sample Stat   F3-Strata Stat  F4-Analysis   F5-Exit
```

Press F2 to continue and clear the help window. The screen returns to the strata statistics. Press F4 to see the results of the error analysis.

```
EAU GALLE - 1981 - STATION 20

                         ERROR ANALYSIS

     STRATUM     % VARIANCE      % N      % OPTIMUM

        1           85.3         16.0        52.5
        2           10.7         36.0        27.9
        3            0.8         16.0         5.1
        4            3.2         32.0        14.5


     VARIANCE WITH EXISTING DESIGN       3.97E+01

     VARIANCE WITH OPTIMAL DESIGN        1.96E+01



F1-Help  F2-Sample Stat  F3-Strata Stat  F4-Analysis  F5-Exit
```

Press F1 for help.

```
EAU GALLE - 1981 - STATION 20

                         ERROR ANALYSIS

     STRATUM     % VARIANCE      % N      % OPTIMUM

        1           85.3         16.0        52.5
 ===== Help - Error analysis ==========================================
 The %Variance column gives the relative contribution of each stratum
 to the overall stratified sample variance. The %N column shows how
 the samples were distributed among the strata. Using the observed
 distribution of variance among strata (the %Variance column), error
 analysis suggests an optimal distribution of samples among the
 strata (the %Optimum column). The reported "Variance with optimal
 design" is the error variance that would result if the optimal
 design was adopted for future sampling (if conditions do not
 dramatically change over time).
 ================================================ F2 - Continue =====

F1-Help  F2-Sample Stat  F3-Strata Stat  F4-Analysis  F5-Exit
```

Press F2 to continue and clear the help window.

```
EAU GALLE - 1981 - STATION 20

                    ERROR ANALYSIS

     STRATUM     % VARIANCE     % N     % OPTIMUM

        1           85.3        16.0       52.5
        2           10.7        36.0       27.9
        3            0.8        16.0        5.1
        4            3.2        32.0       14.5


     VARIANCE WITH EXISTING DESIGN       3.75E+01

     VARIANCE WITH OPTIMAL DESIGN        1.93E+01


 F1-Help   F2-Sample Stat   F3-Strata Stat  F4-Analysis   F5-Exit
```

At any time during the program you can switch between the output windows. Press F2 to return to the sample statistics screen.

```
EAU GALLE - 1981 - STATION 20

STRATIFIED SAMPLE STATISTICS

          MEAN       3.62E+01
      VARIANCE       1.85E+02
ERROR VARIANCE       3.97E+01


 F1-Help   F2-Sample Stat   F3-Strata Stat  F4-Analysis   F5-Exit
```

Press F4 to return to the error analysis screen.

```
EAU GALLE - 1981 - STATION 20

                    ERROR ANALYSIS

     STRATUM     % VARIANCE     % N      % OPTIMUM

        1           85.3        16.0        52.5
        2           10.7        36.0        27.9
        3            0.8        16.0         5.1
        4            3.2        32.0        14.5


     VARIANCE WITH EXISTING DESIGN       3.97E+01

     VARIANCE WITH OPTIMAL DESIGN        1.96E+01



F1-Help  F2-Sample Stat  F3-Strata Stat  F4-Analysis  F5-Exit
```

The results of the error analysis indicate that the error variance could be reduced to less than 50 percent (19.6/39.7 = 0.494) of its observed value by using the optimal design. The optimal design consists of a redistribution of samples to place more samples in highly variable strata and less samples in strata with less variability. The spring stratum (stratum 1) accounts for over 85 percent of the observed variance (% Variance column), but only 16 percent (% N column) of the samples were allocated to this stratum. The optimal design would allocate just over 52 percent (% Optimum column) of the samples to this stratum. The winter stratum (stratum 4) accounts for only 3 percent of the observed variance, but 32 percent of the sampling effort was allocated to this stratum. The optimal design suggests that only about 15 percent of the samples should be dedicated to this stratum.

Press F5 to exit.

```
Repeat program with new data? (Y or N)
```

At this point you may choose to either run ERROR on another data set or exit from the program.

Example Output File

```
ERROR ANALYSIS

     EAU GALLE - 1981 - STATION 20                          Title

     STRATIFIED SAMPLE STATISTICS

                MEAN      3.62E+01             Statistics for the entire stratified sample
            VARIANCE      1.85E+02
      ERROR VARIANCE      3.97E+01


     STRATA STATISTICS

          STRATUM     N      MEAN      VARIANCE      ERROR VARIANCE

             1        4    7.33E+01    4.85E+03        1.21E+03            Statistics for each
             2        9    4.18E+01    3.42E+02        3.80E+01                 of the strata
             3        4    3.19E+01    4.51E+01        1.13E+01
             4        8    1.39E+01    9.34E+01        1.17E+01

                           ERROR ANALYSIS

          STRATUM     % VARIANCE     % N      % OPTIMUM

             1           85.3        16.0       52.5
             2           10.7        36.0       27.8
             3            0.8        16.0        5.1                 Error analysis
             4            3.2        32.0       14.5


          VARIANCE WITH EXISTING DESIGN      3.97E+01

          VARIANCE WITH OPTIMAL DESIGN       1.96E+01
```

5 Cluster Analysis

Cluster analysis is a multivariate classification technique that may be used to group or identify similar objects or entities. In a data analysis situation (rather than a sampling design evaluation), cluster analysis may be used to group a set of reservoirs according to their trophic state or by the composition of their phytoplankton. The use of cluster analysis in a typical data analysis mode can be found in Gaugush (1986). For the purposes of sampling design evaluation, cluster analysis can be used to identify and possibly reduce redundancies in the sampling design. The use of cluster analysis for this type of application is described more completely in Gaugush (1987).

In the evaluation of a sampling design, cluster analysis can be used to examine the quality of the information being provided by elements of the sampling design. In cluster analysis these elements are referred to as "entities" and may be sampling stations, dates, and/or the strata used in a stratified sampling design. The analysis begins with each entity in its own cluster and proceeds to join similar clusters until all of the entities are in a single cluster. The object, when used to evaluate a sampling design, is to determine if all of the elements of the design are providing independent information. For example, assume that data have been collected for twelve stations in a reservoir and a cluster analysis of the data indicates that the data fall into four clusters each represented by three stations. This implies that some of the stations are redundant (they are supplying essentially the same information). If the sampling program were to be continued (as in a monitoring program), the results of the cluster analysis could be used to reduce sampling effort. Sampling only 1 of the 3 stations from each cluster would result in the use of 4 stations rather than 12.

The CLUSTER program can be used to identify redundancies in sampling programs and suggest ways in which to reduce sampling effort in future studies. CLUSTER uses one of three clustering methods (average linkage, centroid, or Ward's method) to cluster the data; outputs a tabular "history" of the clustering; and produces a dendrogram of the clustering.

Data Set Preparation

The Cluster Analysis program requires that input data sets be prepared prior to its use (i.e., data input during the program is not available). Data sets can be prepared with most text editors and word processing software. The data sets may contain only ASCII characters and none of the special characters used by most word processors for formatting. If you use a word processor to generate your data sets, be sure to save the files in DOS or ASCII format.

Data in CLUSTER input files are organized into four groups:

Group 1 - title
Group 2 - problem size identifiers
Group 3 - entity names
Group 4 - data records

An example data set, EG.CLS, is provided on the SDS distribution diskette and is shown below:

```
EAU GALLE                        Data Group 1
5 3                              Data Group 2
STA10
STA20
STA30                            Data Group 3
STA50
STA60
.069 1.507 44.129
.078 1.503 43.144
.068 1.473 41.155                Data Group 4
.068 1.427 33.800
.070 1.487 46.068
```

CLUSTER does not require strict positioning of data in specific columns, but it does have two simple requirements: (a) each line must start in column 1, and (b) multiple items on a single line must be separated by one blank space. Data Group 1 consists of a single line specifying a title for the data set (maximum of 60 characters). Data Group 2 is a single line with two items. The first is the number of entities in the data set, and the second indicates the number of variables to be used. The CLUSTER program can handle a maximum of 50 entities with a maximum of 10 variables. Data Group 3 provides the names of the entities (one line for each of the entities specified in Data Group 2). Each name can have a maximum of 20 characters. In the example data set, the entities are water quality sampling stations in Eau Galle Reservoir. Data Group 4 lists the data for the variables (one line for each entity and in the same order) to be used in the cluster analysis. In the example data set, these variables are total phosphorus, total nitrogen, and chlorophyll *a* concentrations (from left to right).

Suggestion: use an extension of .CLS for CLUSTER data files. This will distinguish them from other data files.

Program Execution

To run the Cluster Analysis program, simply type "cluster" at the DOS prompt. Be sure your default directory (i.e., the directory that you are in when you enter the above command) contains all of the files provided on the Sampling Design Software disk.

After the above command is entered, the program will prompt you for all of the necessary inputs. Program flow is as follows:

a. Introductory screen.

b. Prompt for input file name.

c. Prompt for output file name.

d. Prompt for clustering method.

e. View output.

f. Exit program.

A documented session presented below provides a more complete view of program flow.

Error Messages

After prompting for the input and output file names, CLUSTER performs an error check on the input data set. If the data set specifies either more than 50 entities or more than 10 variables in Data Group 2, CLUSTER outputs the following:

```
ERROR IN INPUT FILE

EITHER NUMBER OF ENTITIES  50 OR
NUMBER OF VARIABLES  10

EDIT INPUT FILE AND BEGIN AGAIN
```

After displaying the error message the program terminates.

CLUSTER also performs an error check on Data Group 4. If the standard deviation of any of the variables is zero, CLUSTER outputs the following:

```
ERROR IN DATA

STANDARD DEVIATION FOR VARIABLE  j  IS ZERO

THIS MEANS THAT VARIABLE  j  IS THE SAME FOR
ALL ENTITIES AND WILL SERVE NO PURPOSE IN THE
CLUSTER ANALYSIS - DELETE THE VARIABLE FROM THE
INPUT FILE AND BEGIN AGAIN
```

As the error message states, a variable without variance (standard deviation equal to zero) does not add information to the cluster analysis. After displaying the error message, the program terminates.

Documented Session

This example execution of CLUSTER uses the EG.CLS data set provided on the SDS distribution diskette. These data were derived from studies conducted on Eau Galle Reservoir in west-central Wisconsin. The entities are five water quality stations within the reservoir. Stations 10 and 50 (STA10 and STA50) are littoral stations located in two different coves. Station 40 (STA40) is an inlet station. Station 30 (STA30) is located over the old river channel, and Station 20 (STA20) is located over the deepest portion of the pool. These stations were routinely sampled, and the data in Group 4 of EG.CLS are station means for total phosphorus, total nitrogen, and chlorophyll *a* in the epilimnion (0 - 3 m) for one growing season (April - September).

The object of the analysis is to determine if any of the stations are redundant. If two or more stations are supplying the same information, the possibility exists for reducing the number of stations. Reducing the number of stations brings about the obvious reduction in costs without reducing the information derived from the sampling program.

Entering the command "CLUSTER" at the DOS prompt begins the program.

Cluster Analysis

Sampling Design Software – Version 2.0

Developed by
Dr. Robert F. Gaugush
Environmental Laboratory
USAE Waterways Experiment Station

(Press any key to continue...)

Created using Turbo Pascal, Copyright Borland International 1984, 1989

After pressing any key, the program prompts for the input file name. For this session enter EG.CLS.

```
Input data file name? eg.cls

Provide the file name of your data file. Paths are accepted.
```

CLUSTER then prompts for the output file name. Use EG.OUT for this session.

```
Output data file name? eg.out




Provide a file name of your output data file. Paths are accepted.
```

At this point CLUSTER prompts for the method to be used in the cluster analysis. Help windows are available by pressing F1, F2, F3, or F4.

```
CLUSTERING METHOD:  AVERAGE LINKAGE (A)
                           CENTROID (C)
                              WARDS (W)

            Enter choice of method...


F1 - General help
Specific help:  F2 - Avg linkage  F3 - Centroid  F4 - Wards
```

Press F1 and the following is displayed.

```
CLUSTERING METHOD:  AVERAGE LINKAGE (A)
                           CENTROID (C)
                           WARDS    (W)

== Help - Clustering methods ==========================

Three methods (average linkage, centroid, and Wards) are available
to use to cluster the data. Select a method by entering the letter
associated with the desired method.
                                       == F2 - Continue ==
```

Press F2 to continue, and the help window is removed.

```
CLUSTERING METHOD:  AVERAGE LINKAGE (A)
                           CENTROID (C)
                           WARDS    (W)

          Enter choice of method...


F1 - General help
Specific help:  F2 - Avg linkage  F3 - Centroid  F4 - Wards
```

Press F2 for the Average Linkage help window.

```
        CLUSTERING METHOD:  AVERAGE LINKAGE (A)
                                   CENTROID (C)
                                     WARDS  (W)

 ═══ Help - Clustering method: Average linkage ══════════════
 This clustering method has been found (along with Wards method) to
 be one of the more robust approaches to clustering data. In average
 linkage the distance between two clusters is the average distance
 distance between pairs of observations, one in each cluster. This
 method tends to produce clusters with small variance and is somewhat
 biased toward producing clusters with the same variance.
 ═══════════════════════════════════════════ F2 - Continue ═══
```

Pressing F2 (continue) again would remove the help window and restore the method selection screen. For the sake of brevity, assume F2 was pressed followed by F3 for the Centroid help window.

```
        CLUSTERING METHOD:  AVERAGE LINKAGE (A)
                                   CENTROID (C)
                                     WARDS  (W)

 ═══ Help - Clustering method: Centroid ═════════════════════
 This clustering method uses the distance between the centroids or
 means of the clusters. This method is more robust to the presence of
 outliers in the data than either the average linkage or Wards
 methods.  In other respects, the centroid method may not perform as
 well as the other two methods.
 ═══════════════════════════════════════════ F2 - Continue ═══
```

Again assume F2 was pressed to return to the method selection screen and then F4 was selected to bring up the help window on Wards method.

```
CLUSTERING METHOD:  AVERAGE LINKAGE (A)
                           CENTROID (C)
                              WARDS (W)

== Help - Clustering method: Wards ==

This clustering method, although robust, tends to join clusters with
a small number of observations and is biased toward producing
clusters with generally the same number of observations. This method
is also sensitive to the presence of outliers in the data.
                                             == F2 - Continue ==
```

Press F2 to return to the method selection screen.

```
CLUSTERING METHOD:  AVERAGE LINKAGE (A)
                           CENTROID (C)
                              WARDS (W)

            Enter choice of method...


F1 - General help
Specific help:  F2 - Avg linkage  F3 - Centroid  F4 - Wards
```

Press A to select the average linkage method.

```
File: eg.out

Cluster Analysis

    EAU GALLE  .

     ID
    NUMBER      ENTITY

       1        STA10
       2        STA20
       3        STA30
       4        STA50
       5        STA60


    Average Linkage Method used for clustering


  F1-Help F2-Exit Movement Keys: Home, End, PgUp, PgDn, Up and Down Arrows
```

At this point the cluster analysis is complete and you can view your output file (in this case EG.OUT as indicated in the first line). The cursor movement keys (Home, End, Page Up, Page Down, up arrow, and down arrow as indicated on the last line) allow you to browse through the output file. Press Page Down.

```
File: eg.out

    Stage     Clusters Joined     Distance

     1           1     5          6.080E-01
     2           1     3          1.219E+00
     3           1     2          3.191E+00
     4           1     4          6.000E+00


    The distances are segmented into the following
    classes for the Linear dendrogram

    CLASS       LOWER BOUND      UPPER BOUND

       1         6.080E-01        8.237E-01
       2         8.237E-01        1.039E+00
       3         1.039E+00        1.255E+00
       4         1.255E+00        1.471E+00
       5         1.471E+00        1.686E+00

  F1-Help F2-Exit Movement Keys: Home, End, PgUp, PgDn, Up and Down Arrows
```

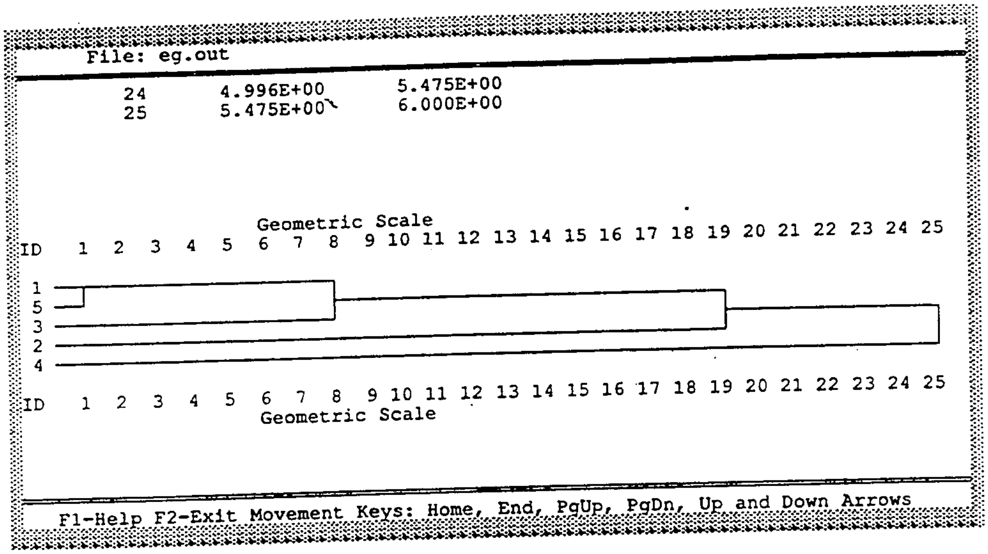
The next 20 lines of the output file are displayed. Press Page Down again.

```
File: eg.out

     6       1.686E+00      1.902E+00
     7       1.902E+00      2.118E+00
     8       2.118E+00      2.333E+00
     9       2.333E+00      2.549E+00
    10       2.549E+00      2.765E+00
    11       2.765E+00      2.981E+00
    12       2.981E+00      3.196E+00
    13       3.196E+00      3.412E+00
    14       3.412E+00      3.628E+00
    15       3.628E+00      3.843E+00
    16       3.843E+00      4.059E+00
    17       4.059E+00      4.275E+00
    18       4.275E+00      4.490E+00
    19       4.490E+00      4.706E+00
    20       4.706E+00      4.922E+00
    21       4.922E+00      5.137E+00
    22       5.137E+00      5.353E+00
    23       5.353E+00      5.569E+00
    24       5.569E+00      5.784E+00
    25       5.784E+00      6.000E+00

F1-Help F2-Exit Movement Keys: Home, End, PgUp, PgDn, Up and Down Arrows
```

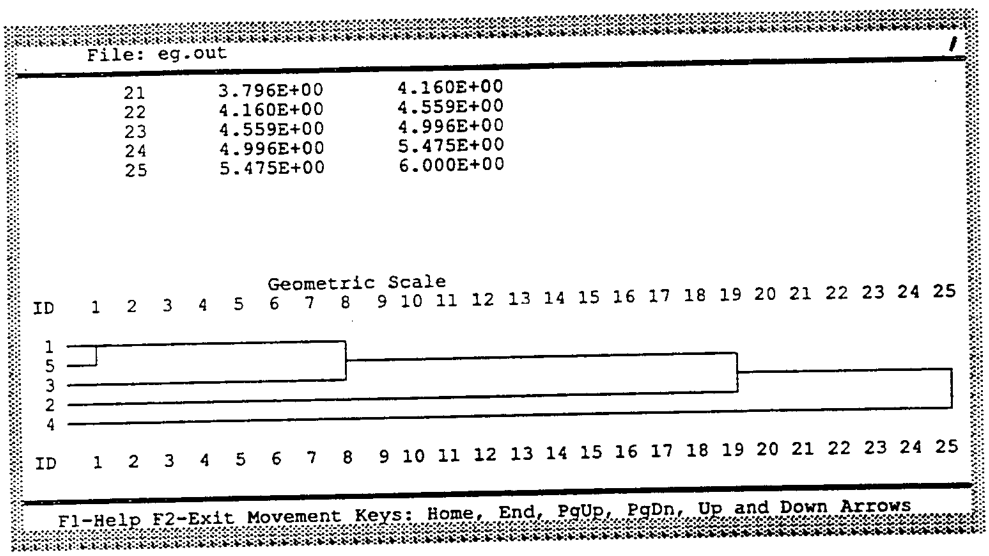
Again the display moves 20 lines down. Press Home.

```
File: eg.out
Cluster Analysis


     EAU GALLE


       ID
     NUMBER      ENTITY

        1        STA10
        2        STA20
        3        STA30
        4        STA50
        5        STA60


     Average Linkage Method used for clustering

F1-Help F2-Exit Movement Keys: Home, End, PgUp, PgDn, Up and Down Arrows
```

The display returns to the top of the output file. Press End.



The display moves to the bottom of the output file. Press the up arrow three times.



The display moves up three lines. The other movement keys operate in a similar manner. Press F1 for help.

```
    File: eg.out
         21        3.796E+00        4.160E+00
         22        4.160E+00        4.559E+00
         23        4.559E+00        4.996E+00
         24        4.996E+00        5.475E+00
         25        5.475E+00        6.000E+00
 ==== Help - Cluster analysis output ====================================

  Short descriptions of various portions of the output are available.

           F3 - Entity and ID numbers

           F4 - Stages of clustering                                      24 25

           F5 - Distances

           F6 - Dendrogram

           F7 - Linear vs. geometric scales for the dendrogram

 ====================================================== F2 - Continue ===  24 25

   F1-Help F2-Exit Movement Keys: Home, End, PgUp, PgDn, Up and Down Arrows
```

A help menu window is displayed over the output file. Press F3.

```
    File: eg.out
         21        3.796E+00        4.160E+00
         22        4.160E+00        4.559E+00
 ==== Help - Entity listing =============================================

  This section lists the ID numbers that have been assigned to the
  entities in the data set. Entities can be stations, dates, depths,
  reservoirs, etc. This listing will be necessary to interpret the
  dendrogram.

 ====================================================== F2 - Continue ===

           F4 - Stages of clustering                                      24 25

           F5 - Distances

           F6 - Dendrogram

           F7 - Linear vs. geometric scales for the dendrogram

 ====================================================== F2 - Continue ===  24 25

   F1-Help F2-Exit Movement Keys: Home, End, PgUp, PgDn, Up and Down Arrows
```

A second help window appears describing the association between ID numbers and the entity names in the data set. Pressing F2 (Continue) removes both help screens and restores the output screen. Press F2 to continue followed by F1 for the help menu, and then press F4 for help on the clustering stages.

```
File: eg.out

   21        3.796E+00        4.160E+00
   22        4.160E+00        4.559E+00
 Help - Clustering stages

 This section of the output provides a tabular display of the data
 used to develop the dendrogram. At each stage of the clustering, two
 clusters are joined (shown in the "Clusters Joined" column) to form
 a new cluster. The "Distance" column provide a measure of the
 relative similarity of the members of the cluster. The smaller the
 distance, the greater the similarity.
                                          F2 - Continue        24 25

          F5 - Distances

          F6 - Dendrogram

          F7 - Linear vs. geometric scales for the dendrogram
                                          F2 - Continue        24 25

 F1-Help F2-Exit Movement Keys: Home, End, PgUp, PgDn, Up and Down Arrows
```

Press F2 to continue followed by F1 for the help menu and F5 for help on the distance classes.

```
File: eg.out

   21        3.796E+00        4.160E+00
   22        4.160E+00        4.559E+00
 Help - Distances

 The range of relative distance (presented in the output describing
 the clustering stages) is divided into 25 discrete classes. This is
 necessary to accommodate the techniques used to develop the graphical
 depiction of the dendrogram.
                                          F2 - Continue
                                                               24 25
          F4 - Stages of clustering

          F5 - Distances

          F6 - Dendrogram

          F7 - Linear vs. geometric scales for the dendrogram
                                          F2 - Continue        24 25

 F1-Help F2-Exit Movement Keys: Home, End, PgUp, PgDn, Up and Down Arrows
```

Press F2 to continue followed by F1 for the help menu and F6 for the dendrogram help window.

```
File: eg.out

      21       3.796E+00       4.160E+00
      22       4.160E+00       4.559E+00
  Help - Dendrogram

 The graphical display from a cluster analysis is referred to as a
 dendrogram because of its tree-like appearance. At the "trunk", all
 of the entities have been joined into a single cluster (at the far
 right of the dendrogram). At the far left, each of the "branches"
 represents a single entity and each cluster has only one entity.
 Moving from left to right, clusters are joined until all of the
 entities have been combined into a single cluster.                      24 25

 The ID values listed along the left margin correspond to those
 assigned to the entities in the data set. The values (1 -25) along
 the top and bottom of the dendrogram correspond to the criterion
 values and provide a relative measure of the similarity between
 members of a cluster. Clusters at the left are composed of more
 similar members than clusters at the right.

                                             F2 - Continue              24 25

 F1-Help F2-Exit Movement Keys: Home, End, PgUp, PgDn, Up and Down Arrows
```

Press F2 to continue followed by F1 for the help menu and F7 for help on the scales used for depicting the dendrogram.

```
File: eg.out

      21       3.796E+00       4.160E+00
      22       4.160E+00       4.559E+00
  Help - Linear vs. geometric scales

 Dendrograms are output using both a linear and geometric scale for
 the relative distances between members of a cluster. This is done
 because if the range of relative distances is very large, the plot
 algorithm gets "confused" when drawing the left side (where the
 relative distances are at a minimum) of the dendrogram using a
 linear scale. When the range of relative distances is large and a
 linear scale is used, there is too much detail on the left side of
 the dendrogram for the algorithm to deal with.                          24 25

 When the distance range is large (> than two orders of magnitude)
 the dendrogram plotted on a geometric scale will provide a better
 representation of the clustering.

                                             F2 - Continue
                                             F2 - Continue              24 25

 F1-Help F2-Exit Movement Keys: Home, End, PgUp, PgDn, Up and Down Arrows
```

Press F2 to continue and F2 again to exit the program.

Using the dendrogram (the entire output file is presented in the next section) one can see that the two littoral stations (ID numbers 1 and 5) are very similar and are clustered together in the first stage. The inlet station (ID number 4) is very different from all of the other stations and is only grouped with the rest at the last stage. With this information it may be possible to reduce sampling effort at this reservoir by sampling only one of the two littoral stations currently being sampled.

Example Output File

```
Cluster Analysis

     EAU GALLE
```

Title provide in input data set

```
     ID
     NUMBER       ENTITY

       1          STA10
       2          STA20
       3          STA30
       4          STA50
       5          STA60
```

ID numbers associated with entity names

```
     Average Linkage Method used for clustering
```

Method used

Stage	Clusters Joined		Distance
1	1	5	6.080E-01
2	1	3	1.219E+00
3	1	2	3.191E+00
4	1	4	6.000E+00

"History" of the clustering

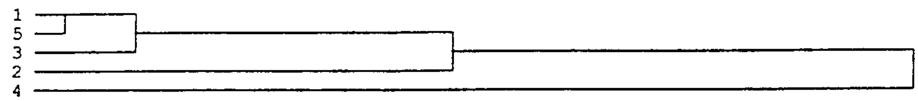
```
     The distances are segmented into the following
     classes for the Linear dendrogram
```

CLASS	LOWER BOUND	UPPER BOUND
1	6.080E-01	8.237E-01
2	8.237E-01	1.039E+00
3	1.039E+00	1.255E+00
4	1.255E+00	1.471E+00
5	1.471E+00	1.686E+00
6	1.686E+00	1.902E+00
7	1.902E+00	2.118E+00
8	2.118E+00	2.333E+00
9	2.333E+00	2.549E+00
10	2.549E+00	2.765E+00
11	2.765E+00	2.981E+00
12	2.981E+00	3.196E+00
13	3.196E+00	3.412E+00
14	3.412E+00	3.628E+00
15	3.628E+00	3.843E+00
16	3.843E+00	4.059E+00
17	4.059E+00	4.275E+00
18	4.275E+00	4.490E+00
19	4.490E+00	4.706E+00
20	4.706E+00	4.922E+00
21	4.922E+00	5.137E+00
22	5.137E+00	5.353E+00
23	5.353E+00	5.569E+00
24	5.569E+00	5.784E+00
25	5.784E+00	6.000E+00

The range in distance between the last stage and the first stage of the clustering is divided into 25 equal classes for displaying the dendrogram.

Dendrogram displayed using a linear scale

```
                        Linear Scale
ID   1  2  3  4  5  6  7  8  9 10 11 12 13 14 15 16 17 18 19 20 21 22 23 24 25
```



```
ID   1  2  3  4  5  6  7  8  9 10 11 12 13 14 15 16 17 18 19 20 21 22 23 24 25
                        Linear Scale
```

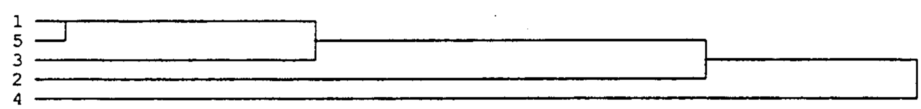
```
The distances are segmented into the following
classes for the Geometric dendrogram
```

CLASS	LOWER BOUND	UPPER BOUND
1	6.080E-01	6.663E-01
2	6.663E-01	7.302E-01
3	7.302E-01	8.003E-01
4	8.003E-01	8.770E-01
5	8.770E-01	9.611E-01
6	9.611E-01	1.053E+00
7	1.053E+00	1.154E+00
8	1.154E+00	1.265E+00
9	1.265E+00	1.386E+00
10	1.386E+00	1.519E+00
11	1.519E+00	1.665E+00
12	1.665E+00	1.825E+00
13	1.825E+00	2.000E+00
14	2.000E+00	2.191E+00
15	2.191E+00	2.401E+00
16	2.401E+00	2.632E+00
17	2.632E+00	2.884E+00
18	2.884E+00	3.161E+00
19	3.161E+00	3.464E+00
20	3.464E+00	3.796E+00
21	3.796E+00	4.160E+00
22	4.160E+00	4.559E+00
23	4.559E+00	4.996E+00
24	4.996E+00	5.475E+00
25	5.475E+00	6.000E+00

The range in distance between the last stage and the first stage of the clustering is divided into 25 classes using a geometric scale.

Dendrogram displayed using a geometric scale

```
                      Geometric Scale
ID   1  2  3  4  5  6  7  8  9 10 11 12 13 14 15 16 17 18 19 20 21 22 23 24 25
```



```
ID   1  2  3  4  5  6  7  8  9 10 11 12 13 14 15 16 17 18 19 20 21 22 23 24 25
                      Geometric Scale
```

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Sampling Design Software consists of four interactive programs (DECMATRX, VARCOM, ERROR, and CLUSTER) that assist users in the development of water quality sampling programs. The user's manual describes data entry and program use and provides examples. Example screen displays are also provided.

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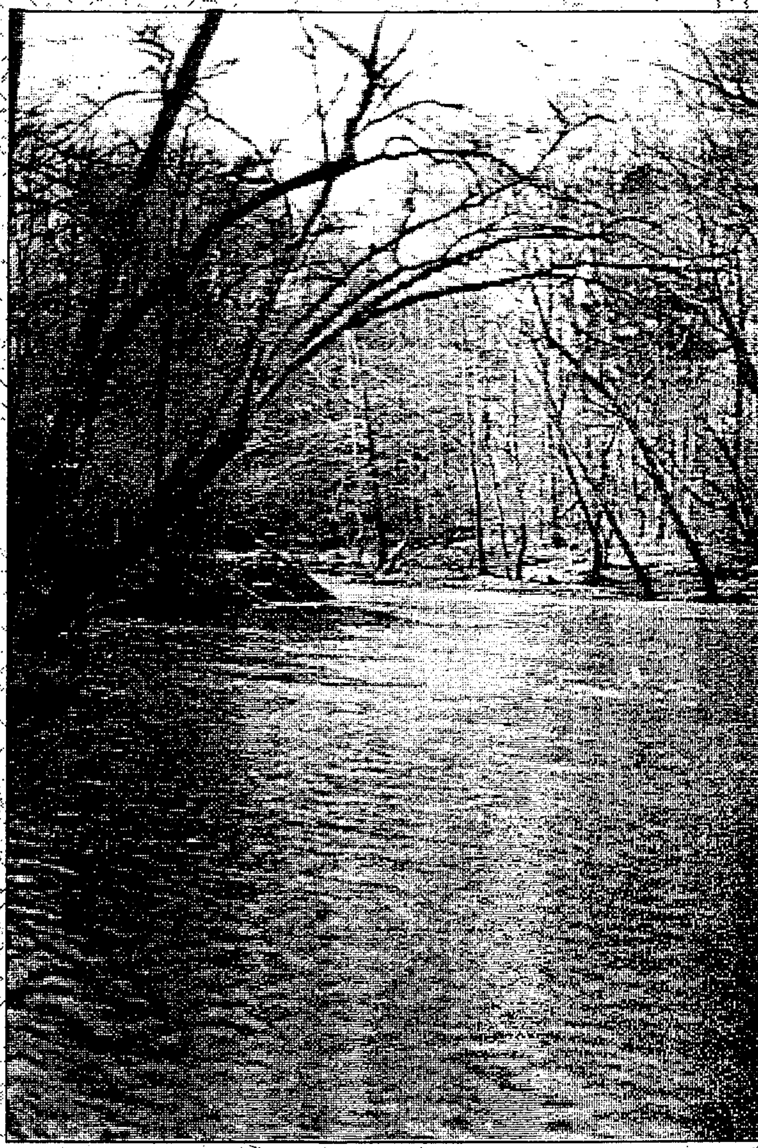
Appendix 2.2A

THE NONPOINT SOURCE MANAGER'S GUIDE TO WATER QUALITY AND LAND TREATMENT MONITORING

The Nonpoint Source Manager's Guide to Water Quality and Land Treatment Monitoring



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Disclaimer

Abstract

The purpose of a nonpoint source (NPS) land treatment project is to restore or protect the beneficial use or ecological integrity of a water resource. Watershed and water quality monitoring may be required to document the sources and impacts of pollutants and to track the effectiveness of their control. Efficient monitoring helps to document those changes in water quality variables and land treatment directly related to project objectives and activities. Monitoring to support the manager's information needs is a step-by-step process that requires documentation of the problem, analysis of project objectives, determination of approach, and development of a design before monitoring begins. This guide was written to help managers oversee a water quality monitoring project. This guide discusses monitoring to evaluate current conditions, to identify the water quality problem, to detect trends and impacts, and to document water quality improvement associated with land treatment. It also provides guidance on use of existing data in monitoring program design. For formulating an estimate of the monitoring program budget, this guide provides options in the monitoring approach based on level of detail and presents relative costs for some procedures.

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Important Messages

- The monitoring program should be based on clear management objectives.
- Careful investigation and documentation of the water quality problem pays off by increasing monitoring efficiency and value of results.
- Pollutant load monitoring has a high information value, but the procedure can be expensive.
- Periodic evaluation of selected variables is the most direct route to an answer on project impact. However, factors that are not related to land treatment efforts, such as watershed and system inertia, typically confound the detection of short-term trends.
- Biological monitoring and habitat evaluation can be meaningful, cost-effective approaches for assessing resource condition and project impact.
- Monitoring a treatment and a control site before, during, and after land treatment improves the chances of detecting trends or impacts.
- Monitoring a treatment and a control site in a paired watershed design also improves the chances of meaningful results.

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Development and implementation of a monitoring program that supports a narrowly defined objective increases the likelihood that results will be relevant and useful. A thorough understanding of the water quality problem, monitoring objectives, and expected results will help the manager make informed decisions while overseeing the total water quality project.

1

Overview of Monitoring Program

Introduction

Audience and Purpose. This monitoring guide was written to help managers oversee a water quality monitoring project. It is intended to be used as a simple framework to assist managers in developing a program for nonpoint source (NPS) pollution monitoring. Defining management objectives and documenting the water quality problem are crucial to building a successful project. Rarely does a monitoring program yield meaningful results without clear directions from carefully developed objectives and a thorough investigation of the water quality problem.

In addition to establishing the framework for monitoring, this guide was also meant to be used by the managerforgeneral reference. This guide provides enough detail so anew manager can use it as a training tool to improve his or her knowledge forcommunication with thescientist orstatistician. Theguideshould beconsulted periodically throughout the implementation of the monitoring program to check for deficiencies or the need for reallocation of effort.

Monitoring. Monitoring is the best method for evaluating water quality and its response to land treatment and other factors. Development and implementation of a monitoring program that supports a narrowly defined objective, such as problem identification or trend detection, increases the likelihood that results will be relevant and useful.

Water quality problem identification monitoring should seek first to specify pollutants and conditions responsible for the impairment to the designated use. Once the water quality problem is identified, the severity of the problem can be assessed. Clearly identifying the specific pollutant and assessing the problem assists land treatment staff in identifying critical areas and targeting BMPs.

Water quality monitoring is essential for determining project results and evaluating the effectiveness of land treatment. Adequate and effective land treatment and water quality monitoring for NPS pollution control projects are required to:

- document progress towards water quality goals;
- determine needs for further treatment;
- maintain the interest of project participants and staff;
- develop and transfer technology;
- reduce the number of inconclusive studies conducted;
- assure credibility; and
- address increasing information needs.

The manager should become familiar with the essential features of an effective monitoring program. A thorough understanding of the water quality problem, monitoring objectives, and expected results will help the manager make informed decisions and oversee the total water quality project. Because the manager may be the only person involved in the monitoring program who has a big-picture perspective of the overall program, he or she plays a key role in sustaining a coordinated monitoring program that is effective for its intended purpose.

The Relationship Between Management and Monitoring

Management objectives may include restoring or protecting the uses of a water resource or improving its ecological condition. In turn, a monitoring objective must be related to the management objective and defined so sampling will support the information needs of the manager. The monitoring objective specifies the approach for monitoring a water quality variable, measuring pollutant loading rates, or evaluating other measures of ecological integrity. Monitoring can document pollutant sources or impacts or can help to justify the expenditure of private or public funds on remediation or protection. Nonpoint source monitoring generally employs a fixed station network with long-term systematic sampling to evaluate factors important to management.

The Differences Between Point and Nonpoint Source Pollutant Monitoring

Nonpoint source monitoring stations should be located downstream and near major pollution sources, directly below areas where targeted and comprehensive land treatment is planned, because subtle impacts are difficult to detect. More stations or a more detailed monitoring program may be required to assess diffuse NPS pollutants or combinations of PS and NPS pollutants.

A different approach to planning and design is generally required for NPS monitoring compared with traditional point source (PS) monitoring. Typically PS pollutants are diluted by the receiving stream such that high stream flows result in low pollutant concentrations. Point source discharges may also vary with the industrial process, time of day, and day of the week. Runoff and other land-based pollutant transport mechanisms may have limited effect on PS pollutants but in-stream physical, chemical, and biological processes remain important. Therefore, stations for PS load monitoring are generally located near and downstream from the known outfall.

Because runoff and snowmelt drive NPS pollutant transport, the high variability of the process reflects such factors as weather, land use, and watershed characteristics. Runoff and high stream flows can result in high pollutant concentrations. Nonpoint source pollutant concentrations vary by source type (land use), location of source, transport mechanisms, and they are influenced by trapping in the watershed and in-stream processes. Careful placement of monitoring stations is required to account for these factors. Nonpoint source monitoring stations should be located downstream and near major pollution sources, directly below areas where targeted and comprehensive land treatment is planned, and where improvements in water quality due to land treatment are expected to be greatest, because subtle impacts are difficult to detect.

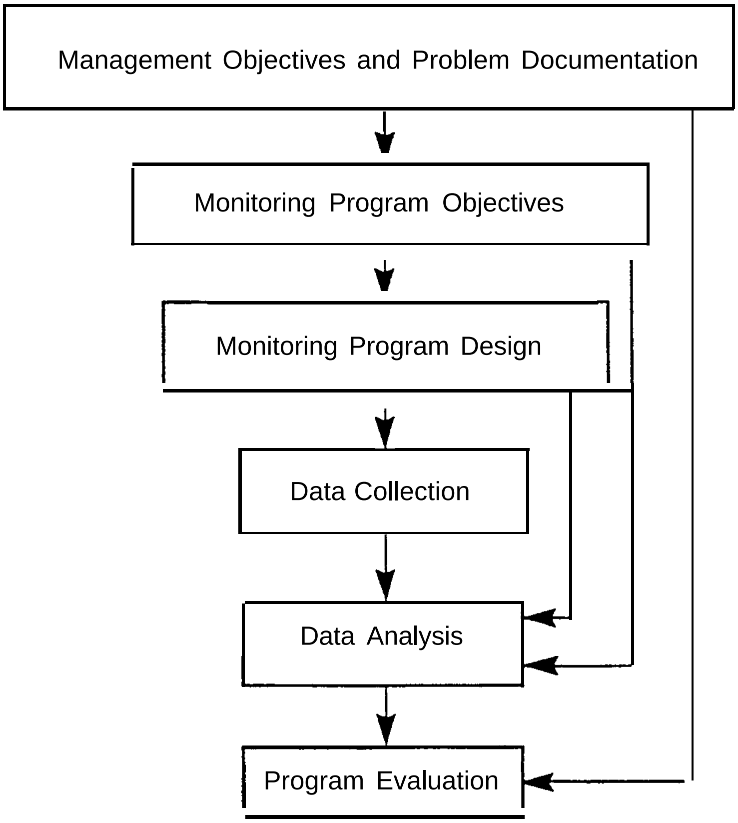
Upstream or ground water fluxes may affect the placement of both PS and NPS monitoring stations. More stations or a more detailed monitoring program may be required to assess diffuse NPS pollutants or combinations of PS and NPS pollutants. Measuring the effect of pollution control, therefore, requires careful assessment of major pollutant sources and contributing factors.

The most effective approach to reduce NPS pollution is to target and treat critical areas with a system of best management practices (BMPs). Critical areas are sources in the watershed that affect the availability or mobility of pollutants or areas which, when treated with BMPs, have the greatest potential to improve the ecological condition of the water resource. Maas et al. (1985, 1987) provide guidance on critical area determination for improving water quality.

Overview Of a Monitoring Program

As suggested in Figure 1.1, the monitoring program supports management objectives and begins by documenting the water quality problem. From problem documentation, monitoring objectives can be developed as the basis for the monitoring program design. Data collection is the action stage that proceeds according to program design. Data analysis employs statistical methods to summarize the findings and to determine trends or effects due to treatment. Program evaluation provides guidelines to assess and document findings. We suggest this order of the process to improve efficiency and results. For example, monitoring should not begin before the management and monitoring objectives are set and the monitoring design and analysis plans are in place. Data analysis at regular intervals throughout the monitoring program is part of a feedback loop that provides timely information for making refinements in monitoring program

Figure 1.1 The development of a nonpoint source control monitoring program.



Monitoring should not begin before the management and monitoring objectives are set and the monitoring design and analysis plans are in place.

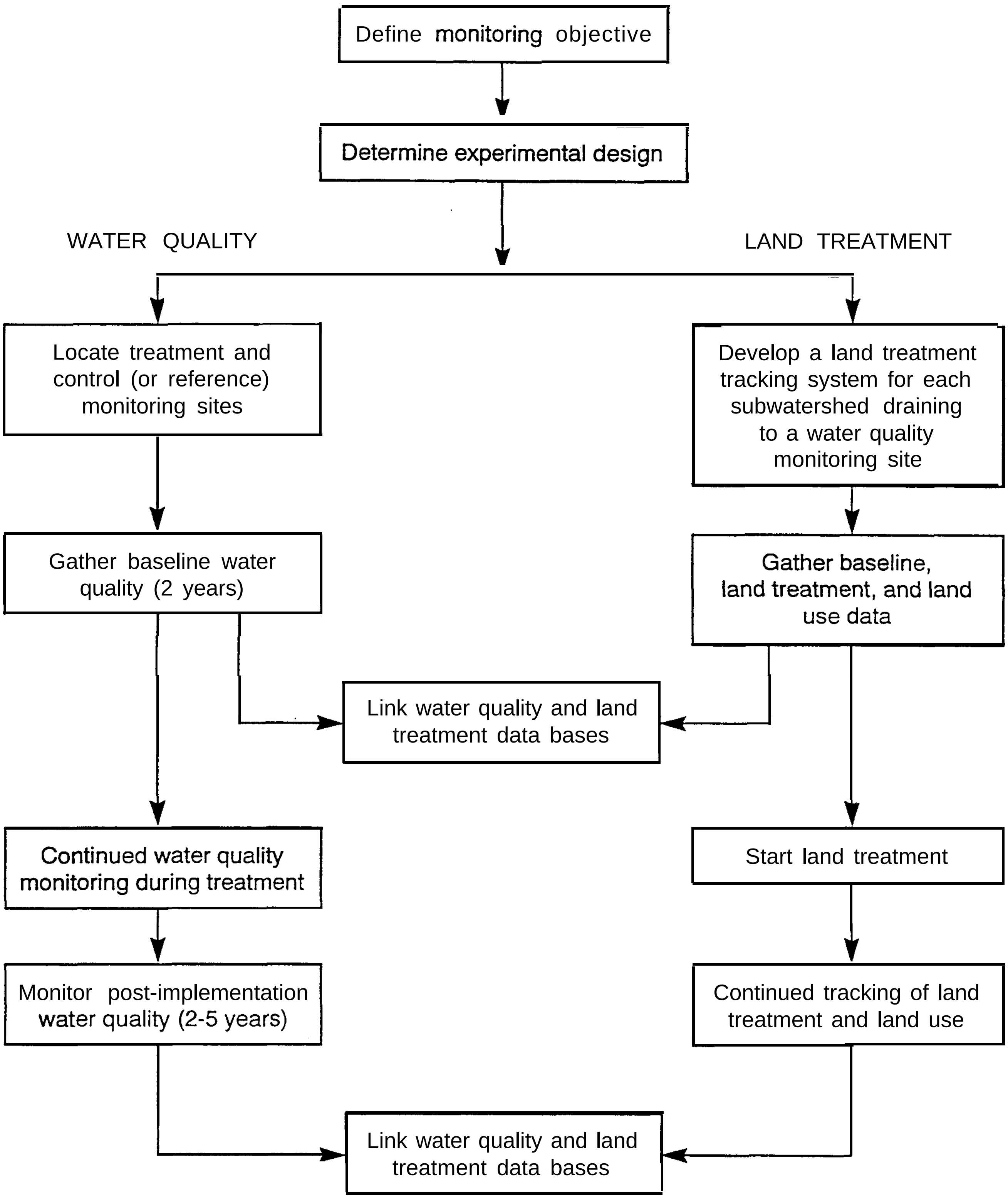
design or objectives. In the program evaluation stage the feedback loop may serve to update management objectives and the problem statement.

Monitoring program design is the most detailed part of the overall monitoring program. Design provides the format for most water quality and land treatment monitoring activities. Water quality and land treatment monitoring must be coordinated to maximize the chance of meaningful results. In order to provide the manager with a sense of the nature of the coordination needed, an overview of monitoring program design is provided in Figure 1.2.

Monitoring program design, as shown in Figure 1.2 and discussed in chapter 4, begins by defining the monitoring objective. Once the objective is defined the experimental design (e.g., upstream/downstream and pre- and post-BMP, and paired watershed) is determined. Based on the experimental design, separate but coordinated parallel water quality and land treatment activities are specified.

The next activity is to locate water quality monitoring sites and then develop the land treatment tracking system for each subwatershed that drains to a water quality monitoring site. Baseline water quality and land treatment datashould be collected for two years prior to treating critical areas in the watershed with BMP systems. During baseline datacollection and at regular intervals throughout the monitoring program, the water quality and land treatment data bases should be linked (see section 4.7). Water quality and land treatment monitoring continue on a parallel course until monitoring and management objectives have been met. Prior to final analysis the water quality and land treatment data bases must be linked to evaluate project effectiveness.

Figure 1.2 Land treatment and water quality monitoring program design.



Water quality and land treatment monitoring begins by defining management objectives for the water resource. The water quality problem must also be documented with reliable data. Management objectives andproblem documentation are necessary for further management actions such as development of a monitoring program. A monitoring program yields meaningful results only with clear direction from carefully developed objectives and a thorough investigation of the water quality problem.

2

Management Objectives and Problem Documentation

Monitoring supports project objectives by providing information to track progress and to evaluate project effectiveness. Management objectives are usually related to protecting a threatened water resource, restoring designated water body use, or achieving a water quality standard.

Problem identification monitoring uses a site-specific plan to identify pollution sources and impacts during both baseflow and storm conditions.

Carefully defining and documenting the water quality problem is one of the most important steps for NPS pollution control and water quality monitoring. An effective approach is to implement a problem identification monitoring program lasting six to 18 months. Problem identification monitoring uses a site-specific plan to identify pollution sources and impacts during both baseflow and storm conditions; monitoring may be most effective during the seasons of greatest pollutant loading (spring runoff, snowmelt) and during the season when impairments are noted (algal blooms, shellfish closing).

For documentation, in-project use, and communicatingwith the public, a problem statement should be written. The problem statement summarizes the results of a thorough effort to investigate and document present or potential negative impacts on water quality. The statement draws together the history, causes, and significant contributing factors affecting the quality of the water resource, including pollutants, their sources and timing, background fluxes, and habitat and site factors that influence the **problem.** The statement also establishes the basis for the implementation of the land treatment program.

Accurate and complete problem assessment is instrumental to achieving water qualify goals.

The level of problem assessment depends on the nature of the impairment of the water resource, diversity of pollutant sources, hydrologic transport system, and size of the watershed. Accurate and complete problemassessment is instrumental to achieving water quality goals. An evaluation of the problem and land use upgradient from the water resource provides much of the information to specify the monitoring program objective.

2

Nonpoint source pollution control monitoring typically suffers from lack of focus, often resulting in inconclusive studies. Successful monitoring programs require long-term systematic measurement of both primary and explanatory variables related to the management objective. The project manager should work with the water quality and land treatment agency to assure that program objectives are narrow enough to address the management objective and the water quality problem.

3

Monitoring Program Objectives

The monitoring program objective is developed to address the water quality problem and the overall management objective. Objectives should be comprehensive, non-overlapping, and relevant, but defined narrowly enough to provide focus. A substantial amount of time may be necessary to specify monitoring objectives, but the initial effort should improve long-term program efficiency. This chapter discusses biological, habitat, physical, and chemical variables for NPS pollution control monitoring. The discussion of variables is detailed because variable selection is important for the development of the monitoring objective. The level of detail needed for several types of monitoring objectives is also discussed. Chapter4 provides examples of how to formulate aspecific monitoring objective based on the monitoring design.

The Water Quality Monitoring Approach

Level of detail and whether the monitoring program will focus on trends in variables, pollutant loads, or other attributes will be discussed below. The monitoring approach must also consider the minimum detectable change (MDC) required to show a significant difference or trend.

Level of Detail

Objectives and budget dictate the level of monitoring detail. Levels differ primarily in the skill, intensity, time, resources, and equipment necessary. Different levels provide the manager with options based on resources and objectives. In this guide, information on levels is cumulative such that the discussion of level II builds on the information obtained for level I. Examples of level I and level II monitoring objectives are given in Table 3.1.

Level I monrtoring is generally most useful to evaluate current water quality conditions and to document water quality problems.

Level I is the basic, minimum level of monitoring, at relatively low cost, for assessing conditions and problems or determining trends in easily measured variables. Level I monitoring is generally most useful to evaluate current water quality conditions and to document water quality problems. However, despite their low cost, a level I analysis is sufficient and defensible under the right conditions. If the objective is to evaluate current conditions, the analysis should focus on anoverall assessment of theecological condition, beginningwith habitat. Biosurveys and physical/chemical analyses may also be needed to determine biological potential. Problem documentation or problem identification monitoring is an essential first step of a watershed management program. Specific pollutant constituents should be documented as causing the water quality problem. Clearly documenting specific pollutants causing the problem and monitoring for their impact helps to avoid monitoring pollutants which are not vitally important. Carefully documenting the problem assists project staff in clearly defining land treatment and monitoring objectives of the project.

Table 3.1 **Monitoring Objectives and Level of Detail.**

General Motiitoring Objective	Level I	Level II
Evaluate Current Conditions	X	
Problem Documentation	X	
Standards Violations	X	
Trend Detection (variables)	X	X
Impact Assessment		X
Causality		X
Load Monitoring (loads)	(X?)	X
Trends		X
impact		X
Causality		X

Standards violations is an important objective; however, because a careful examination of this topic is beyond the scope of this guide, standards violations will not be discussed herein. For level I trend detection, a large and persistent change in a variable with respect to background is required to evaluate program effectiveness.

Level II trend detection of load monitoring is usually best suited to demonstrate a trend, impact, of a cause-and-effect relationship between a management action and a response variable.

Level II monitoring is more intensively detailed, with more comprehensive data collection and higher cost. Compared to level I, level II may involve a higher sampling frequency, sampling more variables, sampling variables that are expensive to analyze, or sampling at a greater number of locations. Level II trend detection or load monitoring is usually best suited to demonstrate a trend, impact, or a cause-and-effect relationship between a management action and a response variable (Table 3.1). Load monitoring is used to measure the change in pollutant mass loading rate.

Monitoring trends in pollutant concentrations or in biological/habitat variables may be the most direct route to an answer on treatment program effect on designated use. Sensitivity may be low if there are not enough samples or explanatory variables. Load monitoring is used to determine the pollutant mass loading rate. Load monitoring requires more frequent samples as compared to determining trends in concentrations or biological/habitat variables. Often automatic sampling is required, which increases equipment and analysis costs. Discharge and concentration measurements are essential for load calculations. Load monitoring usually requires a level II monitoring effort.

Load monitoring may be useful to quantify the effect of land treatment at a subwatershed or project area scale. Load monitoring may have informational value if the response in the receiving water is expected to be slow and improvements are not likely to occur for many years. For instance, measuring the effectiveness of animal waste control on one tributary of a large lake with several important tributaries (such as Lake Champlain) may require monitoring phosphorus loads, since in-lake monitoring is unlikely to measure the change in the mean phosphorus concentration or trophic state measures of the lake.

A pollutant budget may be a useful decision tool to determine variables and frequency of monitoring and expected information from load monitoring. The budget accounts for a mass-balance of a pollutant and water input by source, including ground water and atmospheric deposition, all output, and changes in storage. The budget may show the magnitude and relative importance of controlled and uncontrolled sources (atmospheric deposition, resuspension from sediments, streambank erosion point sources, septic tanks).

Minimum Detectable Change and Making Program Decisions

The most defensible measure of land treatment project performance is a well-designed and well-implemented monitoring program that examines statistically the relationship between the project's pollution control activities and a change in water quality. Planning an adequate monitoring program considers extent of treatment, relative magnitude of sources, system variability, and the minimum detectablechange(MDC) (Spooneret al. 1987a) needed in a water quality variable to document a statistically significant change.

An analysis of historical data to determine the MDC may serve to estimate the amount of time needed (number of years or seasons) to show a significant trend. For a monitoring program underway, the variability and trends observed may be calculated at regular intervals to determine if the sampling program can realistically meet thequantitative objectives for trend detection and when to proceed with sampling or when to stop.

Variable Selection

Monitoring for each objective requires a different approach. Monitoring to evaluate current conditions should focus on critical variables related to designated water body use and those variables expected to respond to management activities. For violations of standards, the choice of variable is specified by the standard. To assess ecological integrity, monitor a set of variables that show how an ecosystem compares to a control or one that has a composition, structure, and function essentially unimpaired by human activities (Karret al. 1986). For trend detection, the response variable and explanatory variable must be carefully selected to show treatment effect and account for changes in system variability.

Variability and Explanatory Variables

Sources of variability include climate, weather, watershed characteristics, and human activities. Variability may be in daily, seasonal, year-to-year patterns, or have some random component. Measuring and accounting for sources in variability increases monitoring sensitivity and reduces the MDC.

Explanatory variables such as those in Table 3.2 can account for the influence of climate, hydrology, land use, and other factors. Land treatment variables are also important as explanatory variables. The appropriate explanatory variable or set of variables is directly related to the primary variable/pollutant of concern. Incorporation of explanatory variables into the study increases the analyst's ability to isolate true water quality trends due to land treatment. For example, antecedent precipitation, stream discharge, or water table depth may be used to quantify the hydrologic factors affecting changes in stream phosphorus concentrations. Stream discharge and estuarine salinity may be used to explain either increases in fecal coliform counts due to transport in runoff or decreases due to die-off from high salt concentrations. Temperature determines solubility of dissolved oxygen (DO), making it an important explanatory variable for DO. Biochemical oxygen demand (BOD) can deplete DO and may be an explanatory variable for a DO monitoring program. Both suspended solids and chlorophyll a can affect Secchi depth transparency, making them appropriate explanatory variables. For monitoring trout abundance, one explanatory variable may include the percent fines in substrate sediment because substrate composition affects reproductive success. Similarly, the area of undercut banks is a measure of hiding cover to escape from prey. Relevantevents that could affect monitoring results, such as droughts, floods, and storms, or fishery management and harvest, should be tracked and documented. The last row of Table 3.2 lists a generic primary water quality variable y and a land treatment variable x to show that land treatment variables should be measured along with the water quality explanatory variables.

Table 3.2. Example Primary Variables and Explanatory Variables for Trend Monitoring.

Primary Variable	Explanatory Variable
Total phosphorus	Antecedent precipitation, stream discharge, water table depth
Fecal coliform	Stream discharge, estuarine salinity
Dissolved oxygen	Water temperature, biochemical oxygen demand
Secchi depth	Suspended solids, chlorophyll a
Trout abundance	Percent fines in sediment, area of undercut banks
Water quality variable y	Land treatment variable x

One approach to identifying appropriate explanatory variables is through a statistical analysis of a historical data set. Explanatory variables should be selected because they measure factors in the ecosystem that are thought to effect the primary variable(s) of concern. A check should be made to assure appropriate selection by verifying that the selected explanatory variable and the primary variables are statistically correlated (e.g., using linear regression techniques).

For some monitoring programs, variables or metrics may be summarized and combined into an index. An index contains less information and therefore less explanatory power than the original data, but it may be more easily used and understood by the public or the decision-makers. Indices are chosen for their ecological meaning and ability to summarize information on community structure, function, or response to pollution.

Monitoring to Detect Ecosystem Impacts

Monitoring decisions are most efficient if based on the watershed as the functional unit of the ecosystem. The hydrologic basis is particularly important for assessing the impact of land use, BMPs, and runoff-driven NPS pollution. Even PS problems require watershed information to determine their impact.

Chemical vs Biological Monitoring

Project activities are expected to affect physical, chemical, and biological variables; therefore, an integrated approach that accounts for ecosystem components is desirable. The timing and magnitude of response to remediation is generally difficult to estimate. Monitoring results to track compliance with water quality standards are unlikely to be directly applicable to ecological assessments. In addition, biological monitoring cannot identify specific contaminants or their concentration. Therefore, an integrated physical, chemical, and biological monitoring approach may be necessary to document ecosystem impacts.

Monitoring Costs

Data on the cost of monitoring is very limited, but the cost of water quality monitoring can vary significantly for many reasons. One way to report cost is to provide an estimate of the number of hours required to perform a task for a monitoring event at a single station.

Lenat (1988) reports that for macroinvenebrare monitoring, qualitative sampling requires at least one experienced biologist on the team, and six person-hours in the field and four hours for identification with no time required for laboratory picking. For two kick samples, 1.5 hours were required to collect the sample, nine hours to pick the sample, and 10 hours for identification.

Plafkin et al. (1989) in the USEPA Rapid Bioassessment Protocols for Use in Streams and Rivers have found that sampling riffles, runs, and pools at each site, with effort proportional to each of these major habitat types, requires generally one-two hours. Gear, size, and complexity of the site are factors that affect sampling time. Times were not given for sample processing.

Costs forvarious types of physical, chemical, and biological monitoring have been reported by Ohio EPA (1989). Since the cost of labor is difficult to estimate and it is unclear if labor costs are equal for different procedures, comparisons with the figures above are difficult to determine. For a basic lake monitoring protocol, Wedepohl et al. (1990) provide a table of variables, general sampling information, and a general cost estimate for the sampling program.

Land Treatment and Land Use Monitoring

Land treatment and land use monitoring are used to track where and when BMPs are implemented and how well they are adhered to. The purpose is to track treatment strength in time and space. Watershed management variables (e.g., land treatment, land use) are explanatory variables as discussed in section 3.1.4.

Monitoring BMP implementation and land use in critical areas is necessary to track treatment progress. Also noncritical area treatment and land use can be important and should be monitored, but probably at a lower level of effort.

Land treatment and land use monitoring should relate directly to the pollutants or impacts monitored at the water quality station. Since the impact of BMPs on water quality may not be immediate or implementation may not be sustained, information on relevant watershed activities will be essential for the final analysis.

Level I Land Treatment and Land Use Monitoring

To track land treatment implementation, explanatory variable(s) must be selected that will accurately reflect the desired land treatment effect. Land treatment impacts can be expected in three areas:

- source area (field, confined animal operation, forested tract, urban area),
- delivery area between source and receiving ecosystem,
- direct ecosystem effects.

For monitoring source *urea,* the number of acres treated and untreated with BMP systems should be tracked. It is not correct to simply add up the total area treated for each practice. For instance, terraces and nutrient management practices may be applied on a lo-acre field. The total number of acres treated would be 10 because the single source area was treated with two BMPs. The acres treated should not be double-counted as 20.

If no tillage (NT) or crop residue management is selected for reducing erosion, the number of critical acres that have 30% area! residue coverage (or some other standard) prior to planting should be tracked. Residue varies greatly for NT, and tracking residue density is a better measure than estimates based on type of equipment used for tillage.

For phosphorus problems due to animal waste, both the number of animal units in critical areas and the tons of manure produced and treated should be tracked. Both structural and management systems should be monitored to determine if the waste management system is being operating properly and that manure applications and fertilizer use matches an approved nutrient management plan. Waste storage facilities may help for proper timing of waste applications to fields but they do not prevent farmers from over-applying nutrients.

For practices designed to reduce *pollutant delivery*, the number of acres treated by the practice may be reported. Buffer strips, field borders, and sediment basins are installed at the edge of the source area to reduce pollutant delivery. The acres treated by these practices refer to the field watershed or source area contributing runoff to the BMP. If the source area is also treated with conservation tillage or nutrient management it is important not to double-count the total acreage treated by combined source area and pollutant delivery reduction BMPs.

For level I land treatment and land use monitoring, watershed and subwatershed summaries should include acres served by BMPs on an annual basis.

Direct ecosystem effects include the activities in the riparian area and in the resource. The extent of cattle grazing or other animal use of the stream should be tracked. Crossing the stream with agricultural or logging equipment can have an important effect, and these events should be documented.

The land use in critical and noncritical areas should be known. Example land use/ treatment variables are included in Table 3.3. For each variable the extent of activity and location of activity are important.

For level I land treatment and land use monitoring, watershed and subwatershed summaries should include acres served by BMPs on an annual basis. These annual drainage area summaries should emphasize explanatory variables that relate directly to the pollutant or condition of concern. Aerial photographs may be useful to track land use and BMP implementation in rural areas. Also aerial photographs and city planning maps may be used to categorize urban land use based on percent impervious area.

Table 3.3 Land Use and Land Treatment Explanatory Variables

Agriculture

- ❑ The animal unit density per subwatershed
- ❑ Area receiving manure per subwatershed and amount and timing of application
- ❑ Tons of manure treated with BMPs
- ❑ Area receiving commercial fertilizer and application rate
- ❑ Stream miles with direct livestock access
- ❑ Area of each crop
- CI Rotation and tillage for cropping systems
- ❑ Area receiving pesticides, pesticides used, and application rate

Forestry

- ❑ Area clearcut
- ❑ Area harvested during high soil moisture conditions
- ❑ Area prepared for planting
- ❑ Extent of road building (distance and slope)

Urban

- ❑ Land use (e.g., residential, commercial, industrial)
- ❑ Area under construction or land-disturbing activities
- ❑ Area with storm sewers not being treated

Individual source areas should be tracked for /eve/ II /and treatment monitoring.

Level II Land Treatment and Land Use Monitoring

The frequency of level II data gathering should be decided for land treatment and land use monitoring. Intensive management practices for pesticides or nutrients/ animal waste require monitoring at a greater frequency compared with monitoring the installation of structural practices such as manure storage, roofing, grassed waterways, or terraces. Monitoring cropping type and rotation, tree planting, and Conservation Reserve Program areas may be less frequent.

Individual source areas should be tracked for level II land treatment monitoring. For agricultural data a farm operator survey should be developed and used to gather data. Coffey et al. (1991) and Meals et al. (1991) have developed farm operator surveys for level II land treatment tracking. Farmers should be interviewed at least on an annual basis to track cropping system, animal operation and waste management variables. Source area aggregation for the analysis with water quality data should be at the subwatershed level above the appropriate water quality monitoring station.

The use of a geographic information system (GIS) is essential for level II land treatment tracking. GIS systems are available for the personal computer and some software packages are menu driven and are compatible with other packages.

Sampling Locations

Initially, the point of designated use or some location critical to ecological condition should be monitored. Samples should be reasonably representative of the volume of water that is most meaningful to address the monitoring objective.

3

The sample should be a good subset of the population of interest, unbiased by edge-effects or anomalies. Monitoring station selection is problem-specific, but some general station attributes for each type of water resource are suggested below.

Stream Monitoring Locations

Locating a stream monitoring station may be difficult since several factors influence placement.Theeffect oftributary pollutant loading, dilution effects, and lateral gradients should be considered. Point source pollutant influences can also impede NPS monitoring activities. Monitoring a stream reach (length of stream) with several stations may be necessary. However, a reach may be monitored by a single station if variability for the constituent of concern is low. Problem documentation monitoring during high and low flows at several locations will provide information on site variability and can serve as the basis to select stream monitoring stations.

Wetland Monitoring Locations

Wetland functions are highly interrelated and can be quite complex. Hydrology and hydraulic loading rates of pollutants are very important for evaluating wetland functions and response to management. Fixed stations are needed to evaluate changes in hydrology, pollutant concentrations, and biological variables (Hammer 1992).

Monitoring at the inlet and outlet are important if a pollutant budget or information on loading and wetland treatment efficiency are needed. Monitor pollutants at selected wetland stream channel stations and one or more nonchannel stations to characterize water quality gradients and patchiness within the wetland.

Water levels should also be determined for each water quality sample. The area1 extent of the permanent pool is an important measurement for wetland function and this usually varies with season and other factors.

Vegetation monitoring is important for assessing overall health and may be accomplished with an analysis of area1 photographs. Vegetation sampling includes monitoring along quadrants, transects, or bisecting the wetland at random locations.

Reservoir Monitoring Stations

A horizontal gradient of pollutants with high concentration near tributary headwaters decreasing to the outflow is common for reservoirs and their tributary arms. If it clearly reflects the condition of the designated use, a single station over the greatest depth may be the preferred sampling point. Additional stations may be located in the mainstem, down gradient of a major tributary, or over the deepest water in a tributary arm, depending upon designated use, additional management objectives, and knowledge of the pollutant budget. The depth of the sample is also a concern for managing depth of withdrawal from the reservoir to control tailwater quality.

Lake Monitoring Stations

For pelagic and profundal water column monitoring in a lake with low shoreline development (i.e., regular-shaped lakes, not reservoirs), a single station at the deepest part of the lake may be sufficient for assessing whole lake conditions. Generally few stations are needed for lakes that mix continuously and do not stratify. For a lake with more complex morphometry, lake mixing and circulation patterns should be evaluated to determine if individual bays, or subbasins of significantly different mean depth, are hydrodynamically distinct. Based on the location pollutant influx and areas of use or critical habitat conditions, the minimum number of stations, with one at the deepest part of the subbasin(s) of interest, should be chosen.

Basin mixing and variable type should be evaluated prior to determining sampling depth. For unstratified lakes (lakes with a uniform temperature from surface to sediments), a surface (one may be enough), mid-depth, and near-bottom samples may be appropriate. An alternative protocol is to take an integrated sample from just above the bottom to the surface. For stratified lakes, one surface sample, one in the metalimnion, one at the mid-depth of the hypolimnion, and one near the bottom of the sediments may be required (Wedepohl et al. 1990).

Water Quality Monitoring Variables

Level I can be used for evaluating current conditions, problem documentation, frequent violations in standards, trend detection in some cases, or measuring large impacts. Many state agency monitoring programs address level I objectives.

Trend detection for level I can be performed when background variability is low and the level of treatment, pollutant control, and restoration accomplishments are high. The watershed should be relatively small (e.g., less than approximately 30,000 acres) and all or nearly all of the critical area pollutant sources must be treated for a sustained period. Level II monitoring, which is more detailed to quantify and explain greater variability, should be used otherwise. If the objective is to determine an impact or a cause-and-effect relationship, then level II monitoring is needed.

Level I trend detection employs the use of relatively inexpensive methods, such as measuring Secchi depth, grab sampling for chemical concentrations, and measuring simple habitat variables. On the other hand, these measurements reflect a complex of factors that may be difficult to interpret. It is helpful to measure appropriate explanatory variables to account other sources of variability.

Level I Water Quality Monitoring

The following subsections describe variables common to level I monitoring of or biology, habitat, and chemical/physical characteristics of a water resource.

Biological Monitoring

Biological monitoring provides the most direct measure of use attainment related toaquaticlife. Organisms respond toanaggregateofstressfactors, includingthose not monitored by chemical or habitat protocols. Chemical monitoring and bioassay alone may fail to directly assess pollutant-induced degradation or partial restoration of biota. In addition, the public may understand the purpose or progress of biological monitoring more easily than other methods (Zaroban 1988).

However, biological monitoring may be relatively field-intensive and it requires a trained staff, with knowledge of local biota and their habitat, in order to obtain high information content and to maintain quality control. Sampling bias can make protocols subject to errors. The effects of fish stocking or other management activities may confound analysis of treatment effect and natural regeneration. Many techniques are available for biological data analysis but some require modification to considerregional and seasonal variability, including life cycle and behavioral aspects (Zaroban 1988).

Choosing the taxonomic group (e.g., fish, macroinvertebrates, etc.) should be based on monitoring program objectives and designated use. Selection may consider spatial and temporal variability, length of the life cycle, extent of the home range, and level of taxonomic expertise required for analysis. Other considerations include ease of sampling, cost, known ecology of taxa, economics ofimportance, or the ability of tissue to accumulate pollutants for bioassay (Hellawell 1986).

Advantages of monitoring more than one taxonomic group are discussed by Plafkin et al. (1989) in the USEPA Rapid Bioassessment Protocols for Use in Streams and Rivers. However, monitoring more than one taxonomic group could be redundant if objectives do not directly relate to the variables being measured. Several taxonomic groups are reviewed and relevant sampling issues are discussed below.

Coliform Bacteria

Coliform Bacteria. The transmission of waterborne diseases from both PS and NPS pollution continues to be a public health threat. Bacteria or other organisms should be considered in a monitoring program for water supply, contact recreation, and shellfishing where pathogens are a threat. The coliform group of bacteria is often found with other organisms that pose a more serious risk to health. Coliform bacteria are easily detectable and they are not generally present in unpolluted waters (National Academy of Sciences 1977). Fecal coliform (FC) and fecal streptococcus (FS) bacteria are found in the intestinal tract and feces of humans and other animals and may signal the presence of a pathogen such as *Salmonella* (Thomann and Mueller 1987).

Compared with other organisms, bacteria are highly variable. They have a relatively short lifespan, cells drift substantially, and cell counts can change rapidly due to changes in water quality. Bacteria should be monitored at the point of designated use, such as a water supply intake, shellfishing grounds, or recreation area for the duration and frequency specified by state or local standards. A health department should be consulted for pollutant source identification, monitoring, or interpretation. Laboratory quality control and quality assurance is essential for reliable coliform bacteria counts.

Explanatory variables for monitoring bacteria include: temperature, salinity, sunlight, predation, effectsof nutrients or toxins, and time of travel, distance from source, settling, or resuspension from sediments (Thomann and Mueller 1987).

Phytoplankfon

Phytoplankton. Screening-level monitoring may be done with careful, systematic observation. Periodic examination of algal taxa and biomass are basic techniques for lake monitoring. Some taxa are less desirable and indicate problems. Forexample, dominance by blue-green algae can be responsible for noxious blooms and surface scums. Excessive algal production reduces transparency and degrades lake quality.

Sampling gear decisions are based on the water column location where samples should be taken. A trap encloses a volume of water at a discrete depth. Traps may be preferred since nets are size-selective and smaller taxa will be missed. However, towing a net at a constant depth provides a horizontal sample while a vertical tow samples plankton from a given depth to the surface. Phytoplankton vary horizontally and vertically and throughout seasons. Therefore, care should be taken when sampling to avoid missing plankton concentrated in a thin strata.

Chlorophyll a concentration is a measure of algal biomass and trophic state. Chlorophyll a is expected to decrease if lake nutrient loading rates are decreased.

Although chlorophyll *a* may be quite variable, a well-developed monitoring protocol can achieve acceptable levels of error in estimating a mean growing season concentration or some other summary statistic. Potential explanatory variables to monitor include phosphorus, nitrogen, suspended sediment, or transparency, depending upon the dynamics of the problem and land use. The entire photic zone should be sampled when monitoring lake chlorophyll (Lind 1985; Wedepohl et al. 1990) through the use of a tube sampler or by combining samples from several depths. Monitoring a single station over the deepest point may be the most desirable and the most efficient. Samples may also be taken near the water supply intake or at any other point of designated use.

For lakes and reservoirs with complex morphometry, more than one station may be needed. An additional station may be selected to monitor a lake segment that functions hydrodynamically as an individual basin with regard to mixing and stratification. For reservoirs with a strong horizontal pollutant gradient, stations may be located along the main stem or near the confluence of a tributary arm and the main stem. Typically, however, adding a station does not contribute a significant amount of information beyond that which is obtained from the original station.

Macrophytes

Macropbytes. Nuisance stands of macrophytes in shallow areas of lakes and streams can interfere with a water supply intake, impair recreation, and decrease aesthetic enjoyment. Monitoring macrophyte species and area1 extent should occur at the same relative time of the growing season at a regular interval (e.g. every year or every other year).

Several techniques are available for macrophyte monitoring. Visual surveys or photographs taken in a standardized manner can be used to track the remediation or the gross extent of a problem. Harvesting, herbicide applications, and draw-downs should also be tracked for their effect.

Macrophyte mapping procedures can employ survey techniques. Langland and Pesacreta(l986) used sonar (fish locator mounted on a boat) to estimate the cross-sectional area of macrophytes along a transect. Geissler (1988) used areal photographs for macrophyte sampling.

Benthic Macroinvertebrates

Benthic Macroinvertebrates. Monitoring benthic macroinvertebrates can identify a problem or provide data to determine use attainability, assess trends, or determine impact. Macroinvertebrates are sensitive to natural and human disturbances and are an important food for fish.

Lenat (1988) and Plafkin et al. (1989), in the USEPA Rapid Bioassessment Protocols for Use in Streams and Rivers, provide methods for stream macroinvertebrate monitoring on natural substrates. Methods vary by the type and number of habitats (either multiple or single) sampled, skill required for field evaluation and laboratory identification, and data analysis. Riffle or riffle/run habitat surveys are the most common. Here a kick net (Lenat 1988; Piafkin et al. 1989), Surber sampler (Lind 1985), or some other device may be used to dislodge and collect benthos from an approximated or measured area of the riffle substrate.

Multiple habitat surveys include sampling riffles, streambanks, leaf packs, rocks and logs. Snags (fallen iogs and branches) may also be sampled in sandy streams (Lenat 1988). Sampling multiple habitats can collect more taxa and the technique is potentially more likely to detect subtle impacts, but care must be taken to follow a standard operating procedure.

In sand-bottom streams, snags and shoreline habitats may not support enough organismsforasample(i.e.generally greateror muchgreaterthan 100 organisms). Where natural substrate monitoring does not provide a sufficient collection, use of artificial substrate samplers such as Hester-Dendy multiplates (Hester and Dendy 1962)and rock-filled baskets (Henson 1965), anchored in thewatercolumn for a standardized period, may provide an adequate sample.

Many metrics (biological indices) are available for summarizing and reporting macroinvertebrate monitoring results. Metrics quantitatively describe community structure, function, or health. A computerized data base such as BIOS is useful for data storage, retrieval, and calculating metrics. Other metrics may be calculated using SAS (Plafkin et al. 1989). Recent studies (Resh 1988; Szczytko 1989; Davis and Lubin 1989) have evaluated macroinvertebrate data sets from California, Wisconsin, and Ohio to assess statistical properties such as metrics and the variability and distribution of macroinvertebrate variables.

Zooplankton

Zooplankton. The four major groups of zooplankton are protozoa, rotifers, and two sub-classes of crustaceans, the cladocerans and the copepods. Protozoa have high variability even in microhabitats (Hellawell 1986) and little is known about their population dynamics and productivity (Wetzel 1983). The life histories and population dynamics of planktonic rotifers and crustaceans have been studied to a greater extent. Rotifers and crustaceans may help to regulate or respond to the trophic status of a lake. They can feed selectively on phytoplankton (based on palatability or size) and zooplankton populations may be regulated by the feeding activities of planktivorous fish. Horizontal variability in zooplankton may be due to the effects of wind, weather, and their avoidance of shoreline areas (Wetzel 1983).

Fish

Fish. Fish monitormg results can be related to designated use. Fisheries variables are the result of management impacts on the fishery, natural conditions, and the interaction with lower taxon in the food chain. In addition, the public may be able to understand fish monitoring results more easily than other taxonomic groups. Quantitative fish sampling may be difficult due to nonrandom distribution, gear selectivity, and efficiency; however, creel surveys may be useful if carefully designed. On the other hand, Hendricks et al. (1980) discuss some of the disadvantages of monitoring fish.

Hocutt and Stauffer (1980) provide descriptions of methods for stream, reservoir, and lake fish sampling. Protocols also vary by species so that more than one type of gear or approach may be required to sample game and nongame species representative of the stock. Testing is essential to sampling protocol development.

Karr et al. (1986) developed a multihabitat stream fish survey to evaluate individual, community, and zoogeographic factors. For data analysis, 12 metrics were developed and combined into the Index of Biotic Integrity (IBI). The IBI may be used for the development of stream and lake fish monitoring programs for regions beyond Illinois, where the method was developed. Examples of regional applications of the IBI were given by Leonard and Orth (1986) and Steedman (1988).

Habitat Variables

Habitat includes the complex of biotic and abiotic conditions required for life, growth, health, and reproduction of an aquatic community. Habitat variables are an explanatory variable for a primary biological variable, but habitat variables are also important for characterizing ecological integrity. The composition of the biotic community may be due to the extent of species range, stocking, habitat limitations, natural fluctuations, pollutants, other factors, or a combination of these. Habitats can be degraded by human activities or by natural forces. However, habitats have also been shown to respond both positively and rapidly to management (Platts et al. 1987).

The ecological requirements for sustaining community life stages may be considered when determining habitat variables. Knowledge of the biological community and field training are required to monitor habitat effectively and consistently.

Historical impacts are likely to influence present conditions or restoration efforts. Important events include dam construction, unusual water level fluctuations, grazing, channelization, dredging, or drainage of marshes. Debris snagging (removal of branches and tree trunks to improve navigation) is also an important habitat modification. Degradation may restrict or delay remediation.

Stream Macroinvertebrate and Fish Habitat

Stream Macroinvertebrate and Fish Habitat. Plafkin et al. (1989) have developed and tested methods for stream macroinvertebrate and fish habitat evaluation. Variables for substrate, flow, channel morphology, and riparian cover are scored by a weighted rating scheme through observation and professional judgment. Fisheries habitat procedures for four levels of detail are provided by the US Forest Service (USFS 1989). Aerial photos and maps provide information for level I habitat monitoring, and ground truthing can verify findings. The USFS methods (their level 2) are geomorphic and hydrologic ratings based on few measurements. The habitat at the managed resource should be monitored along with a reference site to determine if changes are due to impact or natural variability.

lake and Reservoir Macroinvertebrate and Fish Habitat

Lake and Reservoir Macroinvertebrate and Fish Habitat. Although lakes and reservoirs share many important features, food web interactions may be distinct in reservoirs compared to lakes and may require a different monitoring approach. Particularly notable are the hydraulic features of lakes and reservoirs. Compared with lakes, reservoir surface waters may not be as well mixed horizontally, resulting in higher pollutant concentrations near inflows and a gradient along the main stem or a tributary, with lower concentrations near the dam. Variability in water quality can affect the food chain, habitat, physical or behavioral features in the fish community.

Because reservoirs are more recent features of the landscape, they have less developed predator-prey relationships. Compared with lakes, thesynchronization in production of fish and their plankton food source may be faulty. Water level fluctuations affect littoral habitat and food stability. Ecological systems may not be in equilibrium, presenting some problems for assessment and the understanding of complex interactions (Noble 1986).

Lake Macroinvertebrate Habitat. Variation in lake benthic communities can be very high due to substrate type, chemistry of the sediment-water interface, vertical migrations, wind, food availability, predation, and daily vertical migrations. The link between water chemistry and benthic communities has not been well established. Fish predation is an important regulator of benthic population structure and dynamics (Wetzel 1983). Descriptive studies may prove useful for understanding variability and developing a monitoring design. Hypothesis testing and careful experimental design studies may be required to link benthic population variables with their habitat.

Lake Fish Habitat

Lake Fish Habitat. Methods are not well developed for evaluating lake and reservoir habitats for fisheries. Special habitat requirements for individual species should be the focus. Due to the large surface area for most lakes, only a subset of the most productive habitat should be monitored. One approach is to select a bay or shoreline area that has important features for spawning and hiding and monitor these areas during the critical times of the year. Substrate, weed beds, depth, cover, and temperature are some candidate explanatory variables. Other habitat types to consider are areas with a current, such as inlets and outlets, ledges, and channels.

Chemical and physical monitoring should relate to the fish population variable of concern, specifically minimum DO, temperature, transparency, and pH. Heiskary and Wilson (1988) list water quality and habitat features needed for lake trout, smallmouth bass, largemouth bass, walleye, and northern pike. Monitoring hypolimnetic DO during late summer and before turnover or under ice cover in shallow bays is important for **some** species. Carline (1986) reviews morphoedaphic indices and regression equations used by fisheries managers to estimate fish yield or other variables for lakes and reservoirs. However, these relationships are expected to provide information for first-cut estimates for fish population variables and not quantitative conclusions for use attainment or trend detection.

Fish population variability may be much greater than the change expected due to impact alone. Particularly strong or weak year classes can mask the effects of land treatment. Lake habitat monitoring can account for some population variability not explained by differences in year class. Types of variables include hydrologic, substrate, cover, water quality, and food required for a fish community.

Riparian and Shoreline Habitat Evaluations

Riparian and Shoreline Habitat Evaluations. Riparian ecosystems, which consist of the stream bank and flood plain, are a complex of the environment near flowing water and the environment's organisms (Ewing, 1978). Lake and reservoir shorelines are also sometimes considered part of a riparian ecosystem. Riparian environments have a great influence on aquatic life, and their restoration may be less costly and can provide more immediate benefits to a fishery than stream enhancements such as installing flow modification structures (Platts et al. 1987). Riparian areas of perennial and ephemeral streams, estuaries, and other water bodies may also function as pollutant buffers. Land use, shoreline and overstory vegetation, and soil characteristics are common features of a riparian habitat evaluation.

Methods for monitoring lake shoreline habitats are very similar to monitoring methods for riparian evaluations. Effective riparian monitoring will consider aquatic life requirements, the impacts of land use in the watershed, pollutant sources, and buffering features of the riparian and shoreline environment. Level I riparian monitoring variables and methods are given in Plafkin et al. (1989).

Physical and Chemical Variables

Monitoring the physical and chemical properties of water becomes more meaningful when matched with the time and space scales of the problem. One of the factors affecting water quality problems is the type of water resource (e.g., river, estuary, lake). Different water resource types **have** different hydraulic characteristics which affect their ability to dilute and flush pollutants. Another important factor affecting water resource response is the type of pollutant.

Monitoring the physical and chemical properties of water becomes more meaningful when matched with the time and space scales of the problem.

Wediscussthelakeexampletoillustratethegeneraleffectofapollutantonasingle water resource type. The fate pollutants in other water bodies may be similar on a relative scale but important variations are likely to occur in time and space. Figure 3.1 shows the approximate time and space scales for the fate of different types ofpollutants in lakes. In the lower left, thermal jets and bacteria are relatively short-lived problems with minimal spatial impact. Monitoring short-lived pollutants (e.g. bacteria) therefore requires more monitoring stations to characterize localized conditions and a higher sampling frequency to detect impact compared to monitoring more persistent pollutants.

In contrast, the problems represented in the upper right, such as nutrients, have slower reactions and are more likely to affect the whole lake for an extended period (Chapra and Reckhow 1983). Therefore, monitoring the impact of a persistent pollutant generally takes longer and stations need not necessarily be close to the pollutant source. Sampling frequencies may be lower when monitoring whole-lake problems. Individual level I physical and chemical variables and how to monitor them are addressed below.

Temperature

Temperature. A lake temperature profile may be used to determine the extent of thermal stratification. Inflow plumes of uniform density made up of suspended sediment or other pollutants can be located by monitoring temperature. Careful chemical analyses, DO, and specific conductance measurements also include monitoring temperature.

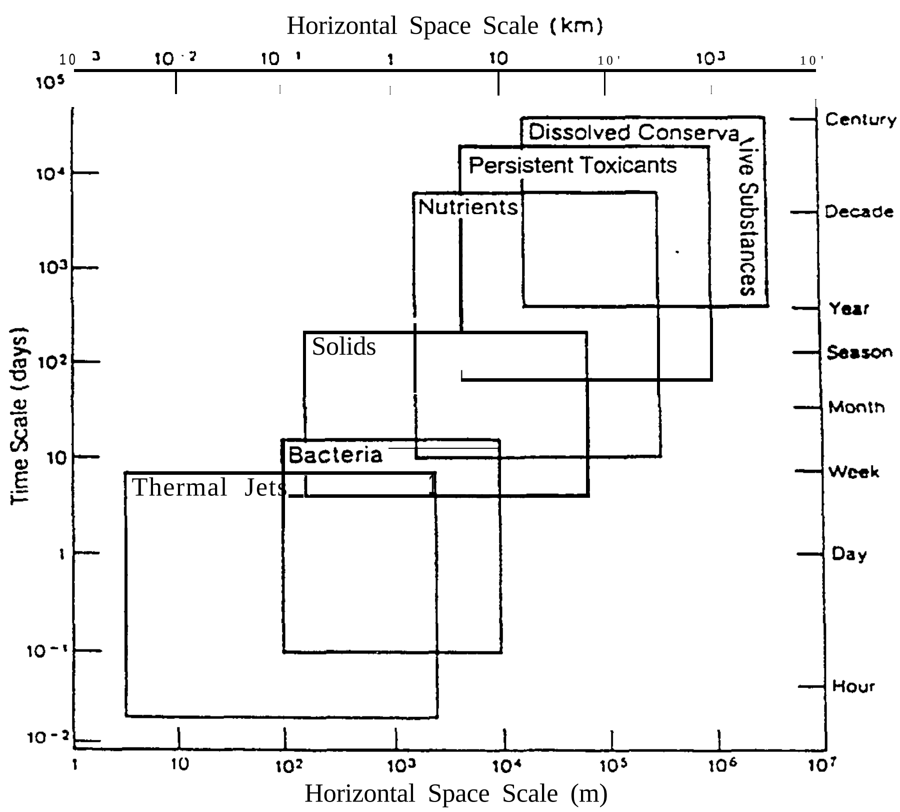


Figure 3.1 Appropriate time and space scales for water quality problems. *(after Chapra and Reckhow 1983)*

Sedimentation

Sedimentation. The rate of sediment-induced storage loss in lakes and reservoirs can be measured with a series of sedimentation or bathymetric surveys. Transects and sampling points are often established perpendicular to the main axis of tributary inflow or the main axis of the lake. Some base strata (original lake or reservoir bottom) should be established to track the rate (cm/yr) of deposition. A long pole may be used in shallow areas to measure sediment depth. For deeper areas, sonar or a SCUBA diver can record sediment depth measurements. Although measurements every year may not be needed, the same transects and stations should be monitored periodically. Major changes in land use, BMPs, or streambank erosion that could increase sedimentation should also be monitored. McIntyre and Naney (1990) used the Cs-137 tracer to estimate rates of sedimentation for Reelfoot Lake in Tennessee.

Transparency

Transparency. An important and obvious property of water is its transparency. The transparency of water to light is critical for aesthetic enjoyment, sight-feeding fish, and recreational uses such as swimming. Transparency decreases as algal and nonalgal dissolved and particulate matter increase water column turbidity. The Secchi disk is an easy, low-cost measure of transparency that provides a rough measure of trophic state. Measured periodically throughout the year, Secchi disk readings can be tracked along with other seasonal changes related to mixing, alga1 succession, and decay, and the changes in suspended sediment due inflows or resuspension. Nonalgal turbidity such as suspended sediment, detritus, dissolved material, and color, and the light attenuation properties of different types of algal cells can impair the interpretation of Secchi disk data and other trophic state relationships.

Turbidity

Turbidity. The reduced transmission of light due to scattering or absorption by suspended solids such as silt and clay particles can be measured by several types of instruments, including the nephalometer and spectrophotometer. Vertical illumination may be measured by the submarine photometer. Methods are available in APHS (1980) and Lind (1985).

Phophorus

Phosphorus. The choice in monitoring phosphorus constituents is based on the source of the pollution problem and the expected advantage of tracking additional variables. Monitoringortho-phosphate is a basicconstituent since it represents the fraction available for plant growth. Total phosphorus is useful for comparison with other measures of trophic state, the development of nutrient budgets, and its application to lake modeling.

Nitrogen

Nitrogen. Nitrogen monitoring generally accompanies phosphorus monitoring for surface water studies. Many eutrophication problems have been linked either in part or wholly to nitrogen. Total Kjeldahl nitrogen (TKN) is a measure of organic nitrogen and ammonia and is typically a large constituent. Ammonia nitrogen, while often in low concentration, may exceed 10 mg/l in the anaerobic hypolimnion of an eutrophic lake. Nitrate is also useful to determine total nitrogen and for an assessment of the likelihood of blue-green algae.

Level II Water Quality Monitoring

The monitoring of treatment program effectiveness when variability is high or when the change to be detected in a variable is subtle requires more detail and careful selection of variables, explanatory variables, and monitoring design. Objectives for level II monitoring include detecting trends, impacts, or causality with water quality variables or pollutant loads.

Detection of a trend, impact, or causality under level II conditions will be rare without adequate pre-, during, and post-implementation data.

Detection of a trend, impact, or causality under level II conditions will be rare without adequate pre-, during, and post-implementation data.

All of the variables discussed for level I monitoring apply to level II monitoring objectives. However we discuss additional variables that are most likely to apply to a level II monitoring study in this section.

Biological Variables

Level II biological monitoring generally requires more detailed measurements than are typically found in level I. Approaches for monitoring biological variables and their explanatory variables are given below.

Pathogens, Viruses, and Intestinal Parasites

Pathogens, Viruses, and Intestinal Parasites. Monitoring of the coliform group of bacteria may not provide sufficient information on the safety of water supply, contact recreation, or shellfish potentially contaminated with pathogens from animal waste effluent or urban runoff. Animal waste sometimes contains disease-causing agents that can be persistent for an extended period Of time. Monitoring specific disease-causing agents and their pathways may be necessary to document abatement.

Pathogenic Bacteria. Many factors are thought to influence the survival of a pathogenic bacterium from the time it leaves the colon of the host animal to the time the pathogen can reach the water course. As the amount of time in waste storage increases, die-off rates also increase, reducing the risk to water when the manure is spread on the field or pasture. Temperature and the treatment of the waste also affect the lifespan of the bacteria.

Viruses. Animal waste is known as a vector for viruses; however, the study of viruses is not very well developed, nor are monitoring methods highly refined. Virusesaremeasuredorcountedfromcellcultureoran infected animal (Thomann and Mueller 1987).

Pathogenic Protozoa. Incidence of intestinal protozoa such as *Giardia lambia* is on the rise, and most often humans are reported as the host. In addition, ofconcem to human health are other pathogens in this group, such as amoeba and nematodes.

Phyfoplankton

Phytoplankton. Variables for monitoring phytoplankton in lakes include taxonomy, biovolume, density, chlorophyll *a*, productivity, and the algal growth potential test. Phytoplankton are sensitive to the physical and chemical changes that occur in the photic zone. Wind can move phytoplankton to the leeward side of the lake, making phytoplankton distribution patchy. Seasonal succession can also hinder trend detection. With careful choice of variables, explanatory variables, and sampling design, phytoplankton monitoring can show impact.

Algal Taxonomy. Phytoplankton species composition may respond to changes in nutrients, light, and other physical, chemical, and biological features of the water column. Species and abundance information can be used to track nuisance blue-greens or other important species for detecting trends.

Biovolume. Biovolume is one of the variables, along with density and chlorophyll a, to monitor phytoplankton production. Biovolume (mm^3/m^3) is the volume of living algal material in a unit area (NCNRCD 1992). The volume of individual cells measured as part of the procedure to calculate biovolume (NCNRCD 1992). Biovolume and density measurements may also help to track algal blooms.

Density. Algal density is the number of units or individual algae in a sample (NCNRCD 1992). A unit may be defined as a single cell, a filament, or a colony. When monitoring biovolume and density, both the numberofcells and the number of units are recorded.

Chlorophyll a. Tracking chlorophyll *a* can provide important evidence for lake restoration programs. This variable is-part of many state lake quality standards (NALMS 1988). Chlorophyll has also been empirically related to other trophic state variables (see Reckhow and Chapra 1983; Coffey et al. 1989; Reckhow et al. 1991).

Phaeophyton is a degradation product of chlorophyll that also absorbs light at the same wavelength, impeding the measurement of chlorophyll. The concentration of phaeophyton is greatest during bloom conditions. Accurate lab analysis accounts for phaeophyton by subtracting it from the chlorophyll concentration.

Productivity. Phytoplankton productivity, along with that of macrophytes and periphyton, generates the vast majority of a lake's organic matter from carbon dioxide, water, and nutrients. Where phytoplankton dominate production, changes in nutrient input should change productivity. Methods to determine productivity are fairly well developed, but results are only as representative as the conditions of the test. Lind (1985) gives the methods for the light and dark bottle oxygen production-consumption and the ^{14}C carbon dioxide uptake technique to estimate productivity.

The light and dark bottle technique may be appropriate for monitoring eutrophic lake productivity where a high sensitivity is not needed. The method is the least expensive and can detect a change of photosynthesis of 20 mg $C/m^3/hr$ (Strickland 1960).

The ^{14}C technique is suited for studies in oligotrophic lakes, where the method must be more sensitive to detect a change on the order of 0.1 to 1 mg $C/m^3/hr$ (Wetzel 1983). Changes in photosynthetic activity of species may be due to changes in nutrient composition or inhibition by a herbicide. The impacts of herbicides are difficult to detect in the field.

Productivity is expressed on an area1 basis for the entire lake. Samples should be taken for various depths within the photic zone to estimate total productivity for the water column (Lind 1985). Spatial gradients and the effects of season should be considered when designing the sampling program.

Algal Growth Potential Test_ The USEPA (Raschke and Schultz 1987) has developed the algal growth potential test (AGPT) to determine the potential for nutrients in water or sediment to support or inhibit growth. The test can provide information on the bioavailability of nutrients and algal response to nutrient constituents or changes in constituents. The AGPT can be used for pollutant source identifying and tracking controls through monitoring the bioavailability of phosphorus in tributary or streambank suspended sediments, water body sediment, or other locations in the water column. Algae in the AGPT may respond more quickly and may be less influenced by confounding factors than response variables higher in the food chain.

Periphyton

Periphyton. In streams and shallow areas of lakes, monitoring periphyton may be used to detect changes in chemical conditions (herbicides) or productivity. Periphyton include protozoa, rotifers, nematodes, bacteria, diatoms, and blue-green algae, along with detritus attached to sediment, sand, rock, or other plants.

Variables to consider include the number of taxa and their abundance. For estimating periphyton biomass or production, collections can be made using an artificial substrate such as a glass slide. Also periphyton on rocks or logs may also be sampled, but estimating production with this method is more difficult. Periphytoncollected on artificial substrate may not mimic natural populations, but they can be useful when an inadequate sample size is available on natural substrates.

Macrophytes

Macrophytes. Estimation of area1 macroinvertebrate biomass can be completed by a diver taking standard area plot collections of plant tissue (Canfield and Duarte 1988; Kelly 1989). Species are determined and the sample is dewatered in a standardized manner and weight. Area1 biomass should be estimated by its density and theextent ofthe species. Variation inspeciesand biomass may be due to depth, wind effects, substrate, time of year, and water column nutrients or light.

Macroinvertebrates

Macroinvertebrates. A qualitative protocol for macroinvertebrate sampling and data analysis is presented by Lenat (1988). The protocol is based on monitoring multiple habitats at a stream station. More taxa are potentially collected than with the kick net or Surber sampler.

The qualitative collection protocol uses coarse mesh samplers to monitor riffles, snags, streambanks, and leaf packs. A fine mesh sampler may be used to process samples from rocks, logs, and sand, and visual collections consist of picking macroinvertebrates from large rocks or logs. Macroinvertebrates are separated from organic matter and picked in the field or preserved. A description of the method and testing of the protocol with kick samples and a water quality index (chemical data) are presented by Lenat (1988).

Zooplankton

Zooplankton. Level II zooplankton work may to answer questions about zooplankton grazing and their effect on algal productivity, or zooplankton as a source of food for fish. Patterns of seasonal succession are likely to play an important role. The relative importance of zooplankton on lake productivity should be assessed to determine the proper emphasis of this group in the efforts and budget of project monitoring.

Fish. The fish species, type of water resource, habitat conditions, and variable of interest will determine the best monitoring methods. Hocutt and Stauffer (1980) provide examples of fish monitoring methods in streams, reservoirs, and lakes.

Other fish variables may be the presence/absence or abundance of individual species, density orbiomass. Recruitment success and population size structure can be determined yearly based on the length-frequency distribution

The IBI (Karr et al. 1986) may also be used for level II monitoring, especially if individual metrics are chosen or developed. Generally a level I procedure and enumerations could replace ratings to produce more quantitative data.

Fish spawning environment. The reproductive success of fish can depend heavily upon the condition and quality of the spawning substrate and interstitial and overlying water. Recent testing of embryo survival in simulated spawning environments has been conducted- for stream salmonids habitats of the West (Burton and Harvey 1990).

Bioassay. Biological systems may be used to assess current conditions or to estimate the effect or impact of pollutants on single or multiple species. Bioassays are a very systematic means of determining the effect of a chemical concentration or other attribute of the aquatic environment on the survival, growth, reproduction, or physiology of an organism or group of organisms. When the findings of the work have direct meaning to management objectives, the resul ts can be very useful and can be used as the basis for clear direction for setting goals and even regulation.

The benefits of extending the findings from bioassay from one species to another or interpreting confounding influences are less obvious. Maltby and Calow (1989) reviewed the application of bioassays and found that the responses observed in particular systems were not transferable or relevant to others, and that the mechanism of the response should be part of the theoretical framework for designing the bioassay.

Habitat Variables

Macroinvertebrate and fish habitat assessment for streams and lakes, and their riparian and shoreline areas, are discussed below. More direct measurements and fewer ratings are suggested for level II.

Stream Macroinvertebrate Habitat

Stream Macroinvertebrate Habitat. The physical and chemical quality of the stream and its substrate are the major features of macroinvertebrate habitat. Initially monitoring hydrologic properties and substrate quality are priority habitat variables. Some protocols also include organic matter and interstitial water chemistry. Wiederholm (1984) discusses lake and stream macroinvertebrate response to various pollutants.

Hydrologic Parameters. The stability of an aquatic environment is dependent upon the presence, discharge, and velocity of water. Both low flows (e.g., low precipitation, low water table, or withdrawals) and major runoff events should be tracked. Water velocity (distance moved per unit time) and depth may have an important influence on the structure of benthic communities (Osborne and Hendricks 1983). The lnstream Flow Incremental Methodology (IFIM) is a technique for recommending flows for stream management. The IFIM was developed by the US Fish and Wildlife Service with the primary objective of assessing the changes in fish-standing crop and species composition due to changes in streamflow (Bovee 1978). Some IFIM studies have been applied to benthic organisms (Gore and Judy 1981).

Newbury (1984) discusses stream hydrologic habitat assessment and identifies potentially important variables to consider.

Substratum Habitats. The substrate consists of parent material, human trash, and organic matter such as leaves, branches, logs, grass, filamentous algae, moss, etc. Macroinvertebrates live on the substrate and are especially adapted for clinging and attaching to it. The substrate functions as a place for burrowing, escapement, protection from current, or a place to construct a case or deposit eggs. Minshall (1984) provides a list of potentially useful substrate variables for evaluating macroinvertebrate habitat. Chapman and McLeod (1987) provide a detailed literature review of the importance and measurement of substrate variables.

Sediments should also be disturbed to determine and document the presence and extent of odors, oils, and deposits (Plafkin et al. 1989). Sewage, petroleum, and chemical odors or anaerobic conditions should also be documented. Past anaerobic conditions may be indicated by the blackened condition of the undersides of streambed rocks. The general extent of sediment oils and sediment deposits such as sludge, sawdust, paper fiber, sand, or relict shells should be noted (Plafkin et al. 1989). Any abnormalities should be evaluated to determine if further investigation is needed on pollutant sources and impacts.

Course Particulate Organic Matter (CPOM). Plant debris (e.g., leaves, twigs, bark) that accumulates in areas of slower moving water may be sampled for stream benthos in the shredder group. Shredders are particularly sensitive to toxins that often adsorb to CPOM (Plafkin et al. 1989).

Interstitial Water. The substrate-water interface is critical as macroinvertebrate habitat. An evaluation of watershed land use or the water column may prompt chemical analysis (nutrients, metals, toxins) of the quality of interstitial water within the substrate and just above the sediment-water interface. Sampling depths may be based on substrate characteristics and known habitat requirements. Methods for collecting interstitial water are provided by Simon et al. (1985).

Water Column Parameters. An assessment of water column physical and chemical constituents may be basic information for macroinvertebrate community monitoring. Depending upon objectives, useful water column variables include: temperature, dissolved oxygen, pH, conductivity, transparency, turbidity, color, nutrients, alkalinity, conductivity, metals, pesticides, and toxins.

Stream Fish Habitat

Stream Fish Habitat. To monitor the effect of land treatment on stream fish, evaluate ecological conditions that will support the fishery and site potential. Stream fish habitat or riverine-riparian habitat includes the riparian vegetation and the designated use of the land and water in the stream channel.

Land use management practices may cause fish population changes, but it is often difficult to show causality. Changes in fishery management and angler harvest also impair trend detection. Assessment of impact requires site-specific evaluation of habitat conditions and fish population fluctuations both before and after treatment (Platts and Nelson 1988). Many habitat features influence fish communities, and the variables and methods for their measurement can vary widely. Stochastic events such as storms or drought should also be tracked since they can regulate the structure of stream fish assemblages (Schlosser 1985).

Riverine-Riparian Community Classification. A system of classification provides a basis for resource categorizing (Youngblood et al. 1985) and monitoring along with BMP selection and application (Platts 1989). While unique riverine-riparian communities may exist, the development of monitoring protocols is likely to benefit from some method of classification. The classification can standardize monitoringand provideaframeworkforcommunication between thescientist and decision-maker.

The riverine-riparian classification is based on a system of geoclimatic factors. The classification is hierarchical and may be described in the context of mapping scales as in Lotspeich and Platts (1982). The ecoregion is the largest mapping scale, with successively smaller divisions such as geologic district, land type, land forms. The lowest is the vegetation type. The concept is based on an integrated land-aquatic classification that is used within ecoregions or when ecoregion mapping units do not match the desired characteristics.

Riparian-riverine community classification may be defined by riparian vegetative type. Managers assume that the constituent communities of a taxonomic unit will respond in the same way to similar management (Platts et al. 1987). Differing from forest or rangeland terms, riparian communities are classified by present rather than climax community type. Platts et al. (1987) identifies field and office methods for riparian community classification.

Transects, Maps, and Aerial Photography for Habitat Measurements. Monitoring along a ***transect*** is useful for consistent collection and organization of stream habitat data. Figure 3.2a illustrates the arrangement of transects for stream treatment and control measurements perpendicular to the main direction of flow. In the figure, the livestock exclosure, where livestock are excluded, is the treatment. Figure 3.2b shows the use of cross-sectional transects (transect 35 and 26) and medial transects (transects ED, DC, etc.) for stream reach habitat measurements. Figures 3.2c and 3.2d show types of measurements made for a detailed cross-sectional survey. As in Figures 3.2c and 3.2d, some stable feature of the landscape, such as a stake, or fence post may be used to mark the beginning and end of each transect and as a reference for future data collection.

Drawing a diagrammatic ***map*** of a stream reach may require more measurements at the onset than transects, but more details may be quantified, and stream features that remain unchanged in the original map may be retained for future evaluations. An example of an idealized stream section is given in Figure 3.3a, and a diagrammatic map with habitat areas determined by planimetry, or a computer-aided digitizing tablet, is shown in Figure 3.3b. Photographs may also be used for documenting habitat conditions not easily described or measured.

Habitat requirements for the entire range of the species must be considered, not just the monitoring station (Hendricks et al. 1980). For restoration of a fishery in a second-order stream impaired by sediment, monitoring only sediment delivery to that reach may not quantify all relevant aspects of restoration. If spawning habitat and upstream macroinvertebrate food sources are not protected, then the downstream fishery may not recover (Karr and Dudiey 1981). Tributary streams may also be important as spawning areas for some lake fishes.

Aerial photos can be used to identify many characteristics of habitat for a large area. However, ground sampling is necessary to supplement aerial photos through ground truthing and identification of some species. Platts et al. (1987) provide a list of variables and methods for monitoring with aerial photos. Habitat variables to monitor grazing impacts include areas covered with vegetation and bare soil, stream width, stream channel and streambank stability, and width and area of the riparian zone (Platts et al. 1987).

Fish Habitat Models. Several models have been developed to aid in the evaluation of stream fish habitat. Understanding the limits of the model through a review of assumptions, the development data set, and the geographic range will help to avoid misapplication. Habitat models are not likely to be able to estimate fish abundance or biomass since populations may be limited by the impact of pollution or other nonhabitat factors. However, habitat models can provide a standardized framework for consistent habitat monitoring and modeling.

The US Fish and Wildlife Service has proposed the use of the Habitat Evaluation Procedure (HEP) (Terrell et al. 1982) and the Instream Flow Incremental Methodology (IFIM) (Stalnaker 1982). The models share a component called PHABSIM that is based on the assumption that fish population fluctuations are driven by physical habitat variables such as depth, velocity, substrate, and cover. If the

Figure 3.2a Transects for habitat evaluation perpendicular to the main direction of stream flow. ***(after Platts and Nelson 1985)***

A.

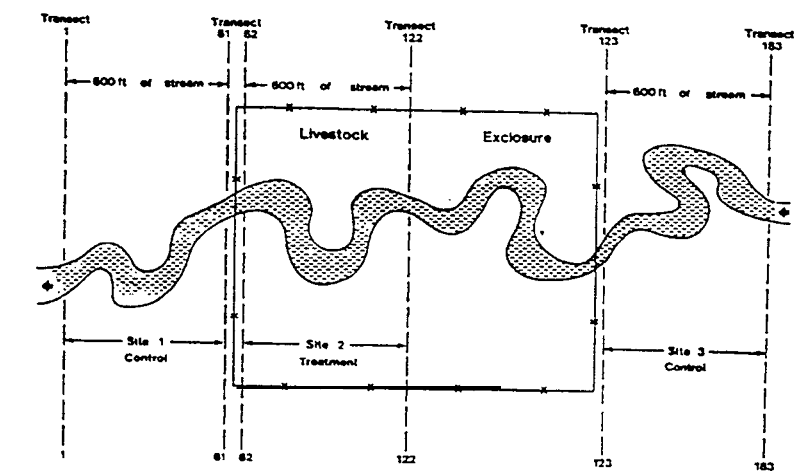
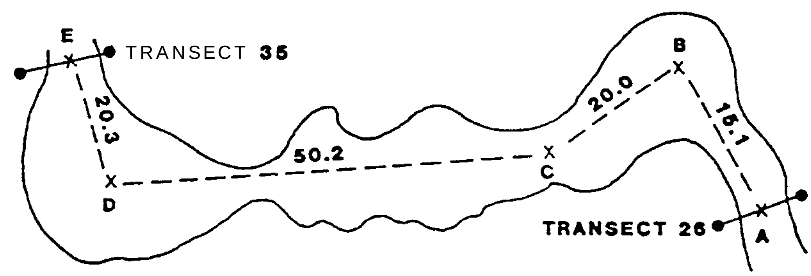


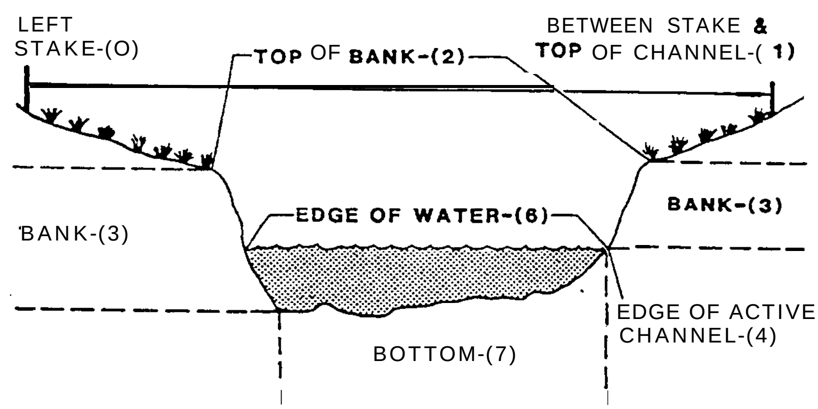
Figure 3.2b Medial transects for habitat evaluation. ***(after Platts et al. 1985)***

B.

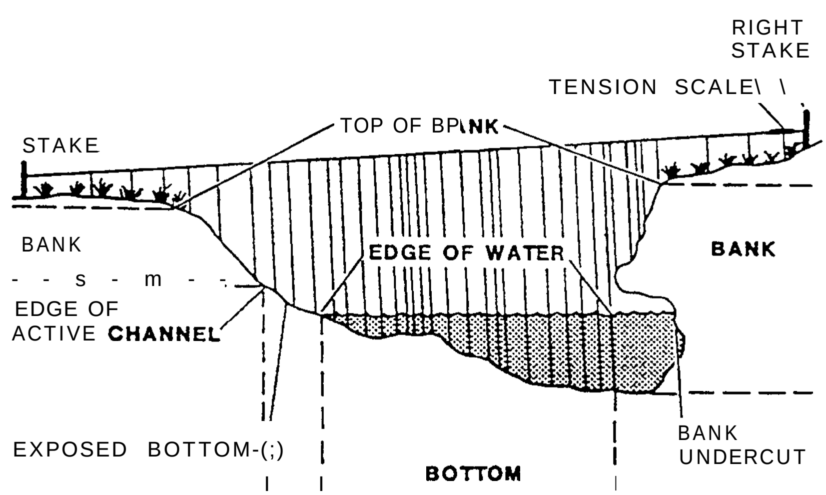


Figures 3.2c,d Cross-sectional transects for detailed channel measurements. ***(after Platts et al. 1985)***

C.



D.



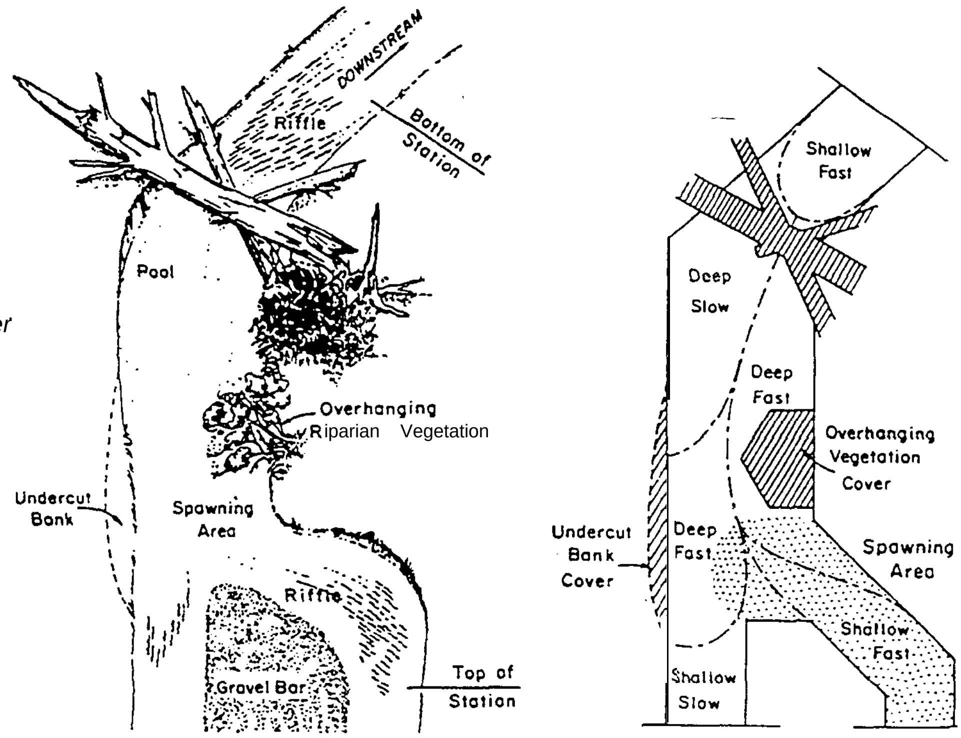


Figure 3.3a, b Stream section diagrams to illustrate detailed mapping analyses of fish habitat. *(after Oswood and Barber 1982)*

habitat model variables remain stable or show little fluctuation, then the model assumes fish populations will also remain stable. Mathur et al. (1985), Karr et al. (1986), and Platts and Nelson (1988) have reviewed models based on PHABSIM and have found limitations to their use.

The US Forest Service COWFISH model (Lloyd 1986) was designed for use in the western United States for estimating past and current livestock impacts on riparian and instream conditions. The model is not intended to estimate fish population variables or to replace current models developed for that purpose; however, the results of COWFISH may be included in the Habitat Suitability Index (Hickman and Raleigh 1982) to estimate optimum and existing catchable fish populations. The geographic range for model development is Nevada, Utah, Montana, and Idaho. The input variables include the extent of streambank undercut, vegetation overhang, and bare soil or trampling.

Instream variables include cobble embeddedness, width, and depth. Stream gradient and soil type are also considered. Shepard (1989) used COWFISH to evaluate livestock impacts in Montana and found the model produced both reasonable and imprecise estimates of catchable trout depending upon the species composition of the stream.

Instream Habitat Parameters. Analysis of some life cycle and biotic interactions may be necessary if impacts go beyond effects of pollutants and habitat. Monitoring life cycle conditions needed for spawning, embryo survival, young-of-the-year hiding, and the requirements for juveniles and adults may be useful, with some evaluations focusing on the requirements for one or more life cycle stages. Predator-prey relationships may also be assessed. Habitat requirements for forage fishes may be a part of habitat analysis.

Biotic interactions may determine extent or absence of a species or community independent of environmental quality or management. These interactions may be important for tracking the effects of pollutants or controls on fish populations. Other interactions to consider are competition, predation, disease, and parasitism.

Energy and **organic matter processing in the stream ecosystem.** Organic energy sources for stream fauna of terrestrial origin include leaves, branches, tree trunks, other organic matter, and algae. The process of organic matter generation and cycling regulates food availability, which in turn helps structure the stream community. Species and groups of benthic macroinvertebrates are specialized in their ability to consume organic matter for a given particle. Empirical studies of the fisheries habitat quality and community attributes may be used as aids in determination of variables (Oswood and Barber 1982) size, type, and origin (Minshall et al. 1985).

Lake and Reservoir Fish Habitat Evaluations

Lake and Reservoir Fish Habitat Evaluations. Few methodologies are available for the assessment of lake habitat quality. Critical conditions to consider are reproduction, hiding, and food in each identified habitat zone.

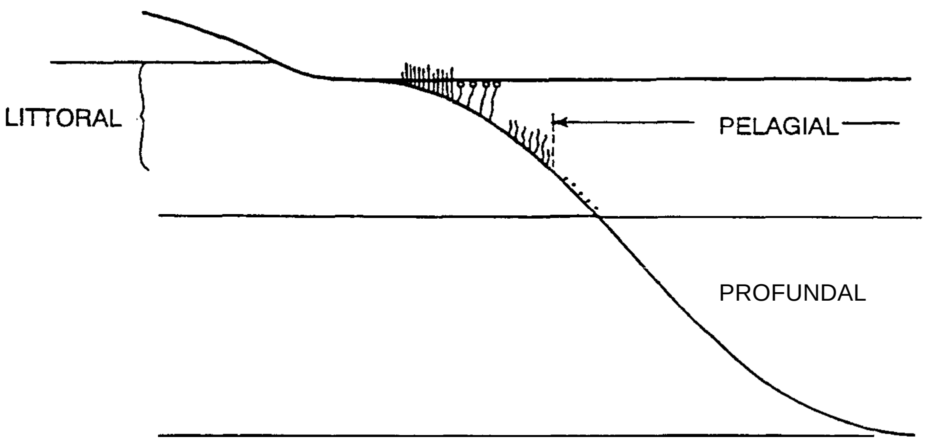
Lake habitat zones are typically demonstrated by depth and by the presence or absence of vegetation attached to the bottom. A simplified cross-sectional view of lake habitat zones is given in Figure 3.4. A lake may be divided into three zones: the littoral, pelagic, and profundal. The littoral zone extends from the shoreline toward the open water to the point where the bottom becomes devoid of vegetation. The pelagic zone is the free open water exclusive of the littoral zone. Below the pelagic zone is the profundal zone which, if it exists, is devoid of vegetation.

Littoral Zone. The shallow area around the perimeter of the lake, which can extend to the middle of shallow lakes, supports a wide range of heterogeneous and patchy habitats. Littoral flora may also include wetland species of macrophytes and periphyton. Production by this group of plants can be substantial and can exceed that of the open-water pelagic zone. Lake bottom characteristics may in part determine the species composition of macrophyte stands. Light availability is important for regulating macrophyte species production and composition (Wetzel 1983). In turn, macrophyte and periphyton production may influence chemical and physical properties of the lake as well as tht composition, and production of insects, larger invertebrates, and fish. While more studies are needed, fisheries managers will benefit from evaluating littoral habitat composition and function.

Pelagic Zone. The habitat of the pelagic zone is generally more homogeneous and less patchy compared with the littoral zone, although gradients and discontinuities are common. This open-water surface layer of lakes produces free-floating algae called phytoplankton. The health or condition of a lake may be measured by phytoplankton speciescomposition orbiomass. Lake response to reduced nutrient loading is first expected in the phytoplankton community. However, inhibiting factors such as food-web interactions between phytoplankton, zooplankton, and fish, for example, can mask the effect of controls. Therefore, food web components and their variability should measure ecosystem impact.

Profundal Zone. Compared with the littoral zone, the profundal zone is more homogeneous and generally contains fewer benthic animal taxa. The profundal zone is essentially devoid of light and photosynthetic activity. Particulate matter (plankton, detritus) from the pelagic zone falls through the water column into the

Figure 3.4 Lake habitat zones.
(after Wetzel 7983)



profundal zone and then to the sediments, where it settles and decomposes. As a lake becomes more productive, the process is accelerated, and larger amounts of organic matter are deposited, increasing the depth of the sediments. Decomposition of organic matter increases oxygen demand, which reduces hypolimnetic oxygen concentration, sometimes below the levelcritical for fish and other aquatic animals. Loss of hypolimnetic oxygen decreases benthic animal taxa richness and numbers.

Interaction of Lake Zones. Understanding the interaction between the three lake zones is important for assessing impact. Wetzel (1983) has documented the biogeochemical linkages between the littoral and pelagic zone. The organic and sediment materials that fall from the pelagic zone into the profundal zone may become recycled back to the pelagic zone during turnover.

Riparian and Shoreline Habitat Evaluations

Riparian and Shoreline Habitat Evaluations. Platts et al. (1983, 1987) provide comprehensive guidance on riparian habitat monitoring.

Riparian Vegetation. Plants growing on a streambank and flood plain influence streamside and instream conditions. Riparian plants stabilize shoreline areas, supply organic matter for organisms, reduce water velocity in'streams, provide cover and food for fish, and intercept, control. and store solar radiation inputs to the stream environment (Platts et al. 1987).

Several riparian vegetation variables may be used for evaluation based on a numerical rating system. They include vegetative use by animals (such as grazing) and the effects of cattle crossings, vegetative overhang, streambank stability, and streamside cover. Detailed guidance on electronic forage analysis is also provided by Platts et al. (1987) for determining forage vegetative production and use by grazing animals in large areas.

Riparian Soils. Platts et al. (1987) define riverine geomorphic terms and the processes that affect the distribution of sediments. They also describe soil characteristics such as soil genesis, morphology, and taxonomy.

Measurements Above the Water Column. The vegetation of the shoreline and the canopy affect shoreline stability, channel roughness, and running water out-of-bank velocity. Both vegetation and topography affect shading, light intensity, and heating effects on the water column. Light and heat affect many water quality variables, such as water temperature, aquatic plant production, and dissolved oxygen. Platts et al. (1987) provide information on the use of instruments to assess vegetative and topographic features above the water column.

Streambanks. Ray and Megahan (1978) developed a procedure for measuring streambank morphology, erosion, and deposition. Detailed streambank inventories may be recorded and mapped to monitor present conditions or changes in morphology through time.

Platts et al. (1987) provide methods for evaluating and rating streambank soil alteration to assess the effect of land-use changes on streambankstability and how bankstability could affect fish. Other measurements that are likely to be important for fisheries habitat evaluations include streambank undercut, stream shore water depth, and stream channel bank angle.

Organic **Matter.** The size, type, and amount of organic material available as food for macroinvertebrates and other levels of the food web in the stream environment can be determined. Organic matter ranges in size from fine particles to whole trees, and stability from living green plants to highly decomposed and refracting residue. Platts et al. (1987) provide details on classification, measurement, and mapping of organic matter for riparian evaluations.

Chemical and Physical Monitoring

The discussion of level I monitoring adequately describes the level of detail needed for the monitoring of most chemical and physical variables. Monitoring these variables for level II increases complexity of design. Monitoring pesticides, chemically contaminated sediment, and sedimentation are the main themes of level II chemical and physical monitoring.

Pesticides

Pesticides. Compounds likely to be a threat or to cause a known impairment should be monitored to determine the level of contamination. Pesticide detections are generally of concern. Also where standards are violated, then the risk to human and aquatic health should be evaluated.

Because analytical procedures must be targeted to a specific pesticide or its metabolite, the county health department should be involved in initial problem assessments. Thereafter, state labs used for assessing environmental health should be consulted. In addition, some out-of-state labs may be able to compete with the quality control, quality assurance, and the costs of an in-state private lab.

Chemically Contaminated Sediment

Chemically Contaminated Sediment_ Impairment may be documented by comparing contaminated sediments with sediments in reference areas or by a relating sediment contamination to some biological effect. Methods for problem identification and monitoring are provided by USEPA (1988) in a review of the present state of numeric- or chemical-specific methods and the more general descriptive methods.

A compendium on monitoring sediment quality provides an overview of methods that are used to assess chemically-contaminated sediments (Tetra Tech 1989). Numeric methods for toxicity and tissue testing are given, along with descriptive methods using benthos, to assess sediment quality using benthos. Sediment chemical contamination, sediment toxicity, and benthic community structure are assessed in the sediment quality triad procedure.

Sedimentation

Sedimentation. McIntyre et al. **(1989)** and McIntyre and Naney (1990) provide an example of using Cesium-137 isotope tracers to determine sedimentation rates for different periods of land use from 1880 to recently. Historical sedimentation may serve as a baseline for comparison to measured rates for trend detection.

Pollutant Loading Rate Monitoring

Monitoring the loading rate is very useful measure for evaluating current conditions, trends in pollutant loading, or evaluating the effect of land treatment. The loading rate or the mass of pollutant exported per unit time (e.g., lb/yr) is a basic measurement for eutrophication studies and pollutant budgets. Loading rates are directly comparable to one another but they can be vary significantly from year to year.

The three major tasks for determining pollutant loads are:

1. measuring water discharge (cubic feet per second);
2. measuring pollutant concentration (milligrams per liter); and
3. calculating pollutant loads (multiplying discharge times concentration over a year).

The primary difference between level I and level II load monitoring is the type of sampling gear, time required, and overall cost. Level I load monitoring does not require continuous stream gaging to measure discharge. Grab sampling is used to obtain water samples to measure concentration for level I. Level II load monitoring requires continuous stream gaging to measure discharge and an automatic sampler to take water quality samples.

Level I Pollutant Loading Rate Monitoring

Where there is a lower variability in discharge and where peak flows are not extreme, as in the case of PS and irrigation return flows, level I load monitoring may be employed. For other cases with high variability in discharge, large errors in the loading estimate should be expected.

Several methods may be used to determine Level I stream discharge measurements. Sampling sites should have a stable stream bed and a natural downstream control. A current meter may be used to measure stream velocity using either a rotating propeller or cup wheel. Because stream velocity varies by depth in the channel and the location, several measurements must be made to measure instantaneous velocity and calculate the average velocity. Using a measurement of the cross-sectional area, and multiplying times average velocity, the total discharge can then be calculated.

A staff gage or tape measurements (distance from bridge to water level) used to determine water level elevation may also serve to determine level I stream discharge. To calculate discharge based on water level elevation, a stage discharge relationship is developed from detailed measurements of the stream bed and known discharges for several stream elevations. The resulting stage discharge relationship or rating curve can be used to estimate discharge based on elevation of the water surface at the time of sampling.

To sample pollutant concentrations a grab sampling technique may be used. The concentration sample should be taken at the same location in the stream for each and every sample. Sampling depth and sample handling protocols should be developed. Overall a predetermined schedule should be developed for sampling both discharge and concentration. Wedepohl et al. (1990) provide several methods to calculate pollutant loads.

Level II Pollutant Loading Rate Monitoring

Level II load monitoring is essential for small watersheds with high peak flows, and in situations where a continuous record of discharge and automatic sampling of pollutant concentration is needed. Level II load monitoring requires a complex and typically expensive sampling protocol to measure discharge and pollutant concentration. However, there are good references for measuring loads, such as Rantz (1982), Brakensiek et al. (1979), and Wedepohl et al. (1990). The US Geological Survey is a direct source of information on stream discharge measurements.

Continuous discharge measurements for level II require instruments to record stage upstream from either a natural control or a structural control such as a weir or flume. Automatic samplers are used to collect concentration samples at a regular interval (e.g., eight or 24 hours).

*Monitoring objectives, pollutantsources, **and budget** dictate much of the design. The need for monitoring a spatial control and the need to quantify conditions before, during, and after land treatment comprise the remainder of design decisions. Finally, the manager should verify that the monitoring committee's design will address the monitoring objective.*

4

Monitoring Program Design

A **time series** *must be obtained to document changes in water quality due to land treatment. Measurements should either be taken at regularly timed intervals (e.g., every 7 or 74 days) or for specified periods and for a sufficient length of time using comparable, consistent methods.*

For trend detection, the monitoring objective should be translated into a testable statistical hypothesis.

The monitoring program design is the framework for sampling, data analysis, and the interpretation of results. Typically, the objective of a NPS pollution control project is to document changes in water quality that are related to the NPS controls. Monitoring both the water quality and the land treatment/land use in a project can provide valuable feedback regarding the impact of land management on water quality. This chapter emphasizes land treatment and water quality monitoring designs to meet the objectives of detecting trends and/or direct impacts of land treatment on water quality; in addition, objectives for evaluating current conditions and problem documentation are discussed.

A time series must be obtained to document changes in water quality due to land treatment. Measurements should either be taken at regularly timed intervals (e.g., every 7 or 14 days) or for specified periods and for a sufficient length of time using comparable, consistent methods.

The components of a time series are both deterministic and random. The deterministic component changes in a predictable manner and is assumed or known without error (e.g., time, seasonal cycles, or treatment strength). The random component is measured with error and consists of unexplained factors that hinder the detection of the trend. To detect a trend, the random component and complex deterministic factors such as cycles (e.g., climatic or life cycle), and the dependence of one observation on the next (serial correlation), must be taken into account.

Improvement from NPS control occurs gradually, and few, if any, agricultural NPS control studies have shown a step trend in the receiving water. More often the change is incremental and subtle, and visual detection of a change can result in false conclusions: claiming progress where none has occurred, or failing to detect small but real improvement. Therefore, NPS monitoring to detect changes in water quality due to land treatment requires an experimental design to isolate land treatment effects.

For trend detection, the monitoring objective should be translated into a testable statistical hypothesis to provide structure to the experimental design. The null hypothesis states that no change is expected. The monitoring survey is designed, using the principles of experimentation, to test the null. If the design is sound and statistical testing shows the null hypothesis to be false, then a change can be inferred. Otherwise, the monitoring survey should conclude that the objective was not met, or detection of change was overcome by extreme variability. In either case, with a sound objective, well-formulated hypothesis, and careful design, the monitoring survey may be expected to produce valuable information.

If it is not obvious that the management action is likely to cause an observable change (e.g., when there are uncontrolled sources, inadequate treatment, or variability masks the detection of treatment), a more sophisticated monitoring design, making use of a carefully chosen set of spatial and temporal controls, may be needed to provide evidence of an impact. In other cases the magnitude of change expected may be too small to detect. Failure to think through the design can result in wasted data collection and inconclusive results.

Below are methods for specifying objectives for incorporation into the experimental design of the monitoring program. Analysis of existing data can provide information on system variability which is useful for developing the design. Reducing the MDC will increase the chances of statistical significance and improve the power of the test.

Formulating a Specific Monitoring Objective

A monitoring objective should be narrowly and clearly defined to address a specific problem at an appropriate level of detail. Spatial and temporal information related to the problem is essential for implementing a successful monitoring program. The monitoring objective specifies, where appropriate, the primary variable(s), the degree of causality or other relationship, and the anticipated result of the management action. Example monitoring objectives include:

- to evaluate current conditions in Long Creek by analyzing ecological integrity and suitability of the creek as a water supply;
- to document the water quality problems in Highland Silver Lake by identifying specific pollutant constituents, their magnitude, sources, and impacts on the designated uses of Highland Silver Lake;
- to detect the trends in the dissolved oxygen concentrations in Hope Creek due to the municipal treatment plant upgrade;
- to evaluate the impact of critical area manure management practices on the frequency of algal blooms in Green Lake;
- to determine the effect of implementing BMPs on sediment and nutrient loads entering Grand Lake from the Grand River watershed.

Monitoring Objectives

The discussion of monitoring objectives serves as a framework for the monitoring program design discussed below.

Evaluation of Current Conditions

The purpose of assessing current conditions or ecological integrity is to evaluate the overall health of the aquatic resource, to determine if the designated use is being attained, and to evaluate the ecological potential of the resource. The USEPA Rapid Bioassessment for Use in Streams and Rivers (Plafkin et al. 1989) provides a method for collecting and integrating habitat, water quality, and biosurvey data to evaluate current conditions. Habitat is an important determinant of ecological potential and provides the basis for further ecological investigations.

Knowing current conditions helps the manager understand the potential for remediation of the water resource. For example, in an agricultural watershed, severe streambank and cropland erosion may have caused a stream bed to be filled with sand and silt. The biological potential would be limited by habitat impairments that reduce fish reproduction capability. Even with extensive implementation of BMPs on cropland and streambanks, the stream may take a long time to flush excess sediment and achieve an improved habitat condition.

In general, a water resource in a predominantly urban or agricultural watershed has a lower potential habitat condition than one in a forested watershed. The overall ecological condition of the resource will be limited by the present and the potential habitat conditions.

Problem Documentation

Problem identification and the careful documentation of the water quality problem with monitoring are essential for projects interested in improving water quality through the implementation of BMPs.

Carefully designing and documenting the water quality problem is one of the most important steps for NPS pollution control and water quality monitoring. An effective approach is to implement a problem identification monitoring program lasting6 to 18 months. Problem identification monitoring uses asite-specific plan to identify pollution sources and impacts during the seasons of greatest pollutant loading (e.g., spring runoff, snowmelt) and during the season when impairments are noted (e.g., algal blooms).

Carefully designing and documenting the water quality problem is one of the most important steps for NPS pollution control and water quality monitoring.

Problem Documentation Monitoring Stations

There are three types of problem documentation monitoring stations: a) tributary, b) main stem stream, and c) wetland or lake. A mixture of station types (depending upon the situation and cost) may be useful to document the problem.

Tributary stations should be located immediately below suspected pollution sources. Tributary monitoring helps to identify pollution sources and their magnitude or to assess habitat limitations. Tributaries may serve as a source of food for fish or they may provide critical habitat for the managed water resource. Monitoring the main stem stream (primary drainage channel) alone is inadequate to identify sources of pollution because the main stem stream dilutes and assimilates tributary inputs. making identification of pollutant sources areas difficult.

Tributary stations are especially useful for identifying pollution sources such as point sources, animal lots, mobile home parks, quarries, construction sites, and cropland. Monitoring above and below sources may be needed and is encouraged if discrete source inputs can be identified and it is necessary to characterize different sources. Pollutant constituents that match the potential source should be sampled, along with pollutants that affect the managed resource.

Main **stem stream stations serve** to show the aggregate of upstream and tributary effects. Consider chemical, biological, habitat, and streambank analyses that match the impairment or threat to designated use. Main stem stations should be located close to suspected sources. Monitoring above and below tributary or point sources of pollution serves to evaluate their impact. Main stem monitoring shows the extent that dilution and assimilation affects pollutants and stream quality.

Wetland and lake stations should be selected to match the location of the impairment or threat to designated use.

Problem Documentation Sample Timing

Both baseflow and storm conditions should be monitored to identify the problem and its source using chemical/physical monitoring. Baseflow water chemistry and discharge samples should be taken at approximately 28-day intervals or more often. Monitor especially during the time of the year when the problem is noted. All baseflow samples need not be at low flow or at a regular interval. The purpose is to characterize low flow conditions. Guidance on the timing of biological monitoring should be available from the state water quality agency.

Storm sampling can be used to document the magnitude of hydrologic and pollutant impacts. Monitoring should coincide with runoff events associated with agrichemical applications, manure applications, irrigation season, or other activities thought to be responsible for the water quality problem. For animal lots with minimal control of waste, the timing of the storm is not critical, since the problem should be relatively easy to detect.

Storm samples should be taken during the rise, peak, and falling stream levels during runoff events. Seasonal and climatic factors should also be considered. If snowmelt is substantial, monitoring during this time is important. Also consider historic rainfall patterns. Drought conditions will most likely be unrepresentative so problem documentation monitoring may have to be extended to represent typical wet weather and pollutant loading conditions.

Examples of Problem Documentation Monitoring

Water quality problem identification monitoring should seek first to specify pollutants and conditions responsible for the impairment to the designated use. Once the water quality problem is identified, the severity of the problem can be assessed. Clearly identifying the specific pollutant and assessing the problem assists land treatment staff in identifying critical areas and targeting BMPs.

The source ***of bacterial contamination*** in shellfish or recreational waters may be difficult to locate. Die-off for bacteria is relatively rapid in cool seasons (an hour to a week or more), and sources such as animal and human waste can generally be defined quickly with a thorough survey and careful monitoring below suspected watershed pollutant sources.

The Utah and the Oregon RCWP projects monitored above and below dairies to determine the magnitude of the bacterial contamination (Spooner et al. 1991).

Sources ***of sediment pollutants*** are often more widespread and more difficult to identify than sources of bacteria. For instance, sediment can originate from cropland, ditches, gullies, roads, forests, and streambanks. Sediment can also re-enter the water column as a result of scouring in streams and recirculation in lakes. A sediment survey and sediment budget are needed to identify watershed sediment sources, determine sediment delivery, and quantify the relative contribution of each source.

In the Idaho RCWP project, streambed quality and trout reproductive capacity were reduced by siltation, and transparency was reduced by high suspended sediment concentrations. At the onset of the project, agricultural sources were identified as the primary cause of reduced streambed quality. Further analysis showed streambank erosion was also a major contributor of sediment load. The influx of sediment from streambank erosion made it difficult to document the effectiveness of cropland BMPs. From the project estimates, the sediment contributions from the two major sources, streambank erosion and irrigation return flow, were similar in magnitude when the project began. In contrast, from 1987 to 1990, monitoring indicated that streambank erosion contributed two to over five times the amount of sediment added from cropland in the subbasins during the May through August irrigation season.

In the Illinois RCWP project, turbidity, siltation, and nutrients were thought to threaten Silver Lake, the water supply for the city of Highland. Sediment survey results showed that siltation was low, which meant there was little threat of rapid loss of lake storage capacity. Analysis of lake turbidity indicated that alga1 production was limited more by light than by nutrients. It was found that turbidity, which increased the cost of water treatment, was due mostly to suspended soil particles. Monitoring demonstrated that loading of fine particle natric soils and their resuspension from lake sediments was the primary factor causing lake turbidity. To target pollutant sources, the project placed special emphasis on keeping natric soils in place and reducing their delivery into the lake.

Nutrient sources can be the most widespread and the most difficult nonpoint sources to identify and quantify. Watershed sources include commercial fertilizer, animal waste, soil reserves, and atmospheric deposition. Streambeds and lake sediments can release stored nutrients into the water resource, as well.

For the Vermont RCWP project, significant phosphorus loading to St. Albans Bay originated from a point source, bay sediment, and a wetland adjoining the bay. Project area soils also contributed part of the total phosphorus load. A budget of all major phosphorus sources was needed to determine the potential for reducing lake or bay phosphorus levels.

The Minnesota RCWP project found high nitrate levels in project area domestic wells. Sources of nutrients included animal operations and cropland. The topography is karst limestone with extensive sinkhole formations. Sinkholes were thought to be a primary source of conveyance to ground water until lysimeter studies showed rapid leaching of nitrate from fertilized cropland. Further study indicated that cropland should be targeted for treatment and sinkholes should be given a lower priority.

Detecting Trends

Physical, chemical, and biological variables in the receiving water may undergo extreme changes without the influence of human activity. Understanding and monitoring the factors responsible for variability in a local system are essential for detecting the improvements expected from management actions. Simple point estimates taken before and after treatment will not confirm an effect if the natural variability is typically greater than the changes due to treatment. Therefore, knowledge of the variability and the distribution of the variable is important for statistical testing. Greater variability requires a larger change in order to determine that an observed change is not due solely to random events (Spooner et al. 1987b). Examination of historical data sets can help to identify the magnitude of natural variability and possible sources.

Management actions may not be detectable as a change in a mean value but rather as a change in variability. Platts and Nelson (1988) found that a carefully designed study was required to isolate the large natural fluctuations in trout populations so that the effects of land use management could be distinguished. They assumed normal fluctuation patterns were similar between the control and the treatment area and that treatment-induced effect could be distinguished as a deviation from the historical pattern.

Detecting Impact

Monitoring a comparable treatment and control site simultaneously is the most effective design to detect impact. Monitoring a control site provides the data to separate the impact of treatment from the variability shared by both the treatment and control. One option is to monitor similar stream stations in paired watersheds--one in which there is a management action and the other, without. Likewise, a survey of treated and reference lakes may show treatment effects. Implementation can be at the same time or staggered through time to track and account for factors (e.g., climate) that affect all lakes at once.

Using one or more references can account for system variability (e.g., biological response, life cycle, population fluctuations, and hydrologic changes), therefore reducing the time needed to detect improvement, and providingstrongerstatistical evidence of cause-and-effect. Disadvantages include the difficulty of finding a suitable reference site, the need for coordinated monitoring in both systems, and expense.

Showing Causality

To determine causality, a system of a control site and a treatment site is needed. Monitoring a control site is necessary to distinguish changes in a variable due to natural variability from those due to treatment. Mosteller and Tukey (1977) identify four conditions to show causality or cause-and-effect: association, consistency, responsiveness, and a mechanism.

Association is shown by demonstrating a relationship between two variables (e.g., a correlation between the intensity of management and the apparent reduction in pollutant loading).

Consistency can be confirmed by observation only and implies the relationship does not change in different populations (e.g., management action was implemented in several areas and pollutant loading was reduced, depending upon the effect of treatment, in each case).

Responsiveness is shown in an experiment when a treatment is performed and there is a corresponding change in a variable.

A *mechanism* is a plausible step-by-step explanation of how the management action could cause the observed change. For example, conservation tillage reduced the edge-of-field losses of sediment, thereby removing a known fraction of pollutant from runoff to a stream. The result was decreased suspended sediment concentration in the water column.

The Hypothesis

Formulating and testing a hypothesis are central to a meaningful monitoring program for detecting trends and impacts or showing causality. The hypothesis is not needed for the objectives of evaluating current conditions or documenting the water quality problem. The remaining discussion will focus on experimental design objectives. The experimental design is part of an important framework for hypothesis testing and the analysis of results.

The hypothesis is based on the monitoring objective and it provides structure to the design. The null hypothesis (Ho:) is a statement reflecting that no change or no difference can be attributed to the management action. Testing a hypothesis is based on refuting the null hypothesis in order to infer the alternative hypothesis (Ha:).Thealtemative hypothesisemulates the monitoring objective. Forexample:

Ho: The trend in mean annual dissolved oxygen concentration in Hope Creek has not increased significantly due to the upgrade at the municipal treatment plant.

Ha: The trend in mean annual dissolved oxygen concentration in Hope Creek has increased significantly due to the upgrade at the municipal treatment plant.

Ho: The number of algal blooms per growing season in Green Lake has not declined significantly due to manure management in the watershed.

Ha: The number of algal blooms per growing season in Green Lake has declined significantly due to manure management in the watershed.

Ho: No significant reductions of nutrients and sediment loading to Grand Lake have resulted from the implementation of BMPs in the Grand River watershed.

Ha: Significant reductions of nutrients and sediment loading to Grand Lake have resulted from the implementation of BMPs in the Grand River watershed.

Monitoring Design and the Use of Existing Data

Existing data may be used for problem definition, or for a pre-implementation baseline data set if the collection protocol matches the monitoring objective, design, and quality assurance/quality control required for the post-implementation data collection.

Existing data may also be used for assessing concentration/load/biological measurement variability and estimating the number of samples or the time period for the monitoring survey, based on the desired level of significance and error.

Minimum Detectable Change

To determine the required sampling frequency and evaluate monitoring feasibility, the minimum detectable change (MDC) should be estimated from historical records (Spooner et al. 1987a). The MDC is the minimum change in a water quality variable over time that is considered statistically significant. The larger the MDC, the more change in water quality is needed to assure that it was not just a random fluctuation. It may be reduced by accounting for explanatory variables, increasing the number of samples per year, and increasing the number of years of monitoring. Achieving a high level of statistical significance and power when background variability is high requires a large number of samples and a sophisticated monitoring design.

The type of change must be defined in relation to the pollutant constituent and the water quality problem in order to specify the monitoring objective. For BMPs that are directed toward reducing acute impacts, monitoring extreme events may provide evidence of change. However, tracking chronic impacts (e.g., toxins or nutrients) may require a long-term monitoring program.

4

Features of the Experimental Design

The experimental design features of a monitoring program include spatial and temporal coverage, control and reference sites, number of samples needed, preliminary sampling, and properties of estimators.

Experimental Designs

Monitormg designs that include a control and a rrearment are discussed by Spooner et al. (1985) and Spooner (1991).

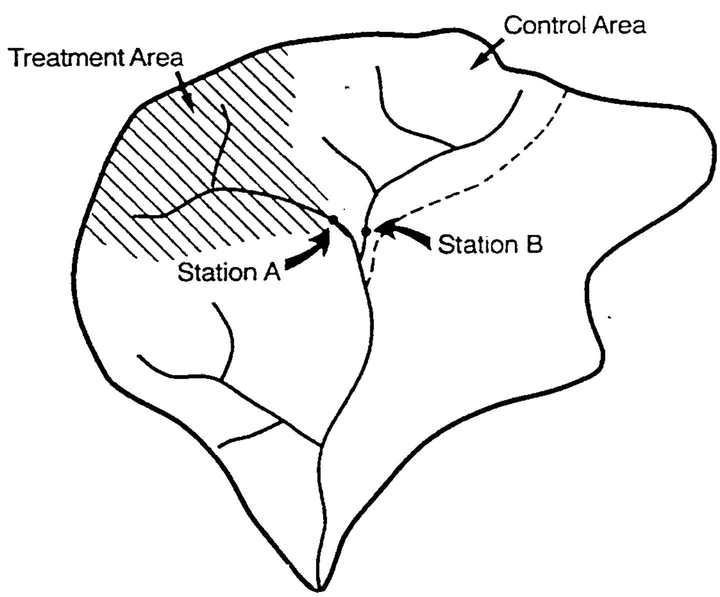
Paired Watershed Design

The most effective practical design for monitoring BMP effectiveness is the paired watershed design.

Due to the presence of an experimental control for year-to-year hydrologic variability, the most effective practical design for monitoring BMP effectiveness is the paired watershed design. This design consists of monitoring downstream from two or more drainages where at least one drainage has BMP implementation (treatment) and at least one does not (control). The paired drainages must have similar precipitation and precipitation response patterns. In addition, the paired watersheds should be relatively homogeneous with similar land use. The two watersheds do not have to be identical, but their paired watershed measurements must be highly correlated. Figure 4.1 shows monitoring stations for an idealized paired watershedstudy. Monitoring station A would be used to monitorthe control site and monitoring station B would be used to monitor the treatment watershed.

Ideally the paired watershed design has the following characteristics: a) simultaneous (i.e., paired) monitoring below each drainage; b) monitoring at all sites prior to any land treatment to establish the relative responses of the drainages (calibration or pre-treatment period); and c) subsequent monitoring where at least one drainage area continues to serve as a control (i.e., accounts for climatological variability) throughout the land treatment period.

Figure 4.1. **Paired watershed design.**



The calibration period is generally one to three years, depending on cropping patterns and the number of runoff events. The calibration period should include a full range of weather conditions to reduce the possibility that a post-implementation event will be out of the range of the calibration equation.

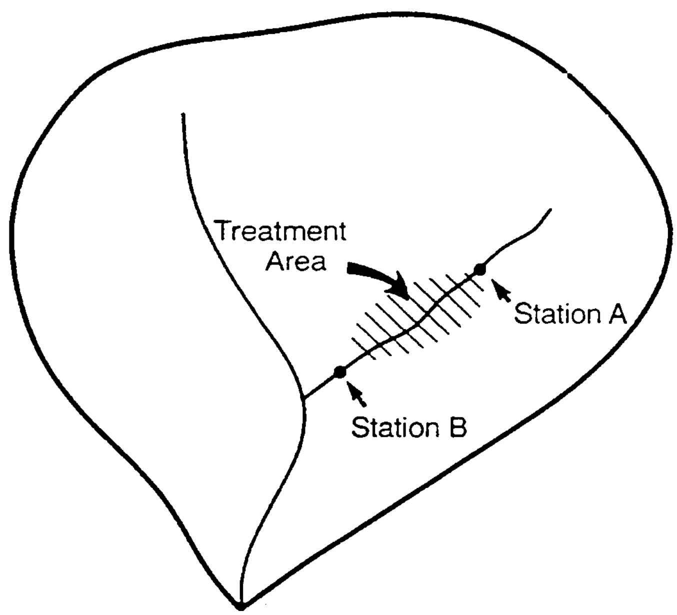
The Vermont RCWP project found that the paired watershed design was the most effective for documenting a linkage between land treatment and water quality changes on a farm field-watershed over a short (3-5 years) time period.

Upstream/Downstream and Pre-and Post- BMP

Monitoring above a site can be used to correct for varying incoming pollutant sources not related to the changes in land treatment in the study area.

Single watersheds can be monitored above and below the pollutant sources. Monitoring above a site can be used to correct for varying incoming pollutant sources not related to the changes in land treatment in the study area. Varying levels of consumptive water use between monitoring points, however, may make interpretation difficult. This technique is applicable to PS monitoring and may also be useful in monitoring the impact of NPS controls when a high correlation exists between concentrations of the pollutant over time measured at the monitoring sites above and below BMP implementation. It should be emphasized that this technique is inappropriate and ineffective unless it is combined with 'before and after' monitoring. The effect of the land treatment cannot be determined unless there is a comparison between the pre- and post-treatment period. Figure 4.2 shows an idealized upstream/downstream and pre- and post- BMP design. Station A would be used to monitor upstream of land treatment and station B would be used to monitor below land treatment.

Figure 4.2. Upstream/down-stream and pre- and post-BMP design.



Year-to-year variability in water quality variable concentrations/loads is often greater than the BMP-induced change in water quality in any given year or season. At least two to three years are required (for both pre- and post- BMP periods) to account for year-to-year variability.

In a pre- and post-monitoring design for monitonng BMP implementation effectiveness with no control watershed, the changes observed over time may be primarily due to climate and therefore very difficult to attribute to the NPS controls. To substantiate a cause-and-effect relationship, the explanatory variable can adjustforchanges in hydrologicand meteorologicvariability betweenseasons and years and should be monitored and used as an explanatory variable in the trend analysis (e.g., in analysis of covariance).

The Idaho, Florida, and Utah RCWP projects found that monitoring upstream and downstream from BMP implementation on a subwatershed scale was effective in documenting water quality improvements associated with the RCWP land treatment.

Multiple Watersheds and Pre- and Post- BMP

Comparison of Multiple Watersheds was a common design in the RCWP. This may be useful when comparingsimilarsubwatersheds, especially when combined with the ***before-after*** and/or the *above and below designs.* Although there is no experimental control, observing water quality changes of similar directions and magnitudes occurring with land treatment changes across several watersheds serves to substantiate the evidence for BMP effectiveness. For this design to be truly effective, approximately one-half of the subwatersheds need to remain untreated for the entire monitoring period to be used as comparisons. An effective design would allow for about 15 treated and 15 untreated subwatersheds over several years.

The multiple watershed approach was used successfully in the Utah, Florida, and Vermont RCWP projects. Detection of predicted water quality trends and patterns over multiple drainage areas improves documentation that the changes in water quality were attributed to BMPs.

Coverage Through Time

Baseline monitoringduringpre-land treatment implementation is usually required to detect a trend or impact or to show causality. Two years of pre-implementation monitoring and two to five years of post-implementation monitoring are typically needed. Less time may be needed for edge-of-field studies, when hydrologic variability is known to be less than typical for largeragricultural systems, or when a paired watershed design is used. Sufficient baseline data are required for impact assessment because:

- historical or baseline monitoring is fundamental to the study of the problem, system function, and variability;
- NPS control projects have difficulty detecting a statistically significant treatment effect, in part attributable to insufficient baseline; and
- adequate historical or baseline data may be the most reliable and significant design of the monitoring program if a control is not monitored successfully.

Watershed Site Monitoring

There are three spatial scales for watershed monitoring, edge-of-field, subwatershed, and watershed outlet. Criteria for selecting the spatial scale are the monitoring objective, the location and intensity of treatment, funding, and availability of sampling equipment.

Edge-of-Field

Monitoring pollutant export from a single-field watershed is the most sensitive scale since the direct effects of implementation can be detected without pollutant trapping in afield border or stream channel. Edge-of-field monitoring is also ideal for demonstrations and pilot studies. However, edge-of-field results may not be directly extrapolated to larger areas (e.g., subwatersheds).

Subwatershed

Monitoring a subwatershed by taking samples close to pollutant sources and treatment can be useful for observing the aggregate effect of implementation on a group of fields or several farms. Subwatershed monitoring networks measure the aggregate effects of treatment and nontreatment runoff as it enters an upgradient tributary or the receiving water body. Subwatershed monitoring can also be used for targeting critical areas.

Paired subwatersheds are often monitored when tightly controlled experimental conditions are desired. A pre-implementation hydrologic calibration monitoring survey of one to three years may be required. Each watershed is monitored in order to develop a precipitation-runoff model to estimate its relationship with hydrologic response and pollutant export.

Watershed

Monitoring at the watershed scale is appropriate for assessing total project area pollutant load using a single station. Depending on station arrangement, both subwatershed and watershed outlet studies are very useful for water and pollutant budget determinations. Monitoring at the watershed outlet is the least sensitive of the spatial scales for detecting treatment effect. Sensitivity of the monitoring program decreases with increased basin area and decreased treatment extent or both. In addition, nontreatment effects such as hydrologic variability and nonhomogeneous land use increase MDC.

Control and Reference Sites

Monitoring comparable treatment and *control* sites is an important feature in a monitoring design. Monitoring a control site provides the data to separate the impact of treatment from the variability shared by systems. One option is to monitor similar stream stations in similarly paired watersheds--one in which there is a management action and the other, without. Likewise, a survey of treated and reference lakes may be used to show treatment effects. Implementation can be at the same time or staggered through time to track and account for factors (e.g., climate) that affect all lakes at once.

Monitoring comparable treatment and control sites is an important spatial feature in a monitoring design.

Using one or more reference sites can account for biological or habitat variability, therefore reducing the time needed to detect improvement and providing stronger statistical evidence of cause-effect. Disadvantages include expense and the difficulty of finding a reference similar in most features except for implementation and the need for coordinated monitoring in both systems.

The reference site should be part of an ecosystem with the best attainable habitat and biological components (Plafkin et al. 1989). Reference system conditions should be similar to the treated area in almost every respect except for the treatment. The reference and the treatment system should be in the same ecoregion and their watersheds should have similar geography, soils, and land use. Best professional judgment should be used to determine if the ecosystem to be treated has the potential to achieve the quality of the reference ecosystem.

Streams and Rivers

For monitoring streams or rivers, a paired or an upstream-downstream configuration of a network or a control and treatment station should be considered. Both streams should have similar land use, be of the same order, have similar hydrologic regime, and be close enough to have approximately the same rainfall.

Lakes and Reservoirs

The lake or reservoir reference system should be similar in basin shape, size, and hydraulicdetention times to the treatment water body. Treatment and control lakes should mix and stratify similarly. Depending on the monitoring objective, other nontreatment factors such as land use, habitat, and water chemistry may be important.

A cross-sectional study of several lakes within the same region (more than one treatment or more than one reference lake or both) may be monitored to increase the chances that the impact will be detected. Carpenter (1989) discusses the importance of sufficient treatment strength and the advantage of using a network of multiple treatment and reference lakes for impact assessment. Climatic factors influencing the entire network of lakes can be tracked to improve the detection of treatment effect.

For monitoring localized problems on large lakes or reservoirs, a bay or tributary arm with similar morphometric and hydraulic characteristics may be used as a reference site; however, careful definition of differences between sites and the area1 extent of treatment effect must be determined.

Number of Samples Needed

The time between samples or the sampling interval and the number of sampling events or years of monitoring are key elements of the sampling design.

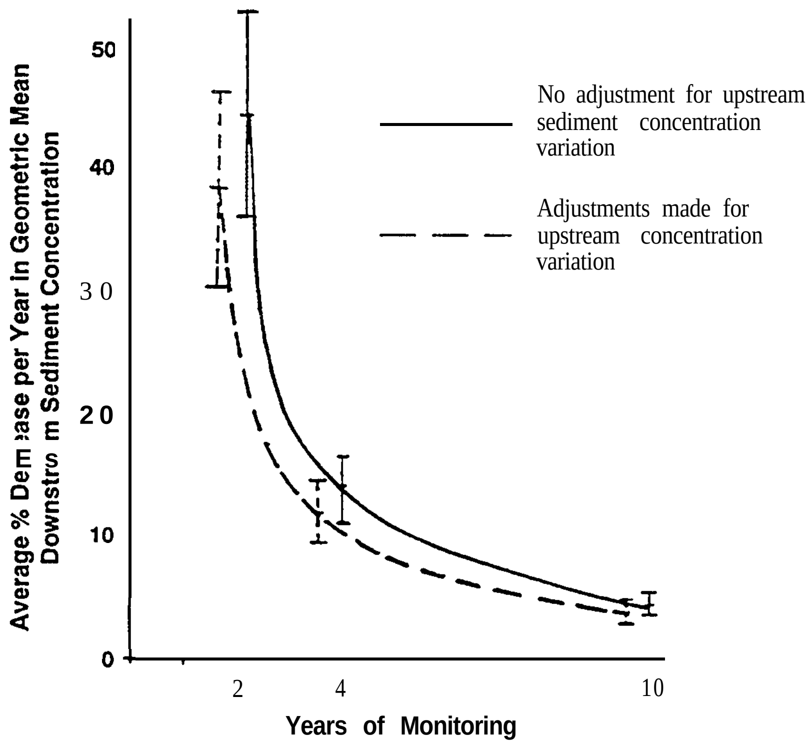
For monitoring the state of biological variables, the length of the life cycle may determine the sampling interval. Level I macroinvertebrate and fish sampling occurs generally one to four times a year, with timing adjusted for flows or reproductive cycles. Level I lake monitoring for water column chemical constituents may be every 14 days depending on the time of year and the objective. Level I grab sampling for stream chemical constituents may be every 7 to 14 days, monthly, or seasonally, depending upon the objective.

Monitoring at regular intervals increases the chance that the monitoring program can detect a trend. Sampling should be repeated within a year for systems where the temporal variability is estimated for the year or season and for a measure of its variability (i.e., mean and coefficient of variation). The extent of repeated sampling within a year is initially specified by the monitoring program objective and planned statistical analysis to test the null hypothesis. Consideration should also be given to the seasonal changes and to the life cycle for biota. Minimum sampling frequency may be two times the length of the life cycle for some biota.

Monitoring at regular intervals increases the chance that the monitoring program can detect a trend.

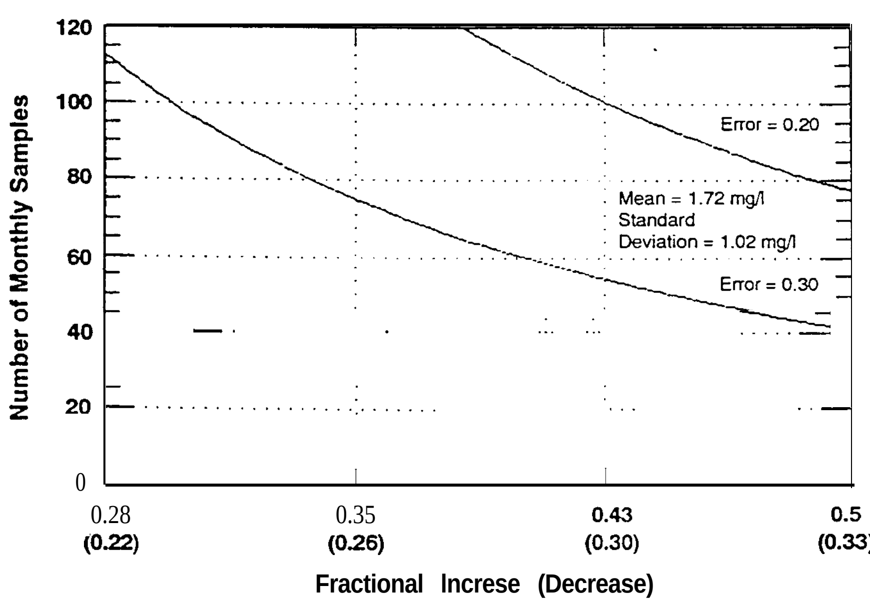
Spooner et al. (1987a) developed a method to calculate the MDC in water quality variables for three RCWP projects. The method was applied to fecal coliform data for Tillamook Bay, Oregon; total phosphorus and fecal coliform data in Snake Creek, Utah; and suspended sediment concentrations in Rock Creek, Idaho subwatersheds. The effect of the MDC with changes in the sampling interval, the explanatory variable, and the total number of sampling events can be determined. The concept of the MDC is illustrated in Figure 4.3 for a two-year, four-year, and a lo-year sampling scheme. Note the decrease in the magnitude of change in suspended sediment concentration required to detect statistical significance as the number of years of monitoring increases.

Figure 4.3 **The average percent annual decrease relative to the initial yearly geometric mean downstream sediment concentration required to detect a significant decrease over a 2-year, 4-year, and 1 O-year monitoring scheme. This annual decrease is calculated by: (total percent decrease)/(number of monitoring years, minus one). The range over all subbasins is shown. 20 samples per year are assumed.** *(Data from Idaho Rural Clean Water Program monitoring data.)*



Reckhow et al. (1989) developed a method to determine the number of sampling events required to detect a statistically significant change of a given magnitude and range of error rates. The example provided in Figure 4.4 shows the number of monthly samples needed to detect a 30% decrease in the total nitrogen concentration at the Neuse River at Smithfield, North Carolina. Along the bottom of the figure the fraction decrease is shown in parenthesis. For an error rate of 30% (0.30) the approximate number of monthly samples required to detect a 30% decrease is 55.

Figure 4.4 Total number of sampling events vs magnitude of linear trend. *(after Reckhow et al.* **1989)**



Preliminary Sampling

Preliminary sampling helps to ensure that the population of interest is being sampled and that its distribution is being evaluated. Preliminary sampling or previous testing helps avoid the problem of collecting large sets of useless data because of ineffective gear, improper sample preparation, or preservation. The target population can be easily missed. especially for biological monitoring.

Properties of Estimators

The goal for sampling is the collection of a time series of data that can be summarized for a single time period with a single estimator. Most often the time series of interest consists of either years or seasons. The estimator is the expected value, or the mean or another estimate of central tendency (e.g., the median). For regression analysis, other parameters, such as slope, may be of interest. The properties of the estimator should be considered so that it relates to the needs of the sampling program.

Statistical analysis involves testing the properties of sample estimators and their data sets. The monitoring objective, design, and the degree to which these assumptions are met by a data set determine the appropriate statistical test.

Normal *Distribution*

Knowledge of the distribution of water quality variables is important for characterization of water quality and also to determine applicable statistical techniques. In addition, much more information (e.g., spread, skewness) is contained in the data distribution as compared to only using point estimates of central tenancy such as the mean or median.

The normal, log normal, and the gamma distributions are common theoretical distributions that water quality variables exhibit. The log normal distribution may be the best for many water quality and hydrologic variables and is widely used in water quality studies. If the logarithms of the random variable are normally distributed, then the random variable itself has a log normal distribution.

No Bias

The sample estimator is a true estimate of the population. For a normal population, the sample mean and the sample median (center value of an ordered set) are unbiased estimators.

Homogeneous Variance

Variability in the y variable at any value of x is independent of x-value and is randomly distributed. The data scatter should be the same for either high or low values along the x axis.

Independence

Time correlation (temporal autocorrelation) is found when the value of one measurement is dependent on the previous measurement. If a value is dependent upon the value of a parameter at another location, then there is spatial autocorrelation. Dependencies such as these must be known and accounted for. Sequential samples taken during a storm are not independent because they are subject to a common influence, the storm flow. This must be considered when analyzing the data.

4

Coverage Through Time

Monitoring for multiple years before and after BMP implementation is essential for successful documentation of a change in water quality at the subwatershed or watershed level. Water quality and land use monitoring prior to BMP implementation is required to establish baseline data for statistical comparisons with post-BMP data. Consistent sampling frequency and sample collection procedures must be maintained across seasons and years.

Year-to-year variability is so large that at least two to three years each of pre- and post- BMP water quality monitoring are required to indicate the improvement in water quality is consistent. Expected changes that remain consistent over time improve the relationship between land treatment or land use and water quality.

Short-term monitoring is seldom effective because climatic and hydrologic variability can mask water quality changes. However, for small watersheds affected by a few relatively large pollutant sources, the required monitoring period may be shorter. Longer monitoring time periods are required for watersheds in which water quality changes occur gradually. For example, there may be a lag time for water quality changes to be observed in response to land treatment in large watersheds and lakes. This lag time may be due to a buffering effect of long hydraulic residence times and recycling of pollutants.

Experimental Design for Linking Water Quality and Land Treatment

A good experimental design for water quality and land treatment monitoring is essential to document a strong relationship between land treatment and water quality changes. Common designs include: the paired watershed design, upstream-downstream sites monitored before and after land treatment implementation, or multiple watershed monitoring.

The paired watershed design is the best **for** documenting BMP effectiveness in the shortest number of years (at least 3-5 years). This design involves the monitoring of two or more similar subwatersheds before and after implementation of BMPs in one of the watersheds. The paired drainages should have similar precipitation runoff patterns.

Matching Land Treatment and Water Quality Data on a Drainage Scale

Land treatment data must be collected on a hydrologic or drainage basis such that the land area being tracked corresponds to the drainage area served by the water quality monitoring station. Being able to match water quality data with land treatment data increases the likelihood of being able to attribute water quality changes to BMPs. The more direct the linkage, the stronger the evidence for the direct effects from land treatment/land use changes on water quality. The land treatment and water quality data bases must be collected and summarized to the spatial scale desired.

The linkage of land treatment to water quality impacts can be made at the farm field, subwatershed, watershed, or project level. The scale of monitoring is a function of the monitoring objective. In general, the larger the drainage area, the harder it is to identify and quantify the linkage. Subwatershed monitoring is the most effective for demonstrating water quality improvements from a system of BMPs. Water quality changes are more likely to be observed at the subwatershed level compared to a larger watershed level. Confounding effects of external factors, other pollutant sources, and scattered BMP implementation are minimized at the subwatershed level. However, it is important to locate a monitoring station at the watershed outlet if changes at the watershed level are to be documented

Matching the Land Treatment and Water Quality Data on a Temporal Scale

The two data bases should be related temporally. Actual implementation of land treatment needs to be recorded at least seasonally or annually. For some studies, land treatment data should be collected more frequently if the effect on water quality is more short-term (e.g., timing of manure or commercial fertilizer applications, timing of construction of a new sediment control basin or lagoon storage structure, or timing of a dairy closure).

Monitoring Explanatory Variables

Accounting for all major sources of variability in the water quality and land treatment data increases the ability to isolate true water quality trends due to BMPs. Correlation of water quality changes and land treatment changes, by itself, is not sufficient to infer causal relationships. There may be other factors not related to the BMPs causing the changes in water quality, such as changes in land use, rainfall patterns, etc. Factoring in explanatory variables yields water quality values that are close to those that would have been measured had there been no change in the climatic variables over time. In addition, the removal of variability in waierquality due to known causes, decreases the error term in the trend analyses and increases the precision of the statistical trend analyses.

All sources of variability in the land treatment and water quality data should be taken into account. Explanatory variables might include changes in animal numbers, changes in cropping patterns, other land use changes, season, stream discharge, precipitation, ground water table depth, changes in known pollutant sources, and impervious land surface. Seasonal effects may be very large due to seasonal land use changes and climatic changes.

4

Quality assurance procedures are needed to ensure data is compatible with monitoring objectives and design. Periodic review ofprotocol implementation and helps to eliminate long-term problems with the methodology or data quality.

5

Data Collection

Quality Assurance

The source of project funding will dictate the nature of quality assurance/quality control required. Projects funded by the US Environmental Protection Agency (USEPA) are now required to submit a detailed Quality Assurance Project Plan (QAPjP) for water quality monitoring (Dillaha et al. 1988). The QAPjP is a written record of plans that account for and assure data quality by specifying all data generation, analysis, storage, and reporting details. Project personnel responsibilities for assuring data quality are also documented. Clark and Whitfield (1993) provide a detailed procedural overview designed for use by the manager.

Quality Control in the Field and Lab

The standard operating procedure (SOP) guidelines that may be used in QAPjP's appendices are detailed "how to" monitoring procedures that should be developed before the start of the project. These are primary references for day-to-day operations to assure consistency through time. SOP manuals can be derived from existing local or state guidance and may be updated as needed.

Periodic evaluation of trends in land ***treatment and*** *water quality serves to trackprogress andprovide information for potential refinements. Statistical analysis with formal hypothesis testing strengthens the quantitative evaluation ofprogress.*

6

Data Analysis

A detailed preliminary analysis using scatter plots and statistical tests of assumptions and the properties of the data set such as the distribution, homogeneity in variance, bias, independence, etc., precede formal hypothesis testing and statistical analysis. From the objective and the properties of the data set, the appropriate statistical test may be chosen to determine a trend, impact, or causality.

Failure to observe improvement may mean that the problem is not carefully documented, management action is not directed properly, the strength of the treatment is inadequate, the monitoring program is not sensitive enough to detect change, or more time is needed.

For trend detection, some of the appropriate tests include Student's t-test, linear regression, time series, and nonparametric trend tests. For an assessment of impact, a careful tracking of treatment is required and the two sample Student's t-test, linear regression, and intervention time series are appropriate statistical tests. Evidence from experimental plot studies, edge-of-field pollutant runoff monitoring, and modeling studies may be used to support the conclusion of causality.

Failure to observe improvement may mean the problem was not carefully documented, management action was not directed properly, the strength of the treatment was inadequate, the monitoring program was not sensitive enough to detect change, orasufficient time has not elapsed to develop the expected changes. A mid-course evaluation, if conducted early enough, provides an opportunity for modifications in project goals or monitoring design.

Changes in sampling design may not be worthwhile unless a sufficiently long time series can be monitored in a consistent fashion. A power analysis may determine that too many samples are being taken and the number could be reduced to save money if the monitoring objectives can be met with fewer samples. If some variables are unneeded (they no longer support objective or no longer support a modified objective) or some stations do not provide sufficient additional information, then they can be dropped.

Tracking water quality trends and informing the public on progress increase the likelihood for attaining the desired level of land treatment implementation. Reevaluation of BMPs and refining the land treatment program are likely as more information is gained on the water quality problem.

7

Program Evaluation

Continual Tracking of the Treatment Program

Efficient and timely water quality and land use data analysis facilitates interim evaluation of project effectiveness and adjustments in land treatment. Short-term effects may not be detected, depending upon system response time and the detail of the monitoring program. Long-term monitoring may be required to show treatment effect.

Land treatment and other land-use changes need to be documented on a seasonal or yearly basis relative to each water quality monitoring station.

Reevaluating the Effectiveness of BMPs and Land Treatment Adjustments

When trends in water quality are absent and system variability has been adequately incorporated into the analysis, a careful review of the land treatment program accomplishments is warranted. If land treatment strength is insufficient to reduce pollutant export substantially, then there are several important aspects of the treatment program that should be evaluated. Survey current land treatment activities to assure BMPs are being continued and that they are being maintained. Land use changes near tributaries may have an important impact on total critical area treatment impacts. Also evaluate the level of treatment. Perhaps more critical area pollutant sources should be treated. In any event, the cause of the deficiency in land treatment should be determined and documented for use in developing future plans to manage the watershed.

Conveying Results to the Public

A well-informed public is an asset to monitoring and resource management activities. If citizens are aware of the problem and the need for pollution control, then they are likely to support monitoring. The public will also want to be informed of results in a timely manner. Carefully prepared newspaper articles or press releases are very effective in communicating results. Additional information should be available to interested citizens and the project manager or project personnel should be available to answer questions.

The manager in a nonpoint source (NPS) project is in a unique position of understanding the framework of a successful monitoring program and organizing and managing the resources required to meet the objective. While the manager must ask difficult questions, he must also help keep the team working together for the long-term good of the project.

8 Conclusions

This monitoring guide provides a simple framework to assist the manager in developing a program for NPS monitoring. The work begins with defining management objectives and documenting the water quality problem. Rarely does a monitoring program yield meaningful results without clear direction from carefully developed objectives and a thorough investigation of the water quality problem.

Sometimes the steps of the monitoring program cannot be taken in order. For instance, data collection may have begun even before monitoring objectives and monitoring design have been defined. Here the feedback loop can be implemented to refine the direction of the monitoring program. The manager should call a meeting of the project staff and the monitoring agency to discuss monitoring objectives and design. Even if the agency is well known for its ability to conduct surveys, the manager needs documentation on monitoring objectives and monitoring design to assure validity and to allow for modification.

The feedback loop may also be applied to other issues of oversight. The manager may be the only person involved in the monitoring program who has a big-picture perspective of the overall monitoring program. The biologist may be primarily concerned with taxonomy and the water chemistry lab director may have concerns related to instrument operation. The manager may be the only person who can regularly review project monitoring activities. Thus, the manager plays a key role in sustaining a coordinated monitoring program that is effective for its intended purpose.

In addition to establishing the framework for monitoring, this guide was also meant to be used by the manager for general reference.The guide provides enough detail so a new manager can use the guide as a training tool to improve his or her knowledge for communication with the scientist or statistician. The guide should be consulted periodically throughout the implementation of the monitoring program to check for deficiencies or the need for reallocation of effort.

9

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Glossary

bathymetric — measurements of lake basin, such as water depth, sediment depth, relief of bottom, or volume.

biomass — mass or weight of biological material.

community-an aggregate of organisms that form a distinct ecological unit. A community may be comprised of plant or animal life or a combination. The spatial scale of a freshwater community may be as vast as the pelagic zone (open water) phytoplankton of Lake Michigan or as highly localized as the algae attached to a submerged log. Since plants and animals occur on the same habitat and have many interrelations, they comprise the biotic community.

baseline — initial or existing conditions or flux before treatment or impact.

benthic — living on the bottom or at the greatest depths of a body of water.

bioassay-a test procedure that measures the response of living plants, animals, or tissues to a sample that usually contains a pollutant. For example, algae may be exposed to a predetermined concentration of atrazine in the lab or some other controlled environment. The results of the experiment may be used to estimate the potential response of the organism to stress from pesticides in the natural environment.

biota -the animals and plants that live in a particular location or region.

concentration -mass per unit volume such as milligrams per liter.

coliform bacteria-a type of bacteria that ferments lactic acid, producing a gas. Fecal coliform bacteria are found in the intestinal tracts of mammals. The presence of high numbers of fecal coliform bacteria in a water body can indicate the recent release of untreated wastewater or the presence of animals and may indicate the presence of pathogens.

conductivity — a measure of the conducting power of a solution. Expressed in micromhos per centimeter at 25 degrees C.

correlation coefficient — a ratio used to describe the fit between a regression equation and a set of data. As the correlation coefficient (R^2) approaches 1, the fit of the regression equation improves.

explanatory variables-statistical term for a variable that helps to explain the variability in the dependent variable. For instance, temperature may be an explanatory variable for dissolved oxygen because it may be used to explain part of the variability in dissolved oxygen.

designated use — use of the water resource is designated by the state water quality agency. Uses include, but are not limited to, water supply, navigation, recreation, and aesthetics.

detritus — nonliving dissolved and particulate organic material from the metabolic activity and death of terrestrial and aquatic organisms.

discharge — volume of water per unit time moving past a fixed point.

ecoregion — areas of relative homogeneity in ecological systems or in relation ships between organisms and their environments.

embeddedness (cobble embeddedness) -the amount of fine sediment that is deposited in the spaces between larger stream bottom particles.

epilimnion — uppermost, warmest, well-mixed layer of a lake during summer time thermal stratification. The epilimnion extends from the surface to the thermocline.

export — mass of pollutant lost from unit area per unit time (e.g., kg/ha/yr).

eutrophic — nutrient-rich or fertile body of water.

feedback loop -a process of nonpoint source management based on implemen tation of best management practices (BMPs). BMPs are identified through a planning process and applied by land managers for site-specific conditions. The effectiveness of a system of BMPs is evaluated through water quality monitoring. The results may be used to refine the problem statement or change monitoring or management plans.

flux- the rate at which a measurable amount of material flows past a designated point in a given amount of time.

geomorphology — the study of the landforms of the earth and the processes that shape them.

habitat — a specific type of place occupied by an organism, a population, or a community.

hypolimnion — lower, cooler layer of a lake during summer thermal stratifica tion.

impervious-a surface that cannot be easily penetrated. For instance, rain does not readily penetrate asphalt or concrete pavement and roofing and runs off rather than infiltrating.

littoral zone-the upper portion of the water column of a lake or stream that has sufficient light intensity to support the growth of plants.

load-mass inputs per time (e.g., kg/year).

macroinvertebrate — invertebrate aquatic animals large enough to be seen without a microscope. In streams and lakes these are usually immature forms of insects but also include worms, snails, clams, crustaceans, etc.

macrophytes — rooted and floating aquatic plants, commonly referred to as waterweeds. These plants may flower and bear seed. Some forms. such as duckweed and coontail, are free-floating without roots.

metrics -are generally specialized biological variables that can be combined with a rating and used in an index. Metrics are a means of quantifying individual biological attributes.

morphoedaphic index- a regression equation using water quality variables to estimate fish biomass.

morphometry-measurements of the physical structure of a watershed or water body (e.g., length of streams, slope, depth, shoreline length).

multiple regression — a regression model developed with two or more variables.

pathogen -a disease-causing agent, especially viruses, bacteria, or fungi. Pathogens can be present in municipal, industrial, and nonpoint source discharges.

pelagic zone- the open area of a lake, from the edge of the littoral zone to the center of the lake.

phytoplankton — microscopic algae that float freely in open water of lakes and oceans.

pool-portion of the channel with greater than average water depth, slow water velocity, and no surface turbulence; often wider than average.

profundal zone — the deep-bottom water area beyond the depth of effective light penetration. All of the lake floor beneath the hypolimnion.

quadrant-one section of a water body that has been divided into quarters for the purpose of sampling.

rapid bioassessment — refers to several protocols developed by USEPA and several states to examine the biological community of a stream, taking less time than conventional methods.

riffle-portion of the channel with shallower than average water, relatively high gradient, and greater than average current velocities, racing over stones to create much surface turbulence.

run — portion of the channel with water of average width, depth, and current velocity, with little or no surface turbulence.

salinity-a measure of the quantity of dissolved salts, such as in seawater.

shellfish -an aquatic animal, such as a mollusk (clams and snails) or crustacean (crabs and shrimp), having a shell or shell-like exoskeleton.

shoreline development-the ratio of the length of the shoreline divided by the circumference of a circle equal to the area of the lake. Nearly circular lakes have a low shoreline development (near 1). More elongated lakes have a larger value for shoreline development, and as the number of bays or tributary arms increases, the shoreline development increases.

step-wise regression — regression procedure where a computer introduces variables and records the corresponding correlation coefficient after each variable is introduced.

stratification -arrangement of lake water masses into separate, distinct horizontal layers due primarily to temperature. Also dissolved or suspended solids.

substrate- the material making up the bed or bottom of a stream or other body of water.

suspended solids — organic or inorganic particles that are suspended in and carried by water. The term includes sand, mud, and clay particles as well as solids in wastewater.

taxon -singular for taxa. The name applied to a group (e.g., organisms, soils) in a formal system of classification or taxonomy.

thermocline — in a thermally stratified lake, the middle iayer, characterized by a rapidly declining a 1 degree C decrease for each vertical meter of the water column.

transect — a sample area, usually in the form of a long continuous line.

trophic state — the degree of eutrophication of a lake. Transparency, chloro phyll a, phosphorus concentration, amount of macrophytes, and quantity of dissolved oxygen in the hypolimnion can be used to assess trophic state.

use attainability — a type of beneficial use analysis that is a multi-faceted assessment of the physical, chemical, biological, and economic factors that affect the attainment of the use of the water resource.

variable — term used to describe a quantity that has no fixed value. Variables include, but are not limited to, distance, mass, chemical concentration, or biological attributes.

watershed -the geographic region contributing to a water body. The area contained within a divide above a specified point on a stream. It may also be termed drainage area or drainage basin.

zooplankton — microscopic animals which float freely in lake water, graze on detritus particles, bacteria, and algae, and may be consumed by fish.

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Environmental Protection
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Nonpoint Source Control Branch
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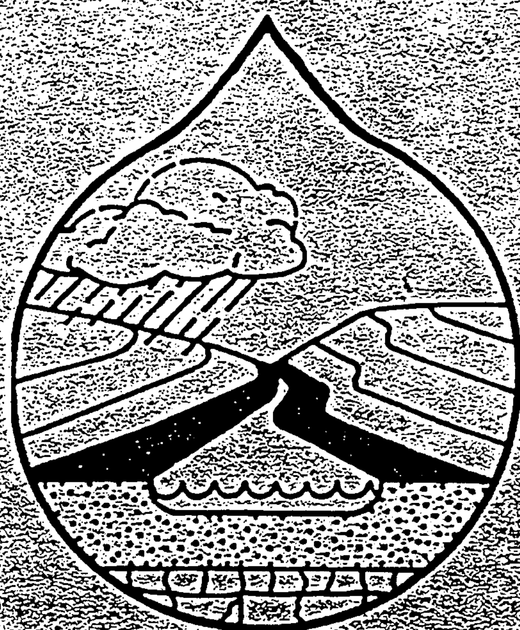
EPA EVALUATION of the EXPERIMENTAL RURAL CLEAN WATER PROGRAM



EVALUATION

of the

EXPERIMENTAL

RURAL CLEAN WATER PROGRAM

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2.2.6 Water Qualify Monitoring, Evaluation, and Reporting

Water quality monitoring, evaluation, and reporting are needed to refine and transfer NPS pollution control technology to future efforts. Monitoring documents water quality changes due to land treatment practices, whereas evaluation and reporting conveys the results. The following lessons provide an overview of the essential elements of a successful monitoring program.

2.2.6.1 Administration

- ***Lesson: NPS*** control projects whose primary objective is to document water quality changes resulting from BMP implementation should be funded only when there exists a firm long-term (six to 15 years) commitment to water quality monitoring and evaluation from a responsible agency. Effective and responsible administration is essential to maintain and support such long-term activities.

  Example: The commitments to water quality monitoring from the South Florida Water Management District in Florida; Division of Environmental Quality in Idaho; Department of Environmental Quality in Oregon; USGS in Pennsylvania; Utah Mountain Land Association of Government in Utah; tbe University of Vermont in Vermont; Department of Environmental Control in Nebraska; the Minnesota Pollution Control Agency in Minnesota; and the Department of Environment and Natural Resources in South Dakota contributed greatly to the success in water quality monitoring in these projects.

  Example: The Louisiana RCWP project was unable to determine if the implemented BMPs improved the water quality of the bayou because of the seeming lack of funding commitment from the water quality monitoring agency.

  Example: Water quality monitoring funding for the Virginia project ceased at the end of the 10-year project period As a consequence, scheduled post-project water quality monitoring was canceled due to lack of funds and the effectiveness of the BMPs will remain unanalyzed.

- Lesson: The agency responsible for water quality monitoring should be involved in selection of the project and the preparation of the proposal in order to ensure the on-going commitment of the agency to the project.

  Example: The Oregon Department of Environmental Quality was instrumental in project planning for the Oregon RCWP project They provided a high level of commitment, including water quality monitoring and data analysis.

  Example: The USGS in Pennsylvania, using money from a USEPA grant, was involved initially in project planning and continued to be the lead water quality agency for the duration of the project

- Lesson: For water quality projects, monitoring activities should be coordinated with water resource management activities. Management activities such as biological controls, dredging, or lake drawdown can significantly alter lake chemistry and hydrology, making the detection of trends due to BMPs difficult or impossible to detect.

  Example: In Virginia, ground water high in phosphorus was pumped into project area reservoirs when the reservoirs were low, thus confounding detection of phosphorus trends.

  Example: In the Iowa RCWP project, lake drawdown and rotenone treatment of carp impeded the detection of lake water quality trends, further complicating the linkage of water quality to land treatment

  Example: Stocking of trout during the project period in the Minnesota project confounded the detection of trends in monitored fisheries variables that may have been attributable to RCWP

2.2.6 Wafer Qualify Monitoring, Evaluation, and Reporting (continued)

2.2.6.2 Water Quality Problem Definition and Problem Assessment

- *Lesson:* **Carefully** defining the water quality problem is one of the most important steps for NPS pollution control and water quality monitoring. An effective approach is to implement a probiem identification and assessment monitoring program lasting six to 18 months. Problem identification monitoring uses a site-specific plan to identify pollution sources and impacts during both base flow and storm conditions; monitoring may be most effective during the seasons of greatest pollutant loading (spring runoff, snow melt) and during the season when impairments are noted (growing season algal blooms). Clearly identifying the specific pollutant and assessing the problem assists land treatment staff in identifying critical areas and targeting BMPs.**

Example: The Florida, Idaho, Nebraska, Oregon, Pennsylvania, Utah, and Vermont projects had ample visual and analytical evidence of receiving water problems.

Example: In Iowa, heavy sediment and a blanket of corn stalks covering a recreational lake surrounded by farmland helped make the problem and its source especially clear.

Example: In Massachusetts, where both intensive dairy farming on small acreages and booming residential development were taking place adjacent to an estuary containing important shellfish resources, the source of the problem needed to be more clearly documented to generate community support for project activities.

Example: South Dakota's project required several intensive monitoring programs to gain a thorough understanding of the water quality problem and its causes in complex interactions between the surface and ground water sources feeding the target lakes.

Example: In Idaho, streambed quality was reduced by siltation caused by high suspended sediment concentrations, which then contributed to loss of trout reproductive capability. At the onset of the project, agricultural sources were identified as the primary cause of reduced streambed quality. Further analysis showed streambank erosion was also a major contributor of sediment load. Influx of sediment from streambank erosion made documentation of the effectiveness of cropland BMPs difficult. Based on project estimates, sediment contributions from two major sources, streambank erosion and irrigation return flow, were similar in magnitude when the project began. In contrast_ fro5 1987 to 1990, monitoring indicated that streambank erosion contributed 2 - 5 times the amount of sediment added from cropland in the subbasins during the inigation season The problem of streambank erosion will continue to mask instream benefits from the land treatment.

Example: In the Illinois RCWP project, turbidity, siltation, and nutrients were thought to threaten Silver Lake, the water supply for the city of Highland. Sediment survey results showed that sedimentation rates were low which meant there was little threat of rapid loss of lake storage capacity. An analysis of lake turbidity indicated that algal production was limited more by light than by nutrients. It was found that turbidity, which increased the cost of water treatment, was due mostly to suspended soil particles. Monitoring demonstrated that loading of fine particle natric soils and their resuspension from lake sediments was the primary factor causing lake turbidity. In order to target pollutant sources, the project placed special emphasis on keeping cropland natric soils in place or reducing their delivery into the lake.

Example: To accurately interpret ground water monitoring results, a thorough understanding of project area geology was essential for the Minnesota RCWP project Project personnel found that performing a geologic investigation was critical even though it was time-consuming, expensive, and occurred after the start of the project The critical area and BMP emphasis were changed to address the identified ground water problem_ Monitoring plans were enhanced to consider ground water and pesticides

Example: The Florida project benefited greatly from several years of water quality assessment monitoring performed in the late 1970's by ARS and the South Florida Water Management District These data helped document the water quality problem and sources.

2.2.6 Wafer Quality Monitoring, Evaluation, and Reporting (continued)

- ***Lesson:*** Source of bacteria causing contamination in shellfish or recreational waters are generally not difficult to locate. Die-off for bacteria is relatively rapid and sources can generally be located by monitoring below suspected animal waste sources.

Example: The Utah and the Oregon projects monitored above and below dairies to determine the magnitude of the bacterial contamination.

Example: Subwatersheds with dairy *operations in the* Vermont project were monitored to determine the relative magnitude of bacterial pollutant sources. Bacteria counts decreased significantly after dairy sources were treated.

Example: The Alabama project, with few animal operations, documented dramatic decreases in fecal coliform levels in the lake as operators closed or improved animal waste management

- ***Lesson:*** Nutrient sources of pollution can be the most widespread and difficult to identify and quantify. Sources include commercial fertilizer, animal waste, soil reserves, and atmospheric deposition. Streambeds, lake sediments, and ground water can also release stored nutrients.

Example: In Vermont, significant phosphorus (P) loading to St Albans Bay was believed to originate from hay sediment, an adjoining wetland, and agricultural runoff. Area soils also contributed to the total watershed P load_ A budget of all major sources was needed to determine potential for reducing lake or bay P levels.

Example: Sources of high nitrate levels in domestic wells in Minnesota included animal operations and cropland. The topography is karst limestone with extensive sinkhole formations. Sinkholes were thought to be a primary conveyance to ground water until lysimeter studies showed rapid leaching of nitrate from fertilized cropland. Further study indicated that cropland should be targeted for treatment.

Example: Monitoring in South Dakota showed that animal operations contributed significantly to nutrients in surface water and fertilizers ap plied to cropland affected ground water.

- Lesson: Sources of sediment are often more widespread and difficult to isolate than bacteria sources. Sediment can originate from cropland, ditches, gullies, roads, forests, and streambanks and can re-enter the water column via scouring in streams and recirculation in lakes. Sediment surveys and budgets are needed to identify sources, determine delivery, and quantify relative contributions of each source.

Example: A survey of sediment sources and monitoring of streambanks in the Vermont project indicated that one subwatershed contributed the most sediment to the St Albans Hay and sediment delivery was not as much of a problem as previously thought

Example: The Tennessee/Kentucky project had high erosion rates in areas with steeply sloping cropland and targeted these areas for critical area treatment Huge gullies were also identified as significant, but sediment delivery from these sources was not estimated. Overall, the effectiveness of the critical area designation is questionable since the relative magnitude of gully and cropland sediment sources is not known.

The Illinois project found that both the watershed and lake sediments were sources of the turbidity problem in Highland Silver Lake.

Streambank erosion was a significant source of sediment in the Idaho and Nebraska projects. Identification and treatment of streambank erosion in the Nebraska project was key to documenting and treating the problem. The Idaho project would have benefited from increased emphasis *on* streambank *erosion* control.

2.2.6.3 Monitoring Objectives

- ***Lesson:*** Objectives should be clear and should provide a general guide **for the** experimental design of the water quality and land treatment monitoring program. The primary objectives of NPS watershed projects should be evaluation of use support status, trend detection, or impact assessment.

2.2.6 Wafer Quality Monitoring, Evaluation, and Reporting (continued)

■ **Lesson:** Monitoring objectives for trend detection or impact assessment should identify the water quality variable and the reason the variable is expected to change with time.

Example: The water quality monitoring objective in Florida precisely stated the water quality variabl: (total phosphorus) being monitored and the changes that should occur in that variable (50% reduction in phosphorus concentration at the project outlet). That variable was to evaluate the effectiveness of agricultural BMPs for reducing phosphorus loads to Lake Okeechobee, as measured by changes in water quality concentrations and loads in the tributaries and basin outlet.

Example: The Idaho RCWP project had realistic, quantitative goals for reducing sediment However, water quality goals also should have been developed to achieve the designated uses established by the state for Rock Creek. The lack of goals directly related to use-support hindered the initial establishment of a water quality monitoring design that could directly document progress towards use-support goals. However, the project did establish an extensive biological and habitat monitoring program that documented changes in beneficial use support in Rock Creek.

■ ***Lesson:*** Trend detection and impact assessment may be the most important objectives for long-term watershed projects_ Other objectives, such as storm event sampling for load calculations or hydrograph-pollutant relationships, may be useful; however, these objectives are auxiliary and should be addressed in addition to, not instead of, the predetermined and scheduled sampling for the primary objective(s).

Example: In *the* Tennessee/Kentucky project, the majority of the water quality objectives addressed water quality problems and the sources of the pollutants, not water quality trend detection. As a consequence, the water quality information which was gathered, although useful for identifying pollutants' sources, was unable to demonstrate changes in water quality.

2.2.6.4 Water Quality Monitoring Plan

■ Lesson: Projects should invest in the planning and design of the water quality monitoring program. The monitoring plan should be developed based on the monitoring objectives. The monitoring plan should include the monitoring design, agency roles, laboratory procedures, quality assurance and quality control, data storage, reporting requirements, personnel needed, and costs.

Example: The Vermont project is a model of how a project can plan and implement a moni*toring* program. The project implemented short-term, intensive monitoring on a field-scale to document the effectiveness of a specific BMP, while at the same time monitoring for *a* longer term on a watershed and subwatershed scale to evaluate the effectiveness of a combination of many different BMPs.

2.2.6.5 Water Quality Monitoring Designs

■ Lesson: The most (statistically) effective protocol for detecting long-term trends includes collection of samples on a regularly spaced predetermined time schedule.

Example: The Idaho RCWP project used regularly-timed sample collection (at 14-day intervals) to document a decrease in suspended sediment concentrations.

Example: The Utah, Vermont, and Florida projects used regularly-timed sampling to document water quality improvements.

Example: After changing their water quality design from trend determination to storm sampling, the Oregon RCWP project personnel found that trends were difficult to quantify from storm samples of fecal coliform data. Samples for trend detection should have been collected on a regular, predetermined schedule.

2.2.6 Wafer Quality Monitoring, Evalua tion, and Reporting (continued)

■ ***Lesson:*** Trend detection is more effective if monitoring focuses on collecting samples at a relatively high frequency and analyzing them for a small number of relevant variables. Use of the entire list of variables employed to measure general conditions in ambient monitoring programs should be avoided. Variables measured should respond directly to the implementation of BMPs and should reflect the water quality problem.

Example: Vermont project personnel indicated that they could have saved money, effort, and data storage and management by reducing the number of variables analyzed for at *some sam*pling stations.

■ ***Lesson:*** The monitoring design should include sampling an experimental control. Controls may be either a site above an installed BMP or a paired watershed in which BMPs have not been implemented.

Example: The Utah project used an up stream/downstream comparison before, during, and after BMP implementation to show reductions in phosphorus concentration below a dairy that installed a waste management system.

Example: The Idaho RCWP project effectively utilized the upstream/downstream strategy with monitoring before, during, and after BMP implementation ova a ten-year period to document the effectiveness of sediment reduction BMPs.

Example: Upstream/downstream monitoring stations were located in the tributaries and on Long Pine Creek (Nebraska project) to document water quality improvements from irrigation water management and streambank stabilization.

■ ***Lesson:*** The most effective experimental design for documenting BMP impacts on water quality is the paired watershed design, in which two watersheds with similar physical characteristics and, ideally, land use, are monitored for one to two years to establish pollutant- runoff response relationships. Following this initial calibration period, one watershed receives treatment and monitoring continues in both watersheds for **one** to two years. This experimental design accounts for many factors that may affect response to treatment; as a result, the treatment effect can be more effectively isolated.

Example: The Vermont project, which used a paired watershed experimental design, demonstrated the effectiveness of reducing nitrogen and phosphorus concentrations in field runoff by properly timing manure application

■ ***Lesson:*** Trend monitoring stations established to collect baseline data for a before-after monitoring approach must remain fixed and must be downstream from sites planned for installation of BMPs. Each station must remain fixed during and after implementation to assure a valid comparison with the pre-implementation baseline data. Baseline data should be collected for a period of time sufficient to characterize pre-BMP implementation conditions.

Example: The Virginia RCWP project had access to baseline water quality data that had been collected three years prior to implementation. This allowed for a thorough characterization of the water quality problem and targeting of appropriate BMPs.

Example: The Florida, Oregon, Idaho, Nebraska, Pennsylvania, Vermont, and Utah RCWP projects had adequate pre-BMP monitoring with fixed stations below sites planned for installation of BMP monitoring, which was essential for documenting water quality conditions before BMP implementation

2.2.6 Water Quality Monitoring, Evaluation, and Reporting (continued)

- ***Lesson:*** Post-BMP implementation water quality data must be collected for at least two to three years in order to assess the effectiveness of BMPs.

  Example: Post-BMP multiple-year monitoring, along with adequate pre-BMP monitoring, was effective in demonstrating water quality changes that could be associated with land treatment in the Idaho, Florida, Oregon, Vermont, and Utah RCWP projects. It is also expected to be a useful technique in the Nebraska RCWP, which is now conducting its post-BMP water quality monitoring.

  Example: As a consequence of reduced funding, the planned post-project evaluation of the monitoring data in the Virginia project was canceled and the effectiveness of BMPs will not be documented.

- ***Lesson:*** Long-term monitoring (six to 10 years) with grab samples taken every two weeks *is* sufficient to document water quality trends in a stream that exhibits at least a 40% change in pollutant concentrations.

  Example: The Idaho, Florida, and Utah projects documented greater than 40% change in their pollutant concentrations using grab samples taken two times per month.

- ***Lesson:*** Laboratory and field quaiity assurance and quality control (QA/QC) programs that include data evaluation and verification for precision and accuracy are essential elements of a successful water quality monitoring program.

  Example: The Alabama and Oregon RCWP projects found that QA/QC for fecal coliform analysis was especially important because of rapid die-off and the high natural variability of the data

  Example: The Idaho and Florida projects implemented extensive QA/QC procedures for their chemical and biological data field and lab collection and analysis techniques.

- Lesson: Use of constructed wells for monitoring ground water is preferable. If existing wells must be used, and are found to be contaminated, the possibility that the contamination results from poor construction or leaking rather than ***as*** a result of genera1 aquifer conditions must be considered.

  Example: In the Minnesota RCWP project, vadose zone monitoring was used to document that the high level of pesticide contamination in wells was due primarily to point sources of pesticides (commercial pesticide application services).

  Example: Sampling of irrigation and domestic wells in the Nebraska RCWP project resulted in inconclusive results, partially because of local contamination and lack of information about well construction.

  Example: The South Dakota RCWP project utilized wells constructed for the RCWP. Although expensive, the project had an effective water quality monitoring program in which the results were directly related to the RCWP.

2.2.6.6 Spatial and Temporal Considerations for Monitoring

- Lesson: Monitoring is needed at the field, farm, or subwatershed level to assess the effects of BMP systems. Short-term intensive monitoring studies of individual BMPs should be included to help understand physical processes and to provide a basis for assessing the Ionger-term, overall effectiveness of the project.

  Example: The Minnesota RCWP project used vadose *zone* sampling to determine that splitting the application of nitrogen did little to reduce soil nitrate levels

  Example: South Dakota used a master field site (research) and several farmers' field sites to determine the effectiveness of BMPs.

  Example: The Vermont project *used* monitoring at the subwatershed level to document that increasing the percentage of animals under BMP waste management decreased fecal coliform leveis in the monitored streams.

2.2.6 Wafer Quality Monitoring, Evaluation, and Reporting (continued)

■ ***Lesson:*** Reference stations characterizing attainable conditions are needed in order to evaluate the health of aquatic biota and habitat potential.

> *Example: The Idaho* RCWP established reference sites in the headwaters of the watershed in order to quantify attainable conditions for trout habitat in the project area.

■ ***Lesson:*** The start-up date of monitoring should coincide with the beginning of an easily identified annual period to avoid partial and, therefore, nearly useless collection of part of a year of data. However, establishing sampling procedures, QA/QC, and data management systems is encouraged prior to the formal data collection period.

> ***Example: The*** Vermont RCWP project team found that some of their data were unusable because of a partial year of monitoring data that did not coincide with other data

■ ***Lesson:*** Grab sampling conducted at seven- or 14-day intervals over a sir- to 10-year time period can be used on a watershed scale to document water quality changes and provide valuable feedback.

> **Example: The** Utah, Florida and Idaho projects were able to document water quality improvements using weekly or bi-weekly grab sampling in their water quality monitoring efforts.

> *Example:* Grab sampling was an integral part of the monitoring program in the Vermont project Sampling bi-weekly sampling was conducted during the summer months, sample collection frequency decreased to monthly for the winter months.

2.2.6.7 Variables

■ ***Lesson:*** Significant land use activities should be identified and accounted for in the monitoring program, particularly when such activities are located immediately upstream of a monitoring station.

> *Example: In* Alabama, sudden increases in fecal coliform levels were not understood until project personnel located a beaver dam upstream of the monitoring station.

> *Example:* In Idaho, non-cropland activities in the project area also affected pollutant loading to the impaired water resources. Activities included: expanded fish hatchery production, illegal gravel mining, chaining the irrigation canal systems to remove unwanted vegetation, forest tires, and the construction and operation of a new hydroelectric generating plant

■ ***Lesson:*** Direct measures that evaluate how well a water resource supports various uses (water supply, fish spawning, and habitat) should be used whenever possible.

> ***Example: In Minnesota, water*** chemistry ***and spring*** adult trout and fall fingerlings were sampled each year at two non-stocked **brook** locations. Results from the fish sampling demonstrated more improvement in water quality in the fish populations than the water chemistry.

> ***Example: The Idaho*** and Nebraska projects utilized biological and habitat monitoring program designs that facilitated documentation of use impairments and water quality improvements. Biological and habitat monitoring included surveys of fish and macroinvertebrates, habitat assessment, and embryo survival for trout spawning.

2.2.6 Wafer Quality Monitoring, Evaluation, and Reporting (continued)

■ **Lesson: Explanatory variables (discharge, seasons, upstream pollutant concentrations, precipitation) should be monitored to ensure accurate interpretation of monitoring results. Adjustment for hydrologic and meteorologic variables is important when quantifying impacts of land treatment or land use on regional water quality. This procedure renders water quality values that are closer to those that would have been measured had there been no change in climatic variables over time. In addition, hydrologic and meteorologic explanatory variables can be used to account for water quality variability.**

Example: Adjustments for precipitation in water quality trend analysis were made by the Florida, Idaho, and Pennsylvania projects.

Example: Stream discharge measurements were taken concurrently with water quality sampling and accounted for in the data analysis in the Florida Idaho, Maryland, Michigan, Oregon, Pennsylvania, Utah, and Vermont projects

Example: In Oregon, fecal coliform reduction initially seemed to be 70%, and staff believed their water quality goal had been reached. However, saline concentrations strongly affect fecal coliform. After adjustment of data for salinity levels through covariate analysis, fecal coliform levels had only decreased by 40% and personnel realized more dairies needed BMPs.

Example: Idaho, Florida, and Utah effectively utilized upstream pollutant concentrations to adjust concentrations downstream of land treatment to account for incoming concentrations.

■ **Lesson: When sediment is a major pollutant, at least some bedload sampling should be performed during high runoff periods to avoid seriously underestimating overall sediment loading.**

Example: Idaho RCWP project personnel believed that significant sediment movement occurs in the bedload and that they may have underestimated sediment loading by only measuring suspended sediment in the water column

■ ***Lesson:*** **Changes in land use, difficulties in tracking BMP implementation, and many other factors may hinder documentation of the impact of BMP implementation on water quality within a particular project or watershed area.**

Example: The Michigan project has been unable to document any real BMP effects due to confounding factors such as low level of BMP implementation, difficulty in assessing the effects of the sub- basin areas that do not have BMPs, large variations in sources and transport of sediment and nutrients over time, and accuracy of estimates of BMP implementation area.

Example: In the Virginia project, beneficial effects of BMP implementation may not be immediately apparent because the project began after major point sources and some nonpoint sources were removed. An improving trend was already in effect in the estuaries. Manipulation of the water supply lakes for water withdrawal and storage of pumped ground water may have confounded results.

Example: Draining of Prairie Rose Lake (Iowa project) and direct manipulation of the fish population may have obscured some water quality results. Water clarity was highest in 1982-83, following draining of the lake and restocking of fish in the fall of 1981 in an attempt to improve the fishery. Since then water clarity has deteriorated to pre-RCWP levels. Reduction in sediment delivery due to adoption of conservation practices may have improved water clarity, but algal density has increased, apparently because of greater light penetration Monitoring data are highly variable. Factors such as desorption of nutrients from bottom sediment and ground water or runoff contributions of soluble nutrients were not addressed. After correcting for both precipitation and chlorophyll *a* there is no significant trend over time.

Example: There is strong evidence that two dairy closures in the Otter Creek sub-watershed (in September 1980 and 1986) in the Florida (Taylor Creek - Nubbin Slough) project resulted in a decrease in total phosphorus concentrations in Otter Creek and at the main discharge to Lake Okeechobee from the project area (Station S-191). These dairy shutdowns resulted in a masking effect for evaluating impacts of BMPs implemented along this tributary.

Example: Upon completion of CM&E activities, the Illinois RCWP project recommended no additional field site monitoring because of the large amount of data needed to explain variability attributable to variables other than differences in BMP implementation.

2.2.6 Water Qualify M oniforing, Evaluation, and Reporting (continued)

2.2.6.8 Data Manaaement and Analysis

■ **Lesson: Data management is crucial to the success of a monitoring program. Computerized storage is essential. All data should be stored in a central project file and reviewed frequently for efficient integration and subsequent evaluation of hydrologic, water quality, and land management variables.**

Example: Much of the RCWP project data was stored in STORET, a data storage and retrieval system used by USEPA.

Example: Oregon RCWP personnel, after evaluating their data mid-project, re-analyzed their data using covariate analysis. The new results gave them a much better understanding of the effectiveness of BMPs. Subsequently, there was an increase in the number of farms targeted for BMP implementation.

Example: The Vermont RCWP project reported that quarterly analysis and review of the water quality data helped continually refine both the sampling program and the data storage systems.

■ ***Lesson:*** **Methods of data analysis should be determined early in the project planning process to ensure that data sufficient for the anticipated analysis are collected. Data management, quality assurance, and analysis techniques should be clearly defined prior to monitoring.**

Example: In Alabama, many water quality indicators were measured Some of these indicators were dropped (pesticide and nutrient monitoring except for nitrate) and others were sampled erratically. By the end of the project, only two variables (nitrate and fecal coliform) were used in the final data analysis.

2.2.6.9 Feedback

■ **Lesson: Monitoring information has been very effective in educating the public on water quality and beneficial use support.**

Example: The Utah, Florida, Oregon, Idaho, and Vermont projects had strong water quality monitoring programs emphasizing pre- and post-BMP monitoring and above- and below-site sampling. Combined with large land treatment efforts, these monitoring programs resulted in documentation of water quality improvements.

Example: In the Utah project, animal waste **management systems** reduced phosphorus concmtrations by 75% and nitrogen and fecal coliform by 40 to 90%. These BMPs reduced the impact of agricultural activity on Deer Creek, an important water supply for Salt Lake City, Utah The project served as a model project to protect valued natural resources and stimulated creation of projects in adjacent watersheds.

Example: Water quality monitoring documented that animal waste management systems installed on Oregon dairies reduced bacterial contamination of oyster beds by about 40 to 50%. Sites in Tiliamook Bay restricted to shellfishing based on Food and Drug Administration classification decreased from 12 in 1979-80 to one in 1985-86.

Example: Vermont project personnel used water quality monitoring to demonstrate that increasing the percent of animals under BMP waste management decreased fecal coliform **levels in the** monitored streams.

Example: Biological and habitat monitoring was utilized in Idaho and Nebraska to directly moni*tor* fish habitat in streams. This information was shared with the public in relation to the RCWP projects' impacts on the quality of recreational fishing in the project area water resources.

Example: Monitoring mformation was used successfully in Oregon, Alabama, Minnesota, Vermont, Idaho, Utah, and Nebraska to inform local producers and citizens of the impact the RCWP project was having on their environment

■ **Lesson: Water quality monitoring can provide feedback in defining critical areas needing priority treatment.**

Example: Water quality monitoring was utilized in the Utah, Nebraska, and Florida projects to identify critical areas needing high levels of attention for land treatment, water quality monitoring, and evaluation of water quality changes.

Appendix 2.4A

Evaluation of Methods for In Situ Monitoring of Releases from Hydro power Projects

by Michael C. Vorwerk, William E. Jabour, and Joe H. Carroll

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Evaluation of Methods for In Situ Monitoring of Releases from Hydropower Projects

by Michael C. Vorwerk, William E. Jabour, and Joe H. Carroll

Purpose

This technical note describes and qualitatively compares methods for in situ monitoring of release water quality from hydropower projects.

Background

The areas immediately downstream of hydropower projects are of extreme importance for water supply, recreation, navigation, and aquatic habitat. Increased environmental awareness and concern regarding the impacts of hydropower releases on downstream water quality have resulted in the need for increased monitoring. However, these same areas can present difficulties in effective and representative monitoring. To address these difficulties, a variety of specialized monitoring techniques are presently in use by the U.S. Army Corps of Engineers. The resulting data aid resource managers and hydropower operators in managing projects to minimize detrimental downstream environmental effects while maintaining optimum generation schedules.

Many factors must be considered when designing a hydropower tailrace sampling scheme. Sampling falls in two broad categories—manual and automated. Manual sampling, the most common type of data collection, includes all modes of sampling conducted by individuals with hand-operated equipment. Automated methods of sampling require equipment that can log real-time data independent of a human operator. A description and evaluation of both categories of sampling currently in use will be presented in this technical note.

These procedures have been used during evaluations of releases from Savannah River reservoir projects, St. Stephens Powerhouse, Bull Shoals Dam, sites within the Charleston, Little Rock, and Tulsa Corps districts, and other sites throughout the country.

The first step in monitoring water quality is gaining an understanding of the gradients and dynamics of the parameter of interest. Some parameters, such as temperature, are relatively conservative and change relatively slowly. Others, such as dissolved oxygen, can change quickly as the result of mechanical aeration, moderately fast due to biotic activity and chemical oxygen demand, or slowly from diffusion and temperature-related effects. Thus, the effectiveness of a monitoring location in meeting the needs of a manager depends greatly on the dynamics of the parameter of interest.

Typically, the principal parameters of concern in hydropower release water quality are temperature and dissolved oxygen concentration. Other parameters sometimes of interest include specific conductivity, pH, and turbidity. Both manual and automated sampling methodologies are effective in monitoring each of these parameters. In many situations, both methods are necessary to fully evaluate the release from a project.

Other important considerations include safety of technicians during calibration and use of the system and cost. Ideally, the system should be constructed with minimal cost, take advantage of the natural features of the dam and tailrace, and incorporate readily available off-the-shelf equipment and supplies.

A number of manufacturers offer equipment designed for water quality sampling. Equipment ranges from a basic instrument measuring only temperature and dissolved oxygen concentration with no logging capability, to extremely sophisticated models offering multiparameter monitoring capabilities that can be deployed remotely and can log data for extended time periods.

Another critical decision is to determine whether a manual or remote sampling strategy will be most beneficial. Through an examination of both methods, one can decide whether one, or a combination of both, is most appropriate for the specific site and questions to be resolved.

Manual Sampling

Description

Manual sampling, whether done from the shoreline, bridge, or boat, is the method employed by most individuals and resource agencies in determining water quality conditions in lakes, rivers, and streams. Advantages of manual sampling include the possibility of examining many regions of questionable water quality within a large sampling area. Manual sampling can determine the origin of detrimental water quality, refuges of good water quality, and the vertical, horizontal, and longitudinal progression of water quality zones. Also, a single sampling instrument can be used to determine water quality throughout the entire study area, which is beneficial to those with financial restraints.

The equipment used for manual sampling can be as simple as a hand-held thermometer, but typically a multiprobe water quality sonde is used to provide greater information. A multiparameter sonde can be used to profile multiple depths and provide near-instantaneous measurements of temperature, dissolved oxygen, specific conductivity, and pH. Equipped with a waterproof cable, the sonde is used to sample releases, tailraces, tailwaters, and reservoirs.

Prior to actual fieldwork, development of a carefully designed sampling plan is of the utmost importance. The plan should include a general survey of the study area, with more detailed work to answer the questions being considered. One component of a hydropower release monitoring study is to collect information on conditions in the upstream reservoir, typically through vertical profiling of the water column immediately upstream of the power intake openings. This allows the manager to examine the water quality conditions of water entering the dam, prior to release. Profiles along the upstream face of the dam will reveal any lateral heterogeneities in the lake that might result in variance in releases from different units.

Downstream manual profiling of hydropower releases can require different sampling approaches, depending on the information needs of the resource manager. Fixed-location temporal sampling requires the collection of multiple samples at a given point over a period of time. This affords an opportunity to observe rapidly occurring or short-term changes at a fixed location. Fixed-parcel temporal sampling requires the observer to sample the same parcel of water over time. For releases, this typically involves deploying an inert marker in the stream, then drifting along with the marker in a boat, and repeatedly sampling the same parcel of water over a period of time.

Spatial sampling involves the selection of stations in a longitudinal or lateral arrangement so that spatial patterns of water quality can be identified. This spatial array can then be sampled simultaneously to show the distribution of water quality throughout the region (a "snapshot" of water quality) or temporally to show the change or travel of some water quality parameter.

Successful Example

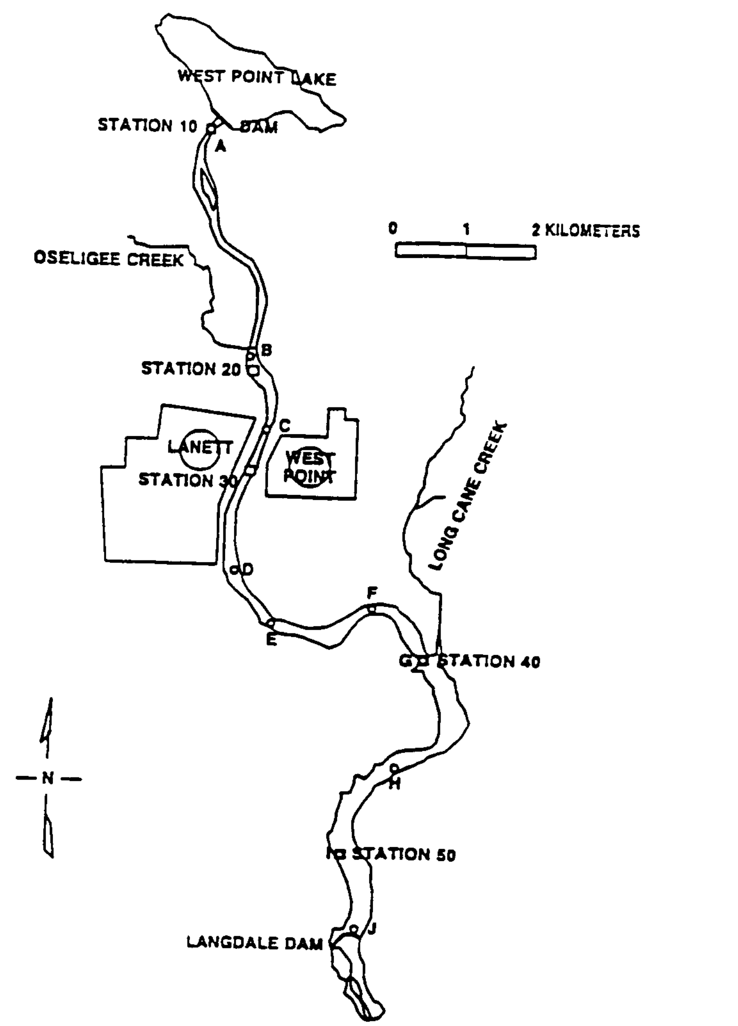
Work conducted in the tailwater downstream of West Point Dam, on the Georgia-Alabama border, illustrates the variety of sampling methodologies often necessary to answer release water quality questions (Figure 1). The study objectives were to determine the dynamics of water quality constituents in West Point Lake releases (Ashby, Kennedy, and Jabour 1992). Because of the variety and short time span of the studies required to explore the water quality of the release, it was determined that manual sampling was the best method for obtaining the required information. Automated sampling would be too costly and would not provide sufficient flexibility to conduct the various studies.

Vertical column water quality profiles of temperature, dissolved oxygen, pH, specific conductivity, and samples of other chemical parameters were collected in the West Point Lake forebay. These measurements provided information on inputs into the dam, and subsequently in the release and tailrace.

Individuals positioned at stations along the river sampled the tailrace prior to, during, and following release. Samples were collected at predetermined time intervals over the release cycle and included measurements of the above in situ parameters as well as water collection for chemical analysis. This sampling strategy provided "snapshot" records of water quality over the length of the tailrace, temporal records of change at each specific station with time, and temporal records of change of the spatial distribution of water quality. Thus, longitudinal and temporal trends in water quality were effectively monitored.

These samples showed the temporal and spatial degradation of water quality during release and the return to ambient conditions of water quality during release. The changes were primarily due to decreased dissolved oxygen concentrations in the water released from the dam, and subsequent re-oxygenation of water throughout the reach of the tailwaters (spatially) and throughout time (temporally).

A second team profiled surface-to-bottom water quality conditions along the downstream buoyline by boat, investigating lateral variability during release. Though the turbulent nature of tailwaters lends itself to being completely mixed, near-dam tailrace water quality often reflects lateral heterogeneities present in the forebay waters.



Legend: Numbers represent spatial sampling locations. Letters represent time of travel temporal locations.

Figure 1. Manual sampling locations for Chattahoochee River below West Point Dam

Still another team conducted a time-of-travel study, drifting downstream at the same rate as a parcel of water. Through close interval sampling of in situ parameters and chemical constituents, changes within that parcel of water were recorded over time and distance.

The West Point Dam study illustrates several of the many release studies that can be undertaken using manual sampling techniques. The primary disadvantages of manual profiling are the labor-intensive nature of the sampling and the fact that the data are taken intermittently. Personnel must be present for data to be collected. When one is concerned with trends over a season, or a longer period, it is often difficult to obtain a sufficient number of manual observations. Manual sampling is of great value in determining water quality at multiple depths and locations, such as was desired in the above study. However, it is not the best tool if close-interval or nearly continuous sampling is desired.

Automated Remote Monitoring

Description

After determining the immediate area of concern within the tailrace through manual sampling, the resource manager will often require a continuous record of real-time water quality data as remedial procedures are employed to improve conditions. As these remedial procedures are implemented, a data set of ambient water quality conditions over time is required. Because of the close intervals required and the necessity of around-the-clock measurements, manual sampling techniques typically prove inadequate. In this situation, the best choice is an instrument that is capable of measuring the desired parameters and logs data remotely. The principal advantage to this design is in the ability of the instrument programmer to determine the desired sampling interval and the overall sampling period. Further, the operator can deploy and recover the data logger as a one-time occurrence, while still collecting a near-continuous data set. This freedom is extremely advantageous.

Successful Example

The tailrace of St. Stephens Powerhouse, located on the Cooper River Rediversion Canal in South Carolina, experienced midsummer fish kills during periods of nonoperation. These kills were believed to be caused by insufficient dissolved oxygen concentrations in the warm, nutrient-rich, and highly productive waters. Manual profiling was used to explore the dissolved oxygen dynamics in the tailwater. Data showed that anoxia developed within the near-dam bottom waters and progressed vertically and longitudinally, eventually encompassing the entire tailrace. This anoxia formation ultimately resulted in near-elimination of what had been a thriving tailrace fishery.

Analysis of the poor water quality conditions resulted in a monitoring and remediation plan. The tailrace was monitored daily via manual sampling from the wing wall near the powerhouse. When oxygen concentrations decreased to less than specified levels, the operator released the more highly oxygenated forebay water to flush the poor water from the canal. The volume of water released was equivalent to the volume contained in the canal, resulting in a near-complete replacement of water within the tailrace. The desired result was achieved; dissolved oxygen concentrations increased rapidly within the study area.

Manual sampling revealed that changes in dissolved oxygen concentrations followed a diel cycle, with concentrations reaching a maximum in midafternoon during peak photosynthesis and a minimum during the early-morning hours of minimal photosynthetic activity. Thus, the most critical periods occurred when personnel were unavailable for manual sampling. This resulted in the decision to install an automated remote monitor system.

The system was installed in a wet well on the wing wall of the powerhouse, with the water quality sonde approximately 1 m above bottom. The sonde was wired into the control room to a PC used to operate the sonde and to store data. Data were recorded at 1-hr intervals. Using this system, the nature of the diel fluctuations of dissolved oxygen dynamics was quantified. The resource manager found that daily fluctuations in dissolved oxygen concentrations were as great as 4.0 mg/L during periods of nonoperation, that is, periods where dissolved oxygen concentration was affected only by natural processes. This determination would have been difficult to achieve through manual monitoring techniques.

The continuous record of dissolved oxygen concentrations allowed the development of a remediation strategy dependent on the actual trends in dissolved oxygen and not on diel fluctuations. A plan was implemented to release lake waters when the dissolved oxygen concentration decreased to less than a specified concentration for a period of 8 hr or longer. This provided enough time for natural cycling to correct any deficit, while still remediating if a deleterious trend in water quality was detected. The details of this system are presented in Water Quality Technical Note CS-01 (Vorwerk and Carroll 1995).

This example shows how a single automated monitor system can be used to reflect the water quality of a large area. Because the area being sampled is at a fixed location and depth and comprises only a small percentage of the entire sample area, the utmost care must be used in determining the location and depth of the remote logger, that is, the representativeness of the sampling location. To determine trends over a larger areas, often more logging instruments must be used.

Representativeness of a Sample Location

Automated remote monitoring deployments inherently require a fixed sample location. Therefore, instrument location is critical to ensure that sampled water is representative of the body of water in question. Manual sampling procedures are most often used to determine this location.

Determining the representativeness of various potential monitoring locations is typically the most difficult task for a resource manager. Experience in manual monitoring provides much insight into finding representative locations. This technical note explores many possibilities, illustrating locations representative and nonrepresentative of releases. It is essential to collect data that are not biased, that is, data collected from water that is not release water but a mixture of release water and some other water. Some sources of this bias are listed below.

- Monitoring release from one unit when several are operating, with lateral heterogeneities existing in the forebay.
- Collecting water within the dam from a location that does not provide completely mixed sample water (heterogeneities caused by vertical stratification in the forebay).

- Collecting samples downstream of the dam which are affected by eddy currents returning downstream (not release) water to the monitor location.
- Monitoring the release from a location where all dam-induced processes are not complete (for example, turbine aeration and boil aeration).
- Monitoring the release from a location distant enough from the dam that photosynthesis and respiration influence the sample water. In shallow tailwaters, primary production can contribute large amounts of oxygen to the release.

In some cases, the optimum location (that which best represents the release or answers the question of interest) is not feasible for deployment because of limited access or equipment constraints. In these situations, careful consideration must precede the selection of an alternate location. The following discussion illustrates possible locations through short case studies and examples. The advantages and disadvantages of each deployment and equipment type are discussed. The locations include lake forebay, penstock, draft tube, and tailwater deployments (Figures 2 and 3).

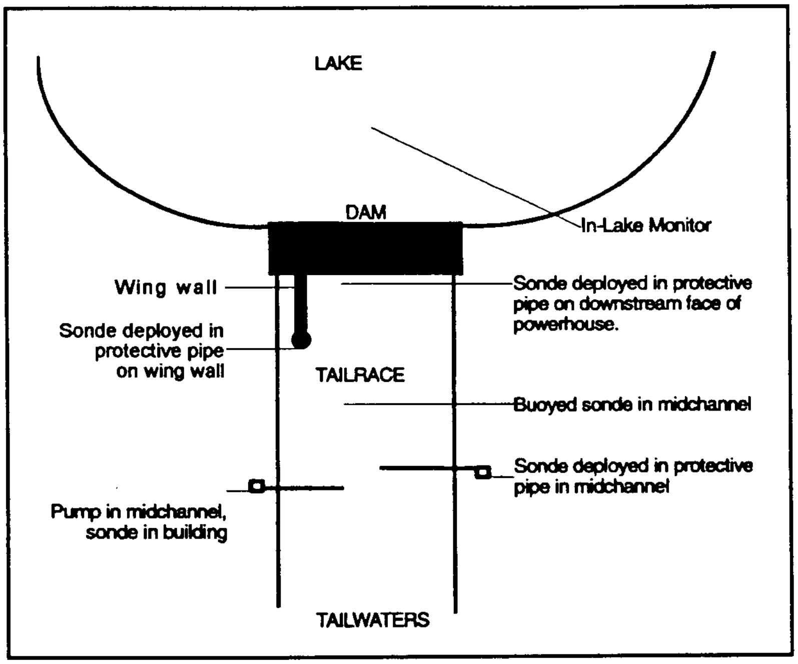


Figure 2. Plan view of dam monitoring locations

In-Lake Logging Units

In-lake logging units can be used to determine water quality conditions in the near-dam region of a lake. These data reflect water quality conditions prior to release and can be used to predict release values. A typical setup includes one or more water quality sondes measuring temperature, dissolved oxygen concentration, specific conductivity, and pH. The sondes are attached to a cable suspended from an anchored buoy. This system can provide a continuous record of water quality conditions in the forebay. However, one drawback is the lack of accessibility to the sondes for data downloading and maintenance, which results from the need for a boat and windlass large enough to retrieve the buoy, anchor, and sondes. Also, the operator cannot access real-time data.

Because in-lake water quality changes slowly (scale of days to weeks), it is typically adequate to use a boat crew and manual sampling to determine forebay conditions on a routine schedule. This manual system is used at Richard B. Russell Lake to provide data for predicting release dissolved oxygen concentrations. A more versatile but costlier alternative is to use a radio-linked data transmitting station mounted on a buoy. The water quality sondes can be connected to the radio transmitter, which allows the operator to view real-time information and to transfer data.

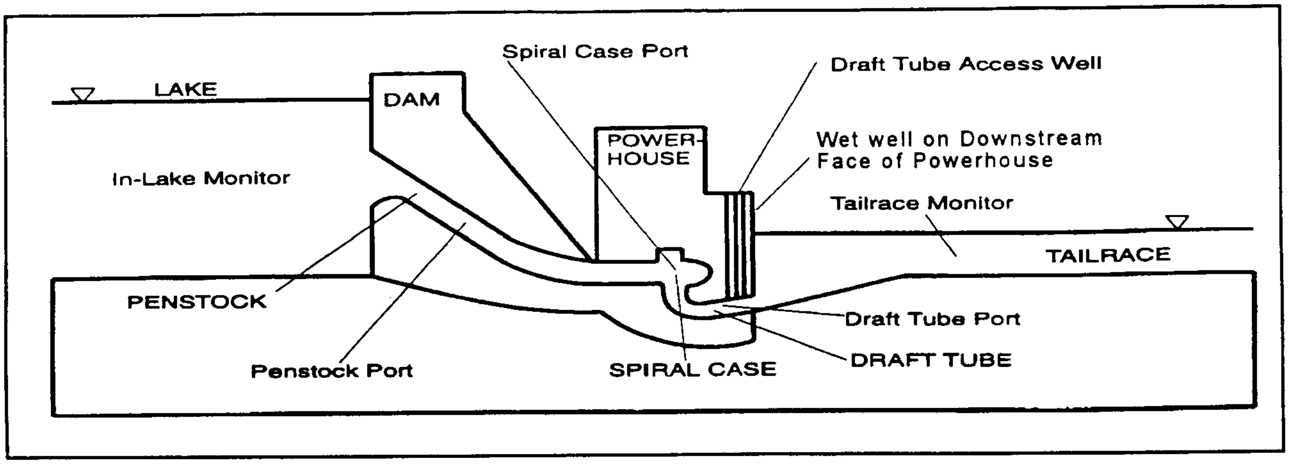


Figure 3. Cross section of dam monitoring locations

In-Lake Automated Profilers

In-lake automated profilers developed by Duke Power are presently undergoing testing (personal communication, John Knight, Duke Power, Huntersville, NC). These prototype units are self-contained lake-profiling, data-recording, and transmitting stations. These stations automatically lower a water quality sonde to specified depths and intervals, providing a continuous record of lake water quality conditions. The units are connected to cellular phones, allowing data to be downloaded remotely via modem. Thus, real-time in-lake water quality data are available. While this type system may become standard in the future, current drawbacks include no commercial production, significant initial purchase costs, and vandalism problems.

Tapping Water from Penstock

The first post-dam locations to consider for sampling are the dam's penstocks, if equipped with ports for sampling water. These ports can be plumbed to water quality sondes equipped with flow-through cells. This location is useful in determining penstock intake water quality conditions prior to any effects, such as turbine venting, that result in increased dissolved oxygen concentration in the downstream releases. However, some studies performed by Jim Ruane of the Tennessee Valley Authority and Steve Wilhelms of the U.S. Army Engineer Waterways Experiment Station have shown that if the forebay waters are not homogeneous, as is the case during stratified conditions, water sampled from the penstock may not be well mixed and thus not representative of the release. The taps are plumbed into a water quality sonde flow-through cell. The sonde can be connected to a data-recording and sonde control computer, or to a radio or satellite link if the dam is a remote site.

This type of monitoring system has been installed at Bull Shoals Powerhouse. It provides information about intake water quality prior to turbine venting. The access provided to real-time data is often critical in maintaining the downstream White River trout fishery.

Tapping Water from Spiral Case

Taps from the spiral case can provide information about near-turbine water quality. Often, cooling water for the generators is drawn from the spiral case. A typical installation directs water from a cooling line into a water quality sonde flow-through cell. Because of the mixing effect of the turbine and the additional travel time or distance from the face of the dam, water is assumed to be well mixed by this point and representative of the intake water. In the absence of turbine venting processes, this water should be representative of the release. However, if turbine venting is occurring, the effects of aeration are not complete by the time the water passes through the spiral case. This installation also provides easy real-time access to water quality information. This system was used at Richard B. Russell dam to monitor release dissolved oxygen concentrations.

Plumbing to Central Location

Water from the cooling line of each unit (as in the above example) is plumbed via solenoid-controlled valves to a central mixing chamber. The solenoid switches allow waterflow from each unit only during turbine operation. Because units contribute water to the mixing chamber only during respective turbine operation, a representative release can be sampled through this installation. Thus, any lateral heterogeneities present in the forebay are proportionally sampled. Drawbacks to this system are the cost and time necessary to install the piping, solenoids, and mixing chamber. Benefits, beyond the laterally representative sampling, include the ability to use a single sonde to monitor the release from all units. This eliminates any cross-calibration problems that could occur if multiple sondes were used to monitor multiple units. Installation costs, therefore, could be offset by moneys saved in purchasing a single sampling instrument. Further, since the operator must communicate only with one sonde, data collection and communication are minimized.

As in the above example, this method is not appropriate if any turbine venting or other water quality alteration occurs downstream of the turbine. This setup allows real-time data access and is presently in use at Richard B. Russell dam.

Tapping Water from Draft Tube

Water can be tapped from the draft tube, typically through ports immediately below the turbines. The water is plumbed to a sonde with a flow-through cell and passed to a drain. Because of the proximity of the turbine, travel time is insufficient for changes due to turbine action (for example, turbine venting in the water). This location is typically accessed in the penstock gallery, and thus the damp environment may be inhospitable to electronic equipment. Because of these drawbacks, this location, while allowing real-time access, is not recommended for most purposes.

Draft Tube Access Port

The draft tube access port is located on the draft tube deck. This port is designed to allow access to the draft tube after dewatering. Using a wet well, a sonde can be deployed in the access port and used to record water quality of the release. This location is sufficiently distant from the turbines so that most changes due to turbine venting can be detected. The sonde is typically wired directly into the powerhouse, where a data collection and sonde control computer is located. Thus, the operator has real-time access to information. The advantages of this location include ease of installation and access, representativeness of release water quality, and relatively low cost. Drawbacks are that aeration due to post-powerhouse processes, such as boil or weir aeration, is not measured. This system, in conjunction with a penstock monitor, is used at Bull Shoals Dam to determine the efficiency of turbine venting.

Downstream Face of Dam

Some post-powerhouse processes can be monitored by mounting a protective pipe vertically on the downstream face of the dam. The lower section of pipe is perforated to allow water access to the sonde sensors. Use of a pipe, instead of strapping the sonde to the face of the powerhouse, allows the sonde to be easily retrieved. In this installation, the sonde is lowered into the pipe and wired directly into the powerhouse, where a data collection and sonde control computer is located, allowing real-time data access. Drawbacks include a relatively difficult installation (divers must attach the wet well to the powerhouse face) and nonrepresentative data during periods of nonoperation of immediately adjacent units. When generation is composed of units not adjacent to the wet well, swirling eddy currents of tailrace or tailrace/release water may be measured. The tailrace of Richard B. Russell Lake is presently being monitored with a string of thermistor cables located in a wet well mounted on the downstream face of the powerhouse.

Sonde Deployed in Midchannel

A data logging sonde can be deployed, via buoy and anchored cable, in the full flow of releases. If a sonde is located downstream a sufficient distance, a representative portion of each releasing unit may be monitored. If the release does not fill the channel (plug flow), return currents can be entrained into the release causing the sonde to sample a mixture of release and other water, that is, the sonde monitors nonrepresentative water. Because of the midchannel location, this deployment necessitates a boat for retrieval and data downloading. Retrieval may be difficult or potentially hazardous during release, and real-time data access is not possible. The greatest advantages are ease of deployment and low installation cost, making this type deployment desirable for limited budgets and short-term studies.

Sonde in Protective Pipe on Wing Wall or Bank

A sonde can be deployed in a protective pipe mounted on a wing wall of the tailrace. This deployment allows the resource manager to monitor water in which most post-powerhouse effects (boil aeration, turbine venting) have occurred. The sonde communications cable is typically wired into the powerhouse, where real-time data can be accessed by the operator. This location can also be used to monitor tailwater conditions during periods of no release. One drawback of this location is that, if multiple turbines are present, the water quality of the unit nearest the wing wall may be the only one accurately monitored. Thus, any lateral heterogeneities in release would not be represented. This location can also be affected during generation by eddy currents when the unit nearest the wing wall is not operating. A wing wall deployment is in place at St. Stephens Powerhouse (as detailed earlier in this technical note) and at Norfolk Dam.

Sonde Deployed in Protective Pipe in Tailwaters

For this deployment, the sonde is placed in a near-horizontal protective pipe extending into midstream. The end of the pipe is perforated to allow water to flow across the sonde sensors, while protecting the sonde from physical damage. The data cable runs out of the pipe to a terminal that is housed in a weatherproof case. The terminal can be satellite or modem linked to a data-recording computer. Another option is to run the data communications cable directly to a computer that is housed in a nearby structure or building. The computer can be remotely accessed via modem for real-time data. The advantages of this deployment are that the sonde is deployed centrally in the current, and the water sampled is representative of the tailwaters. The disadvantages are difficulties in deploying the sonde, increased fouling of the probes, and risk of vandalism. Further, if return eddy currents are present or the sonde is not in full flow, sample bias will be recorded. This type of system is in place in the White River, below Bull Shoals Dam.

Sonde Deployed in Building—Water Plumbed to Unit from Midstream

This deployment involves a pipe extending into the tailwater with a submersible pump deployed at its base. The pump is plumbed into a building, where the sonde (fitted with a flow-through cell) and the data-recording computer are located. Sample water is pumped from a point in the channel assumed to be representative of the tailwaters. The drawbacks of this location are long-term pump maintenance and possible bias of sample water. Bias can occur if the pump location is nonrepresentative or if ambient conditions affect transported water prior to measurement by the sonde. This method is relatively secure from vandalism, and because the computer can be connected to a phone line, real-time remote data are available. This system is in use at J. Strom Thurmond Dam and Hartwell Dam on the Savannah River.

Communications for Automatic Remote Monitoring

Communications, relaying the collected data to the database, plays a central role in automated remote monitoring systems. Communication can be accomplished using either one- or two-stage processes. In one-stage communication, the data are transmitted via cable from the water quality sonde to the user. For two-stage communication, information is first transmitted from the water quality sonde to an interim data collection point, such as a computer or relay station. This information is in turn transmitted to users via modem, radio link, or satellite link.

Strategies

One-stage communication can be quite simple. A logging sonde or other water quality instrument can be deployed to log data. At the end of the study period, the sonde is retrieved and downloaded. This strategy is best employed for short-term studies. A second strategy is to connect a computer to a sonde using a data communications cable. This allows the computer to control the sonde and record the data. This computer might be located in the operator's office or control room where the operator can query the sonde for real-time data.

Two-stage communication allows greater versatility. If the data-recording computer is at a remote site, for example, a remotely operated dam, modems may be used to communicate from the monitor site to the central control room. Using this method, an operator can access real-time information, monitoring the releases from several remote sites from a central location.

Radio links can also be used for two-stage communications. Commercially available radio links can be used to control sondes and send data to a central receiving station, which stores the data. A similar method employs satellite linkages to transmit data to a central location. While these two methods are necessary for remote applications and applications having large amounts of electromagnetic interference (limiting the ability to use wire to carry the signals), their cost is substantially greater than modem communications discussed in the above paragraph.

Interference

When data transmission wires carry signals long distances (>15 m), electromagnetic interference can cause signal loss; weak, garbled signals; and incorrect information. This problem is often extreme in monitoring hydropower releases because signal transmission wires are often located near areas of high electromagnetic radiation (generators, switch yards, high-voltage transmission lines, and transformers). Several potential solutions exist, which vary in cost and installation difficulty.

A shielded cable can be used instead of the normal data transmission cable (typically telephone line). A greater degree of protection can be gained by enclosing the cable in grounded metal conduit. A second method is to use fiber optic cable and modems. With fiber optics, the signal is carried by light and thus is not susceptible to magnetic interference. A third method is to use commercially available radio links, which have built-in error correction capabilities in the software.

Depending on the severity of interference, one of these methods should be appropriate. A good strategy is to begin an application with shielded cable, the least expensive solution, and then employ a more expensive fix as necessary.

Conclusions

An ideal sampling plan for hydropower release monitoring would include both manual and remote methods. However, if a compromise must be achieved, the resource manager must determine whether the release water quality problem requires short-term intermittent or long-term continuous data sets. Manual sampling is valuable when used in an exploratory manner. Manual sampling can determine if a degradation of water quality exists, the location of the worst and best water quality, and any gross changes with time or operation schedule. While labor intensive, manual sampling affords the best overall view of a system while recognizing the limited nature of the data due to the number of measurements. Automated remote monitoring is the best choice when a continuous record of water quality is required. A more thorough analysis of hydropower release conditions can detect short-term changes (daily or during project operation), as well as long-term changes in water quality (over a season or year). However, due to the fixed nature of automated sampling, the responsible individual must be absolutely certain that the data logger is placed in a location where representative water will be measured. Many factors must be considered prior to proper implementation of a sound and appropriate hydropower release sampling strategy.

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Point of Contact

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WATER OPERATIONS TECHNICAL SUPPORT

APPENDIX 2.4B

Remote Monitoring of Hydroprojects: Design, Installation, and Verification of Remote Monitoring Systems

by: John W. Lemons, Michael C. Vorwerk, Joe H. Carroll, and William E. Jabour

US Army Corps
of Engineers

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Remote Monitoring of Hydroprojects: Design, Installation, and Verification of Remote Monitoring Systems

by John W. Lemons, Michael C. Vorwerk, Joe H. Carroll, and William E. Jabour

Purpose

This technical note describes methods for monitoring water quality at hydroprojects. A water quality manager can apply the techniques described herein to design a site-specific water quality monitoring system that provides information for water quality problem-solving.

Background

Remote monitoring systems are important tools for lake managers, hydropower operators, and others concerned with hydroproject-influenced water quality. Remote, automated water quality monitors provide temporal data sets that are used for determining water quality trends under various operational and seasonal conditions. Data collected via remote monitors can be used to identify areas of management concern and are valuable for developing and calibrating predictive models.

The usefulness of data collected by remote monitors depends on how effectively the sampled water represents the parameters of concern for the area. Many variables affect the representativeness of monitoring locations, including lateral, longitudinal, and vertical heterogeneities in the water; equilibration times of the water quality instruments; and hydrological, biological, and physicochemical processes within the sample areas.

This technical note describes the processes involved in designing and deploying automated, remote monitoring systems and analyzing the data they generate. It is not intended as an exhaustive review of the subject, but highlights the more critical steps in developing monitoring systems. Where appropriate, case studies are cited.

Although the primary purpose of this technical note is to describe the installation and maintenance of *automated* remote monitoring systems, the ideas presented have application to manual sampling programs as well. The ultimate goal of any monitoring program should be to collect pertinent, representative data. The flow diagram presented as Figure 1 is a generic guideline for implementing a monitoring program. It is meant to organize the ideas that are discussed in this technical note, and not as a “recipe” for designing and installing automated monitors.

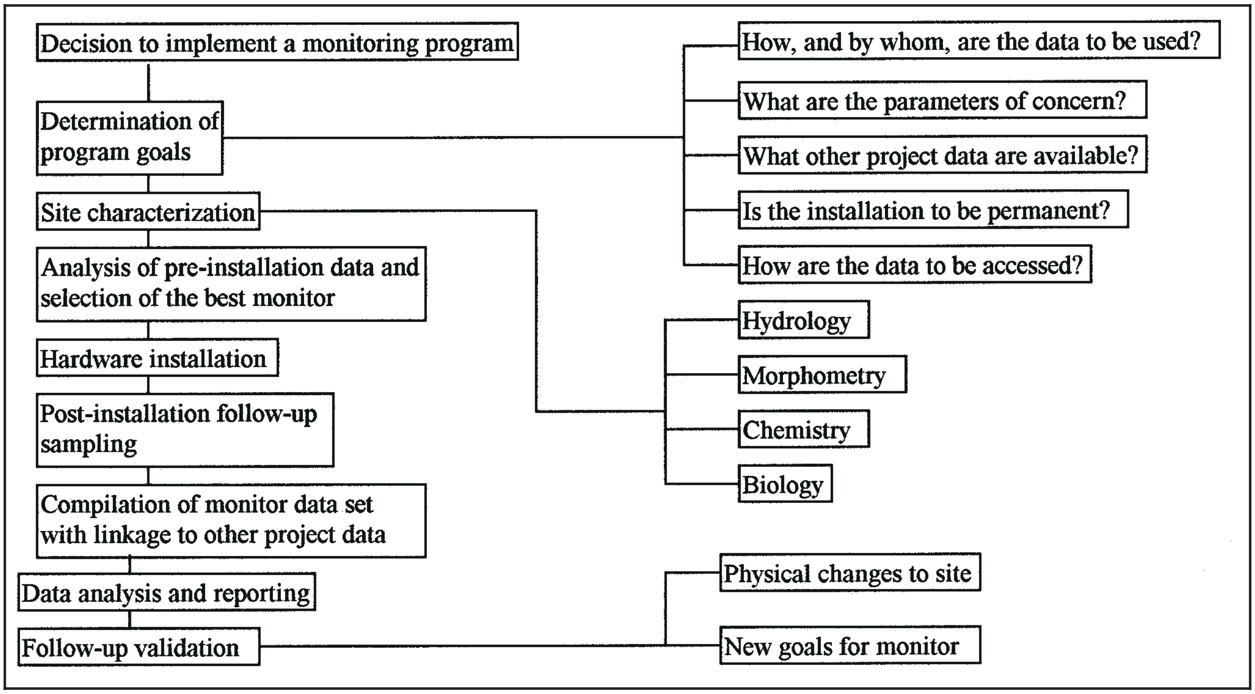


Figure 1. Flow diagram for water quality automated monitor system design

Preinstallation

Goals

The first step in implementing any monitoring program is to determine its goal. Potential questions may include the following:

- Why are the data needed?
- Who will need access to the data?
- Are the data needed real-time or at some other level of frequency?
- What type of sampling interval will be required?
- What is the time frame from data collection to data reporting?

The objective at this stage is to determine what will be expected of the monitoring program. The answers to these questions influence subsequent decisions regarding equipment and location, and are necessary to prevent the implementation of what has been characterized as a “data-rich but information-poor” monitoring program (Ward, Loftis, and McBride 1986).

The answers to questions such as those listed above help managers determine if automated monitoring is needed to attain the goal of the program. Grab sampling may be better suited to a temporary monitoring program or one having a long sampling interval. In a lengthy or permanent installation or one requiring a short sampling interval, grab sampling quickly becomes cost prohibitive, and automated, remote monitors are both more appropriate and effective.

Site Characterization

After the decision to install an automated monitor has been made and the water quality parameters to be measured have been identified, the next step in the preinstallation process is to characterize the study area. This may be accomplished with short-term manual sampling. A working knowledge of the parameter(s) to be measured is essential to identify the most representative deployment site. In addition, the hydrology, morphometry, flow patterns, climate, chemistry, and biology of the site determine the optimum monitoring location. Characterization of the area should include identifying any lateral, longitudinal, and vertical heterogeneities. Sampling should be conducted under the conditions that will be experienced by the monitor; that is, if the monitor is to measure hydropower release water quality, then preinstallation sampling should be conducted during release periods.

Four general areas need to be considered in deploying hydroproject monitors: the forebay, the area within the hydroproject's physical structure, the tailrace, and the tailwater. Preliminary areas of study would depend on the monitoring objective. For example, the preliminary study area for a release water quality monitor for a hydropower dam may be the tailrace. An installation for monitoring the effectiveness of water quality improvement measures may be located upstream for pretreatment conditions and downstream for posttreatment conditions. A monitor for evaluating hydroproject operation on downstream habitat may be located in the tailwater some distance downstream of the project.

Regardless of the monitoring program's goal, certain locations will probably be apparent as logical starting points for consideration. Secondary consideration may focus on accessibility for calibration and maintenance; however, the most convenient location is not always the most representative one, and greatest emphasis should be placed on data quality.

Many relatively inexpensive water quality instruments that are capable of internally storing data are commercially available. These instruments allow project planners to experiment with various site locations via short-term deployments. These data can then be combined with grab data to provide temporal and spatial representations of the daily and seasonal variations for the area. Careful analysis of the available data is crucial during the preliminary stages of developing a monitoring program, to prevent future problems regarding data validity and defensibility. Often, a logical location for the monitor may be apparent; however, peculiarities of the site, particularly with respect to flow patterns, may preclude installation of the monitor in this area. The logical location provides a starting point for the validation stage of the preinstallation process.

Conservative water quality measures (such as temperature or specific conductance), which are not easily affected by biota, may be used as "tracers" to track parcels of water. Comparing conservative parameters cannot conclusively validate the representativeness of a potential location but can eliminate a nonrepresentative one. Several case studies will be presented to further develop these ideas.

Ice Harbor Example

Ice Harbor Dam is located on the Columbia River immediately upstream of McNary Dam and immediately downstream of Lower Monumental Dam along the Oregon/Washington border (Figure 2). Spilling operations conducted for fish passage, as well as flood control, often lead to dissolved gas concentrations that are supersaturated with respect to the atmosphere. Supersaturation of dissolved gases in water may have severe detrimental impacts on fish. As a result, extensive studies to measure dissolved gas concentrations and dynamics have been conducted at the U.S. Army Corps of Engineers projects in the Columbia River Basin.

Data gathered during transect studies in support of the total dissolved gas monitoring program illustrate how data gathered for other purposes can be used to plan an automated monitor installation. The results from these lateral transects are displayed in Figure 3. Two monitors were previously installed in the Ice Harbor tailwater (indicated as the labeled points in Figure 3); however, they were neither designed nor intended to reflect the extent of the variation in total dissolved gas concentrations in the area.

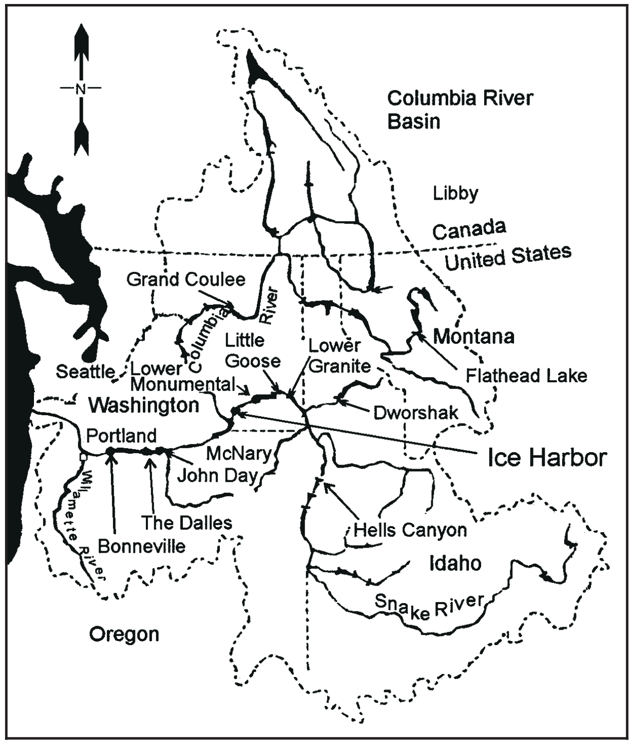


Figure 2. Columbia River basin

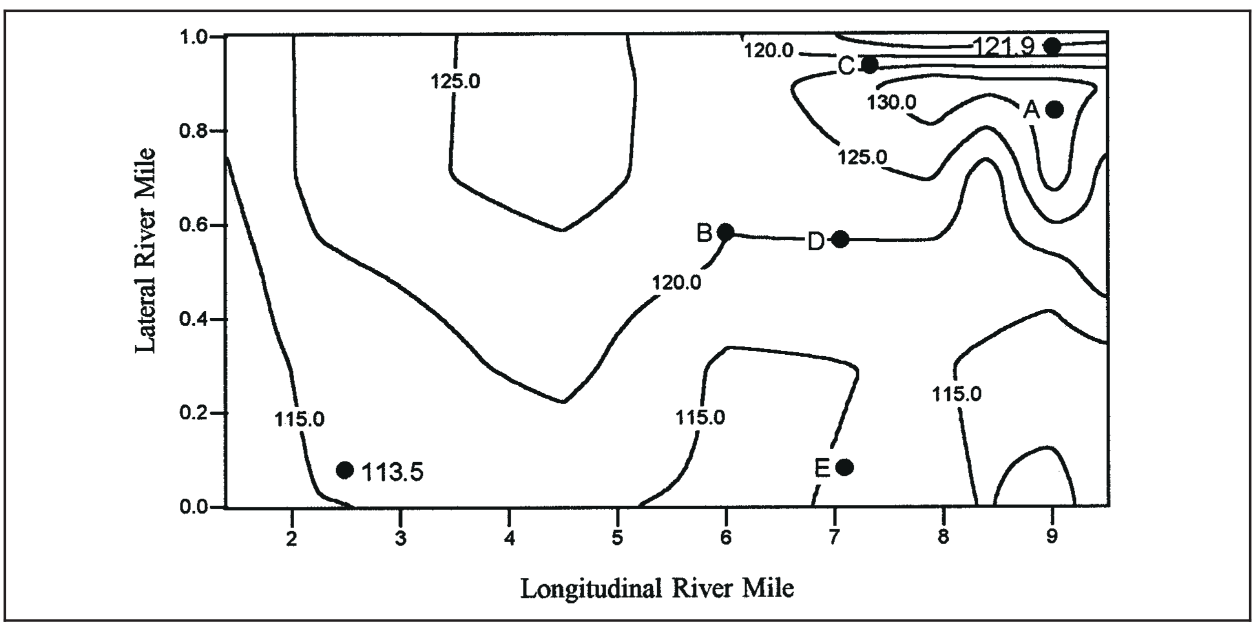


Figure 3. Contour plot of Ice Harbor total dissolved gas transect data

The goal of the monitoring program dictates the deployment design. If the goal of the program were to measure critical total dissolved gas concentrations, then a single monitor near the area of highest total dissolved gas concentrations may be sufficient (point A, Figure 3). If concerns were for the mean total dissolved gas concentrations for the area, a single monitor located near midchannel may be appropriate (point B). However, if the program's goal were to map the total dissolved gas concentrations for the tailwater, a single fixed monitor would be inappropriate, and an alternate plan would have to be developed, involving numerous fixed positions (for example, points C, D, and E in Figure 3). This example highlights the need for good planning and preinstallation sampling in the early stages of developing a monitoring program.

Monitor Equipment

Hardware

Data requirements and available funding will dictate the hardware selected for the monitor installation. Water quality instruments equipped to measure most parameters of concern are commercially available. However, these instruments vary with respect to accuracy, precision, data presentation, and expense.

Consideration should be given to the design limitations of the instrument when selecting water quality equipment. For example, if the purpose of the monitor were to record dam release dissolved oxygen (DO) concentrations for mitigation and the requirement was to remain within 0.5 mg/L of a target DO concentration of 5.0 mg/L, then oxygen probes with an accuracy of less than ±0.5 mg/L would be inadequate.

Deployment/retrieval monitoring is used for thermal monitoring and special studies at Richard B. Russell Reservoir on the Savannah River. For this application, water quality instruments with data logging capabilities are deployed, and the data are retrieved later. If data are needed real time, a computer/modem system can be used. Relatively inexpensive, reliable water quality sondes interfaced with a personal computer/modem can be obtained for less than $5,000 (1996). Commercially built data collection platforms are available, and most can be tailored to fulfill the design requirements of the site. With computers and other data platforms, the operator achieves greater flexibility with respect to how the data are stored and accessed.

As a general rule, equipment should be selected based on the following factors:

- Instrument accuracy, precision, and resolution desired.
- Instrument deployment requirements.
- Deployment method (deploy/retrieval, computer/modem, incorporation with existing equipment, etc.).
- Fouling concerns and required calibration and maintenance regimens.
- Instrument expense and monitoring program budget constraints.

Software

Off-the-shelf data collection platforms include software or programming instructions that allow them to be configured to communicate with a variety of instruments. Additionally, personal computer communications packages can communicate with water quality equipment and store and transmit data; however, design flexibility is generally less. BASIC software programs (Microsoft Corporation) can be developed as an alternative to off-the-shelf communications packages and afford the user control over communication protocol and data storage format (Vorwerk, Moore, and Carroll 1996). The data storage format is an important design consideration because it facilitates integration of the final monitor data set with other pertinent data sets (for example, hydroproject operation data) and allows real-time data presentation to better fit project requirements.

Location Validation

Postdeployment data validation is a crucial final step in the monitor installation process, as this evaluates the representativeness of the monitor location. Although postvalidation may seem unnecessary if care was taken during preinstallation sampling, the installation itself may have a measurable impact on how the water quality is represented by the equipment. A dam release monitor could be installed in the tailrace of a project, with water pumped to it from an area determined to reflect the area of management concern during generation periods. Subsequent calibration visits may confirm that the sensors are operating well within the manufacturer's specifications. From this, it may be assumed that the monitor is accurately representing the parameters of concern. If, however, the water were being warmed as it passed from the tailrace through the pipe to the monitor, it would actually reflect the water within the sample chamber and not the tailwater. Likewise, changes in the physical structure of a site or introduction of water quality improvement measures may alter the representativeness of an established monitor. These concerns must be addressed via postdeployment verification studies.

Data Interpretation

After the monitor is in place and recording representative water quality data, the next concern is how the data should be used. Raw monitor data are of little use if they are not presented in a manner that facilitates interpretation. Off-the-shelf spreadsheet and database programs such as Excel (Microsoft Corporation), SAS (SAS Institute, Inc.), and SPSS (SPSS, Inc.) expedite data analysis and reporting by facilitating the linkage of monitor data with other project data. Data must undergo vigorous error-detection and filtering processes prior to analysis. Raw monitor data must be edited to remove machine characters, usually artifacts of the data collection software, before they can be properly imported into analysis software packages.

Water quality sensors typically exhibit some degree of response drift as a result of the sensors' chemical reactions (for example, oxidation of DO probes). Sensor drift can also result from biological activity. For example, algal growth on DO probes may decrease the reported DO concentrations by inhibiting oxygen diffusion across the sensors' membranes. Routine calibration may reduce the degree of sensor drift; however, postdeployment corrections for sensor drift can further improve data accuracy.

For the Savannah River monitors where dam release DO concentrations are the primary concern, frequent calibration visits (at least once a week) during summer months reduce the degree of drift resulting from biological activity. Calibration drift is assumed to be linear, which allows corrections to be based on the degree of drift per hour for the period between calibrations. Each reading is then corrected for drift by adding or subtracting this value to it, with the drift at the time of the first calibration being equal to zero. The causative factors leading to drift vary depending on the site, the parameters being measured, and the equipment being used. (The instruments used for monitoring the Savannah River hydroprojects have a resolution of ±0.2 mg/L; therefore, drift must be >0.2 mg/L before corrections are made.) Drift must be determined for each site and should be factored into the data set prior to its incorporation with other project data (Whitfield and Wade 1993).

Data should be incorporated with other project data prior to final analysis. By combining the available data into a comprehensive project data set, "windows of reflectiveness" can be better identified and data interpretation will be more accurate. For example, the release monitor at Hartwell Dam, a Corps project located on the Savannah River (Figure 4), is deployed in the tailrace (Figure 5). It consists of a submersible pump and pipeline to pass water from the tailrace to a water quality sonde in a nearby building. Because it samples water from the tailrace, the monitor represents release water quality only during periods when Hartwell Dam is releasing water. Data for periods of nonrelease reflect the tailwater conditions only in the area localized around the sample intake line.

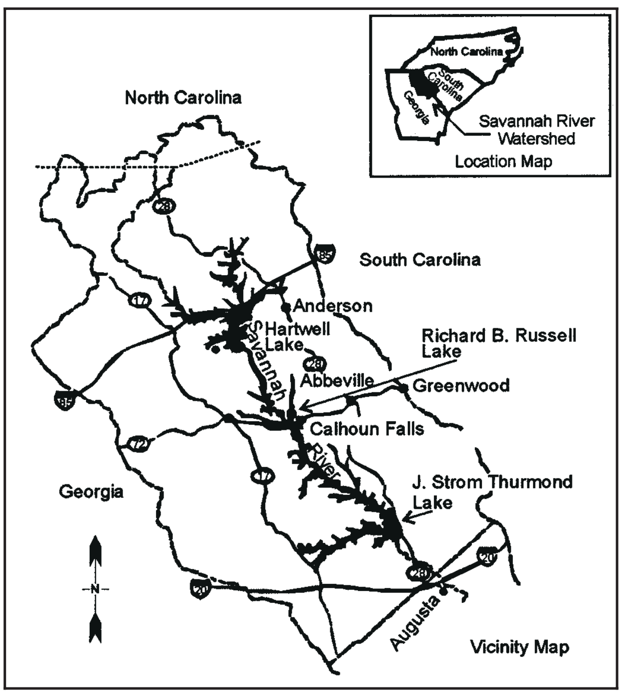


Figure 4. Savannah River basin

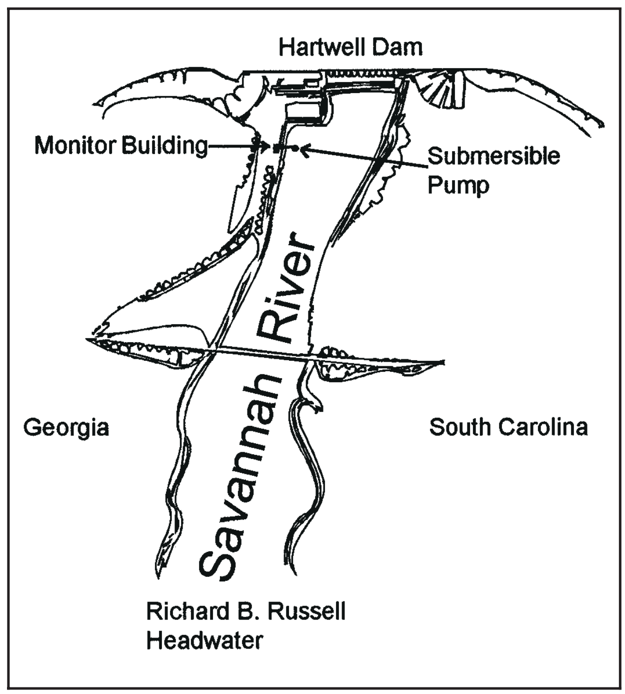
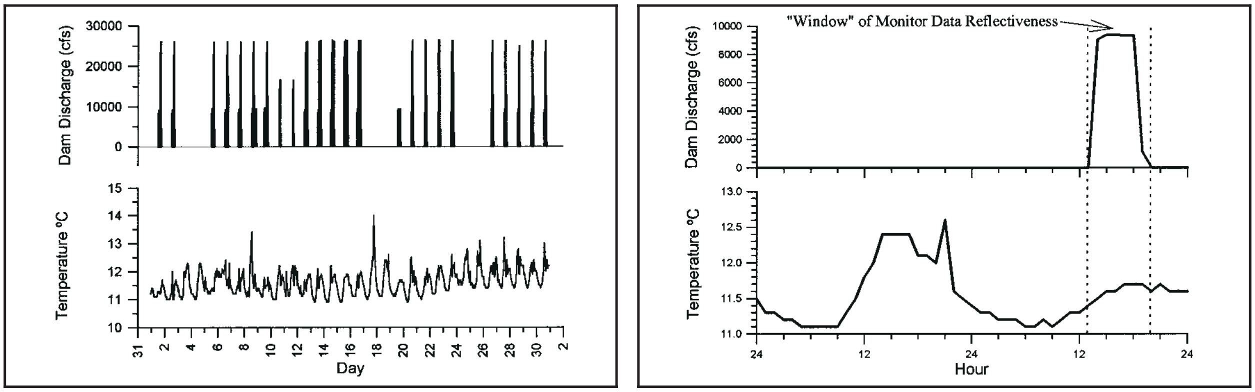


Figure 5. Hartwell Dam release monitor

Representative periods are readily apparent when both the monitor and operations data are incorporated (Figure 6). Data falling outside the “window” that defines representative periods, generally resulting from changes in project operations, are not included in final reporting as they are not reflective of the parameters of concern.



a. May 31-June 30, 1995

b. June 18-19, 1995

Figure 6. Hartwell Dam operation and release temperatures

A large equilibration period (longer than 20 minutes) may be required by some instruments before accurate measurements are possible. This is especially true for gas measuring instruments such as DO or total dissolved gas sensors. Instrument and design limitations such as these should be considered during the final analysis, particularly in situations where rapid changes are experienced.

Case Studies

Continuous, automated monitors are presently being used by the Corps to monitor the release water quality of the hydropower projects on the Savannah River forming the Georgia/South Carolina border, the tailwater conditions during periods of no release at St. Stephen Dam on the Cooper River in South Carolina, the effectiveness of turbine venting procedures at Bull Shoals Dam on the White River in Arkansas, the total dissolved gas concentrations at various projects throughout the Columbia and Snake River systems, and at other projects throughout the United States. The monitoring goals, parameters of concern, and available funding vary significantly from project to project; however, the overall goal—to collect representative data—is common to all. The case studies discussed below demonstrate some of the techniques that have been used to ensure sample reflectiveness at various projects.

Richard B. Russell Dam

Richard B. Russell Dam is a Corps generation/pumped storage project located on the Savannah River between the Corps reservoirs of Hartwell and J. S. Thurmond (Figure 4). The Russell monitor measures release water quality for the purpose of maintaining a release DO concentration of 6.0 mg/L. The Corps operates an oxygen injection system in Russell forebay to maintain this concentration during the summer months when hypolimnetic DO concentrations approach anoxia. The 6.0 mg/L DO concentration requirement applies to the release and not to the tailrace or tailwater conditions; therefore, the sampled water must reflect the Russell release and not the conditions of the Thurmond headwater.

The monitor was originally located in the tailrace, where follow-up studies later demonstrated that flow patterns caused the monitor to be less reflective of Russell Dam release water than the ambient tailwater conditions (Figure 7). Temperatures and DO concentrations were measured at various points in the tailrace and the dam, and were compared with the temperatures of the water sampled by the original tailrace monitor. For comparison, temperature was selected over DO, since it was a more conservative parameter and as such was deemed to be less susceptible to exterior influences (Vorwerk and Carroll 1994).

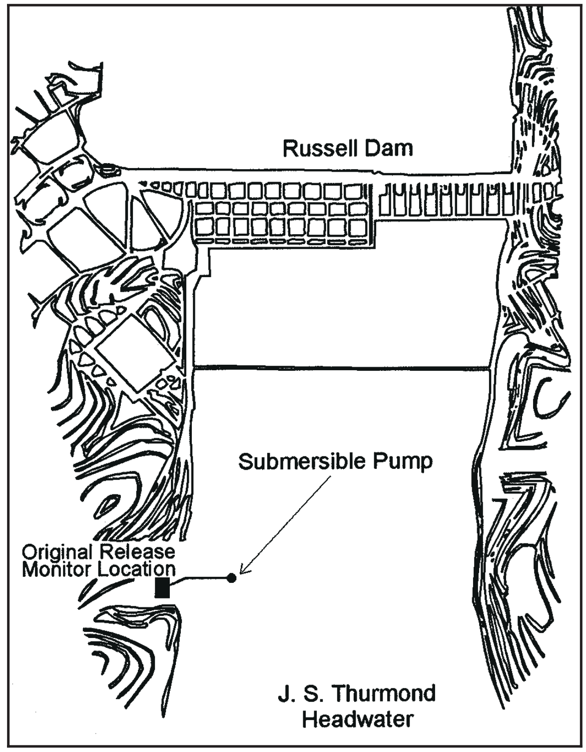


Figure 7. Richard B. Russell original downstream monitor

The lacustrine tailwater region at the Russell project prevented the deployment of the tailrace monitors that had been successful for other Savannah River monitors. A mixing chamber system containing a water quality sonde was implemented such that water passage was controlled by solenoid switches. The switches were configured to restrict water passage to periods of turbine operation. This system (Figure 8) allowed representative water to be sampled with a single in-dam unit. While the monitoring goal (to measure release temperatures and DO concentrations) was the same for the Savannah River monitors, specific characteristics unique to each site had to be considered in determining where to locate the monitors.

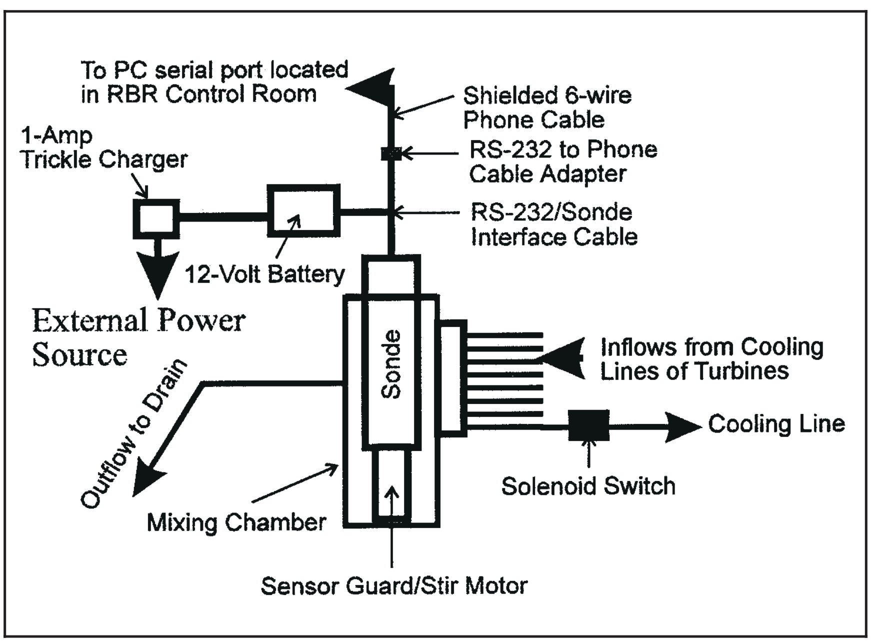


Figure 8. Richard B. Russell piping gallery monitor

Bull Shoals Dam

At Bull Shoals Dam on the White River, Arkansas (Figure 9), the goal for the monitoring program was to determine the efficiency of turbine venting operations conducted in order to increase downstream DO concentrations. Two of the seven Bull Shoals units had turbine venting capability, and penstock monitors had previously been installed to measure the pretreatment water quality. In situ sampling demonstrated that locating the posttreatment monitors in or near the draft tube exits would best represent the release water quality. The draft tube access ports were chosen for their proximity to the draft tube exits and because they afforded easy access for calibration and maintenance. The concern was to isolate the monitors from the release of the other units to accurately identify the DO increase resulting from individual turbine venting.

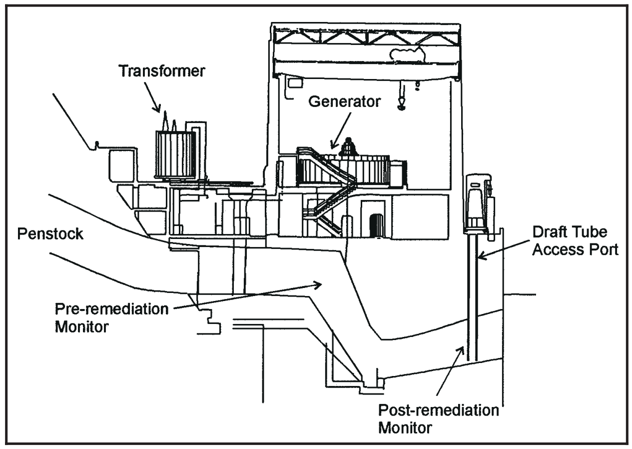


Figure 9. Bull Shoals powerhouse, White River, Arkansas

St. Stephen Dam

St. Stephen Dam is a Corps power project located near St. Stephen, SC. The dam rediverts water from Lake Moultrie back to the Santee River (Figure 10). A fish kill during spring 1991, which was attributed to insufficient DO concentrations during nonrelease periods, prompted evaluation of the DO dynamics surrounding the project. It was determined that releasing water when the DO concentrations were low caused dilution of the poorly oxygenated canal water with well-oxygenated reservoir water and prevented DO-related fish kills. The monitor program implemented at St. Stephen was designed to measure the tailrace DO concentration during periods of no release. Real-time monitoring data were used to indicate when critically low DO concentrations occurred so water could be released, thus minimizing the potential for a fish kill. Manual sampling indicated that the monitor should be placed near the bottom of the canal and near the dam, since anoxic conditions were realized in these areas first. A monitor attached to the wingwall downstream of the dam (Figure 11) represented “worst-case” conditions (Vorwerk and Carroll 1995).

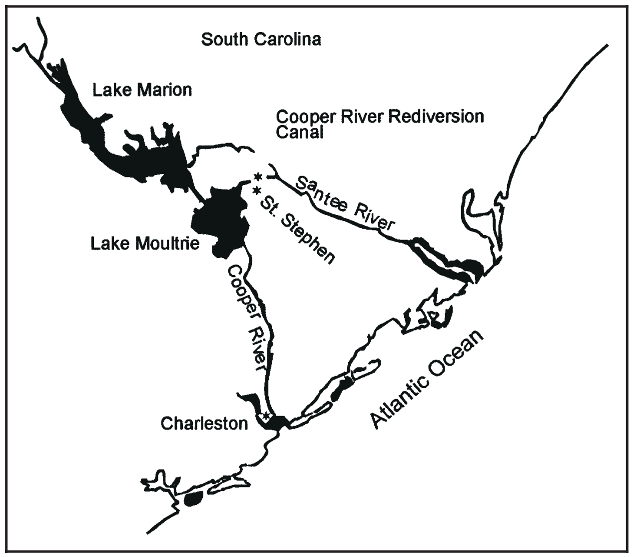


Figure 10. St. Stephen Dam vicinity map

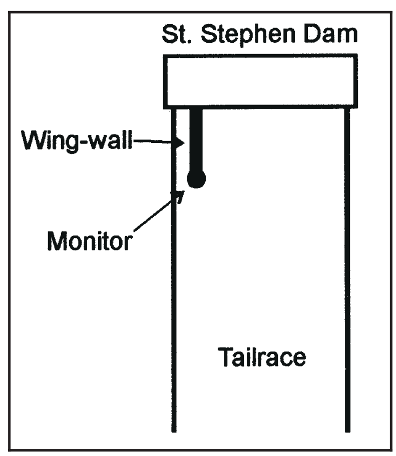


Figure 11. Schematic of the St. Stephen Dam tailrace monitor

Conclusions

Remote, automated monitors are valuable water management tools. Large gains continue to be made with respect to water quality instrumentation, which reduces the need for costly equipment and labor-intensive sampling regimes. Too often, however, the assumption is made that deployment of a fixed monitoring system is sufficient for generating the desired data with little (if any) forethought devoted to outlining the goals of the monitoring program. Without clear goals, it is impossible to design a preinstallation program to determine the most appropriate location for the fixed monitor. Data density without data quality is of no use to project managers.

By clearly defining the objectives of the monitoring program prior to beginning data collection, and characterizing the study site with respect to the physicochemical and biological attributes of the system, it becomes possible to design and install an automated, fixed location monitor that supplies data representative of the parameter(s) of management interest. Data should be analyzed as they are collected, especially during the critical preinstallation sampling, as it may be necessary to redesign the sampling approach to better address the questions to be answered or address new questions that arise during the study.

Incorporating all available data (including project operations, meteorological, and historical data for the project of concern) helps managers to address issues and collect data that may require intensive sampling efforts to obtain. Valuable information may be realized from historical data sets that may have been neglected otherwise. The monitoring program should remain focused on the objectives that were outlined at its inception.

Periodic evaluation of the monitor's performance should be a routine component of the analysis process, especially when structural or operational modifications to the project or monitor occur. Reevaluations of this nature are imperative for ensuring representative data collection.

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WQRP

WATER QUALITY RESEARCH PROGRAM

APPENDIX 2.4C

Statistical Verification of Mean-Value Fixed Water Quality Monitor Sites in Flowing Waters

by: Michael Vorwerk, Joe Carroll,
and John Lemons

Water Quality Technical Note AM-03
January 1998

US Army Corps
of Engineers

Statistical Verification of Mean-Value Fixed Water Quality Monitor Sites in Flowing Waters

by Michael Vorwerk, Joe Carroll, and John Lemons

Purpose

This technical note describes a method for verifying the representativeness of mean-value and extreme-value water quality monitoring locations. Recommended techniques are illustrated using data collected with the total dissolved gas monitoring system on the Columbia and Snake Rivers. This technical note shows how statistical techniques can be applied to the design of monitoring systems to ensure that data collected are representative and thus scientifically defensible.

Background

Water quality managers must carefully choose the locations for fixed water quality monitors, to ensure that the data they collect accurately reflect water quality conditions of the water of interest. Often, a monitor site will experience some spatial or temporal bias, and data collected there will not represent the release or river in question.

For rivers and hydroproject releases, bias may be the result of combined spill and generation releases (Lemons, Vorwerk, and Carroll 1996), releases into lacustrine tailwaters (Vorwerk and Carroll 1994), generation drawing water from a forebay with heterogeneities (Lemons and others 1996), point sources of pollution, or other processes (Vorwerk, Jabour, and Carroll 1996). A monitor system intake may be located in some portion of a flow and accurately measure its water quality, while not reflecting the quality of other portions (Figure 1). Thus, to provide usable data for operation, regulatory, or background monitoring needs, a manager must verify the representativeness of monitor sites with regard to the monitoring program goals.

This verification must include quantification of the spatial and temporal similarity between water quality data gathered at the monitor site and in the stream or river in question. Flowing water monitor systems can be designed to create temporal records of water quality information as either means or extreme values (Ward 1979). Different verification techniques are necessary for each of these designs. This technical note discusses the techniques necessary to verify mean-value monitor systems.

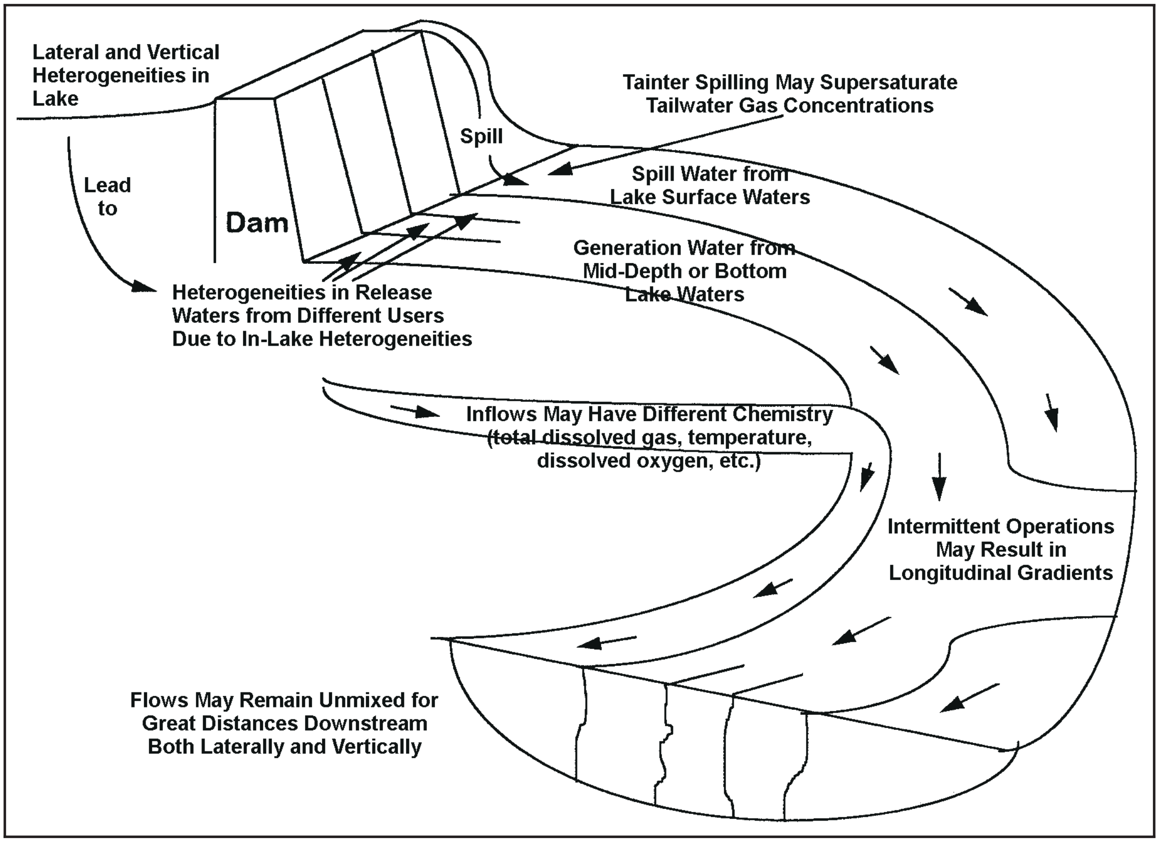

Figure 1. Possible sources of heterogeneities in flowing water

To obtain mean values of water quality parameters in flowing water, the analyst must have some knowledge of the mixing processes that are present. *In situ* data are needed for the verification. If the stream is turbulent and well mixed, it may be the case that any location can accurately represent the quality of the water. If the stream is not well mixed and has heterogeneities in water quality, the data must be flow weighted.

Flow-weighted data allow one to calculate the mass transport of parameters through the cross section of the stream in time. Some examples of flow-weighting include temporal quantification of dissolved oxygen mass or average dissolved oxygen concentration moving down a river, a record of average total dissolved gas saturation, mass transport of nutrients, or a record of average temperature. The important aspect is that the value of the parameter of interest is averaged across the area of the channel cross section with respect to velocity.

Any verification must be both qualitative and quantitative. This technical note describes approaches for statistically quantifying and verifying the adequacy of monitoring sites for measuring the average water quality at river transect. Total dissolved gas data collected from the Columbia and Snake Rivers are used to illustrate these techniques. The statistical methods provided will allow users with a basic knowledge of statistics to design and implement studies to verify the representativeness of their own monitor locations. A review of statistics with water quality applications can be found in Gaugush (1986). It should be noted that, although this technical note is based on the use of automated fixed water quality monitors, the procedure described can be applied to manual monitoring as well.

Approach for Mean Data

Data Collection and Preparation

The basic approach to verifying the representativeness of a monitor site is to compare matched pairs of observations from the monitor and averaged from the flow (Figure 2). These pairs must be taken over as many different times, flow conditions, and water quality variations as possible.

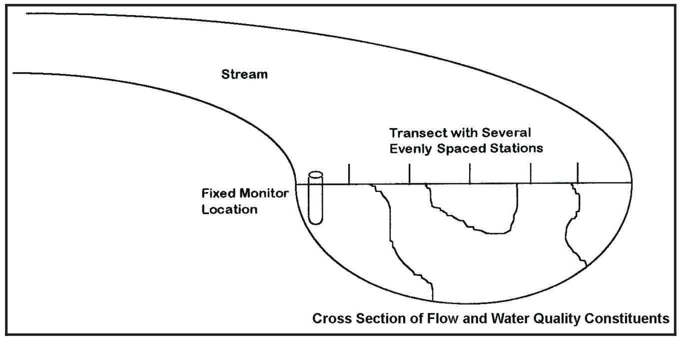

Figure 2. Cross section of flow with evenly spaced sample stations along a transect

The observations in the flow must be distributed so they adequately describe water quality conditions across the stream. For wider streams and rivers or for more highly variable water quality conditions, more sample locations are necessary. The sample values from the stream are averaged with an area-weighted average. If velocities vary greatly in the stream cross section, the data averaging must also be flow-weighted. The next section provides details on this weighting.

In practice, data are often limited, and the only available option is averaging the transect data with a simple arithmetic average, and then carrying out the statistical comparison. However, if the stations are not evenly spaced or if the water column has lateral or vertical heterogeneities in water quality or velocity, then a flow-weighted average should be calculated.

Flow-Weighted Data

The following method can be used to calculate a flow-weighted average. For each sample station, *i*, and depth, *z*, with velocity $U_{i,z}$ and water quality parameter value $P_{i,z}$, assign an area $A_{i,z}$ that the information gathered at that location represents (Figure 3). The area can be difficult to calculate and is most often approximated from depth soundings, maps, surveying techniques, global positioning equipment, and "best-guess."

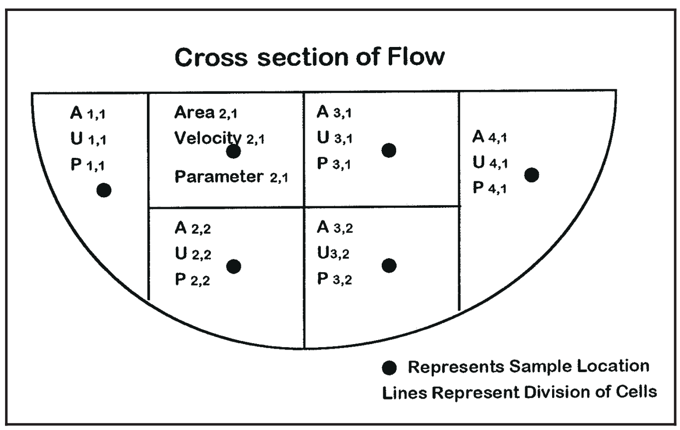

Figure 3. Hypothetical sample scheme for flow-weighting data

The transect flow-weighted average of the parameter *P* can then be expressed as

$$\overline{P} = \frac{\sum_{all\,i} \sum_{z=0}^{z\,max} A_{i,z} U_{i,z} P_{i,z}}{\sum_{all\,i} \sum_{z=0}^{z\,max} A_{i,z} U_{i,z}} \tag{1}$$

Averaging should be carried out for each sampling time. Again, only in the most carefully designed and executed studies is such information available. More typically, an analyst may have three to seven lateral measurements along a transect to compare with fixed monitor information. In this case, the analysis can be performed, but the analyst must be aware that those limited data lessen the weight that may be given to any conclusions.

Statistical Comparison

At this point, the verification data set should contain n pairs of data ($X_{m,j}$, $X_{s,j}$), each containing a monitor observation X_{mj} and an average stream value X_{sj} for time j, where m and s indicate that the observation came from the monitor or stream, respectively. Next, one tests the relationship between the two locations using a paired t-test (following Hines and Montgomery 1980). This test assumes that the samples each come from a normally distributed, independent distribution. However, moderate departures from normality should not adversely affect the analysis (Pollard 1977). The difference between each pair of observations, $D_j = X_{mj} - X_{sj}$, should come from a normally distributed independent distribution.

To verify that the data come from a normally distributed population, either of two methods can be used. The easiest method is to plot the data on normal probability paper or use a statistics software package to generate a normal probability graph. A second method is to use a quantitative test such as the Kolmogorov-Smirnov test or Lillefore’s test. Further details of these tests can be found in Hines and Montgomery (1980) and Pollard (1977). Within this technical note, normal plots are used; these were generated using SPSS (SPSS, Inc., Chicago, IL), a statistical analysis software package.

Once it has been determined that the data come from a normal or nearly normal distribution, one can begin the comparison by stating the hypotheses. The null hypothesis is that the mean of the differences between pairs, μ_D , is zero. This implies that monitor value agrees with stream values and is representative. The alternative hypothesis is that the mean of the differences is not zero; that is, the monitor values do not agree with stream values and are not representative. This is stated as follows:

$$H_0 : \mu_D = 0 \tag{2}$$

$$H_1 : \mu_D \neq 0 \tag{3}$$

These hypotheses are tested with the following statistic:

$$t_0 = \frac{\overline{D}}{S_D / \sqrt{n}}, \tag{4}$$

where

$$\overline{D} = \frac{\sum_{j=1}^{n} D_j}{n}, \tag{5}$$

$$S_D^2 = \frac{\sum_{j=1}^{n} D_j^2 - \frac{1}{n}\left(\sum_{j=1}^{n} D_j\right)^2}{n-1}, \tag{6}$$

and we reject H_0 if $t_0 > t_{\alpha/2,n-1}$ or if $t_0 < t_{\alpha/2,n-1}$. The confidence level, α, is typically taken to be 0.05 and is the type I error, or the probability of rejecting H_0 when H_0 is true. If H_0 is rejected, we conclude that the fixed monitor system does not represent the water quality of the stream at the α confidence level. If H_0 is not rejected, we conclude that "we have not found sufficient evidence to reject H_0" (Hines and Montgomery 1980). This may be because the monitor site accurately represents the stream or because the sample size (that is, the number of comparisons) is so small that not enough data are available to make the stronger conclusion to reject H_0. So, for verification, we need a large enough sample size to minimize the type II error (that is, the probability of accepting H_0 when H_0 is false).

Similarly, one-sided hypotheses can be tested as follows:

$$H_0{:}\mu_D \le 0,\ H_1{:}\mu_D > 0,\ \text{reject } H_0 \text{ if } t_0 > t_{\alpha,n\text{-}1} \tag{7}$$

$$H_0{:}\mu_D \ge 0,\ H_1{:}\mu_D < 0,\ \text{reject } H_0 \text{ if } t_0 < t_{\alpha,n\text{-}1} \tag{8}$$

Determining the Power of the Test

The rejection of the null hypothesis is considered a "strong" conclusion because we control the type I error (choice of α), or the probability of rejecting H_0 when H_0 is true. On the other hand, the acceptance of the null hypothesis is considered to be a "weak" conclusion, because we do not control the type II error (β), or the probability of accepting H_0 when H_0 is false.

Thus, to determine the meaning of our conclusion when we accept the hypothesis that a monitor represents the flow, we must determine the type II error. For the monitor location to be acceptable, the type II error must be acceptably small.

To estimate the type II error, or β, a statistic d is calculated, and with α and n, β can be determined from operating characteristic charts available in statistics books (Hines and Montgomery 1980, p 604). Using Equations 5 and 6, we calculate d as follows:

$$d = \frac{|\overline{D}|}{S_D} \tag{9}$$

Once β is found, the probability of correctly accepting H_0 is the power, namely $P = 1 - \beta$. Because we want only to correctly accept H_0, we desire the power to be as close to 1 as possible. The question then becomes, What's good enough?

Since we typically choose α to be 0.05, it seems reasonable to attempt to hold β to a similar probability. However, because we have no direct control over β, probabilities less than 0.2 are probably sufficient. Thus, we consider comparisons with the power greater than 0.8 to be acceptable.

If we are designing a verification study, pilot studies, such as the one described in examples 1 and 2, provide *a priori* knowledge of $\overline{D}$ and S_D. This information can be used to design the verification study with a sample size large enough to ensure that the power is as great as desired. This is accomplished through increasing the sample size until the desired value for β is achieved on the operating characteristic curve.

Example 1: Columbia River Camas/Washougal Station—Hand Calculation

The following example illustrates this method with data from the Camas/Washougal total dissolved gas monitoring station (CWMW) on the Columbia River. To assist smolt in their downstream migration, the U.S. Army Corps of Engineers spills surface water from projects on the Columbia and Snake Rivers. This spillage causes air to be driven into the water column to depths where it causes gases in the water column to be supersaturated with respect to surface saturation. This supersaturation can be detrimental to fish, so the Corps monitors spill gas concentrations in the rivers. Thus, this system is designed to determine the extreme total dissolved gas concentrations resulting from spilling water. This information is used for compliance and in project operations.

To determine if these monitors could be used to determine the flux of total dissolved gas in the river, the statistical verification studies presented in this technical note were carried out. The verification is based on comparing monitor data with data collected at eight transects near the CWMW monitor site (river mile 122) on 3 days (Table 1). The stations on the transects were approximately evenly spaced, so the data for each transect were simply averaged together to obtain an average total dissolved gas concentration at that transect.

Table 1
Average Total Dissolved Gas as Percent Saturation, Columbia River Transects and Camas/Washougal Monitoring Station Fixed Monitor

Date	Transect Mile	Percent Saturation		No. Samples
		Transect Average	Monitor	
18 May 95	119.9	115.1	113.4	5
25 May 95	121.2	118.1	115.5	5
25 May 95	121.6	119.0	117.3	5
25 May 95	122.1	119.4	118.5	5
25 May 95	119.9	117.0	113.4	7
27 Jul 95	121.2	112.1	109.8	32
27 Jul 95	121.6	116.0	111.9	15
27 Jul 95	122.1	112.9	109.5	15

Figures 4 and 5 show normal probability plots of the transect and fixed monitor system data, respectively. Ideally, the data would be randomly distributed along the normal distribution line, with points close to and on either side of the line. Though the transect data in Figure 4 do not appear to be completely random about the normal line, they are sufficiently normal for this

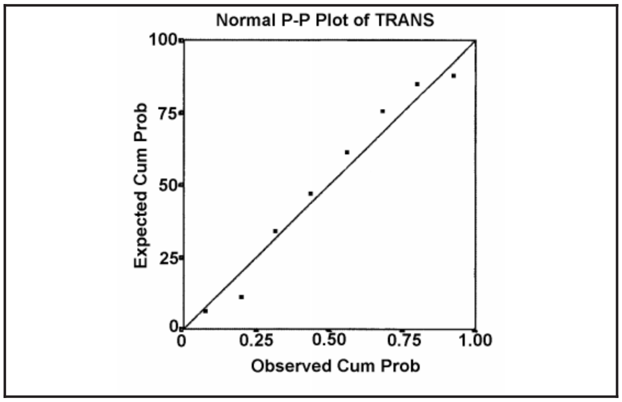

Figure 4. Normal probability plot of transect data. (Straight line plots the normal distribution; square symbols are the observed data.)

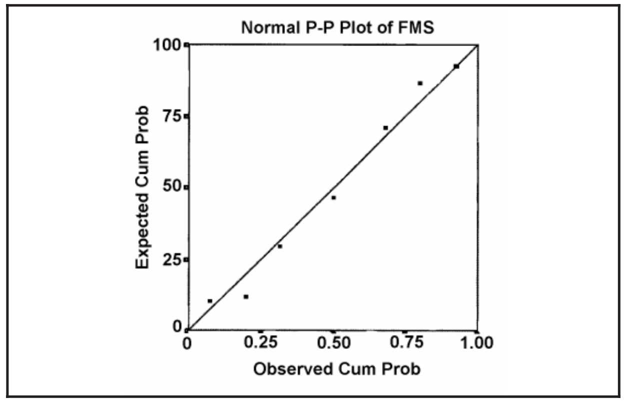

Figure 5. Normal probability plot of fixed monitor station data. (Straight line plots the normal distribution; square symbols are the observed data.)

analysis. The data essentially fit the normal distribution line, but show a trend to be above the line for higher cumulative probabilities and below the line for lower cumulative probabilities. We conclude that the data are approximately normally distributed.

Figure 5 suggests that the fixed monitor data are also normally distributed. Note that the data in Figure 5 are somewhat more randomly distributed on each side of the normal line, with fewer "runs" or continual observations on one side or the other of the normal line. This graphically based determination is subjective. To lessen subjectivity, tests as discussed above can be used (Kolmogorov-Smirnov, Lillefore's, etc.), but the analyst is often forced to use whatever data are available.

Because the data were collected for another study and not specifically for monitor verification, the transect locations did not coincide exactly with the monitor location. For our comparison, all transects that were within 3.5 km of the fixed monitor station were selected. The number of samples varied with transect mile and date. The May samples had five or seven evenly spaced measurements at a constant depth of 4.6 m. July samples had multiple depths and five to seven sample locations. The calculations of the differences, the square of the differences, and the totals of the two sites are depicted in Table 2.

Table 2
Differences (D), Squares of Differences (D^2), and Totals for Data Specified in Table 1 (Sample Size, n = 8)

Date	Transect Mile	D	D^2
18 May 95	119.9	1.7	2.9
25 May 95	121.2	2.6	6.8
25 May 95	121.6	1.7	2.9
25 May 95	122.1	0.9	0.8
25 May 95	119.9	3.6	13.0
27 Jul 95	121.2	2.3	5.3
27 Jul 95	121.6	4.1	16.8
27 Jul 95	122.1	3.4	11.6
Total		**20.3**	**60.1**

Figure 6 is a normal probability plot of the differences between the transect and fixed monitor data pairs. Though the data show some tendency to be lower than the normal plot for low probabilities and higher than the normal plot for high probabilities, the data appear to be approximately normally distributed.

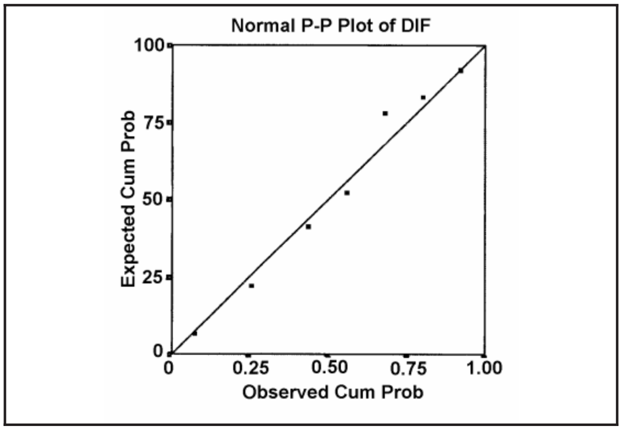

Figure 6. Normal probability plot of the differences between transect and fixed monitor station pairs of observations. (Straight line plots the normal distribution; square symbols are the differences.)

Equations 2 and 3 were used to test whether the data collected at the fixed monitor site represent the water quality within the river. First, the parameters necessary for the test statistic were calculated.

The mean difference (Equation 5) was

$$\overline{D} = \frac{\sum_{j=1}^{n} D_j}{n} = \frac{20.1}{8} = 2.5 \tag{10}$$

The variance was estimated using Equation 6:

$$S_D^2 = \frac{\sum_{j=1}^{n} D_j^2 - \frac{1}{n}\left(\sum_{j=1}^{n} D_j\right)^2}{n-1} = \frac{60.1 - \frac{1}{8}(20.1)^2}{8-1} = 1.4 \tag{11}$$

The test statistic, t_0, was then calculated using Equation 4:

$$t_0 = \frac{\overline{D}}{S_D / \sqrt{n}} = \frac{2.5}{1.2 / \sqrt{8}} = 5.9 \tag{12}$$

Next, the test statistic calculated in Equation 12 was compared with $t_{\alpha/2, n-1}$. This value can be found in various statistics books in the Students' *t* table or *t* distribution table (Hines and Montgomery 1980, p 596). For $\alpha = 0.05$ (our choice) and $\upsilon = n - 1 = 7$ (determined by the sample size of 8), the value of $t_{\alpha/2, n-1} = t_{0.025,7} = 2.365$ (from tables). Then, since

$$5.9 = t_0 > t_{\alpha/2, n-1} = t_{0.025,7} = 2.365 \tag{13}$$

we rejected H_0 and concluded that the difference between the transect values and the fixed monitor station values was not zero. The fixed monitor did not adequately represent the water quality in the river at this location.

We tested the hypothesis that the transect % TDG values (total dissolved gas as percent saturation) were greater than the fixed monitor % TDG values using Equation 7. We hypothesized that $H_0{:}\mu_D \leq 0$ with alternative $H_1{:}\mu_D > 0$. We rejected H_0 if $t_0 > t_{\alpha,n-1}$. Again, using $\alpha = 0.05$ and $\upsilon = n - 1 = 7$ (determined by the sample size of 8), the value of $t_{\alpha,n-1} = t_{0.05,7} = 1.895$.

Also, since $5.9 = t_0 > t_{\alpha,n-1} = t_{0.05,7} = 1.895$, we rejected H_0 and concluded that the difference between the transect values and the fixed monitor station values was greater than zero. The fixed monitor consistently recorded total dissolved gas percent saturation values that were less than the average of those actually present in the river at this location during this study. Thus, we conclude that the fixed monitor system does not accurately represent the flux of total dissolved gas in the river.

To avoid tedious hand calculations, software packages are useful for calculating the paired t-test statistics for the data sets. Two commonly used packages are SPSS (SPSS Inc., Chicago IL) and SAS (SAS Institute, Inc., Cary, NC).

Example 2: Columbia and Snake Rivers Fixed Monitoring System

The technique employed in the above example can be used to look at an entire monitoring system. Though the fixed monitoring system is designed to determine extreme concentrations of total dissolved gas in spill waters, here we explore the potential of each station for use in monitoring the average total dissolved gas concentration in the river. The system consists of monitors at 26 sites. As in example 1, these fixed monitor sites were compared with transect data collected during 1995.

Again, because the transect study was designed to aid modelers and not strictly to verify the fixed monitor system, adequate data were not available for each location. The analysis shown here was intended only to provide insight into the representativeness of the monitoring system. Details, such as verifying normality, have been omitted. The results presented here might best be used to design future, more rigorous verification studies.

Data Collection

Transects within 3.5 km of each fixed monitor site were used for comparisons to fixed monitor data. This created a larger data set than if only transects that were adjacent to the monitor sites were used. Larger data sets reduce the type II error, that is, the probability of accepting H_0 when H_0 is false. The paired test requires at least two pairs of data for each site. This constraint eliminated three stations, leaving 23 for further possible analysis.

Results

Because of the number of comparisons that were desired, SPSS was used to analyze the data. A paired t-test was run on each of the 23 fixed monitor sites and their comparable transect data. The results of these analyses are shown in Table 3.

Table 3
Verification of Fixed Monitor Station Location with Transect Data

Station	FMS Mean*	Trans. Mean*	Dif. Mean*	Dif. s.d.*	T Value	d.f.	2-Tail Sig.	Relationship**
BON@	108.5	111.3	–2.8	0.4	–16.1	3	0.001	FMS > Transect
CWMW@	113.7	116.2	–2.6	1.1	–6.6	7	0.000	Transect > FMS
HPKW#	113.7	116.0	–2.3	4.0	–0.8	1	0.565	Accept Null Hypoth.
IDSB@	126.8	120.9	5.9	6.8	3.1	12	0.009	FMS > Transect
IDSW@	126.9	120.9	6.1	6.0	3.7	12	0.003	FMS > Transect
IHR#	111.8	111.9	–0.1	0.5	–0.3	2	0.794	Accept Null Hypoth.
JDA@	107.8	106.0	1.8	0.2	15.2	1	0.042	FMS > Transect
JHAW@	109.8	106.5	3.3	3.7	2.4	6	0.053	FMS > Transect
KLAW#	109.8	110.2	–0.5	0.5	–2.0	3	0.152	Accept Null Hypoth.
LGNW	109.3	108.9	0.4	1.8	1.0	13	0.362	Accept Null Hypoth.
LGS@	107.3	108.1	–0.7	0.0	–33.7	1	0.019	Transect > FMS
LGSW	110.7	113.7	–3.0	7.8	–1.5	13	0.169	Accept Null Hypoth.
LMNW@	117.6	113.9	3.8	0.8	13.7	7	0.000	FMS > Transect
MCPW@	117.9	115.4	2.4	2.6	3.7	15	0.002	FMS > Transect
MCQO#	113.9	112.7	1.1	1.4	1.6	3	0.202	Accept Null Hypoth.
MCQW#	112.0	112.7	–0.7	2.2	–0.6	3	0.569	Accept Null Hypoth.
SKAW@	112.9	114.1	–1.2	1.2	–2.8	7	0.026	Transect > FMS
TDA#	106.0	106.2	–0.1	0.8	–0.2	1	0.852	Accept Null Hypoth.
TDAB#	105.8	106.2	–0.3	0.7	–0.7	1	0.621	Accept Null Hypoth.
TDTO@	112.0	115.5	–3.5	1.3	–8.6	9	0.000	Transect > FMS
WANO#	106.5	106.6	–0.1	0.2	–0.5	2	0.682	Accept Null Hypoth.
WRNB	113.5	114.0	–0.5	0.9	–1.4	6	0.227	Accept Null Hypoth.
WRNO	114.4	114.0	0.5	0.8	1.7	6	0.147	Accept Null Hypoth.
AGGR. FILE	114.8	113.9	0.9	4.6	2.6	156	0.012	FMS > Transect

* Variable is total dissolved gas percent saturation.
** Decision made at alpha = 0.05 significance level.
@ Additional study recommended.
# Additional data collection recommended.

The “Relationship” column was created by comparing the “T value” column (t_0) with values from a Students’ t table using the degrees of freedom in the “d.f.” column. First, we tested to see if the difference was zero. If this was not rejected, we labeled the “Relationship” column “Accept Null Hypoth.”

If the null hypothesis was rejected, Equations 7 and 8 were used with the appropriate values from the Students’ t table to determine whether the transect data were greater or lesser than the fixed monitor station (FMS) data. These results were labeled in the “Relationship” column as “Transect > FMS” or “FMS > Transect,” respectively.

For 11 of the 23 stations, the statistical tests rejected the hypothesis that the FMS and transect data were equal. This means that data collected at these FMS sites did not reflect the water quality conditions occurring across the river.

These stations, which had nonequivalent FMS and transect comparisons, are marked with an ampersand. It is recommended that further analysis be conducted on these stations to determine if the fixed monitor system needs to be moved, modified, or increased in scope. It is possible that the differences detected occur uniformly, allowing a simple addition or subtraction from the FMS data to then accurately represent river conditions. If the variance is large temporally or spatially, these stations should be relocated. To ensure the validity of these conclusions, it is generally accepted that a sample size of at least seven is necessary.

At the remaining 12 stations, the null hypothesis that the FMS and transect data were equal was accepted. This may be because the FMS adequately represents the transect, or simply because the limited data did not provide sufficient evidence to reject the null hypothesis. Thus, further analysis is needed to determine whether the monitors represent the flow.

Determining the Power of the Test

Using Equation 9, we calculated the statistic d for each station where the null hypothesis was accepted. These results are shown in Table 4. The table shows that in no case is the power greater than 0.32. Thus, we conclude that in each case where the conclusion of the test was to accept the null hypothesis (fixed monitor data represents water quality conditions in the river), there are insufficient data to make a reasonable statistical decision.

We next calculated the necessary sample size for each of these 12 stations to obtain the desired target power of 0.8. These values are shown in Table 5. With the exception of stations KLAW and MCQO, the sample sizes are somewhat unrealistic. This occurs because of the relationships between the sample means and standard deviations.

From Equation 9, $d = \frac{|\overline{D}|}{S_D}$. The power of the test relies on this relationship, in addition to the sample size n . In these other stations, the variance is so large compared with the mean that sample sizes are not reasonable. This implies that the fixed monitors are not located in such a way that their values change uniformly with the flow values. Thus, a first step at improving these monitors would be to place them in locations experiencing more uniform changes with flow and to increase the number of fixed monitor locations across the flow.

Table 4
Calculation of Parameters Needed to Determine *d*, β, and the Power of the Test

Station	$\overline{D}$	S_D	$d = \frac{\lvert\overline{D}\rvert}{S_D}$	*n*	β from Table	Power
HPKW	–2.3	4.0	0.58	2	0.94	0.06
IHR	–0.1	0.5	0.20	3	0.96	0.04
KLAW	–0.5	0.5	1.0	4	0.74	0.26
LGNW	0.4	1.8	0.22	14	0.90	0.10
LGSW	–3.0	7.8	0.38	14	0.76	0.24
MCQO	1.1	1.4	0.79	4	0.79	0.21
MCQW	–0.7	2.2	0.32	4	0.93	0.07
TDA	–0.1	0.8	0.13	2	0.97	0.03
TDAB	–0.3	0.7	0.43	2	0.95	0.05
WANO	–0.1	0.2	0.50	3	0.92	0.08
WRNB	–0.5	0.9	0.56	7	0.72	0.28
WRNO	0.5	0.8	0.63	7	0.68	0.32

Table 5
Determination of Sample Size Needed to Obtain Desired Power of 0.8

Station	$d = \frac{\lvert\overline{D}\rvert}{S_D}$	*n*
HPKW	0.58	28
IHR	0.20	300
KLAW	1.0	10
LGNW	0.22	300
LGSW	0.38	75
MCQO	0.79	15
MCQW	0.32	75
TDA	0.13	400
TDAB	0.43	50
WANO	0.50	32
WRNB	0.56	30
WRNO	0.63	25

Conclusions

This technical note has demonstrated statistical techniques for verifying the representativeness of fixed monitoring systems that monitor mean values of parameters in flowing water. These techniques were illustrated with data collected on the Columbia and Snake Rivers. Based on the criteria detailed in this technical note, a preliminary analysis of the 1995 Columbia and Snake Rivers fixed monitor system data set revealed that none of the fixed monitor systems accurately represented the average river total dissolved gas concentrations. This demonstration was, however, based on limited transect data, which were not specifically collected for the purposes of monitor site verification.

These examples given in this technical note illustrate use of the statistical approach to eliminate the subjectiveness involved in determining whether a monitoring station accurately represents the water quality in a river. The information presented can be used to guide managers to the most problematic locations, so improvements can be made on a “worst-case first” basis. Additionally, pilot studies similar to the ones used to collect the data used in this technical note can be used to help design verification studies to control the power of the test, obtaining the desired trust in the results.

Many other factors, such as cost, ease of accessibility, and equipment availability, contribute to the difficulties in monitor system design and installation. The cost of an intensive analysis like the ones described above may be prohibitive to many water quality managers. However, the ideas presented herein should make the manager more aware of the difficulties involved in collecting representative data and improve the final system design.

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WQRP

WATER QUALITY RESEARCH PROGRAM

APPENDIX 3.1A

Elements of a Model Program for Nonpoint Source Pollution Control

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Deanna L. Osmond, and Frank J. Humenik

Elements of a Model Program for Nonpoint Source Pollution Control

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ABSTRACT

Rural Clean Water Program (RCWP) projects have contributed significantly to the knowledge necessary for reducing nonpoint source pollution and achieving water quality goals. An RCWP evaluation, conducted during 1991 and 1992 by the National Water Quality Evaluation Project (NWQEP) at North Carolina State University, shows that many of the 21 projects were highly effective and others had some effective elements. When expected results were not achieved, the NWQEP attempted to analyze program and project deficiencies that may have affected the outcome. Despite difficulties, the RCWP is the best program to date for evaluating agricultural nonpoint source pollution control methods, and it should serve as a model for developing future programs. RCWP was effective because it had good overall program management and institutional arrangements that encouraged consultation between the U.S. Department of Agriculture (USDA) and the U.S. Environmental Protection Agency, excellent program guidance, and effective technical support for reviewing reports and providing ongoing evaluation.

NWQEP has developed a model program and a model project based on the RCWP and added refinements to strengthen weaker elements identified during the evaluation. The model program includes a technical support group with access to resources for visiting project sites to assist in project selection, monitoring, evaluation, and developing the plan of work. The model program needs technical support from the USDA Agricultural Research Service (ARS) for developing and evaluating best management practices. Increased assistance for monitoring is also needed and should be provided by the U.S. Geological Survey.

NWQEP suggests three project levels, based on complexity and level of monitoring detail. This paper discusses the NWQEP model and lists monitoring protocols that should be incorporated in program guidance to improve the chances of detecting water quality trends. All projects need a preimplementation plan of work development to strengthen the problem definitions, select the critical area, model the watershed to set treatment goals, and establish a means for land treatment tracking. Projects also need a manager, technical support, and core project staff to improve efficiency and encourage accountability.

A growing awareness of agriculture's contribution to the nonpoint source pollution problem, increasing concern about water quality, and pressure from special interest groups have influenced Federal agencies to respond to these concerns by developing a demonstration of nonpoint source pollution control capabilities. In response to the 1989 President's Water Quality Initiative, the U.S. Department of Agriculture (USDA) has developed programs that accelerate soil conservation and best management practice (BMP) implementation on farms, ranches, highly erodable lands, and watershed projects. Implementing these programs produces many important benefits, including increased adoption of soil conservation practices and BMPs that improve water quality. Many programs, however, do not target specific critical area pollutant sources. With only limited targeting of pollution sources (and even less water quality monitoring to document the linkage between land treatment and water quality), our knowledge of the water quality benefits in these measures will not expand appreciably.

New nonpoint source control programs must build on current knowledge to be effective. The evaluation of the section 108a Great Lakes Demonstration projects showed that nonpoint source pollution is more persistent and more difficult to treat than previously thought (Newell et al. 1986). It also showed that using a pollutant runoff model to determine critical areas is an efficient way to use project funds. In addition, the Model Implementation Program (MIP) evaluation demonstrated that a project should target critical areas for treatment to improve the likelihood of success, and that BMPs should be selected and applied to promote water quality results (Natl. Water Qual. Eval. Proj. and Harbridge House, 1983a,b).

Lessons learned from the Rural- Clean Water Program (RCWP) provide critical information about nonpoint source pollution control technologies and approaches for the U.S. Environmental Protection Agency (EPA), U.S. Department of Agriculture (USDA), and other Federal, State, and local nonpoint source pollution control agencies and programs. The RCWP is significant among nonpoint source control programs because it combines land treatment with water quality monitoring to document the effectiveness of nonpoint source controls.

The RCWP has 21 projects located in nearly every region of the United States that address a wide range of water quality problems. The program is unique in that it received a higher level of up-front funding for a longer period (10 to 15 years) than other federally sponsored nonpoint source programs. The longevity and dependability of RCWP funds enhanced efforts to establish a clear link between water quality and land treatment, and several RCWP projects have been able to demonstrate such a link, The publication of RCWP rules and regulations in the *Federal Register* (1980a) provided clear guidelines for RCWP projects, facilitating the overall program evaluation by standardizing many of the projects' administrative and technical aspects. Finally, the approach taken to address water quality problems — providing Federal cost-share funds to producers willing to implement BMPs — makes the RCWP experiment important as a way to evaluate voluntary versus regulatory approaches to the problems of agricultural nonpoint source pollution.

Because of its unique characteristics as an experiment in nonpoint source control, the RCWP is an important source of insights and technology transfer for the many ongoing and future nonpoint source programs, including the 319 National Monitoring Projects, other shorter-term 319 projects, the USDA Demonstration and Hydrologic Unit Projects, the Clean Lakes Program, and State nonpoint source programs, among others. Because so many other nonpoint source programs are being planned and conducted, the need for clear articulation and dissemination of the lessons learned from the RCWP is even more important. To share these valuable lessons in the most effective way possible, the National Water Quality Evaluation Project (NWQEP) has restated them as a set of recommendations for a model nonpoint source pollution control program and project.

NWQEP's evaluation of the RCWP has been conducted to establish a set of recommendations for developing Federal nonpoint source pollution control and water quality programs — programs whose primary goal is to evaluate the water quality improvements from nonpoint source controls. The objectives of the evaluation were to assess

- cooperation among project team members, committees, and agencies;
- agreement between the water quality problem and the choice of solutions;
- project achievements;
- results of monitoring and assessment of project impacts; and
- project findings to compile lessons learned.

Methods

For the program analysis, we reviewed the MIP evaluation (Natl. Water Qual. Eval. Proj. and Harbridge House, 1983a,b) and the section 108a Great Lakes Demonstration Programs (Newell et al. 1986). We also reviewed literature for the USDA President's Water Quality Initiative and the EPA's section 319 Nonpoint Source Program (U.S. Environ. Prot. Agency, 1991). From these reviews we gained valuable insights on methods that could be used to evaluate the RCWP.

Including our own past experience, we used five sources of information to evaluate RCWP projects:-

1. an in-person interview questionnaire for project personnel during site visits,

2. a short answer questionnaire administered to project personnel,

3. a telephone survey of producers who did not participate in the 21 projects,

4. 10-year reports from the RCWP projects, and

5. NWQEP's own 10 years' experience in offering technical assistance to the projects and performing program evaluations.

For site-visit evaluations, an interagency evaluation team (led by a NWQEP member) visited each project. In-person interviews of local and state project staffs using a standardized questionnaire were conducted during site visits (Coffey and Smolen, 1991). Questions were designed to gather specific information on project elements, including State and local coordination, local program administration, information and education, land treatment, and water qualify monitoring and evaluation.

Project staff responses to a short answer questionnaire (Coffey and Hoban, 1992) were used to gather information on project coordination, advisory committees, project effectiveness, Information and Education (I&E), farm operator participation, and BMP implementation. A companion telephone survey of farm operators was used to determine factors that influenced participation and BMP implementation (Hoban and Wimberley, 1992). RCWP projects also produced detailed 10-year reports that provided important insights, findings, and recommendations for each project element.

For each RCWP project, the NWQEP wrote a comprehensive analysis, including

- a project synopsis;
- a section on findings, successes, and recommendations for each of the project elements; and
- a detailed project description.

At the foundation of the analysis were the RCWP regulations and the findings from individual RCWP project evaluations. The results of the RCWP analysis are presented here as a set of recommendations for a model program and a model project, including selected examples from RCWP projects that support the results.

Results

Based on NWQEP's review of agricultural nonpoint source pollution control programs, the RCWP is, to date, the best program available for achieving water quality goals. For example, the RCWP had a set of rules and regulations (Federal *Register*, 1980), technical oversight, and secure, long-term funding. Some projects have documented water quality improvements, and all projects have contributed to a greater understanding of water quality problems and to cooperation among agencies charged with address ing nonpoint source pollution.

The overall RCWP assessment has shown that it was not possible to document water quality benefits for RCWP projects in which

- agricultural activities were not the primary pollution source,
- the area1 extent and magnitude of land treatment was inadequate, or
- the monitoring designs were inadequate to document water quality improvements.

However, each project did have one or more nonpoint source pollution control benefits, including

- development of cooperative relationships among Federal, State, and local agencies necessary to achieve an effective nonpoint source pollution control program;
- achievement of widespread adoption of BMPs to improve water quality under this assistance program;
- visual improvements in water quality associated with the use of BMPs; or
- water quality improvements documented by water quality monitoring.

Therefore, the model program and project described herein builds on the RCWP's structure and essential features, while adding refinements to strengthen weaker components identified during the RCWP evaluation.

Elements of a Mode! Program for Evaluating Nonpoint Source Pollution Controls

Guidance written in the form of regulations must be available to help implement the program (Federal. *Register,* 1980a). The model program's major features (as outlined in the RCWP regulations) will include

- clearly defined responsibilities of Federal, State, and local agencies and landowners or operators;
- criteria for project selection, approval, and implementation;
- contracting requirements for technical and financial assistance to farm operators:
- provisions for project funding and termination;
- requirements for making cost-share payments to participants; and
- plans for program and project monitoring and evaluation.

The model program guidance will include these important features and strengthen water quality and land treatment monitoring, evaluation, and reporting.

Program guidance would also list the roles of project staff at the Federal, State, and local levels, and would help staff understand the responsibilities of interagency counterparts.

The RCWP objectives were to

- achieve improved water quality in the most cost-effective manner possible in keeping with the provision of adequate supplies of food, fiber, and a quality environment;
- help agricultural landowners and operators reduce agricultural nonpoint source pollutants and improve water quality in rural areas to meet water quality standards or goals; and
- develop and test programs, policies, and pro cedures for the control of agricultural nonpoint source pollution.

These objectives can be restated as model program objectives that are relevant, comprehensive, and nonoverlapping. Thus, the model program is to

- achieve improved water quality to restore and protect the designated use of surface or groundwater resources,
- help agricultural landowners and operators reduce agricultural nonpoint source pollutants and habitat perturbations, and
- develop, test, and evaluate policies and programs to control agricultural nonpoint source pollution.

Program Administration and Management

The model nonpoint source pollution control program should be administered by a single department. The USDA would administer the model program in consultation with the EPA Administrator and the Director of the U.S. Geological Survey (USGS). Or, the Secretary of Agriculture could delegate the responsibility of program administration to the ASCS, which has a long history of program administration and leadership that contributed to the RCWP's success. Local project funding would have to be received on time through the State ASCS office. Technical assistance for identifying and documenting the water quality problem through monitoring and evaluation could be provided by EPA. Technical assistance for land treatment and land treatment monitoring would become a joint responsibility of USDA Soil Conservation Service (SCS) and Extension Services. The Extension Service (ES) should be responsible for information, education, and BMP recommendations. The SCS should be responsible for the development of farm plans and structural BMPs.

The Agricultural Research Service (ARS) would also provide technical assistance for developing and evaluating BMPs. Technical assistance for water quality monitoring and linking land treatment data to water quality data would be coordinated by EPA, with additional technical assistance on sampling, instrumentation, and data management from USGS and ARS.

The model program will also need a national coordinating committee to oversee functions currently defined for this committee by the RCWP, including

- developing program regulations and cost-share rates,
- reviewing technical aspects,
- selecting projects to fund based on a technical assessment of likelihood of success,
- developing annual project reviews, and
- reviewing project progress.

The national coordinating committee should have the ability to assign provisional status to projects if State or local program staffs are not meeting minimum performance standards. In addition, the committee should have the authority to terminate projects that fail to meet minimum requirements after two complete years on provisional status.

■ Program Planning. Program planning is necessary to ensure adequate attention to all project elements and stages of deveiopment. Problem identification, the selection of critical areas, and the development of project proposals precede funding (Fig. 1). The first two elements may extend into the first year to allow refinement. Assistance from a technical support group is also needed before funding and throughout the project.

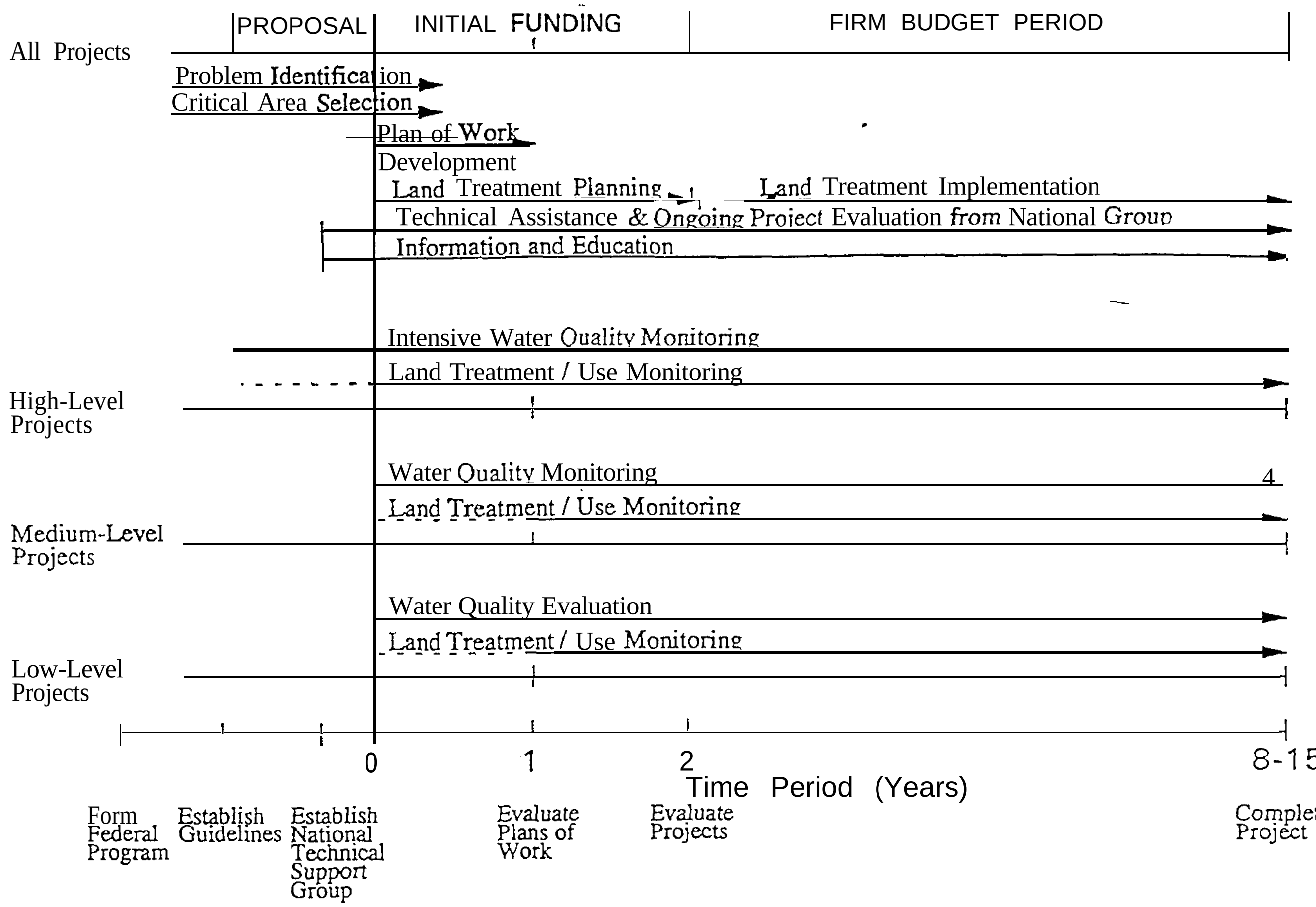

Figure 1.—Model program and project timetable.

Project funding occurs in stages (Fig. 1). The first two years are designated for initial funding only, whereby projects may be terminated at any time. Successful advancement to the firm budget period (after which the project will be funded continuously throughout its life) requires

- detailed and accurate problem identification,
- adequate selection of critical areas based on problem pollutants,
- a detailed plan of work for land treatment and water quality monitoring, and
- demonstrated progress toward key agency cooperation and the development of institutional arrangements.

Land treatment planning before BMP implementation is critical to ensure adequate targeting of resources. We suggest allocating two years for this activity so that technical assistance may be sought if needed. A baseline should also be established for water quality monitoring before BMP implementation. Succeeding land treatment periods are for installing structural and managemenr BMPs. Water quality monitoring must be consistent before, during, and after the implementation periods.

Proaram Technical Support

The model program must be structured so that projects are carefully selected to improve chances for meeting program objectives and obtaining water quality improvements. A national technical support group (outside the administrative organizations and the national coordinating committee) should be in place at program initiation to help develop program and project guidelines. This group should also provide technical assistance during (and after) the planning period for project selection, critical area and BMP designation, watershed modeling, land treatment and water quality monitoring, and project evaluation.

This technical support group must be provided adequate resources for site visits to projects to gain information and develop cooperative relationships before project selection, during BMP and monitoring implementation, and during final project e-:+:x-tion. The support group may respond to program and project technical requests from administrative agencies and individual projects and be responsible for verifying the accuracy and completeness of water quality analyses. Finally, the support group may take responsibility for the final evaluation by emphasizing lessons learned, identifying water quality improvements, and making recommendations for future programs.

Project Selection

Project selection is a key factor for program success. Selection criteria are needed to ensure that all possible projects are evaluated for their potential to contribute to program objectives.- Projects should be selected because they have

- high priority water resources with documented water quality problems, or
- highly valued water resources threatened by documented agricultural nonpoint source pollution (because prevention of severe degradation is often more cost-effective than restoration).

In addition, projects should have the following characteristics:

- water resources having the highest public use value (e.g., recreation or water supply) because these projects can show a significant economic benefit:
- smaller watersheds of less than 30,000 acres because problems in these areas can be more readily identified, are easier to treat, and respond more rapidly to treatment;
- the potential for effective control of nonpoint source pollutants;
- the capability to use water quality models and monitoring to determine if significant pollution reductions are likely with BMP implementation;
- clearly stated objectives and goals related to water quality impairments or conditions threatening designated use;
- the ability to establish and maintain strong interagency cooperation and institutional project coordination;
- well-defined critical areas in which implementation of BMPs targeted to a specific pollutant (or group of pollutants) can be emphasized;
- the potential for a high level of landowner participation in the critical area;
- the potential that Iandowners will accept and implement the necessary BMPs and, perhaps, adopt alternative agricultural systems (e.g., changing ‘from row crops to hayland or pasture), which are integrally tied to water quality improvements and project goals;
- a plan of work development process to obtain baseline monitoring data, determine problems, refine critical areas and develop BMP systems, conduct I&E programs, and document effective project administration staffing and cooperative relationships;
- the ability to conduct an effective I&E program in advance to determine if key BMPs (e.g., fencing or dairy waste use) will be acceptable to farm operators;
- the characterization of the hydrology and pollutant transport system to allow adequate development of water quality goals and monitoring systems; and
- the ability to monitor explanatory variables, such as season, stream discharge, water table depth, precipitation, other hydrologic and meteorologic variables, and land use changes.

The most successful RCWP projects were those that met most or all of these criteria. The Florida, Idaho, Utah, Vermont, and Oregon RCWP projects contained most of these elements and were among the most successful projects in implementing land treatment and documenting water quality improvements as a result of RCWP treatment. For example, the Utah RCWP project was relatively small (700 acres) with a well-defined critical area in which BMPs were targeted to the major source of pollutants (i.e., the dairies). Also, the project had a high level of landowner participation in the critical area. The Utah project’s commitment to a two-year preproject monitoring program proved to be the key monitoring element that helped document substantial water quality improvements.

Several other effective projects contained many of the stated criteria but could have been strengthened if the missing elements had been present. The Nebraska RCWP project, for example, suffered in its early years from the lack of clearly defined water quality and land treatment goals. However, this project developed quantitative water quality and land treatment goals, a critical area definition that included BMPs targeted to sediment and erosion control, and a strong I&E program — resulting in a high level of landowner participation in the critical area.

The Delaware and Maryland RCWP projects were successful but lacked preproject water quality monitoring baselines, which impeded the ability to make quantitative statements regarding water quality improvements. The Iowa project contained most of the suggested components but had only a one-year pre-BMP monitoring database and initially did not understand that the turbidity problem in Prairie Rose Lake resulted not only from incoming sediment but also from resuspended sediment and algal growth.

Although the Massachusetts RCWP project met several key project selection criteria in that the Westport River estuary was a high priority resource with significant economic value (shellfish beds), the' source of the water quality problem was not well documented. This lack of clarity was one of several factors that contributed to a lack of consensus within the community and, therefore, to poor producer participation. The Kansas RCWP project also lacked a clearly documented water quality problem that could be linked to a critical area pollutant source. Careful application of project selection criteria could have prevented the selection of this project and its sub sequent termination three years later.

The Michigan RCWP project had only vague information indicating that the Saline River was a large contributor of nutrients (mainly phosphorus) to Lake Erie. The project had not clearly identified the critical pollutant source or critical area, and the project did not document any water -use impairments. On the other hand, the Pennsylvania RCWP project presented a documented water quality impairment of agricultural origins and had the high visibility of a project that could reduce pollutants entering Chesapeake Bay. However, careful evaluation of project potential would have shown that the large number of small farms and the conservative nature of the farmers would impede BMP acceptance and implementation, thereby limiting the project's potential.

Program Funding

In the RCWP, all funds were identified and made available at each project's initiation so that long-term project planning and budgeting were possible. In contrast, budgets for the current USDA Demonstration and Hydrologic Unit projects must be approved each year. The associated delays have caused work plan uncertainties, budgetary burdens on State and local agencies, and incompatibility with fiscal budget requirements. In the model program, funds should be provided for preproject planning periods, which may last from six months to two years (as defined under "Project Proposal and Plan of Work Develop ment").

Elements of a Model Project for the Evaluation of Nonpoint Source Pollution Controls

The model *program,* which is based on RCWP guidelines, carefully selects the individual *projects* that will be undertaken. The model *project is* based on the outline provided by RCWP regulations. The following discussion of the model project is supported by examples from the RCWP projects. The model project would operate under the primary authority of USDA with consultation and concurrence from EPA ASCS would be the administrative lead agency. SCS should be responsible for the development of structural BMPs, and ES should be responsible for I&E and management BMPs. While agencies supervise project activities, committees would be responsible for setting priorities and coordination. All agencies, committees, and program participants would be guided by model program regulations published in the *Federal Register.*

Project Administration and Management

Because of its management abilities, administration for the model project at the State and local levels should remain with ASCS. To implement the project at the State level, each successful project must have strong administrative and technical support from a State coordinating committee, which also provides a link to the national coordinating committee and the local coordinating committee. The local coordinating committee needs to have strong and continual support from the State coordinating committee, which must establish and maintain open communication lines and a willingness to allow the local coordinating committee to implement the project.

The fundamental project administration and management elements are a local coordinating committee, a county Agricultural Stabilization and Conservation (ASC) committee, a project manager, and project advisory committees. The local coordinating committee should provide guidance for the agencies, community leaders, and citizens to oversee the administrative and technical tasks of a local project. The committee serves many functions, including

- assuring an adequate level of public participation,
- developing a plan of work,
- enlisting the help of needed agencies,
- overseeing information and educational activities,
- determining priorities for water quality plans,

- enlisting the help of one or more agencies for land treatment and water quality monitoring and evaluation,
- developing a plan for critical area selection,
- creating a plan for implementing targeted recommendations,
- establishing a plan for linking land treatment and water quality data and analysis, and
- developing a plan for project reporting.

The Florida, Vermont, Idaho, South Dakota, Pennsylvania, Oregon, Delaware, Utah, Maryland, and Iowa RCWP projects all had strong local committees that contributed profoundly to the success of their projects.

The county ASC committee, elected by county farm operators, is responsible for encouraging project participation and compliance. It can also play a major role in promoting the project. The involvement of the county ASC committee for the Appoquinimink River RCWP project in Delaware was a significant factor affecting participation: BMPs were implemented in 87 percent of the project's critical area.

A project manager is also essential (Brichford and Smolen, 1991). The manager should have a water quality and management background, ideally should work with the project from its inception, and hold the designation for the length of the project. The manager coordinates and monitors all project activities, including project reports, and has the authority to exert pressure on agencies or individuals not performing adequately. The project manager is responsible to the local and State coordinating committees and can report problems and successes directly to the national coordinating committee.

Examples of RCWP projects that used managers with positive results are South Dakota, Kansas, Virginia, and Minnesota. The South- Dakota project hired a temporary, full-time manager during its initial phase to conduct individual visits with farmers to lay groundwork for their participation. The manager also organized project activities and compiled information so that the local coordinating committee could operate quickly and efficiently. The position continued until the last few years of the project. Likewise, Minnesota RCWP project recommendations suggested that a manager should be hired at the program's start who is familiar with all government agencies involved in the project but autonomous. A half-time manager was hired in Minnesota after the project had begun. As a respected area farmer, the project assistant was able to encourage the participation of his neighbors through one-on-one visits, well testing, and newsletter preparation.

Project advisory committees (e.g., administrative, technical, I&E, land treatment and water quality monitoring and evaluation, and modeling) are useful for gaining progress in areas where input from a smaller, more focused group improves decisionmaking. Advisory committees should be -formed, disbanded, or regrouped as needed. For example, an advisory committee comprised of land treatment and water quality monitoring and modeling personnel can help coordinate efforts to link land treatment and water quality information. In the Vermont RCWP, an advisory committee proved to be highly effective; it ensured cooperation among agencies and kept work activities on schedule. Similarly, the key to success in the Florida RCWP project was the implementation of an administrative subcommittee. The subcommittee (comprised of major agencies) met regularly to coordinate project activities.

Project Proposal and Plan of Work Development

Activities before project start-up influence the operation and success of each project and the total program. Pre-project programs and periods are specified in Figure 1 for three different levels of projects based on problem magnitude, monitoring intensity, and project complexity.

Initially, the Federal program administration is formed to develop and publish program and project guidelines. Thereafter, a proposal development period without funding is specified for all three project levels. The national technical support group provides leadership for proposal evaluation and determines which projects will be funded for plan of work development.

The high-level, or most complex, projects are required to have baseline water quality monitoring data or to initiate water quality monitoring during the proposal development period. Monitoring of water quality explanatory variables and land treatment are to continue throughout the total project.

Medium-level projects may begin water quality monitoring during the plan of work development and continue for the total project. Land treatment monitoring will be conducted throughout the project period. Sampling design for water quality and explanatory variables would be less comprehensive at this level than in a high-level project.

Projects at the lower level may require periodic water quality evaluation, such as visual examinations or simple measurements of an unambiguous water quality problem, or a citizens' group may provide monitoring.

After a successful two-year initial funding period, firm budget can be allocated and guaranteed for the duration of the project as long as satisfactory progress continues on the project.

Project Technical Support

To provide technical support for the project's first two or three years, a minimum core project staff must be created using individuals from the cooperating agencies. Core project staff will be responsible for project activities and required to work cooperatively with the project manager. Core project staff and the lead administrative agencies will have primary authority over project technical activities but will also seek input from other agency staff, farm operators, and local groups. Final technical decisions need not require a consensus of local coordinating committee members as long as decisions are consistent with program guidance and recommendations from the national technical support group.

Because they will be accountable for project progress, the core project staff will have a great investment in the project. Agencies must establish a mechanism for accountability and credit for good performance. The minimum core project staff should consist of a land treatment planner, and an I&E specialist. In the Alabama RCWP project where over 100 percent of the critical area was treated with BMPs, an extension agent was instrumental in encouraging producer participation.

A full-time planner will be needed to help develop farm plans, assist in BMP installation, help farm operators maintain practices, and track land treatment. Other core project staff positions beyond the minimum (e.g., an engineer, a water quality monitoring specialist, and an agronomist) may be needed.

When an adequate level of technical capability is not available at the project level, outside help should be employed to assist the project. Core project staff at the local level will enjoy greater freedom of communication and have a larger team of experts for technical support, compared to the limited communication that happens when technical assistance must be sought through line agency procedures. In the Idaho RCWP project, ARS provided valuable research and recommendations regarding the develop ment and evaluation of conventional and new BMPs, particularly conservation tillage and no-tillage.

Because staff turnover can be problematic, incentives should be provided to encourage core project staff to make a minimum commitment of three years to the project. In the Louisiana RCWP project, annual turnover of the SCS soil scientist hired specifically to help implement the RCWP made it difficult to track BMP implementation and maintain consistency.

Problem Definition

Water quality monitoring cannot be left as an afterthought in an effective nonpoint source project Monitoring must be used to identify specific pollutants (and their variability) responsible for the impairment or threat to designated use. Initital problem identification monitoring serves to help the project team understand sources and response characteristics of the affected water resource. The RCWP projects have vividly illustrated that clear identification of the source of the water quality problem and acceptance of this information by the public and producers are crucial to project success.

In Iowa, heavy sediment and a blanket of corn stalks covering a recreational lake surrounded by farmland helped make the problem and its source especially clear. RCWP projects in Utah, Vermont, Florida, Idaho, Nebraska, Oregon, and Pennsylvania also had ample visual and analytical evidence of problems in the receiving waters. In Massachusetts, however, where both intensive dairy farming on small acreages and booming residential development were taking place adjacent to an estuary containing important shellfish resources, the source of the problem needed to be more clearly documented to generate community support for project activities. South Dakota's project required several intense monitoring programs to gain a thorough understanding of the water quality problem and its causes because of complex interactions between the surface and groundwater sources feeding the target lakes.

Refinement of problem definition may occur as the result of new information obtained from water quality monitoring or modeling. Monitoring provides a way to track BMP effectiveness and progress toward water quality goals. Feedback on project effectiveness provided by monitoring is important to land treatment personnel and farm operators. For example, Vermont's RCWP project was able to reduce bacterial contamination enough to reopen public beaches for swimming. This accomplishment was heavily promoted in the news media, which gave the participating farmers pride and an investment in nonpoint source control and their project.

Project Plan of Work and Time Frame

The plan of work is a written strategy used to organize agencies, project staff, and interested parties for project implementation. An effective plan is difficult to write, primarily because the linkage between land treatment and water quality is not known with certainty. A national technical support group is needed to help the project address key obstacles, define the water quality problems, and develop effective land treatment and water quality monitoring strategies.

Project objectives and goals as stated in the plan of work-must be measurable, quantitative, and (for the most part) attainable, given best available information. Project objectives and goals must be critically reviewed to ensure consistency with overall program objectives and goals.

■ Time Frame. A model project should last from 6 to 15 years, depending on size and the ability to implement land treatment. The median project length should be 8 to 10 years, but some projects may need 12 to 15 years to implement enough practices and document results. Larger areas could require long periods to show improvement. Examples of projects that successfully made use of longer time frames are the Idaho, Florida, Oregon, and Utah RCWP projects. The long pre- and post-BMP water quality and land treatment monitoring time frames for these projects, along with high levels of BMP implementation, made it possible to track irrigation water management, sediment control structures, and conservation tillage in the Idaho project, and animal waste management in the Florida, Oregon, and Utah projects. On the other hand, the Pennsylvania RCWP project found that more time was needed than originally expected to establish firmly the reduction in nutrient levels from BMP implementation on experimental sites.

■ Critical Area Definition. Critical areas are pollutant source areas in which the greatest improvement in the water resource can be obtained for the least investment in BMPs (Maas et al. 1987). The effectiveness of a nonpoint source pollution control program is likely to be a function of where, when, and how many BMPs are installed. Therefore, cost-share funding should only be available for the treatment of critical areas. Smolen (1988) reports that in critical areas cause and effect are clear, hydrology is simple, and response time to treatment is short. The Utah, Oregon, and Vermont RCWP projects documented major reductions in bacterial concentrations resulting from land treatment efforts in animal waste management. The project areas exhibit simple surface water hydrology, and treatment occurred in the critical areas. Bacterial populations, especially in surface waters, respond to BMP implementation, thus making bacteria in water a prime candidate to demonstrate project effectiveness.

■ Targeting BMP Systems. BMP systems directed at water quality improvements are far more effective than the installation and maintenance of individual BMPs. In Oregon, for example, the development and use of BMP systems to store and use manure were essential in reducing fecal coliform levels in Tillamook Bay. However, whether a BMP system or an individual BMP is to be used, each should be targeted to control specific pollutants identified in the water quality problem definition and project plan of work.

For example, BMP systems used to control lake sedimentation may be different from and target a different soil particle size than systems used to control lake turbidity. The South Dakota RCWP project targeted its BMPs to a specific problem; consequently, nutrient management was found to be the most effective BMP for reducing nutrient contamination in an area dominated by cropland with only a few scattered animal operations. On the other hand, the Utah RCWP project saw marked improvements in phosphates through animal waste management systems in a watershed totally composed of animal operations.

Implementing the Plan of Work

Federal agencies and committees provide direction and funding to support local administration and coordination of project activities such as I&E and land treatment_ Local committees, however, are responsible for carefully defining project objectives and implementing project activities to meet goals. In addition, local committees receive guidance and sup port from the State coordinating committee and the national coordinating committee.

■ Information and Education. Extension Service should provide leadership for the development, implementation, and coordination of I&E programs for agricultural nonpoint source water pollution control. The local coordinating committee, the county ASC committee, the soil and water conservation district, and SCS should help with I&E efforts to ensure that the I&E message is being received by participants.

During the proposal development, the community and relevant agencies must be informed about problems in the project area, objectives, and design. Local people also need to take part in decisions from the start. An advance I&E effort should be used to ensure that the majority of the population and project staff agree about the problem, its causes, and the treatment approach. The effect of general and farm community support (or lack of support) was clearly demonstrated in several RCWP projects. In the Iowa RCWP project, three public meetings were held to inform the community about the RCWP before the Prairie Rose Lake project ap plication was submitted. This strategy of early community involvement helped the project to a strong start. Delaware producers also participated in the selection of and planning for the Appoquinimink River RCWP project, again contributing to a successful effort with strong producer participation. The Westport River RCWP project in Massachusetts, on the other hand, would have benefited from advanced information and education programming (as well as water quality monitoring for baseline data collection) to address and resolve conflicting views on the source of the water quality problem and the validity of the approach being recommended in the RCWP project.

Informational and educational efforts are take part in stages that change over time. Initially, the I&E team seeks to develop general awareness of the water quality problem and support for the project through mass media and public educational programs. Then, I&E seeks to increase farm operators' knowledge about nonpoint source control and improve their agricultural management skills through educational programs and one-on-one contact. Ultimately, I&E works to modify behavior by promoting the adoption of BMPs for improved management of agricultural chemicals, conservation of irrigation water, use of animal wastes, and conservation of soil.

The I&E message was received and implemented differently by the RCWP projects. For example, in Vermont, the efforts of the local Extension Service office were essential in informing producers and convincing them to participate in the RCWP In Tennessee, every farmer received at least one (and sometimes three) personal visit from an I&E team member to encourage participation. In Florida, field days, demonstration sites, and tours were the most effective methods for promoting land treatment and presenting accomplishments in the RCWP project.

Where fertilizer management and pesticide management are important parts of the BMP program, the I&E staff assists with soil sampling or pest scouting and provides tailored recommendations to project participants. The I&E program develops or strengthens existing commodity associations to sup port integrated pest management and other specialized programs.

Extension Service can also initiate other programs to improve water quality. A good example is the Pennsylvania RCWP project. There the Extension office set up an animal waste trading exchange to enable farmers who wanted animal manure to find farmers who had excess manure. The Nebraska RCWP developed a strong fertilizer testing and management program, along with pest scouting. Both components resulted in a significant decrease in the use of fertilizers and pesticides.

■ Producer Participation. Water quality improvements depend on changes in farm operators' attitudes, knowledge, and BMP implementation. Hoban and Wimberley (1992) surveyed eligible participants and nonparticipants from the 21 RCWP project areas. Their findings on the farm operators' water quality awareness, need for more information, attitudes about water quality problems, adoption of BMPs, and participation in RCWP and other programs provide significant information on ways to improve education and participation in water quality programs. In addition, results from the short answer questionnaire (Coffey and Hoban, 1992) show that cost-share funding was a key incentive to participation.

Other important factors affecting producer participation in RCWP projects included:

- strong leadership within the farm community (as demonstrated in Iowa and Oregon),
- consensus within the farm community and the general public on the source of water quality problems and the importance of water resources (for example, the high value placed on local recreational lakes by the Iowa and Delaware farmers in their projects' critical areas),
- the threat of regulation if the sources of pollution were not voluntarily reduced (as in the Taylor Creek-Nubbin Slough project in Florida),
- economic penalties for producers who did not participate (as in the Oregon RCWP project where producers received lower milk prices from the local cheese cooperative if they were not implementing BMPs), and
- producer perception that BMPs implemented to reach the project goals would also benefit the farming operation (as in Alabama).

Producer participation also came about through other means. Concern for stewardship of the land encouraged many Pennsylvania farmers to participate (many implemented BMPs but refused cost-share funding). In Vermont, a long-standing commitment to keep the community clean was the impetus for participation.

■ Land Treatment The Soil and Water Conservation District (SWCD) participates on the local coordinating committee, prepares applications, and promotes the project. The SWCD, together with the county ASC committee, determines the priority of technical assistance among applicants for water quality plans based on criteria developed by the local coordinating committee. The SWCD also approves water quality plans and revisions.

SCS coordinates technical assistance for BMPs and recommends the appropriate agency for assistance. SCS provides technical assistance for setting priorities among applicants and developing and certifying their water quality plans. The role of SCS as the lead technical agency for land treatment should be retained; however, the contribution that can be made by other agencies and opportunities for interagency cooperation in achieving land treatment goals should be recognized. As a result of the Massachusetts RCWP project, a new approach to farm visits was developed by the local USDA agencies; ASCS and SCS staff members now routinely visit farms together to perform their duties under several USDA programs.

The role of Extension Services should be expanded to emphasize management practices to complement structural practices. For example, during the latter phases of the Pennsylvania RCWP project, most of the land treatment effort was facilitated by the ES through individual contacts and nutrient management plans. For this project, the high number of farms needing animal waste storage facilities and the resistance to installing such facilities made the use of the ES and nutrient management plans the only effective way to reduce nutrients in the area streams.

■ Water Quality and Land Treatment Monitoring. The State water quality agency should participate on the State and local coordinating committees and monitor and evaluate the project's effectiveness. Because Federal assistance is required to encourage consistent and continuous water quality and land treatment monitoring throughout the project period, Federal funding for water quality monitoring must be authorized as a part of the model program. Funding for monitoring is required to document progress, the need for continued treatment, and water quality changes. Funding would be provided to all projects to meet minimum monitoring requirements for both land treatment and water quality.

Greater accountability by the State water quality or other monitoring agency is needed to ensure adequate water quality monitoring. Where applicable, USGS, ARS, local universities, SCS, and Extension Services should provide technical assistance for monitoring program design and implementation. Minimum monitoring protocols for high- and medium-level projects should be reported in the *Federal Register. Projects* would risk cancellation if monitoring efforts fail to meet minimum requirements.

All approved projects should have monitoring to determine BMP application progress and to document trends in one or more variables related to the water quality problem. Stream water quality monitoring requirements for high-level projects should be consistent with the EPA 319 National Monitoring Protocol (U.S. Environ. Prot. Agency, 1991; Spooner, 1992). The protocol requires 20 samples per season at a weekly or biweekly frequency for physical and chemical variables and measurements of explanatory variables (e.g., flow and precipitation) for each sample. If biological monitoring is desired, biological and habitat variables should be monitored one to three times per year. land use and land treatment data must be reported on a drainage basin relative to the water quality monitoring station. In addition, paired watershed studies are strongly encouraged.

The protocol's main objective for high- and medium-level projects is to monitor water quality and land treatment simultaneously to determine if water quality changes can be documented and associated with changes in land treatment. Two features of this objective must be met: (1) detecting significant or real trends in both water quality and land treatment implementation, and (2) associating water quality trends with land treatment trends.

Guidance for minimum monitoring of land treatment and associated water quality changes for the model program and its projects should be maintained and enhanced by EPA and USGS in consultation with other Federal, State, and local agencies. This approach will allow valid technical evaluations of individual projects. For example, the monitoring requirements established by the EPA Clean Lakes Program have been published in the *Federal Regisfer* (1980b). The lack of a complete and uniform database has limited the effectiveness of evaluations of the Model Implementation Program (MIP), RCWP, and (by current indications) the present USDA Demonstration Hydrologic Unit Areas as well as Management Systems Evaluation Areas (MESA) water quality project.

The paired watershed approach involves monitoring two or more similar subwatersheds before and after BMP implementation in one of the watersheds. This design is the most technically sound and reliable method available to document water quality changes in the shortest time period (3 to 5 years). The Vermont RCWP project employed the paired watershed approach successfully and demonstrated that winter storage of manure (instead of winter spreading) was an effective nutrient management strategy.

Land treatment information for high- and medium-level projects should be reported and linked directly to the water quality monitoring data, For example, each observation should be paired hydrologically to a water quality monitoring station on an annual or seasonal basis. All significant land use changes and other nonpoint and point source control efforts should be documented. The monitoring design should include multiyear monitoring of both land treatment or use and water quality before and after BMP implementation.

Several RCWP projects had strong water quality monitoring programs emphasizing pre- and post-BMP monitoring and above and below site testing in combination with a large land treatment effort. These projects were able to document substantial water quality improvements. In the Utah RCWP project, animal ***waste*** management systems reduced phosphorus concentrations leaving the watershed by 75 percent and reduced nitrogen and fecal coliform by 40 to 90 percent. In the Florida RCWP project, fencing, water management, and animal waste management systems reduced phosphorus concentrations in water entering Lake Okeechobee by 45 percent.

In the Oregon RCWP project, animal waste management systems installed on dairies reduced bacterial contamination of oyster beds by about 40 to 50 percent. Sites in the bay restricted to shellfishing based on Food and Drug Administration classification decreased from 12 in 1979-80 to 1 in 1985-86. In the Idaho RCWP project, water management and sediment control BMPs reduced sediment loads in return flows from irrigated land by 70 percent. Trout fishing has been partially restored to this coldwater trout stream.

Likewise, the Idaho and Nebraska RCWP projects realize that a substantial effort would have been saved if they had established clear protocols in the beginning for documenting water quality and land treatment on a subbasin and annual basis such that the two databases could be linked hydrologically and temporally. Both projects have taken the initiative to reconstruct and link the two databases. For these projects, the land treatment databases were the most difficult to reconstruct.

The Vermont project used extensive monitoring of BMP implementation and agricultural activities to establish a link between cows under BMP manure management and bacteria levels in streams. The Minnesota RCWP project used vadose zone monitoring to establish the relationship between agricultural practices, best management practices, and ecological niches to groundwater contamination.

Explanatory variables, which should be monitored in the high-level projects, can include other land-use changes, the seasons, stream discharge, precipitation, groundwater table depth, impervious land surface area, and others. In Alabama, technicians were unable to determine the cause of a sudden increase in fecal coliform levels in a particular stream until they determined that beavers had built a dam upstream of the sampling site. The Florida RCWP project confirmed that the changes in cow numbers and water table depth affected the water quality monitoring results and that documentation and adjustment for these changes allowed valid conclusions to be made regarding changes in water quality.

Evaluation and Reporting

Regular review of progress helps ensure that the project is working toward its goals and that its activities are on track. As part of the evaluation process, regular meetings must be held by the local coordinating committee to keep the project team informed and to coordinate activities. In addition, quarterly meetings of technical groups can help guide the project to water quality improvements. Both the South Dakota and the Vermont RCWP projects used frequent meetings of technical staff to identify needs and document progress.

Annual progress reports on the projects create an opportunity to compile and analyze findings; annual progress reviews by the national coordinating committee and the national technical support group can help projects meet their goals.

Feedback Loop

Regular meetings are a must for project staff, if water quality and land treatment monitoring are to be used to make mid-project adjustments. The project manager can facilitate communication by scheduling local coordinating committee meetings on a quarterly basis. The State and local coordinating committees should meet jointly at least once each year.

Regional workshops should be scheduled to provide information transfers between projects with similar hydrology and agriculture. National workshops are helpful and especially beneficial if all projects are represented. Some of the most important RCWP lessons were learned from projects that were seldom represented at national RCWP workshops

In the early part of the Virginia RCWP project, extremely high levels of coordination and cooperation existed among the different agencies, and communication was excellent. However, after BMP implementation, which occurred about five years into the project, both the State and local coordinating committees stopped meeting, which caused a breakdown in communication between the land treatment and water quality groups.

Conclusion

The Rural Clean Water Program has demonstrated that nonpoint source pollution control programs can be successful in protecting and restoring water resources if they are carefully structured and based on the findings of previous programs. The model program we propose requires administrative and technical support from all levels — Federal, State, and local. The States and their local counterparts need guidance on project implementation. Much of this guidance can best be communicated through program regulations similar to the regulations written for the RCWP *(Federal Register,* 1980a). A national technical support group, independent of designated cooperating agencies, should be in place to help develop program guidance, provide technical assistance, and conduct project evaluations.

Water quality monitoring is required to document the problem and track project effectiveness. We suggest minimum monitoring requirements to guide the development of the monitoring program design. BMP systems must be targeted to treat critical areas and specific pollutants responsible for the present or potential problem. Finally, a project manager and a core project staff (from various coordinating agencies) are needed to implement the project. Greater accountability among project staff and incentives to avoid turnover wjll improve the likelihood of meeting project goals. Information and educational efforts should be expanded to encourage greater adoption of BMPs.

Continual evaluation of programs and projects and full communication of technical information are key factors in controlling nonpoint source pollution and achieving water quality goals.

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APPENDIX 3.1B

Planning and Managing a Successful Nonpoint Source Pollution Control Project

by: Judith A. Gale, Deanna L. Osmond, Daniel E. Line, Jean Spooner, Jon A. Arnold, Gregory D. Jennings, and Frank J. Humenik

Planning and Managing a Successful Nonpoint Source Pollution Control Project

The Rural Clean Water Program Experience

Significant progress has been made in reducing water pollution caused by point sources since the Clean Water Act was passed. However, much work remains to be done to reduce nonpoint source (NPS) pollutants that impair the quality of streams, rivers, lakes, ground water, and other bodies of water throughout the United States.

Many local government officials, as well as citizens, are becoming increasingly interested in taking action to address local water quality problems caused primarily by nonpoint source pollutants. There is also a heightened awareness that water quality problems do not occur in isolation; many activities within a watershed affect the quality of water resources. Surface and ground waters are frequently connected, so management strategies aimed at protecting water quality must often be designed to address the impacts of human activities on a watershed basis for both surface water and ground water.

This fact sheet is designed to provide information to local and state government officials and staff, concerned citizens, educational and technical assistance agencies, landowners, and farmers interested in protecting or restoring water quality. Specific steps are outlined for:

- Deciding whether a water quality project is viable, based upon available information,
- Documenting the water quality problem and its source,
- Defining specific project objectives and goals,
- Involving potential participants and other community members in planning and implementing the project,
- Securing funding,
- Clarifying agency roles and organizing a project,
- Defining the critical area,
- Choosing a land treatment approach, and
- Designing a monitoring and evaluation plan.

Designing a Successful Voluntary Nonpoint Source Pollution Control Project

Choose a Viable Project

The first step in planning a successful nonpoint source pollution control project is to identify a water resource with water quality needing restoration or protection. Focus on a water resource that is valued by the community and a problem that is neither too complex nor too difficult to solve in a reasonable amount of time. Talk to or formally survey community members who live and work in the vicinity of the water resource. Find out whether they believe that there is a water quality problem and if it is of concern to them. For example, find out if the water quality problem impairs recreational uses, such as fishing, swimming, or boating, or aesthetic enjoyment of the water resource.

If the source of the water quality problem is not clear, or if the source is one that cannot be affected by changes in project participants' behavior (for example, if the source is a point source versus agricultural runoff), there may be dissension within the community about the cause of the problem, how best to resolve it, or the value of a NPS pollution control project. Documentation of the problem and its source can help a community come together to support a project designed to address a water quality problem (see next section). If, however, consensus about the existence of a problem cannot be reached, or agencies cannot work effectively together, a project is unlikely to be successful. In such cases, limited resources for addressing water quality problems may be better spent on a different project or program.

If project funds are restricted to one source of nonpoint source pollutants, such as agricultural sources, avoid choosing a watershed that contains major point sources or other nonpoint sources. Pollutants from point sources can mask improvements in water quality brought about by implementation of best management practices (BMPs) aimed at reducing NPS pollution, thus making it difficult to document the benefits of a nonpoint source pollution control project. Other approaches designed to reduce both point and nonpoint source pollutants, such as total watershed management, can be very effective if adequate technical and financial resources are available.

Select a watershed of a size that matches the level of available funding for the project; if funds for installing BMPs are limited, treating most or all of a small watershed (or a subwatershed within a large watershed) will likely result in greater water quality improvements than treating a small land area in a large watershed.

Document the Water Quality Problem

Clearly document the water quality impairment or threat, and the source(s) of the problem. For example, a popular swimming beach at the community lake may have algal blooms (rapid growth of algae) at certain times of year. The results are color changes, odor, and fish kills, which impair swimming and other uses of the lake for recreation. To plan an effective approach to this problem, the specific pollutant(s) causing the blooms must be identified and the source(s) determined. Are nutrients causing the problem? If so, is there too much nitrogen or phosphorus? After identifying the pollutant, find out where it is coming from. Possible sources of nutrients include runoff from animal operations, over-application of fertilizer, septic tank drain fields, sediments in the lake bottom, or discharges from a treatment plant or industry. The source(s) of the water quality problem must be identified before action is taken, so available resources can be targeted to the critical area. Trying to address a problem without knowing the source can result in wasting limited funds and human resources and losing support for future projects.

Existing water quality and other relevant data, such as soils, geology, land use, and weather (and assistance in interpreting such data), should be requested from appropriate agencies, such as the state water quality agency; U.S. Geological Survey; local health department; county planning department; and U.S. Department of Agriculture (USDA) - Natural Resource Conservation

Service, USDA - Consolidated Farm Services Agency, USDA - Extension Service, National Oceanic and Atmospheric Administration, and Soil and Water Conservation District.

If adequate information about the problem and its source(s) has not already been collected, seek technical and financial assistance in designing a water quality monitoring program. Relevant state and federal programs are discussed in the section entitled Obtain Funding.

An effective approach to identifying the exact nature of the problem and its source(s) is to implement a problem identification and assessment monitoring program lasting from six to 18 months. Monitor sites suspected of contributing pollutants or stressors during both baseflow and storm conditions, especially during the seasons when the highest amount of the pollutant enters the water and during the season when water quality problems have been noticed. For example, in winter and spring there is often a great deal of runoff which carries nutrients, sediment, and other pollutants. A walk through the watershed may help identify problem areas with regard to habitat. Creel surveys can identify fishery problems.

Before initiating a project, write a problem statement that: 1) states what the impaired water use is, 2) identifies the location of the problem, 3) specifies the pollutant(s) or stressor(s), and 4) identifies the major or suspected source(s). A written problem statement documents the problem for future reference and clearly conveys the problem and source to participants and community members, thereby contributing to consensus about the problem and the approach being taken to resolve it.

Define Objectives and Goals

Well-defined objectives and goals clearly convey the purpose of the project to potential participants and the public. Objectives and goals also provide a basis for evaluating the project.

Objectives define the overall direction or purpose of the project. Establish objectives that focus the project on achieving water quality changes or meeting water quality standards. Be sure that objectives are measurable and achievable. For example, a workable objective might be "re-opening shellfish beds in Green Creek estuary by 1998."

Goals provide milestones to be met during the course of a project. Establish quantitative goals that provide a way to measure progress. For example, progress toward the goal "reduce the phosphorus load to Blue Reservoir by 45%" can be measured, while achievement of the goal "reduce pollution in the reservoir" is more difficult to evaluate. Set specific goals early with assistance from local agencies, project participants, and community representatives.

Objectives and goals must be tailored to available resources and to the nature of the problem. For example, expecting to reduce eutrophication in a reservoir when the project watershed supplies only 10% of the phosphorus load is unrealistic, as is a goal of reducing nutrient loss from a 500,00-acre watershed with 1,200 producers when resources consist of a $50,000 budget and two staff members.

Involve the Community

Public support and a high rate of participation are key in voluntary nonpoint source projects because of the widespread nature of NPS pollution. The following actions can increase participation:

- Educate potential participants and the community. They need to agree that there is a water quality problem, that it is important to solve it, and that the project will help do so.
- Encourage potential participants to accept responsibility for their contribution to the problem. On-going education about land use impacts on water quality is important as awareness does not necessarily translate into problem ownership or changes in behavior.
- Involve potential participants early in the planning process; involvement fosters a feeling of ownership which often increases participation.
- Find out if federal, state, local, or private funds are available. Financial assistance, such as cost-share funding, is necessary to enable many potential participants to implement BMPs.
- Recommend the lowest cost BMPs that can effectively reduce the pollutant(s) of concern.
- One-to-one contact between project personnel and potential participants is much more effective than mass media for gaining cooperation in a project. Because of their importance in encouraging participation, information and education efforts should be initiated early.
- Provide technical assistance valued by participants, such as soil testing and assistance in designing site-specific affordable BMPs.
- Ask participants to talk with their neighbors about the project and why they decided to become involved.
- Where relevant, notify potential participants that regulations may be instituted if voluntary measures do not improve water quality. This knowledge can provide an incentive for participation.

Obtain Funding

Obtain funds to support each aspect of the project. Cost-share funds that can be used to assist participants in installing BMPs are often critical to the success or failure of a voluntary nonpoint source project. Funding for pre-, during-, and post-implementation water quality monitoring and educational activities is also important.

State cost-share funds may be available to support implementation of agricultural or forestry BMPs for nonpoint source pollution control. Federal programs offering cost-share funds for forestry or agricultural BMPs may be available through the USDA - Consolidated Farm Services Agency. Section 319 funds allocated to each state by the U.S. Environmental Protection Agency (EPA) may be available from a state's water quality agency (nonpoint source program) to support nonpoint source pollution control projects.

Several EPA publications provide information on federal programs for watershed protection (EPA, 1993) and how state and local governments have funded nonpoint source pollution control programs (EPA, 1992).

Clarify Agency Roles and Administer the Project Effectively

Cooperation and coordination among local, state, and federal agencies are essential. Potential participants within the project area must receive clear messages about the project, its purpose, and its value. Conflicting messages from local, state, or federal agencies participating in a project can result in a low rate of participation. Clearly define each agency's role and how agencies will interact to avoid confusion, duplication of efforts, or competition. Urge agency administrators to support the project and encourage inter-agency cooperation. If key agencies cannot agree on the value of a proposed project, or if turf battles seem unresolvable, consider an alternative project choice.

Designate a project manager to coordinate the project and assess progress. Ideally, the project manager should have a background in water resources and project management.

Establish a local coordinating committee, consisting of project participants, agency personnel, and community leaders, to support the project. The committee should set direction, set objectives and goals, assure adequate public involvement, enlist agency assistance, oversee information and education activities, determine priorities for water quality monitoring, and develop plans for critical area selection, choice of BMP systems, and linkage of land treatment and water quality data.

Define the Critical Area

Apply BMP systems to those areas where land treatment will have the greatest effect. Where available, pre-project water quality monitoring and modeling can be used to identify or refine the critical area -- the land area contributing most to the problem. In the absence of such resources, critical areas can be roughly defined based on distance to the water body and its tributaries, or other location or land use characteristics. Within the critical area, significant pollutant sources (such as animal operations, farm fields, or forestry operations) can be prioritized for BMP installation based on the expected impact of each source on the water body.

Choose a Land Treatment Approach

Encourage participants to implement systems of BMPs. Systems of practices often control loss of a pollutant from the critical area more effectively than a single BMP. Resources for assistance in identifying systems to effectively address a particular water quality problem and source include Extension Service, Natural Resource Conservation Service, and Soil and Water Conservation Districts staff.

Design a Water Quality and Land Treatment Monitoring and Evaluation Plan

Water quality and land treatment monitoring and evaluation provide essential tools for assessing project effectiveness. Team members who will conduct and interpret the monitoring effort must be involved from the beginning of the project, not added as an afterthought.

When limited resources are available for monitoring BMP effectiveness, visual observations such as fewer algal blooms, clearer water, or increased recreational use can be helpful in assessing the effectiveness of the project. Monthly monitoring of a few key factors (such as dissolved oxygen or chlorophyll a) can provide useful information.

When funds are available for more extensive water quality monitoring, essential tasks and elements include:

- Developing a monitoring plan based on clearly stated water quality monitoring objectives. Include in the plan: monitoring design, agency roles, laboratory and quality assurance and control procedures, data storage plans, reporting requirements, personnel needs, and costs.
- Collecting sufficient pre-, during -, and post-project data to document water quality changes. In large watersheds with lakes, water quality changes often occur gradually and monitoring for five to 10 years, or longer, may be required to confirm changes that can be linked to land treatment.

Assessing Project Effectiveness

Evaluate data with project objectives and goals clearly in mind. A consistent improving trend in water quality after BMP system implementation may provide evidence needed to attribute water quality improvements to land treatment.

Consider interviewing (pre- and post-project) participants and people who were eligible but chose not to participate in the project to assess the effectiveness of education efforts.

Report successes and failures periodically to provide feedback to project participants and agency staff on the results of their efforts. Make results available to the community to enhance public education and contribute to more effective management of water quality problems in the future.

Keys to Success

Choose a Viable Project

- Choose a water resource that needs restoration or protection and is valued by community members.

Document the Problem

- Document the water quality problem and its source.

Define Objectives and Goals

- Define obtainable objectives and goals.

Involve the Community

- Involve potential participants and the community early in project planning.

Obtain Funding

- Obtain funding for all project aspects.

Clarify Roles and Administer Effectively

- Clarify agency roles.
- Designate a project manager.
- Form a local coordinating committee.

Define the Critical Area

- Define the critical area where treatment will have the most impact.

Choose a Land Treatment Approach

- Apply BMPs that will address the water quality problem.
- Encourage participants to implement systems of BMPs.

Monitor and Evaluate

- Design a water quality and landtreatment monitoring and evaluation program, when possible, to document the effects of BMPs installed.

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APPENDIX 3.1C

Farmer Participation in Solving the Nonpoint Source Pollution Problem

by: Deanna L. Osmond and Judith A. Gale

Farmer Participation in Solving the Nonpoint Source Pollution Problem

The Rural Clean Water Program Experience

The Importance of Producer Participation in Voluntary Agricultural Nonpoint Source Pollution Control Projects

The success or failure of any agricultural nonpoint source pollution control project depends on the participation of many landowners or farm operators. These producers must install or utilize land-based treatments, or best management practices (BMPs), that minimize the movement of agricultural pollutants such as sediment, nutrients, and pesticides to water resources.

The degree of producer participation necessary to protect or remediate water quality will depend not only on the total number of land users employing BMPs in the watershed, but also on several other factors: the location of the producers' farms in the watershed, the types of BMPs selected, the extent of BMP implementation, and the type and severity of the water quality problem.

The first phase in a nonpoint source (NPS) pollution control project is to accurately identify and clearly document the water quality problem, the specific pollutant(s), and the sources of the pollutant(s). Based on the water quality problem assessment, the critical area (land area or areas contributing disproportionately to the water quality problem) should be identified. High-priority project participants are those producers who farm or raise livestock in the critical area of the watershed.

A primary goal of any voluntary NPS pollution control project is to engage a sufficient number of potential participants in the project. The Rural Clean Water Program (RCWP), a nationally recognized nonpoint source pollution control program conducted between 1981 and 1995, established a target voluntary producer participation rate of 75%. Many valuable lessons were learned from the RCWP about how to recruit and retain participants in voluntary NPS pollution control projects. The information presented in this fact sheet is based on these lessons learned.

Farm Structure and Producer Attitudes and Attributes that Affect Project Outcome

An extensive telephone survey of producers farming in the critical areas of the 21 RCWP projects was conducted to evaluate differences between farmers who chose to participate in the RCWP and those who did not (Gale et al., 1993). Farm structure, farm operator characteristics, and water quality awareness and attitudes were assessed.

Participation in RCWP projects was highly correlated with strong economic indicators, such as comparatively larger total acreage farmed, higher gross farm sales, and greater property and farm equipment values. Producers who were employed off-farm, or who received only part of their income from agriculture, were less likely to participate in NPS pollution control projects than were farmers who worked solely on the farm and earned most of their income from agriculture.

Water quality awareness and attitudes were also important in determining participation rates in the RCWP projects. Producers who were more aware of water pollution (in general, in the specific area, or on individual farms) participated in greater numbers than farmers who were less well informed. Producers who received most of their water quality and conservation information from government agencies and farm magazines were more likely to change agricultural practices that affected water quality than producers who did not receive information from these sources.

Many of the results of the farm operator survey were similar to conclusions of previous studies evaluating factors that influence conservation. Farmers who run large-scale operations, are better educated and more willing to take risks, and have access to government information generally participate at a higher rate in conservation programs than producers without these characteristics. Although farm structure and producer characteristics were important factors in determining which farmers chose to participate in the RCWP projects, external incentives also affected participation.

Incentives To Producer Participation

Economic Factors

Financial incentives are extremely important, and may be the most important factor, in obtaining voluntary implementation of BMPs. Financial incentives for voluntary environmental compliance include cost-share funds, tax relief, payment transfers, and government subsidies.

The primary financial incentive in the RCWP projects was federal cost-share funding. Each producer could receive up to 75% of the cost of each recommended BMP implemented (up to a maximum per farm of $50,000).

The cost-share rate for the Alabama RCWP project was originally set at 60%. Few farmers chose to participate until the cost-share rate was raised to 75%. Participation then increased to 100% of the producers in the critical area.

A significant barrier to implementation of BMPs is poor economic status of producers. The farm operator survey (Gale et al., 1993) found a lower rate of participation among farmers who had relatively lower economic indicators. During the early 1980s, many farmers in Oregon were unable to participate in the Tillamook Bay RCWP project because high interest rates limited cash flow, making it difficult for farmers to pay their portion of the cost of installing BMPs. Another hindrance is the high cost of some BMPs, such as animal waste management systems. For many dairy farmers, the maximum cost-share payment of $50,000 was insufficient to make the construction of animal waste storage units economically feasible.

State or local cost-share assistance was offered in some projects as a supplement to federal cost-share funds. To entice absentee landlords to participate in the RCWP, Tennessee and Kentucky officials added 25% to the federal 75% cost-share rate for seeding alfalfa. Producers also received an additional one-time payment of $75 per acre for converting cropland to pasture. Florida dairy farmers participating in the Lake Okeechobee RCWP project received substantial subsidies from the State of Florida to assist them in installing expensive animal waste management BMP systems.

Technology Transfer: The Importance of Information and Education Programs

Information and education (I&E) is an essential component of any agricultural NPS pollution control project. Information should heighten farmers' awareness of water quality problems and approaches to solving them. Education should increase project participation and assist farmers in selecting and maintaining appropriate BMP systems.

Strong and effective I&E programs in many of the RCWP projects (for example, Maryland, Alabama, Nebraska, Idaho, Utah, Vermont, Florida, and Oregon) contributed to high producer participation and, consequently, to water quality improvements.

I&E must begin prior to land-based project activities in order to foster a sense of problem and project ownership on the part of the potential project participants. Delaware and Iowa RCWP project personnel reported that both pre-project meetings to discuss the water quality problem and producer involvement in project planning helped develop strong support for and participation in the project by area farmers.

The most effective way to increase producer participation is one-to-one contact between project personnel and farmers.

On-farm demonstrations can be used effectively to educate farmers about new technologies. Producer participation was increased in the Maryland RCWP project through on-farm demonstrations of BMP installation and maintenance.

To control agricultural runoff, producers must implement additional, often new, BMPs. Technical assistance must help participants with new BMPs, whether the BMPs are structural or managerial. In the Oregon RCWP project, Natural Resource Conservation Service personnel had to modify animal waste storage systems for high-rainfall conditions. Extension Service personnel in Pennsylvania developed nutrient management plans for individual farmers and taught them how to implement the plans. These technical assistance efforts resulted in more effective implementation and maintenance of BMPs. Technical assistance also served to strengthen producers' motivation to participate in the project.

Environmental Concerns

Like air pollution, water pollution from nonpoint sources is a complex issue. It is often difficult for land users to understand how an individual's daily activities can contribute to nonpoint source pollution. Producers are most likely to participate in solving water quality problems when they understand that their own agricultural practices affect the water quality of a local water resource. The farm operator survey showed that the major reason producers did not participate in the RCWP projects was that they did not believe water pollution was a problem. Conversely, twice as many RCWP participants as non-participants stated that they believed water quality was a problem.

Producer participation also depends on farmers valuing the impaired water resource. Because Iowa RCWP project participants valued a recreational lake that was decreasing in size and depth due to sedimentation caused by cropland erosion, they were willing to adopt new agricultural practices.

Environmental regulations, or the threat of regulation, can provide incentives for producers to participate in agricultural NPS pollution control projects. Farmers in the Chesapeake Bay drainage area face possible regulation if voluntary efforts fail to address the NPS pollution problem. As a result, over 50% of the farmers eligible to participate in the Virginia RCWP project were ready to get involved in the project as soon as cost-share funding became available.

Community Support

An impaired or threatened water resource affects the entire community. Nonpoint source pollution control projects must have the support of the whole community. In Oregon, community support of the Tillamook Bay RCWP project was instrumental in achieving 96% participation of critical area dairy farmers. Pressure to participate in the project came from neighbors and a local business. Fecal coliform contamination of the bay, caused by runoff from dairies, threatened the local economy by reducing shellfish harvests. Many of the fishermen losing revenue were relatives and friends of local dairy farmers. These fishermen were able to exert peer pressure on dairy farmers to change their farming practices. In addition, all of the dairy farmers sold their milk to a local cheese-producing cooperative that reserved the right to discount milk prices paid to producers who did not install BMPs. This high level of community support played an important role in the achievement of a very high rate of project participation.

Conclusions

Water quality changes require implementation of BMPs by a large percentage of producers who farm in the critical area of a watershed. However, a high rate of participation does not automatically ensure water quality improvements. Improvements in a degraded water resource, or protection of a threatened water resource, occur as the result of the interaction of many factors: identification of a water quality problem amenable to remediation, documentation of the source of the major pollutant(s), accurate definition of the critical area, correct selection and placement of BMPs, installation of a sufficient number of BMPs in a substantial portion of the critical area, and maintenance of BMPs.

The absolute number of participants necessary to reduce pollutants by a stated amount will vary depending on the pollutant, agro-environmental conditions, and the magnitude of the problem. For some situations, almost 100% producer participation may be required to improve the water resource to its designated use. In the Oregon RCWP project, approximately 60 dairies were considered critical at the start of the project. Dairies having the greatest negative impact on water received cost-share funds to implement BMPs first; then other critical farms were added. However, the project goal of a 70% reduction in fecal coliform counts was not being met. Consequently, additional dairies were classified as critical. By the end of the project, BMPs to control dairy runoff had been implemented on 96% of 109 dairies defined as critical and the project's water quality goals were met. The experience of the Oregon, Florida, and Utah RCWP projects indicates that close to 100% participation is necessary in projects where the major source of the pollutants is animal operations.

Other RCWP projects successfully reduced pollutants with lower participation rates. In Idaho, installation of BMP systems on 75% of the critical area farms resulted in a 75% decrease in sediment and a 68% decrease in phosphorus entering Rock Creek, resulting in better habitat for fish.

While the amount of voluntary participation necessary to successfully address agricultural NPS pollution must be determined for each individual watershed, results from the RCWP suggest that an absolute minimum of 75% participation of critical area farmers is necessary.

Many factors interact to determine the ultimate number of producers who participate in a voluntary NPS pollution control project. Financial incentives are extremely helpful in reducing the economic burden of BMP implementation. Environmental regulations, or the threat of regulations, can also increase participation, although they are most often used as a last resort when voluntary measures have failed. Technical assistance is an important means for helping producers select, install, and maintain appropriate BMP systems. I&E is also an important means for achieving adequate participation and helping potential participants understand how their practices may degrade valuable local water resources. Finally, community support is essential for encouraging and sustaining producers throughout the project period.

Key Points of Farmer Participation

Socio-Economic and Attitudinal Factors Affecting Participation

- Farmers who work solely on the farm or who receive most of their income from agricultural sales are most likely to participate in agricultural nonpoint source (NPS) pollution control projects.
- Project participants are generally more aware of water pollution than farmers who choose not to participate.
- Producers who receive most of their water quality and conservation information from government agencies and farm magazines are most likely to change agricultural practices that affect water quality.

Incentives to Participation

- Financial incentives may be the most important factor in achievement of voluntary implementation of BMPs.
- Financial incentives include cost-share funds, tax relief, payment transfers, and government subsidies.

Economic Factors

- The cost of BMP installation and maintenance serves as a disincentive to BMP implementation.

The Importance of Information and Education Programs

- Information and education programs increase producer participation in agricultural NPS pollution control projects.
- Information heightens farmers' awareness of water quality problems and approaches to solving them.
- Education aids farmers in selecting appropriate BMP systems.
- I&E programs must begin prior to land-based project activities to facilitate development of a sense of problem and project ownership on the part of the potential participants.
- One-to-one contact between producers and I&E specialists is the most effective method to transfer information and increase participation.
- New technologies can be effectively shared with producers through on-farm demonstrations.
- Technical assistance results in more effective BMP implementation and maintenance and better participation in NPS pollution control projects.

Enviromental Concerns

- Producers are most likely to participate in NPS pollution control efforts when they understand that their agricultural practices affect the water quality of a valued local water resource.
- Environmental regulations, or the threat of regulation, can motivate participation by producers in a NPS control project.

Community Support

- The support of the entire community is required for NPS pollution control project to be successful.
- Community members can apply pressure to local farmers to adopt better agricultural practices.

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COLLEGE OF AGRICULTURAL & LIFE SCIENCES

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APPENDIX 4.2A
BEST MANAGEMENT PRACTICES - DEFINITIONS AND DESCRIPTIONS

Appendix

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Appendix A: Best Management Practices — Definitions and Descriptions

Best management practices mentioned in this guidance are listed in alphabetical order below. The NRCS or other code number, if any, is given for each BMP, followed by a short definition. Additional explanatory text about selected BMPs is presented in italicized text below the practice, code, and definition.

Access Road (560): A travelway constructed as part of a conservation plan.

Animal Trails and Walkways (575): A livestock trail or walkway constructed to improve grazing distribution and access to forage and water.

Bedding (310): Plowing, blading, or otherwise elevating the surface of flat land into a series of broad, low ridges separated by shallow, parallel channels

Brush (and Weed) Management (314): Managing and manipulating stands of brush (and weeds) on range, pasture, and recreation and wildlife areas by mechanical, chemical, or biological means or by prescribed burning. (Includes reducing excess brush (and weeds) to restore natural plant community balance and manipulating stands of undesirable plants through selective and patterned treatments to meet specific needs of the land and objectives of the land user.)

Improved vegetation quality and the decrease in runofffrom the practice will reduce the amount of erosion and sediment yield. Improved vegetative cover acts as a filter strip to trap the movement of dissolved and sediment attached substances, such as nutrients and chemicals from entering downstream water courses. Mechanical brush management may initially increase sediment yields because of soil disturbances and reduced vegetative cover. This is temporary until revegetation occurs.

Channel Vegetation (322): Establishing and maintaining adequate plants on channel banks, berms, spoil, and associated areas.

Chiseling and Subsoiling (324): Loosening the soil, without inverting and with a minimum of mixing of the surface soil, to shatter restrictive layers below normal plow depth that inhibit water movement or root development.

Cornposting Facility (317): A facility for the biological stabilization of waste organic material.

The purpose is to treat waste organic material biologically by producing a humus-like material that can be recycled as a soil amendment and fertilizer substitute or otherwise utilized in compliance with all laws, rules. and regulations.

Conservation Cover (327): Establishing and maintaining perennial vegetative cover to protect soil and water resources on land retired from agricultural production.

Agricultural chemicals are usually not applied to this cover in large quantities and surface and ground water quality may improve where these material are not used. Ground cover and crop residue will be increased with this practice. Erosion and yields of sediment and sediment related stream pollutants should decrease. Temperatures of the soil surface runoff and receiving water may be reduced. Effects will vary during the establishment period and include increases in runoff, erosion and sediment yield. Due to the reduction of deep percolation, the leaching of soluble material will be reduced, as will be the potential for causing saline seeps. Long-term effects of the practice would reduce agricultural nonpoint sources of pollution to all water resources.

Conservation Cropping Sequence (328): An adapted sequence of crops designed to provide adequate organic residue for maintenance or improvement of soil tilth.

This practice reduces erosion by increasing organic matter; resulting in a reduction of sediment and associated pollutants to surface waters. Crop rotations that improve soil tilth may also disrupt disease, insect and weed reproduction cycles, reducing the need for pesticides. This removes or reduces the availability of some pollutants in the watershed. Deep percolation may carry soluble nutrients and pesticides to the ground water. Underlying soil layers, rock and unconsolidated parent material may block, delay, or enhance the delivery of these pollutants to ground water. The fate of these pollutants will be site specific, depending on the crop management, the soil and geologic conditions.

Conservation Tillage (329) (NoTill): Any tillage and planting system in which at least 30 percent of the soil surface is covered by plant residue after planting to reduce soil erosion by water; or, where soil erosion by wind is the primary concern, at least 1,000 pounds per acre of flat small grain residue-equivalent are on the surface during the critical erosion period.

This practice reduces soil erosion, detachment and sediment transport by providing soil cover during critical times in the cropping cycle. Surface residues reduce soil compaction from raindrops, preventing soil sealing and increasing infiltration. This action may increase the leaching of agricultural chemicals into the ground water.

In order to maintain the crop residue on the surface it is difficult to incorporate fertilizers and pesticides. This may increase the amount of these chemicals in the runoff and cause more surface water pollution

The additional organic material on the surface may increase the bacterial action on and near the soil surface. This may tie-up and then breakdown many pesticides which are surface applied, resulting in less pesticide leaving the field. This practice is more effective in humid regions.

With a no-till operation the only soil disturbance is the planter shoe and the compaction from the wheels. The surface applied fertilizers and chemicals are not incorporated and often are not in direct contact with the soil surface. This condition may result in a high surface runoff of pollutants (nutrient and pesticides). Macropores develop under a no-till system. They permit deep percolation and the transmittal of pollutants, both soluble and insoluble to be carried into the deeper soil horizons and into the ground water.

Reduced tillage systems disrupt or break down the macropores, incidentally incorporate some of the materials applied to the soil surface, and reduce the effects of wheeltrack compaction. The results are less runoff and less pollutants in the runoff.

Constructed **Wetland (ASCS-999):** A constructed aquatic ecosystem with rooted emergent hydrophytes designed and managed to treat agricultural wastewater.

This is a conservation practice for which NRCS has developed technical requirements under a trial program leading to the development of a conservation practice standard.

Contour Farming (330): Farming sloping land in such a way that preparing land, planting, and cultivating are done on the contour. This includes following established grades of terraces or diversions.

This practice reduces erosion and sediment production. Less sediment and related pollutants may be transported to the receiving waters.

Increased infiltration may increase the transportation potential for soluble substances to the ground water.

Contour Orchard and Other Fruit Area (331): Planting orchards, vineyards, or small fruits so that all cultural operations are done on the contour.

Contour orchards and fruit areas may reduce erosion, sediment yield, and pesticide concentration in the water lost. Where inward sloping benches are used, the sediment and chemicals will be trapped against the slope. With annual events, the bench may provide 100 percent trap efficiency. Outward sloping benches may allow greater sediment and chemical loss.

The amount of retention depends on the slope of the bench and the amount of cover. In addition, outward sloping benches are subject to erosion form runoff from benches immediately above them. Contouring allows better access to rills, permitting maintenance that reduces additional erosion. Immediately after establishment, contour orchards may be subject to erosion and sedimentation in excess of the now contoured orchard. Contour orchards require more fertilization and pesticide application than did the native grasses that frequently covered the slopes before orchards were started. Sediment leaving the site may carry more adsorbed nutrients and pesticides than did the sediment before the benches were established from uncultivated slopes. If contoured orchards replace other crop or intensive land use, the increase or decrease in chemical transport from the site may be determined by examining the types and amounts of chemicals used on the prior land use as compared to the contour orchard condition.

Soluble pesticides and nutrients may be delivered to and possibly through the root zone in an amount proportional to the amount of soluble pesticides applied, the increase in infiltration, the chemist: of the pesticides, organic and clay con-

tent of the soil, and amounts of surface residues. Percolating water below the root zone may carry excess solutes or may dissolve potential pollutants as they move. In either case, these solutes could reach ground water supplies and/or surface downslope from the contour orchard area. The amount depends on soil type, surface water quality, and the availability of soluble material (natural or applied).

Contour Stripcropping (585): Growing crops in a systematic arrangement of strips or bands on the contour to reduce water erosion.

The crops are arranged so that a strip of grass or close-growing crop is alternated with a strip of clean-tilled crop or fallow or a strip of grass is alternated with a close-growing crop [FIGURE 2-4].

This practice may reduce erosion and the amount of sediment and related substances delivered to the surface waters. The practice may increase the amount of water which infiltrates into the root zone, and, at the time there is an overabundance of soil water, this water may percolate and leach soluble substances into the ground water:

Controlled Drainage (335): Control of surface and subsurface water through use of drainage facilities and water control structures.

The purpose is to conserve water and maintain optimum soil moisture to (1) store and manage infiltrated rainfall for more efficient crop production; (2) improve surface water quality by increasing infiltration, thereby reducing runoff, which may carry sediment and undesirable chemicals; (3) reduce nitrates in the drainage water by enhancing conditions for denitrification; (4) reduce subsidence and wind erosion of organic soils; (5) hold water in channels in forest areas to act as ground fire breaks; and (6) provide water for wildlife and a resting and feeding place for waterfowl.

Cover and Green Manure Crop (340): A crop of close-growing grasses, legumes, or small grain grown primarily for seasonal protection and soil improvement. It usually is grown for 1 year or less, except where there is permanent cover as in orchards.

Erosion, sediment and adsorbed chemical yields could be decreased in conventional tillage systems because of the increased period of vegetal cover: Plants will take up available nitrogen and prevent its undesired movement. Organic nutrients may be added to the nutrient budget reducing the need to supply more soluble forms. Overall volume of chemical application may decrease because the vegetation will supply nutrients and there may be allelopathic effects of some of the types of cover vegetation on weeds. Temperatures of ground and surface waters could slightly decrease.

Critical Area Planting (342): Planting vegetation, such as trees, shrubs, vines, grasses, or legumes, on highly erodible or critically eroding areas. (Does not include tree planting mainly for wood products.)

This practice may reduce soil erosion and sediment delivery to surface waters. Plants may take up more of the nutrients in the soil, reducing the amount that can be washed into surface waters or leached into ground water.

During grading, seedbed preparation, seeding, and mulching, large quantities of sediment and associated chemicals may be washed into surface waters prior to plant establishment.

Crop Residue Use (344): Using plant residues to protect cultivated fields during critical erosion periods.

When this practice is employed, raindrops are intercepted by the residue reducing detachment. soil dispersion, and soil compaction. Erosion may be reduced and the delivery of sediment and associated pollutants to surface water may be reduced. Reduced soil sealing, crusting and compaction allows more water to infiltrate, resulting in an increased potential for leaching of dissolved pollutants into the ground water.

Crop residues on the surface increase the microbial and bacterial action on or near the surface. Nitrates and surface-applied pesticides may be tied-up and less available to be delivered to surface and ground water: Residues trap sediment and reduce the amount carried to surface water. Crop residues promote soil aggregation and improve soil tilth

Deferred Grazing (352): Postponing grazing or resting g-razing land for prescribed period.

In areas with bare ground or low percent ground cover, deferred grazing will reduce sediment yield because of increased ground cover; less ground surface disturbance, improved soil bulk density characteristics, and greater infiltration rates. Areas mechanically treated will have less sediment yield when deferred to encourage revegetation. Animal waste would not be available to the area during the time of deferred grazing and there would be less opportunity for adverse runoff effects on surface or aquifer water quality. As vegetative cover increases, the filtering processes are enhanced, thus trapping more silt and nutrients as well as snow if climatic conditions for snow exist. Increased plant cover results in a greater uptake and utilization of plant nutrients.

Dikes (356): An embankment constructed of earth or other suitable materials to protect land against overflow or to regulate water.

Where dikes are used to prevent water from flowing onto the floodplain, the pollution dispersion effect of the temporary wetlands and backwater are decreased. The sediment, sediment-attached, and soluble materials being transported by the water are carried farther downstream. The final fate of these materials must be investigated on site. Where dikes are used to retain runoff on the floodplain or in wetlands the pollution dispersion effects of these areas may be enhanced. Sediment and related materials may be deposited, and the quality of the water flowing into the stream from this area will be improved.

Dikes are used to prevent wetlands and to form wetlands. The formed areas may be fresh, brackish, or saltwater wetlands. In tidal areas dikes are used to stop saltwater intrusion, and to increase the hydraulic head of fresh water which will force intruded salt water out the aquifer: During construction there is a potentia! of heavy sediment loadings to the surface waters. When pesticides are used to control the brush on the dikes and fertilizers are used for the establishment and maintenance of vegetation there is the possibility for these materials to be washed into the surface waters.

Diversion (362): A channel constructed across the slope with a supporting ridge on the lower side.

This practice will assist in the stabilization of a watershed, resulting in the reduction of sheet and rill erosion by reducing the length of slope. Sediment may be reduced by the elimination of ephemeral and large gullies. This may reduce the amount of sediment and related pollutants delivered to the surface waters.

Fencing (382): Enclosing or dividing an area of land with a suitable permanent structure that acts as a barrier to livestock, big game, or people (does not include temporary fences).

Fencing is a practice that can be on the contour or up and down slope. Often a fence line has grass and some shrubs in it. When a fence is built across the slope it will slow down runoff, and cause deposition of coarser grained materials reducing the amount of sediment delivered downslope. Fencing may protect riparian areas which act as sediment traps and filters along water channels and impoundments.

Livestock have a tendency to walk along fences. The paths become bare channels which concentrate and accelerate runoff causing a greater amount of erosion within the path and where the path/channel outlets into another channel. This can deliver more sediment and associated pollutants to surface waters. Fencing can have the effect of concentrating livestock in small areas, causing a concentration of manure which may wash off into the stream, thus causing surface water pollution.

Fence (382A): [ADD DEFINITION]

Fence, Suspension (382B): [ADD DEFINITION]

Fence, Electrical (382C): [ADD DEFINITION]

Field Stripcropping (586): Growing crops in a systematic arrangement of strips or bands across the general slope (not on the contour) to reduce water erosion. The crops are arranged so that a strip of grass or a close-growing crop is alternated with a clean-tilled crop or fallow.

This practice may reduce erosion and the delivery of sediment and related substances to the surface waters. The practice may increase infiltration and, when there is sufficient water available, may increase the amount of leachable pollutants moved toward the ground water.

Since this practice is not on the contour there will be areas of concentrated flow, from which detached sediment, adsorbed chemicals and dissolved substances will be delivered more rapidly to the receiving waters. The sod strips will not be efficient filter areas in these areas of concentrated flow.

Field Border (386): A strip of perennial vegetation established at the edge of a field by planting or by converting it from trees to herbaceous vegetation or shrubs.

This practice reduces erosion by having perennial vegetation on an area of the field. Field borders serve as "anchoring points" for contour rows, terraces, diversions, and contour strip cropping. By elimination of the practice of tilling and planting the ends up and down slopes, erosion from concentrated **flow** *in furrows and long rows may be reduced. This use may reduce the quantity of sediment and related pollutants transported to the surface waters.*

Filter Strip (393): A strip or area of vegetation for removing sediment, organic matter, and other pollutants from runoff and wastewater

Filter strips for sediment and related pollutants meeting minimum requirements may trap the coarser grained sediment. They may not jilter out soluble or suspended fine-grained When a storm causes excess runoff in runoff, the filter may be ***flooded*** *and may cause large loads of pollutants to be released to the surface water. This type of filter requires high maintenance and has a relative short service life and is effective only as long as the flow through the filter is shallow sheet flow.*

Filter strips for runoff form concentrated livestock areas may trap organic material, solids, materials which become adsorbed to the vegetation or the soil within the filter. Often they will not filter out soluble materials. This type of filter is often wet and is difficult to maintain.

Filter strips for overland flow treatment of liquid wastes may effectively filter out pollutants. The filter
may...uding the proper resting time. Filter strips on forest land

runoff. effect may improve the quality of surface water and has little

All of filters may reduce erosion on the area on w
sheet **flow** *through the filter.*
and fibrous are filtered than fine- ciently
grained and soluble substances. Filter strips work for design conditions, but when flooded or overloaded they may release a slug load of pollutants into the surface water:

Floodwater Diversion (400): A graded channel with a supporting embankment or dike on the lower side constructed on lowland subject to flood damage.

Forest Land Erosion Control System (408): Application of one or more erosion control measures on forest land. Erosion control system includes the use of conservation plants, cultural practices, and erosion control structures on disturbed forest land for the control of sheet and rill erosion, gully formation, and mass soil movement.

Grade Stabilization Structure (410): A structure used to control the grade and head cutting in natural or artificial channels.

Where reduced stream velocities occur upstream and downstream from the structure, streambank and streambed erosion will be reduced. This will decrease the yield of sediment and sediment-attached substances. Structures that trap sediment will improve downstream water quality The sediment yield change will be a function of the sediment yield to the structure, reservoir trap efficiency and of *velocities of released water. Ground water recharge may affect aquifer quality depending on the quality of the recharging water. If the stored water contains only sediment and chemical with low water solubility, the ground water quality should not be affected.*

Grassed Waterway (412): A natural or constructed channel that is shaped or graded to required dimensions and established in suitable vegetation for the stable conveyance of runoff

This practice may reduce the erosion in a concentrated flow area, such as in a gully or in ephemeral gullies. This may result in the reduction of sediment and substances delivered to receiving waters. Vegetation may act as a filter in removing some of the sediment delivered to the waterway, although this is not the primary function of a grassed waterway.

Any chemicals applied to the waterway in the course of treatment of the adjacent cropland may wash directly into the surface waters in the case where there is a runoff event shortly after spraying.

When used as a stable outlet for another practice, waterways may increase the likelihood of dissolved and suspended pollutants being transported to surface waters when these pollutants are delivered to the waterway.

Grasses and Legumes in Rotation (411): Establishing grasses and legumes or a mixture of them and maintaining the stand for a definite number of years as part of a conservation cropping system.

Reduced runoff and increased vegetation may Lower erosion rates and subsequent yields of sediment and sediment-attached substances. Less applied nitrogen may be required to grow crops because grasses and Legumes will supply organic nitrogen. During the period of the rotation when the grasses and legumes are growing, they will take up more phosphorus. Less pesticides may similarly be required with this practice. Downstream water temperatures may be lower depending on the season when this practice is applied. There will be a greater opportunity for animal waste management on grasslands because manures and other wastes may be applied for a longer part of the crop year:

Grazing Land Mechanical Treatment (548): Renovating, contour furrowing, pitting, or chiseling native grazing land by mechanical means.

Heavy Use Area Protection (561): Protecting heavily used areas by establishing vegetative cover, by surfacing with suitable materials, or by installing needed structures.

Protection may result in a general improvement of surface water quality through the reduction of erosion and the resulting sedimentation. Some increase in erosion may occur during and immediately after construction until the disturbed areas are fully stabilized.

Some increase in chemicals in surface water may occur due to the introduction of fertilizers for vegetated areas and oils and chemicals associated with paved areas. Fertilizers and pesticides used during operation and maintenance may be a source of water pollution.

Paved areas installedfor livestock use will increase organic, bacteria, and nutrient loading to surface waters. Changes in ground water quality will be minor. Nitrate nitrogen applied as fertilizer in excess of vegetation needs may move with infiltrating waters. The extent of the problem, if any, may depend on the actual amount of water percolating below the root zone.

Hedgerow Planting (422): Establishing a living fence of shrubs or trees in, across, or around a field.

Hillside Ditch (423): A channel that has a supporting ridge on the lower side constructed across the slope at definite vertical intervals and gradient, with or without a vegetative barrier

Improved Water Application (197): [ADD DEFINITION]

Irrigation Canal or Lateral (320): A permanent irrigation canal or lateral constructed to convey water from the source of supply to one or more farms

Irrigation Field Ditch (388): A permanent irrigation ditch constructed to convey water from the source of supply to a field or fields in a farm distribution system.

The standard for this practice applies to open channels and elevated ditches of 25 ft^3/second or less capacity formed in and with earth materials.

Irrigationfield ditches typically carry irrigation rvaterfrom the source of supplying to afield or fields. Salinity changes may occur in both the soil and water: This will depend on the irrigation water quality, the level of water management, and the geologic materials of the area. The quality of ground and surface water may be altered depending on environmental conditions. Water lost from the irrigation system to downstream runoff may contain dissolved substances, sediment, and sediment-attached substances that may degrade water quality and increase water temperature. This practice may make water available for wildlife, but may not significantly increase habitat.

Irrigation Land Leveling (464): Reshaping the surface of land to be irrigated to planned grades.

The effects of this practice depend on the level of irrigation water management. If plant root zone soil water is properly managed, then quality decreases of surface and ground water may be avoided. Under poor management, ground and surface water quality may deteriorate. Deep percolation and recharge with poor quality water may lower aquifer quality. Land leveling may minimize erosion and when runoff occurs concurrent sediment yield reduction. Poor management may cause an increase in salinity of soil, ground and surface waters. High efficiency surface irrigation is more probable when earth moving elevations are laser controlled.

Irrigation Pit or Regulating Reservoir, Irrigation Pit (552A): A small storage reservoir constructed to regulate or store a supply of water for irrigation

Irrigation Pit or Regulating Reservoir, Regulating Reservoir (552B): A small storage reservoir constructed to regulate or store a supply of water for irrigation.

Irrigation Storage Reservoir (436): An irrigation water storage structure made by constructing a dam.

Irrigation System, Drip or Trickle (441): A planned irrigation system in which all necessary facilities are installed for efficiently applying water directly to the root zone of plants by means of applicators (orifices, emitters, porous tubing, or perforated pipe) operated under low pressure (Figure 2-20). The applicators can be placed on or below the surface of the ground (Figure 2-2 1).

Surface water quality may not be significantly affected by transported substances because runoff is largely controlled by the system components (practices). Chemical applications may be applied through the system. Reduction of runoff will result in less sediment and chemical losses from the field during irrigation. If excessive, local, deep percolation should occur, a chemical hazard may exist to shallow ground water or to areas where geologic materials provide easy access to the aquifer

Irrigation System, Sprinkler (422): A planned irrigation system in which all necessary facilities are installed for efficiently applying water by means of perforated pipes or nozzles operated under pressure.

Proper irrigation management controls runoff and prevents downstream surface water deterioration from sediment and sediment attached substances. Over irrigation through poor management can produce impaired water quality in runoff as well as ground water through increased percolation. Chemigation with this system allows the operator the opportunity to mange nutrients, wastewater and pesticides. For example, nutrients applied in several incremental applications based on the plant needs may reduce ground water contamination considerably, compared to one application during planting. Poor management may cause pollution of surface and ground water. Pesticide drift from chemigation may also be hazardous to vegetation, animals, and surface water resources. Appropriate safety equipment, operation and maintenance of the system is needed with chemigation to prevent accidental environmental pollution or backflows to water sources.

Irrigation System, Surface and Subsurface (443): A planned irrigation system in which all necessary water control structures have been installed for efficient distribution of irrigation water by surface means, such as furrows, borders, contour levees, or contour ditches, or by subsurface means.

Operation and management of the irrigation system in a manner which allows little or no runoff may allow small yields of sediment or sediment-attached substances to downstream waters. Pollutants may increase if irrigation water management is not adequate. Ground water quality from mobile, dissolved chemicals may also be a hazard if irrigation water management does not prevent deep percolation. Subsurface irrigation that requires the drainage and removal of excess waterfrom the field may discharge increased amounts of dissolved substances such as nutrients or other salts to surface water. Temperatures of downstream water courses that receive runoff waters may be increased. Temperatures of downstream waters might be decreased with subsurface systems rvhen excess water is being pumped from the field to lower the water table. Downstream temperatures should not be affected by subsurface irrigation during summer months if lowering the water table is not required. Improved aquatic habitat may occur if runoff or seepage occurs from surface systems or from pumping to lower the water table in subsurface systems.

Irrigation System, Tailwater Recovery (447): A facility to collect, store, and transport irrigation tailwater for reuse in the farm irrigation distribution system.

The reservoir will trap sediment and sediment attached substances from runoff waters. Sediment and chemicals will accumulate in the collection facility by entrapping which would decrease downstream yields of these substances.

Salts, soluble nutrients, and soluble pesticides will be collected with the runoff and will not be released to surface waters. Recovered irrigation water with high salt and/or metal content will ultimately have to be disposed of in an environmentally safe manner and location. Disposal of these waters should be part of the overall management plan. Although some ground water recharge may occur; little if any pollution hazard is usually expected.

Irrigation Water Conveyance, Ditch and Canal Lining, Flexible Membrane (428B): A fixed lining of impervious material installed in an existing or newly constructed irrigation field ditch or irrigation canal or lateral.

Irrigation Water Conveyance, Ditch and Canal Lining, Galvanized Steel (428C): A fixed lining of impervious material installed in an existing or newly constructed irrigation field ditch or irrigation canal or lateral.

Irrigation Water Conveyance, Ditch and Canal Lining, Nonreinforced Concrete (428A): A fixed lining of impervious material installed in an existing or newly constructed irrigation field ditch or irrigation canal or lateral.

Irrigation Water Conveyance, High-Pressure, Underground, Plastic (430DD): A pipeline and appurtenances installed in an irrigation system

Irrigation Water Conveyance, Low-Pressure, Underground, Plastic (430EE): A pipeline and appurtenances installed in an irrigation system.

Irrigation Water Conveyance, Pipeline, Aluminum Tubing (430AA): A pipeline and appurtenances installed in an irrigation system

Irrigation Water Conveyance, Pipeline, Asbestos-Cement (430BB): A pipeline and appurtenances installed in an irrigation system

Irrigation Water Conveyance, Pipeline, Nonreinforced Concrete (430CC): A pipeline and appurtenances installed in an irrigation system.

Irrigation Water Conveyance, Pipeline, Reinforced Plastic Mortar (430GG): 4 pipeline and appurtenances installed in an irrigation system.

Irrigation Water Conveyance, Pipeline, Rigid Gated Pipeline (430HH): A rigid pipeline, with closely spaced gates, installed as part of a surface irrigation system.

Irrigation Water Conveyance, Pipeline, Steel (430FF): A pipeline and appurtenances installed in an irrigation system.

Irrigation Water Management (449): Determining and controlling the rate, amount, and timing of irrigation water in a planned and efficient manner.

Management of the irrigation system should provide the control needed to minimize losses of water, and yields of sediment and sediment attached and dissolved substances, such as plant nutrients and herbicides, from the system. Poor management may allow the loss of dissolved substances from the irrigation system to surface or ground water: Good management may reduce saline percolation from geologic origins. Returns to the surface water system would increase downstream water temperature.

The purpose is to effectively use available irrigation water supply in managing and controlling the moisture environment of crops to promote the desired crop response, to minimize soil erosion and loss of plant nutrients, to control undesirable water loss, and to protect water quality.

To achieve this purpose the irrigator must have knowledge of (I) how to determine when irrigation water should be applied, based on the rate of water used by crops and on the stages of plant growth; (2) how to measure or estimate the amount of water required for each irrigation, including the leaching needs; (3) the normal time needed for the soil. to absorb the required amount of water and how to detect changes in intake rate; (4) how to adjust water stream size, application rate, or irrigation time to compensate for changes in such factors as intake rate or the amount of irrigation runofffrom an area; (5) how to recognize erosion caused by irrigation; (6) how to estimate the amount of irrigation runoff from an area; and (7) how to evaluate the uniformity of water application.

Lined Waterway or Outlet (468): A waterway or outlet having an erosion-resistant lining of concrete, stone, or other permanent material.

The lined section extends up the side slopes to a designed depth. The earth above the permanent lining may be vegetated or otherwise protected.

This practice may reduce the erosion in concentrated flow areas resulting in the reduction of sediment and substances delivered to the receiving waters.

When used as a stable outlet ***for*** *another practice, lined waterways may increase the likelihood of dissolved and suspended substances being transported to surface waters due to highjlow velocities.*

Livestock Exclusion (472): Excluding livestock from an area not intended for grazing.

Livestock exclusion may improve water quality by preventing livestock from being in the water or walking down the banks, and by preventing manure deposition in the stream. The amount of sediment and manure may be reduced in the surface water. This practice prevents compaction of the soil by livestock and prevents losses of vegetation and undergrowth. This may maintain or increase evapotranspiration. Increased permeability may reduce erosion and lower sediment and substance transportation to the surface andresultings from the application of this practice may reduce surface water temperature.

Mole Drain (482): An underground conduit constructed by pulling a bullet-shaped cylinder through the soil.

Mulching (484): Applying plant residues or other suitable materials not produced on the site to the soil surface.

Nutrient Management (590): Managing the amount, form, placement, and timing of applications of plant nutrients.

Pasture and Hay Planting (512): Establishing and reestablishing long-term stands of adapted species of perennial, biennial, or reseeding forage plants. (Includes pasture and hayland renovations. Does not include grassed waterways or outlets on cropland.)

The long-term effect will be an increase in the quality of the surface water due to reduced erosion and sediment delivery. Increased infiltration and subsequent percolation may cause more soluble substances to be carried to ground water

Pasture and Hayland Management (510): Proper treatment and use of pasture or hayland.

With the reduced runoff there will be less erosion, less sediment and substances transported to the surface waters. The increased infiltration increases the possibility of soluble substances leaching into the ground water.

Pipeline (516): Pipeline installed for conveying water for livestock or for recreation

Pipelines may decrease sediment, nutrient, organic, and bacteria pollution from livestock. Pipelines may afford the opportunity for alternative water sources other than streams and lakes, possibly keeping the animals away from the stream or impoundment. This will prevent bank destruction with resulting sedimentation, and will reduce animal waste deposition directly in the water: The reduction of concentrated livestock areas will reduce manure solids, nutrients, and bacteria that accompany surface runoff.

Planned Grazing System (556): A practice in which two or more grazing units are alternately rested and grazed in a planned sequence for a period of years, and rest periods may be throughout the year or during the growing season of key plants.

Planned grazing systems normally reduce the system time livestock spend in each pasture. This increases quality and quantity of vegetation. As vegetation quality increases, fiber content in manure decreases which speeds manure decomposition and reduces pollution potential. Freeze-thaw, shrink-swell, and other natural soil mechanisms can reduce compacted layers during the absence of grazing animals. This increases infiltration, increases vegetative growth, slows runoff, and improves the nutrient and moisture filtering and trapping ability of the area.

Decreased runoff will reduce the rate of erosion and movement of sediment and dissolved and sediment-attached substances to downstream water courses. No increase in ground water pollution hazard would be anticipated from the use of this practice.

Pond (378): A water impoundment made by constructing a dam or an embankment or by excavation of a pit or dugout.

Ponds may trap nutrients and sediment which wash into the basin. This removes these substances from downstream. Chemical concentrations in the pond may be higher during the summer months. By reducing the amount of water that **flows** *in the channel downstream, the frequency of flushing of the stream is reduced and there is a collection of substances held temporarily within the channel. A pond may cause more leachable substance to be carried into the ground water*

Precision Land Forming (462): Reshaping the surface of land to planned grades.

Prescribed Burning (338): Applying fire to predetermined areas under conditions under which the intensity and spread of the fire are controlled.

When the area is burned in accordance with the specifications of this practice the nitrates with the burned vegetation will be released to the atmosphere. The ash will contain phosphorous and potassium which will be in a relatively highly soluble form. If a runoff event occurs soon after the bum there is a probability that these two materials may be transported into the ground water or into the surface water. When in a soluble state the phosphorous and potassium will be more difficult to trap and hold in place. When done on range grasses the growth of the grasses is increased and there will be an increased tie-up of plant nutrients as the grasses' growth is accelerated.

Prescribed Grazing (Proper Grazing Use)(528A): Grazing at an intensity that will maintain enough cover to protect the soil and maintain or improve the quantity and quality of desirable vegetation

Planned grazing systems normally reduce the system time livestock spend in each pasture. This increases quality and quantity of vegetation. As vegetation quality increases, fiber content in manure decreases which speeds manure decomposition and reduces pollution potential. Freeze-thaw, shrink-swell, and other natural soil mechanisms can reduce compacted layers during the absence of grazing animals. This increases infiltration, increases vegetative growth, slows runoff, and impioves the nutrient and moisture filtering and trapping ability of the area.

Decreased runoff will reduce the rate of erosion and movement of sediment and dissolved and sediment-attached substances to downstream water courses. No increase in ground water pollution hazard would be anticipated from the use of this practice.

Increased vegetation slows runoff and acts as a sediment filter for sediments and sediment attached substances, uses more nutrients, and reduces raindrop splash. Adverse chemical effects should not be anticipated from the rue of this practice.

Proper Woodland Grazing (530): Grazing wooded areas at an intensity that will maintain adequate cover for soil protection and maintain or improve the quantity and quality of trees and forage vegetation.

This practice is applicable on wooded areas producing a significant amount of forage that can be harvested without damage to other values. In these areas there should be no detrimental effects on the quality of surface and ground water. Any time this practice is applied there must be a detailed management and grazing plan

Pumped Well Drain (532): A well sunk into an aquifer from which water is pumped to lower the prevailing water table.

Range Planting (Seeding)(550): Establishing adapted plants by seeding on native grazing land (does not include pasture and hayland planting).

Increased erosion and sediment yield may occur during the establishment of this practice. This is a temporary situation and sediment yields decrease when reseeded area becomes established. If chemicals are used in the reestablishment process, chances of chemical runoff into downstream waler courses are reduced if application is applied according to label instructions. After establishment of the grass cover, grass sod slows runoff, acts as a filter to trap sediment, sediment attached substances, increases infiltration, and decreases sediment yields.

Rangeland Fertilization (203): [ADD DEFINITION]

Regulating Water in Drainage Systems (554): Controlling the removal of surface or subsurface runoff, primarily through the operation of water-control structures.

Riparian Forest Buffer (Field Windbreak) (392): A strip or belt of trees or shrubs established in or adjacent to a field.

Rock Barrier (555): A rock retaining wall constructed across the slope to form and support a bench terrace that will control the flow of water and check erosion on sloping land.

Roof Runoff Management (558): A facility for controlling and disposing of runoff water from roofs.

This practice may reduce erosion and the delivery of seditnenr and related substances to surface waters. It will reduce the volume of water polluted by animal wastes. Loadings of organic waste, nutrients, bacteria, and salts to surface water are prevented from flowing across concentrated waste areas, barnyards, roads and alleys will be reduced. Pollution and erosion will be reduced. Flooding may be prevented and drainage may improve.

Runoff Management System (570): A system for controlling excess runoff caused by construction operations at development sites, changes in land use, or other land disturbances.

Sediment Basin (350): A basin constructed to collect and store debris or sediment.

Sediment basins will retnove sediment, sediment associated materials and other debris from the water which is passed on downstream. Due to the detention of the runoff in the basin. there is an increased opportunity for soluble materials to be leached toward the ground water.

Soil **and Crop Water Use Data:** From soils information the available water-holding capacity of the soil can be determined along with the amount of water that the plant can extract from the soil before additional irrigation is needed.

Water use information for various crops can be obtained from various USDA publications.

The purpose is to allow the water user to estimate the amount of available water remaining in the root zone at any time, thereby indicating when the next irrigation should be scheduled and the amount of water needed. Methods to measure or estimate the soil moisture should be employed, especially for high-value crops or where the water-holding capacity of the soil is low.

Spring Development (574): Improvin g springs and seeps by excavating, cleaning, capping, or providing collection and storage facilities.

There will be negligible long-term water quality impacts with spring developments. Erosion and sedimentation may occurfrom any disturbed areas during and immediately after construction, but should be short-lived. These sediments will have minor amounts of adsorbed nutrients from soil organic matte):

Stream Channel Stabilization (584): [ADD DEFINITION]

Stream Corridor Improvement (204): [ADD DEFINITION]

Stream crossing (interim): A stabilized area to provide access across a stream for livestock and farm machinery.

The purpose is to provide a controlled crossing or watering access point for livestock along with access for farm equipment, control bank and streambed erosion, reduce sediment and enhance water quality, and maintain or improve wildlife habitat.

Streambank and Shoreline Protection (580): Using vegetation or structures to stabilize and protect banks and streams. lakes. estuaries, or excavated channels against scour and erosion.

Stripcropping, Contour (585): Growing crops ın a systematic arrangement of strips or bands on the contour to reduce water erosion. The crops are arranged so that a strip of grass or close-

growing crop is alternated with a strip of clean-tilled crop or fallow or a strip of grass is alternated with a close-growing crop.

Structure for Water Control (587): A structure in an irrigation, drainage, or other water management systems that conveys water, controls the direction or rate of flow, or maintains a desired water surface elevation.

Subsurface Drain (606): A conduit, such as corrugated plastic tile, or pipe, installed beneath the ground surface to collect and/or convey drainage water.

Soil water outlet to surface water courses by this practice may be low in concentrations of sediment and sediment-adsorbed substances and that may improve stream water quality. Sometimes the drained soil water is high in the concentration of nitrates and other dissolved substances and drinking water standards may be exceeded. If drainage water that is high in dissolved substances is able to recharge ground water, the aquifer quality may become impaired. Stream water temperatures may be reduced by water drainage discharge. Aquatic habitat may be altered or enhanced with the increased cooler water temperatures.

Surface Drainage Field Ditch (607): A graded ditch for collecting excess water in a field.

From erosive fields, this practice may increase the yields of sediment and sediment-attached substances to downstream water courses because of an increase in runoff. In other fields, the location of the ditches may cause a reduction in sheet and rill erosion and ephemeral gully erosion. Drainage of high salinity areas may raise salinity levels temporarily in receiving waters. Areas of soils with high salinity that are drained by the ditches may increase receiving waters. Phosphorus Loads, resulting from this practice may increase eutrophication problems in ponded receiving waters. Water temperature changes will probably not be significant. Upland wildlife habitat may be improved or increased although the habitat formed by standing water and wet areas may be decreased.

Surface Drainage, Main or Lateral (608): An open drainage ditch constructed to a designed size and grade.

Terrace (600): An earthen embankment, a channel, or combination ridge and channel constructed across the slope.

This practice reduces the slope length and the amount of surface runoff which pass& over the area downslope from an individual terrace. This may reduce the erosion rate and production of sediment within the terrace interval. Terraces trap sediment and reduce the sediment and associated pollutant content in the runoff water which enhance surface water quality. Terraces may intercept and conduct surface runoff at a nonerosive velocity to stable outlets, thus, reducing the occurrence of ephemeral and classic gullies and the resulting sediment. Increases in infiltration can cause a greater amount of soluble nutrients and pesticides to be leached into the soil. Underground outlets may collect highly soluble nutrient and pesticide leachates and convey runoff and conveying it directly to an outlet, terraces may increase the delivery of pollutants to surface waters. Terraces increase the opportunity to leach salts below the root zone in the soil. Terraces may have a detrimental effect on water quality if they concentrate and accelerate delivery of dissolved or suspended nutrient, salt, and pesticide pollutants to surface or ground waters.

Tree Planting (612): To set tree seedlings or cutting in the soil

Trough or Tank (614): A trough or rank, with needed devices for water control and waste water disposal, installed to provide drinking water for livestock.

By the installation of a trough or tank, livestock may be better distributed over the pasture, grazing can be better controlled, and surface runoff reduced, thus reducing erosion. By itself this practice will have only a minor effect on water quality; however when coupled with other conservation practices, the beneficial effects of the combined practices may be large. Each site and application should be evaluated on their own merits.

Use Exclusion (472): Excluding livestock from an area not intended for grazing

Livestock exclusion may improve water quality by preventing livestock from being in the water or walking down the banks, and by preventing manure deposition in the stream. The amount of sediment and manure may be reduced in the surface water. This practice prevents compaction of the soil by livestock and *prevents losses of vegetation and undergrowth. This may maintain or increase evapotranspiration. Increased permeability may reduce erosion and lower sediment and substance transportation to the surface waters. Shading along streams and channels resulting from the application of this practice may reduce surface water temperature.*

Waste Management System (312): A planned system in which all necessary components are installed for managing liquid and solid waste, including runoff from concentrated waste areas, in a manner that does not degrade air, soil, or water resources.

Waste Storage Pond (425): An impoundment made by excavation or earth fill for temporary storage of animal or other agricultural wastes.

This practice reduces the direct delivery of polluted water; which is the runoff from manure stacking areas and feedlots and barnyards, to the surface waters. This practice may reduce the organic, pathogen, and nutrient loading to surface waters. This practice may increase the dissolved pollutant loading to ground water by leakage through the sidewalls and bottom.

Waste Storage Structure (313): A fabricated structure for temporary storage of animal wastes or other organic agricultural wastes.

This practice may reduce the nutrient, pathogen, and organic loading to the surface waters. This is accomplished by intercepting and storing the polluted runoff from manure stacking areas, barnyards and feedlots. This practice will not eliminate the possibility of contaminating surface and ground water.. however. it greatly reduces this possibility.

Waste Treatment Lagoon (359): An impoundment made by excavation or earth fill for biological treatment of animal or other agricultural wastes.

This practice may reduce polluted surficial runoff and the loading of organics, pathogens, and nutrients into the surface waters. It decreases the nitrogen content of the surface runofffrom feedlots by denitrifcation. Runoff is retained long enough that the solids and insoluble phosphorus settle and form a sludge in the bottom of the lagoon. There may be some seepage through the sidewalls and the bottom of the lagoon. Usually the long-term seepage rate is low enough, so that the concentration of substances transported into the ground water does not reach an unacceptable level.

Waste **Utilization (633):** Using agricultural wastes or other wastes on land in an environmentally acceptable manner while maintaining or improving soil and plant resources.

Waste utilization helps reduce the transport of sediment and related pollutants to the surface water: Proper site selection, timing of application and rate of application may reduce the potential for degradation of surface and ground water. This practice may increase microbial action in the surface layers of the soil, causing a reaction which assists in controlling pesticides and other pollutants by keeping them in place in the field.

Mortality and other compost, when applied to agricultural land, will be applied in accordance with the nutrient management measure. The cornposting facility may be subject to State regulations and will have a written operation and management plan if SCS practice 317 (cornposting facility) is used.

Water and Sediment Control Basin (638): An earthen embankment or a combination ridge and channel generally constructed across the slope and minor watercourses to form a sediment trap and water detention basin.

The practice traps and removes sediment and sediment-attached substances from runoff. Trap control efficiencies for sediment and total phosphorus, that are transported by runoff, may exceed 90 percent in silt loam soils. Dissolved substances, such as nitrates. may be removed from discharge to downstream areas because of the increased infiltration. Where geologic condition permit, the practice will lead to increased loadings of dissolved substances toward ground water. Water temperatures of surface runoff, released through underground outlets, may increase slightly because of longer exposure to warming during its impoundment.

Water Table Control (641): Water table control through proper use of subsurface drains, water control structures, and water conveyance facilities for the efficient removal of drainage water and distribution of irrigation water.

The water table control practice reduces runoff, therefore downstream sediment and sediment-attached substances yields will be reduced. When drainage is increased, the dissolved substances in the soil water will be discharged to receiving water and the quality of water reduced. Maintaining a high water table, especially during the nongrowing season, will allow denitrification to occur and reduce the nitrate content of surface and ground by as much as 75 percent. The use of this practice for salinity control can increase the dissolved substance loading of downstream waters while decreasing the salinity of the soil. Installation of this practice may create temporary erosion and sediment yield hazards but the completed practice will lower erosion and sedimentation levels. The effect of the water table control of this practice on downstream wildlife communities may vary with the purpose and management of the water in the system.

Waterspreading (640): Diverting or collecting runoff from natural channels, gullies, or streams with a system of dams, dikes, ditches, or other means, and spreading it over relatively flat areas.

Well (642): A well constructed or improved to provide water for irrigation. livestock, wildlife, or recreation.

When water is obtained, if it has poor quality because of dissolved substances, its use in the surface environment or its discharge to downstream water courses the surface water will be degraded. The location of the well must consider the natural water quality and the hazards of its use in the potential contamination of the environment. Hazard exists during well development and its operation and maintenance to prevent aquifer quality damage from the pollutants through the well itself by back flushing, or accident, or flow down the annular spacing between the well casing and the bore hole.

Water-Measuring Device: An irrigation water meter, flume, weir, or other water-measuring device installed in a pipeline or ditch.

The measuring device must be installed between the point of diversion and water distribution system used on the field. The device should provide a means to measure the rate of flow. Total water volume used may then be calculated using rate of flow and time, or read directly, if a totalizing meter is used.

*The purpose is to provide the irrigator the rate of flow and/or application of water, and the total amount of water applied to the **field** with each irrigation*

Wetland Restoration (657A): [ADD DEFINITION]

APPENDIX 4.2B

EXAMPLES OF COSTS FOR BMPs

NORTH CAROLINA AGRICULTURE COST SHARE PROGRAM
AVERAGE COST FOR PROGRAM YEARS 1997 - 1999
NRCS AREA 3

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TABLE 1

Vegetation 1a[1] - For seeding Waterways, Diversions, Field Borders, Filter Strips, Dams, Sediment Basins, and Critical Area Planting.

Components	Unit	Cost Per Unit	Rate Per Acre		Cost Per Acre
Lime (Bulk)	Ton	$ 25.00	2 Tons		$ 50.00
(Bag)	Ton	50.00	2 Tons		100.00
Fertilizer (10-10-10[2])	Cwt	10.00	10 Cwt		100.00
Seedbed Preparation	Acre	50.00	---		50.00
Seed - Fescue	Lbs.	97	60 Lbs.		58.00
Small Grain	Bu.	8:00	1 Bu.		8.00
Small Grain Mulch	Ton	150.00	2 Tons		300.00
			TOTAL:	(Bulk)	566.00
				(Bag)	616.00

Vegetation 1b[1] - For Stripcropping or Cropland Conversion

Components	Unit	Cost Per Unit	Rate Per Acre	Cost Per Acre
Lime'	Ton	$ 25.00	2 Tons	$ 50.00
Fertilizer (10-10-10[2])	Cwt	10.00	7 Cwt	70.00
Seedbed Preparation	Acre	26.00	---	26.60
Seed - Fescue	Lbs.	.97	25 Lbs.	24.00
			TOTAL:	$170.00

1 Fescue used as base vegetation for establishing average cost. Other vegetative types may be used if they meet site specifications but must use base average cost developed for fescue.

2 Applicant may use other than 10-10-10 fertilizer and the NC Agriculture Cost Share Program will pay 75% of $.22 per lb. of plant food based on soil test.

3 NC Agriculture Cost Share Program will pay only bulk rate for stripcropping or cropland conversion.

NRCS AREA 3
AVERAGE COSTS
for
AGRICULTURE COST S&E PROGRAM
PROGRAM YEARS 1997 - 1999

A. INCENTIVE PAYMENTS

I.	Conservation Tillage (see Practice Guidelines, Section V)		$10.00 per acre per year
II.	Conservation Tillage for Tobacco, Cabbage and Tomatoes		
	a.	1st year planting (5 acre limit)	$50.00 per acre
	b.	1st year planting (all acres in excess of 5 acres)	$15.00 per acre
	c.	2nd and 3rd year plantings (no acreage limit)	$15.00 per- acre
III.	Long Term No-till		$75.00 per acre
IV.	Sod-based Rotation		
	a.	4-year sod-based rotation (17 months in sod)	$40.00 per acre
	b.	4-year sod-based rotation (29 months in sod)	$70.00 per acre
	c.	5-year sod-based rotation (41 months in sod)	$95.00 per acre
V.	Stripcropping		$15.00 per acre[1]
VI.	Nutrient Management		$ 6.00 per acre

[1] This incentive payment is to be added to the cost share available under component B. I., Stripcropping.

B. AVERAGE COST LIST FOR BMP COMPONENTS

I. Stripcropping

a. Vegetation establishment (see Practice Guidelines, Section V) — $ 85.00 per ac.

b. Land smoothing (see Practice Guidelines, Section V) (designate elements from Section C, II)

II. Cropland Conversion

a. Conventional - includes grass, trees, and/or perennial wildlife plantings — 170.00 per ac.

b. Christmas tree plantation seeding (see Practice Guidelines, Section V) — X* per ac.

III. Pasture Land Conversion to Trees (Class VII land only)

a. Trees (all species planted for long term timber management. No Christmas trees or ornamentals.) — 85.00 per ac.

b. Competing vegetation control

1. Mowing — 25.00 per ac.

2. Herbicide application — 30.00 per ac.

c. Livestock exclusion (designate elements from Section B, XI,

IV. Critical Area Planting

a. Grading (designate elements from Section C, II)

b. Vegetation (designate elements from Section C, I)

*"X" indicates element is not included in area's average cost list and approval to use this element must be given by the Area Office.

c. Stone (designate elements from Section C, VII)

d. Geotextiles (fabric filter cloth) $2.00 sq. yd.

e. Pipe drops and surface inlets (designate elements from Section C, VI)

f. Animal guard (pre-fabricated flap gate type) 4.00 each

g. Sub-surface drain' (designate elements from Section C, V)

V. Filter Strip

a. Grading (designate elements from Section C, II)

b. Vegetation (designate elements from Section C, I)

c. Pipe drops and surface inlets (designate elements from Section C, VI)

d. Animal guard (pre-fabricated flap gate type) 4.00 each

VI. Vegetated Field Border

a. Grading (designate elements from Section C, II)

b. Vegetation (designate elements from Section C, I)

c. Pipe drops and surface inlets (designate elements from Section C, VI)

d. Animal guard (pre-fabricated flap gate type) 4.00 each

VII. Grassed Waterway

a. Grading (designate elements from Section C, II)

Average Cost Per Unit

b. Land smoothing (see Practice Guidelines, Section V) (designate elements from Section C, II)

c. Vegetation (designate elements from Section C, I)

d. Sub-surface drain (designate elements from Section C, V; gravel filter required, except Area 3)

e. Pipe drops and surface inlets (designate elements from Section C, VI)

f. Animal guard (pre-fabricated flap gate'type) $ 4.00 each

VIII. Diversion

a. Shaping 1.00 L. ft.

b. Land smoothing (see Practice Guidelines, Section V) (designate elements from Section C, II)

c. Vegetation (designate elements from Section C, I)

d. Pipe outlet system (designate elements from Section c, V)

e. Pipe drops and surface inlets (designate elements from Section C, VI)

f. Sub-surface drain (designate elements from Section C, V; gravel filter required, except Area 3)

g. Animal guard (pre-fabricated flap gate type) 4.00 each

IX. Rock-lined Waterway or Outlet

a. Clearing (wooded areas only) X* per ac.

*"X" indicates element is not included in area's average cost list and approval to use this element must be given by the Area Office.

Average Cost Per Unit

b. Grading, excavation (designate elements from Section C, II or III),

c. Stone (designate elements from Section C, VII)

d. Geotextiles (fabric filter cloth) $ 2.00 sq. yd.

e. Vegetation (designate elements from Section C, I)

X. Terrace

a. Shaping 1.00 L. ft.

b. Land smoothing (see Practice Guidelines, Section V) (designate elements from Section C, II)

c. Vegetation (designate elements from Section C, I)

d. Pipe drops and surface inlets (designate elements from Section C, VI)

e. Pipe outlet system (designate elements from Section C, V)

f. Animal guard (pre-fabricated flap gate type) 4.00 each

XI. Livestock Exclusion (see Practice Guidelines, Section V)

a. Fencing

1. Barbed or woven wire (minimum of 4 strands of high tensile) 1.50 L. ft.

2. Electric wire (high tensile and conventional and a minimum of 3 strands) .90 L. ft.

b. Gates 65.00 each

Average Cost Per Unit

XII. Spring Development (from water source to junction box - maximum of 2 spring developments per trough/tank charged to NCACSP)

a_	Excavation for spring development (see Practice Guidelines, Section V)	$ 50.00 per hr.
b.	Plastic pipe and fittings (designate elements from Section C, V)	
c.	Junction box, concrete	40.00 each
d.	Stone (designate elements from Section C, VII)	
e.	Geotextiles (fabric filter cloth)	2.00 sq. yd.
f.	Livestock exclusion (designate elements from Section B, XI)	

XIII. Trough or Tank (from junction box to overflow outlet) (see Practice Guidelines, Section V)

a.	Pipe and fittings (designate elements from Section C, V)	
b.	Watering tanks	75% of actual cost not to exceed $400 charge to NCACSP
c.	Geotextiles (fabric filter cloth)	2.00 sq. yd.
d.	Stone (designate elements from Section C, VII)	
e.	Vegetation (designate elements from Section C, I)	
f.	Animal guard (pre-fabricated flap gate type)	4.00 each
g.	Brass automatic float valve	18.00 each

h.	Livestock exclusion (designate element from Section B, XI)	
i.	Well (must include well head protection)	
	1. Well construction	75% of actual cost not to exceed $500 charge to NCACSP
	2. Well head protection	75% of actual cost not to exceed $500 charge to NCACSP
j.	Pumps (includes all costs associated with pump installation). Applicant must sign statement reflecting the use of the pump is for livestock watering <u>ONLY</u>.	75% of actual cost not to exceed $450 charge to NCACSP
k.	Solar powered water pump (see Practice Guidelines, Section V)	$<u>2000.00</u> each
l.	Windmills	75% of actual cost not to exceed $2,400 charge to NCACSP

XIV. Stream Crossings and Stock Trails

a. Excavation, grading - for FORD TYPE Stream Crossing (based on degree of job difficulty as shown below):

Site Classifications

Degree of Job Difficulty		Depth x Width[4] (sq. ft.)	Price
1.	Low	c 80	$ 600.00
2.	Moderate	80 to 120	800.00
3.	High	> 120	1000.00

[4] Product of average depth of stream channel times the average width of stream channel at crossing location_

Average Cost Per Unit

b. Earth fill and excavation - for PIPE CULVERT TYPE Stream Crossings. (designate elements from Section C, III)

c. Grading and excavation - for STOCK TRAILS

1. Improving existing trail	$ .50	L. ft.
2. Developing new trail	1.50	L. ft.

d Geotextiles (fabric filter cloth)	2.00	sq. yd.

e. Stone (designate elements from Section C, VII)

f. Vegetation (designate elements from Section C, I)

g. Fabrication of metal anchor pins	x*	each

h. Pipe and fittings (designate elements from Section C, V) ** CSP shall be asphalt coated if more than one section is used. Aluminum or PVC pipe may be used for this practice.

i. Livestock exclusion (designate elements from Section B, XI)

For **all waste** management **structures, the CPO must include a detailed sketch of the structure and location of stream system being protected. Signature of proper job approval must be included on the** NCACSP 11-A.

A copy of the waste management plan is no longer required to be submitted with the CPO. However, a statement signed by district staff which certifies that the district CPO file contains a waste management plan that meets NRCS **standards with original signatures and maps of fields to be applied is required. Form NC-ACSP-WMP is located in Section VI of this manual.**

***"X" indicates element is not included in area's average cost list and approval to use this element must be given by the Area Office.**

XV. Components of Waste Management Structures/Systems

a. Clearing (wooded areas only) — $500.00 per ac.

b. Excavation for fill for holding ponds, lagoons, etc. (designate elements from Section C, III)

c. Filter strip for waste water treatment (designate elements from Section B, V)

d. Vegetation (designate elements from Section C, I)

e. Push-off ramp (includes safety rail) — 75% of actual cost not to exceed $4,000 charge to NCACSP

f. Livestock exclusion (designate elements from Section B, XI)

g. Concrete and block (designate elements from Section C, IV)

h. Collection tank for temporary storage and transfer of liquid animal waste (must meet state specifications)

1. 1,000 gallon concrete tank (including installation) — $486.00 each

2. 1,500 gallon concrete tank (including installation) — 599.00 each

i. Geotextiles (fabric filter cloth) — 2.00 sq. yd.

j. Pressure treated lumber (includes fasteners and labor)

a. Boards — 1.70 bd. ft.

b. 4" x 4" post — 1.50 L. ft.
4" x 6" post — 1.75 L. ft.
6" x 6" post — 2.00 L. ft.

k. Pipe and fittings (designate elements from Section C, V)

		Average Cost Per Unit
l.	Bent support	$ 50.00 each
m.	Pump & motor and agitation' system (for waste handling)	75% of actual cost with engineer's approval
	Fiberglass pump housing (Site built protection may be used in lieu of fiberglass housings with approval from the Area Office. 'Cost Share shall be 75% of actual cost not to exceed the current rate for fiberglass pump housings.)	250.00 each
n.	Roof gutters	
	1. Seamless aluminum	
	a. 5"	1.75 L. ft.
	b. 6"	2.00 L. ft.
	2. Assembled sections of aluminum or vinyl	
	a. 5"	1.20 L. ft.
	b. 6"	1.40 L. ft.
	3. Downspout	1.00 L. ft.
o.	Metal Fabrication (includes all structural steel materials, fabrication and installation labor)	75% of actual cost with engineer's approval

XVI. Dry Stack/Litter Storage Facility with roof (Maximum size cost-shared is based on storage required in waste utilization plan, average stacking height of 5 feet.)

a. Totally wooden structure.

1.	Poultry - structure built in combination with concrete slab (includes grading, concrete, roof, framing and all other necessary components of the dry stack).	2.10 cu. ft.

Average Cost Per Unit

2. Dairy/Beef

   a. Wooden structure' $ 7.00 sq. ft.

   b. Concrete (designate elements from Section C, IV)

   c. Grading of site (designate elements from Section C, II)

   d. Earth fill (designate elements from Section C, III)

   e. Stone (designate elements from Section C, VII)

   f. Steel, reinforcing for concrete structures (includes wire fabric and rebar) .74 per lb.

   g. Sub-surface drain (designate elements from Section C, V)

b. Reinforced concrete or block walls

   1. Framing and roof (area within walls and within ends of roofed area) 2.00 sq. ft.

   2. Concrete or block (designate elements from Section C, IV)

   3. Grading of site (designate elements from Section C, II)

   4. Steel, reinforcing for concrete structures (includes wire fabric and rebar) .74 per lb.

c. Metal fabrication structure (includes all structural steel, concrete for footings, framing, grading, and all other necessary components of the dry stack) 1.00 cu. ft.

   1. Concrete for slab (designate elements from Section C, IV)

Average Cost Per Unit

2. Steel, reinforcing for concrete structures (includes wire fabric and rebar) . $ .74 per lb.

XVII. Slurry Storage Structure

a. Mechancial Equipment (agitator, sidemount pump, overtop kit, knife valve) — 75% of actual cost with engineer's approval

b. Engineered Foundation (design, excavation, compaction gravel, crank nozzle installation, sealing strip, floor penetrations. installation, sump forming, concrete foundation & concrete floor, reinforcement steel, steel starter ring etc.) — 75% of actual cost with engineer's approval

c. Slurry tank — 75% of actual cost with engineer's approval

d. Pump (chopper/pit) — 75% of actual cost with engineer's approval

e. Pipe (designate elements from Section C, V)

Invoices for every item must be submitted with the Request for Payment.

XVIII. Poultry Cornposter - Cost includes only lumber and roof. (For site grading, concrete pad, etc., designate elements from Section C) — 3.00 cu. ft.

XIX. Waste Application Systems — 75% of actual cost not to exceed $15,000 lifetime charge to NCACSP

(For .0200 operations $25,000 lifetime charge to NCACSP)

Includes all costs associated with equipment, materials, construction, installation, vegetation, pumps, etc. from the lagoon to and including the delivery system: Cap includes any previous payments to the applicant for pipe, hydrants or other elements of a waste application system. Cost sharing on fencing as related to this practice is not allowed.

Type of system must-be specified on CPO, ie: center pivot, traveling gun, solid set, underground main and hydrant, honey wagon, mobile irrigation system, etc.

Applicant must sign statement reflecting that they are responsible for the maintenance/replacement of all equipment at their expense for the ten year life of the practice.

Invoices for every element must be submitted with the Request for Payment.

XX. Wetlands constructed for animal waste utilization are part of the NCACSP as specified in the December 13, 1990 memo to all Districts. Until "Standards" are established, constructed wetlands will be implemented at 75% of the actual cost and invoices will be included with the request for payment.

XXI. Controlled Livestock Lounging Area

a. Stone (designate elements from Section C, VII)

b. Geotextiles (fabric filter cloth) — $ 2.00 sq. yd.

c. Grading (designate elements from Section C, II)

d. Vegetation (designate elements from Section C, I)

e. Livestock exclusion (designate elements from Section B, XI)

f. Concrete (designate eiements from Section C, IV)

g. Filter strip (designate elements from Section B, V)

h. Sacrifice area shall be installed according to heavy use area (NRCS Standard 561).

Average Cost Per Unit

XXII. To-Be-Closed or Abandoned CAOs — 75% of actual cost with receipts not to exceed $15,000

XXIII. Removal/disposal of animal waste (for abandoned lagoons, lagoons being abandoned as part of a retrofit to meet .0200, and existing lagoons being retrofitted to meet .0200 only)

Payment

75% of actual cost, receipts are required.

XXIV. Heavy Use Area Protection

a.	Stone (designate elements from Section C, VII)	
b.	Geotextiles (fabric filter cloth)	$ 2.00 sq. yd.
c.	Livestock exclusion (designate elements from Section B, XI)	
d.	Grading (designate elements from Section C, II)	
e.	Filter strip (designate elements from Section B, V)	
f.	Vegetation - limited to disturbed fringe area (designate elements from Section C, I)	

XXV. Grade Stabilization Structures

a.	Clearing (wooded areas only)	500.00 per ac.
b.	Earth moving (designate elements from Section C, III)	
c.	Concrete (designate elements from Section C, IV)	
d.	Stone (designate elements from Section C, VII)	
e.	Bent support	50.00 each
f.	Vegetation (designate elements from Section C, I)	

g. Pipe (designate elements from Section C, V)

h. Geotextiles (fabric filter cloth) $ 2.00 sq. yd.

XXVI. Sediment Basin

a. Clearing (wooded areas only) 500.00 per ac.

b. Earth moving (designate elements from Section C, III)

c. Vegetation (designate elements from Section C, I)

d. Stone (designate elements from Section C, VII)

e. Bent support 50.00 each

f. Concrete for riser anchor (designate elements from Section C, IV)

g. Pipe (designate elements from Section C, V)

h. Geotextiles (fabric filter cloth) 2.00 sq. yd.

XXVII. Water Control Structures

a. Flash board riser

The following prices include the flash board riser, cost of welding pipe to riser, installation and vegetation (in accordance with PS 342).

RISER WIDTH	RISER GA	CORRUGATION	PRICE
18"	14	1/2" x 2 2/3"	415.00
24"	14	1/2" x 2 2/3"	460.00
30"	14	1/2" x 2 2/3"	500.00
36"	14	1/2" x 2 2/3"	690.00
42"	12	1/2" x 2 2/3"	790.00
48"	12	1/2" x 2 2/3"	880.00
54"	12	1/2" x 2 2/3"	1125.00

			Average Cost Per Unit
60 "	12	1" x 3"	$1345.00
66 "	12	1 "x 3 "	1423.00
72 "	12	1 " x 3 "	1670.00
78 "	12	1 " x 3 "	1900.00
a4 "	10	1 " x 3 "	2125.00
90"	10	1 " x 3 "	2370.00
96 "	10	1 "x 3 "	2620.00
102"	a	2 1/2" x 9"	2977.00
108"	a	2 1/2" x 9"	3334.00
114"	a	2 1/2" x 9"	3548.00
120"	a	2 1/2" x 9"	3763.00

b. **Corrugated Pipes ***

The following costs include the pipe and installation (includes vegetation in accordance with PS 342)

PIPE DIAM	CORRUGATION	GAUGE	COST/FT
12 "	1/2" X 2 2/3"	16	11.38
15 "	1/2" X 2 2/3"	16	14.17
18"	1/2" X 2 2/3"	16	16.77
24 "	1/2" X 2 2/3"	14	27.95
30 "	1/2" X 2 2/3"	14	34.45
36"	1/2" X 2 2/3"	14	41.28
42 "	1/2" X 2 2/3"	12	66.62
48 "	1/2" X 2 2/3"	12	75.40
54 "	1/2" X 2 2/3"	12	85.15
60 "	1" x 3"	12	112.78
66 "	1" X 3"	12	123.50
72 "	1" X 3"	12	135.20

* A Maximum **of 30 feet of pipe will be paid for per riser without the approval of Area Engineer.**

c. **Sand cement bag** headwall **sand cement bag** $\geq$ **60** lbs. — 2.60 **per bag**

d. Concrete used in lieu. of sand-cement bag headwall and for anti-floatation $108.00 cu. yd.

e. Aluminum headwalls 5.00 sq. ft.

f. Geotextiles (fabric filter cloth) 2.00 sq. yd.

g. Stone (designate elements from Section C, VII)

h. Aluminum anti-seep collars

	Pipe Diameter	Per Collar
1.	12" - 18"	90.00
2.	24"	110.00
3.	30"	125.00
4.	36"	145.00
5.	42"	180.00
6.	48"	205.00
7.	54"	230.00
8.	60"	260.00
9.	66"	300.00
10.	72"	330.00

XXVIII. Roofed Agri-Chemical Handling Facility

a. Building structure (includes roof, posts, siding, labor and related items) 8.00 sq. ft.

b. Floor

1. Concrete and steel (designate elements from Section C, IV)

2. Thor-o-seal to coat sumps 75 % of actual cost with receipts

3. Sealer for floor 75% of actual cost with receipts

	Average Cost Per Unit
c. Chemical storage room	
1. Roof rafters, top plate, plywood top and ceiling, rolled roofing, tank platform, insulation, door, hardware, labor, etc. (cost for outside dimensions)	$9.00 sq. ft.
2. Concrete block (designate elements from Section C, IV)	
d. Tank platform (pressure treated lumber) (designate elements from B, XV, j)	
e. Plumbing items	
1. Interior (includes pumps, valves, tanks, sink, strainers, eyewash, drench shower and triple rinse device)	2950.00 per job
2. Exterior (includes supply pipe and pump necessary to convey water to the facility)	*75%* of actual cost with receipts
f. Electrical (all components)	*75%* of actual cost with receipts
g. Driveway (entrance and exit)	
1. Grading (designate elements from Section C, II)	
2. Geotextiles (fabric filter cloth)	2.00 sq. yd.
3. Stone (designate elements from Section C, VII)	
h. Miscellaneous (fire extinguisher, first aid kit, bulletin board, etc.)	*75%* of actual cost with receipts

Average Cost Per Unit

XXIX. Riparian Buffer

a. Grading (designate element: from Section C, II)

b. Vegetation (designate elements from Section C, I)

c. Pipe drops and surface inlets (designate elements from Section C, VI)

d. Animal guard (pre-fabricated flap gate type) — $4.00 each

XXX. Odor Control Management System

a. Grading (designate elements from Section C, II)

b. Vegetation (designate elements from Section C, I)

c. Pipe drops and surface inlets (designate elements from Section C, VI)

d. Animal guard (pre-fabricated flap gate-type) — $ 4.00 each

e. Pipe and fittings (designate elements from Section C, V)

XXXI. Insect Control Practice Requires approval by the Technical Review Committee. — 75% of actual cost with receipts

XXXII. Streambank Stabilization

a. Vegetation (designate elements from Section C, I)

b. Tree/shrub establishment — 75% of actual cost with receipts

c. Stone (designate elements from Section C, VII)

		Average Cost Per Unit
d.	Land Smoothing (designate elements from Section C, II)	
e.	Earth Fill (designate elements from Section C, III)	
f	Fencing (designate elements from Section B, XI)	
g.	Other components not in average cost as approved by Area engineer or Division P. E.	75% of actual cost with receipts

C. COMPONENTS WHICH ARE COMMON TO TWO OR MORE PRACTICES

I. Vegetation

a. Materials for establishment of perennial grasses and/or legumes - For seeding Waterways, Diversions, Field Borders, Filter Strips, Dams, Sediment Basins, and Critical Area Planting. Price includes seed, lime and fertilizer according to NC Technical Guide Section IV 342-11, and costs associated with applying materials.

1. Accessible with conventional tillage equipment (see Table 1) — $216.00 per ac.

2. Not accessible or practical with conventional tillage equipment due to steepness of slope. Requires Hydro-seeding. — X* per ac.

b. Seedbed preparation (Disking, harrowing and debris removal per NC Technical Guide, Section IV, 342-11-3, not to be used where land smoothing or grading of the site has resulted in satisfactory seedbed.) — 50.00 per ac.

c. Materials for establishment of perennial grasses and/or legumes for seeding strips, cropland conversion and lounging areas (price includes seed, lime and fertilizer according to NC Technical Guide, Section IV, 512-2 and 512-3, and costs associated with application materials). — 144.00 per ac.

d. Seedbed preparation for strips or cropland conversion (disking, harrowing, and debris removal according to NC Technical Guide, Section IV, 512-l. Not to be used where land smoothing of the site has resulted in proper seedbed). — 26.00 per ac.

*"X" indicates element is not included in area's average cost list and approval to use this element must be given by the Area Office.

		Average Cost Per Unit
e.	Small grain mulch (see Practice Guidelines, Section V).	$300.00 per ac.
f.	Mulch netting/installation (see Practice Guidelines, Section V)	.03 sq. ft.
g.	Excelsior matting (includes installation)	.95 sq. yd.
h.	Silt fence (see Practice Guidelines, Section V)	1.00 L. ft.

II. Grading and shaping

a.	Land smoothing of cropland	
1.	Light	60.00 per ac.
2.	Heavy	80.00 per ac.
b.	Smoothing (light tractor disk and blade work)	180.00 per ac.
c.	Light grading (1" to 3" average movement requiring tracked equipment**)	600.00 per ac.
	(minimum per job)	75% of 250.00
d.	Medium grading (3" to 6" average movement**)	900.00 per ac.
	(minimum per job)	75% of 250.00
e.	Heavy grading (greater than 6" average movement**)	1200.00 per ac.
	(minimum per job)	75% of 250.00

** Average depth is based on average cross-sectional depth.

III. Earth fill and excavation

a.	Excavation only (includes cost of spoil removal)	.85 cu. yd.

b. Earth fill (includes excavation, hauling and placement of fill from specified sources not contiguous to the site.)

1.	Material available adjacent to site	$ 1.25 cu. yd.
2.	Material available adjacent to site requiring compaction with sheepsfoot roller	1.50 cu. yd.
3.	Material hauled from a considerable distance off-site (see Practice Guidelines, Section V)	3.75 cu. yd.
4.	Material hauled from a considerable distance off-site and requiring compaction by a sheepsfoot roller	4.00 cu. yd.

IV. Concrete and Masonry

a.	Non-reinforced and reinforced slab work and curbs not requiring extensive forming, includes non-reinforced footings. Cost will be based on concrete volumes delivered at the site and used in construction.	108.00 cu. yd.
b.	Reinforced concrete wall work and other reinforced work requiring extensive forming, includes reinforced footings. Cost will be based on volume of concrete computed from dimensions shown on the plan and specifications.	250.00 cu. yd.

c. Masonry block and brick (prices include cost of block or brick, cement, durowall, sand and labor)

1. Concrete block

a.	6" or 8"	1900.00 per 1,000
b.	12"	2300.00 per 1,000

2. 8" Brick — 510.00 per 1,000

d. Steel, reinforcing for concrete structures (includes wire fabric and rebar) — $ .74 per lb.

V. Pipe and fittings (all prices includes installation)

a. Quick coupling 3/4" or 1" — 16.00 each

b. Polyvinyl Chloride (PVC)

1. PVC pipe

a.	up to and including 1 1/2"	1.50 L. ft.
b.	2"	1.65 L. ft.
c.	3"	2.05 L. ft.
d.	4"	2.65 L. ft.
e.	6"	4.60 L. ft.
f.	8"	8.00 L. ft.
g.	10"	12.00 L. ft.
h.	12"	16.00 L. ft.

2. PVC fittings - elbows, tees, caps, etc.

a.	up to and including 3"	3.00 each
b.	4"	6.00 each
c.	6"	20.00 each
d.	8"	65.00 each
e.	10"	100.00 each
f.	12"	135.00 each

c. Corrugated Polyethylene (CPP) (ASTM-F-405, ASTM-F-667)

1. Non-perforated pipe

a.	4" diameter	1.50 L. ft.
b.	5" diameter	1.80 L. ft.
c.	6" diameter	2.00 L. ft.
d.	8" diameter	2.80 L. ft.
e.	10" diameter	3.30 L. ft.

		Average Cost Per Unit
f.	12" diameter	$ 5.50 L. ft.
g.	15" diameter	7.50 L. ft.
h.	18" diameter	9.75 L. ft.
i.	24" diameter	12.05 L. ft.

2. Perforated drainage tubing (all sizes)

a.	no filter material	1.80 L. ft.
b.	with filter cloth wrap	1.85 L. ft.
c.	with gravel filter	2.10 L. ft.

3. CPP fittings - elbows, tees, etc. (use in conjunction with V. c. 1 or V. c. 2)

a.	4 inch	2.75 each
b.	5 inch	3.85 each
c.	6 inch	6.30 each
d.	8 inch	12.85 each
e.	10 inch	17.45 each
f.	12 inch	22.00 each
g.	15 inch	36.65 each
h.	18 inch	73.65 each

4. Pipe outlet system (Hickenbottom or equivalent)

a.	6" Hickenbottom	20.50 each
b.	8" Hickenbottom	34.00 each
c.	10" Hickenbottom	42.50 each
d.	Rock filter (designate elements from Section C, VII)	

Average Cost Per Unit

5. Stormwater conduit - corrugated exterior, smooth interior (Hancor Sure-Lok 10.8 or equivalent) includes gasketed couplers.

a.	12" diameter	$ 6.30 L. ft.
b.	15" diameter	7.70 L. ft.
c.	18" diameter	10.75 L. ft.
d.	24" diameter	14.00 L. ft.

6. Stormwater conduit fittings - elbows, tees etc. (see note under V. c. 5)

a.	12 inch	106.00 each
b.	15 inch	142.00 each
c.	18 inch	191.00 each
d.	24 inch	290.00 each

d. Corrugated Steel (CSP)
(all corrugated steel to be asphalt coated except as otherwise noted, deduct $1.50 per foot for non asphalt coated CSP)

1. CSP with flanged ends (16 gauge)

a.	6" diameter	9.20 L. ft.
b.	8" diameter	10.90 L. ft.
c.	10" diameter	12.85 L. ft.
d.	12" diameter	14.75 L. ft.

2. CSP with re-rolled ends and hugger-type coupling bands (16 gauge)

a.	15" diameter	16.65 L. ft.
b.	18" diameter	18.70 L. ft.
c.	21" diameter	21.30 L. ft.

Average Cost Per Unit

d. 24" diameter $ 24.30 L. ft.

e. 30" diameter 32.45 L. ft.

f. 36" diameter 49.65 L. ft.

e. Corrugated Aluminum (CAP)

1. CAP with flanged ends (16 gauge)

a. 6" diameter $ 8.80 L. ft.

b. 8" diameter 10.50 L. ft.

c. 10" diameter 12.90 L. ft.

d. 12" diameter 13.20 L. ft.

2. CAP with re-rolled ends and hugger-type coupling bands

a. 15" diameter (16 gauge) 11.80 L. ft.

b. 18" diameter (16 gauge) 13.30 L. ft.

c. 21" diameter (16 gauge) 13.65 L. ft.

d. 24" diameter (14 gauge) 19.10 L. ft.

e. 30" diameter (12 gauge) 29.70 L. ft.

f. 36" diameter (12 gauge) 34.65 L. ft.

f. CAP flanged tee with 1 ft. riser stub (length 20 ft.)

a. 6" x 6" x 8" x 20' (16 gauge) 212.50 each

b. 8" x 8" x 12" x 20' (16 gauge) 277.00 each

g. Reinforced concrete pipe - 4' sections

1. 12" diameter 13.00 L. ft.

2. 15" diameter 14.00 L. ft.

3. 18" diameter 16.00 L. ft.

4. 24" diameter 22.00 L. ft.

Average Cost Per Unit

5. 30" diameter $ 28.00 L. ft.

6. 36" diameter 38.00 L. ft.

h. Pipe risers (based on average cost for 12 ft. high riser with 2 ft. outlet stub)

1. CSP risers

a. 8" to 12" diameter (16 gauge) 17.00 L. ft.

b. 15" to 21" diameter (16 gauge) 27.00 L. ft.

c. 24" to 30" diameter (16 gauge) 40.00 L. ft.

d. 36" to 48" diameter (14 gauge) 84.00 L. ft.

e. 54" diameter (12 gauge) 84.00 L. ft.

2 CAP risers

a. 15" to 18" diameter (16 gauge) 28.00 L. ft.

b. 21" to 24" diameter (16 gauge) 42.00 L. ft.

c. 30" to 36" diameter (14 gauge) 67.00 L. ft.

3. For perforated riser add $3.00 per L. ft. to price of riser above.

i. Trash guards

1. For use with PVC, CSP or Steel

a.	12" diameter	37.00	each
b.	15" diameter	63.50	each
d.	24 18" diameter	74.00	each
	diameter	84.50	each
e.	30" diameter	102.00	each
g.	36" diameter	127.00	each
h.	48 42" diameter	207.00	each
	diameter	236.50	each
i.	60" diameter	396.00	each
j.	72" diameter	566.00	each

		Average Cost Per Unit
2.	For use with CAP	
a.	15" diameter	$105.50 each
b.	24" diameter	143.00 each
c.	30" diameter	235.50 each
d.	36" diameter	254.00 each
e.	48" diameter	292.50 each
f.	54" diameter	330.50 each
j.	Anti-seep collars	
1.	For use with PVC 48" x 48"	52.63 each
2.	For use with CSP or steel	
a.	42" x 42" to 48" x 48"	***65.00*** each
b.	***56" x 56"*** to 72" x 72"	145.00 each
c.	78" x 78" to 90" x 90"	360.00 each
3.	For use with CAP	
a.	48" x 48"	108.50 each
b.	72" x 72"	321.00 each
k.	Valves and gates	
1.	For use with PVC	
a.	Shear gate	188.00 each
b.	8" slide gate	454.00 each
c.	10" slide gate	x*
d.	12" slide gate	1200.00 each
e.	Used slide gate	75% of actual cost (Not to exceed 50% of new valve price

*"X" indicates element is not included in area's average cost list and approval to use this element must be given by the Area Office.

		Average Cost	Per Unit
2. For use with metal pipe			
a. Shear gate for aluminum pipe (with 10' of 3/4" Alum. lift rod)		$145.00	each
b. Shear gate for CSP (with 10' frame and stem)			
1. 6" diameter		271.00	each
2. 8" diameter		413.00	each
3. 10" diameter		454.00	each
4. 12" diameter		850.00	each
VI. Pipe drops and surface inlets (installed)			
a. Corrugated aluminum and corrugated steel pipe (designate elements from Section C, V)			
b. Corrugated plastic pipe (designate elements from Section C, V)			
c. Surface inlet with trash grate			
1. Pipe cost (designate elements from Section C, V)			
2. Removable grate			
a. 24"		44.00	each
b. 30"		53.00	each
c. 36"		59.00	each
3. Rock filter (designate elements from Section C, VII)			
d. Face plate (installed)		75.00	each
VII. Stone			
a_ Gravel (all gradations) (includes ABC or crusher run)		12.00	per ton
b. Rock riprap (includes erosion control stone)		20.00	per ton
	or	30.00	cu. yd.

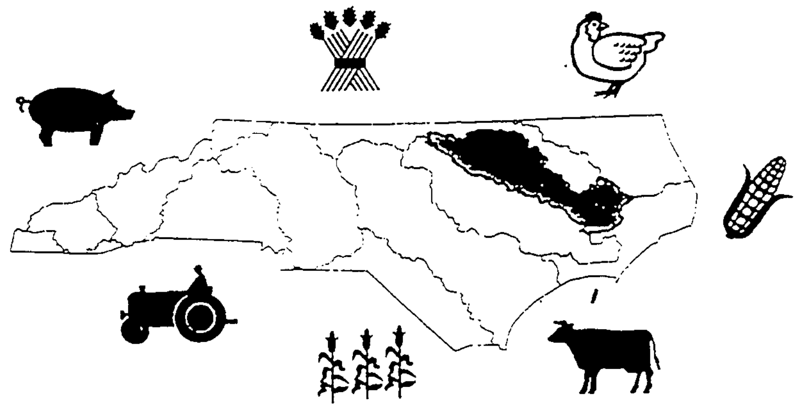
APPENDIX 4.2C

COST EFFECTIVENESS STUDY

COFFEY

RESEARCH TRIANGLE INSTITUTE

Cost-Effectiveness of Agricultural BMPs for Nutrient Reduction in the Tar-Pamlico Basin



Submitted to
The North Carolina Department
of Environment, Health, and
Natural Resources

Prepared by
John P. Tippett
Randall C. Dodd
Center for Environmental Analysis
Research Triangie Institute
Research Triangle Park, NC 27709

January 1995

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January 1995

Approved by

Dennis F. Naugle

Dennis F. Naugle, PhD
Director, Center for Environmental Analysis

Executive Summary

This study was conducted by Research Triangle Institute for the North Carolina Department of Environment, Health, and Natural Resources - Division of Environmental Management. The goal of the study has been to research and develop cost-effectiveness estimates (expressed as $/kilogram of nutrient load reduced) for cost-shared agricultural best management practices (BMPs) in North Carolina's Tar-Pamlico basin.

Specific objectives of the project were to

1. Calculate yearly costs for practices implemented in the Tar-Pa&co basin, incorporating cost-share costs, farmer's contributions, operation and maintenance costs (O&M), area benefit-ted, and practice life expectancies

2. Research the effectiveness of cost-shared agricultural BMPs in reducing surface and subsurface nutrient loads to surface waters relative to typical ***preexisting practices***

3. Where data permit, use the cost and effectiveness information to calculate cost-effectiveness for cost-shared BMPs.

Results

BMP unit costs were calculated for the major cost-shared practices in the Tar-Pamlico basin. These values were based on NC Division of Soil and Water Conservation records and were adjusted to include farmer contributions, O&M costs, area benefitted, and practice life expectancy.

A literature review was conducted to determine the most relevant studies on which to base estimates of BMP effectiveness in the basin. Effectiveness data specific to the Tar-Pamlico basin were available for animal waste management practices and for water control structures. The effectiveness of conservation tillage practices was estimated based on results of the Chesapeake Bay Watershed model for the Southeastern Plains and Middle Atlantic Coastal Plains ecoregions. The effectiveness,@ terracing practices was estimated based on the combined results of two empirical studies in the Chesapeake basin. Vegetated filter strip effectiveness was determined based on two other Chesapeake basin studies that used filter strips of similar size to those cost-shared in the Tar-Pamlico basin. For the remaining practices, only cost data are presented because effectiveness data were not available.

For several practices, cost-effectiveness values are presented as "box and whisker" plots. The range shown in these plots represents the variability in pre-BMP nutrient loading across different sites. The ranges do not capture other sources of variability such as the site-specific variations in BMP cost or effectiveness. Table ES-1 summarizes the cost-effectiveness estimates for practice&or which both cost and effectiveness data were available.

The cost data that are presented in this report represent the direct cost of implementing BMPs. Other "less direct: costs such as (1) opportunity costs from loss of productive land to BMPs and (2) costs of *not implementing* BMPs (e.g., higher fertilizer costs, offsite costs resulting from pollution impacts) are not addressed.

Specific findings of this report include:

- The cost-effectiveness of animal waste management practices is highly dependent upon the preexisting waste management practice on a farm. The range of the cost-effectiveness estimates for any given scenario can be quite wide due to variability in (1) nutrient content of the waste and (2) the crop's fertilization requirement.
- Water control structures are highly cost-effective for nitrogen control, but not for phosphorus control.
- Nutrient management is not cost-shared in the basin, yet it has been shown to be highly cost-effective.
- Relative to other cropland BMPs, conservation tillage is a cost-effective practice for both nitrogen and phosphorus reduction, especially when used in conjunction with nutrient management.
- Relative to other practices, terracing is not cost-effective for either nitrogen or phosphorus reduction.
- Cropland conversion could potentially be very cost-effective, but this depends greatly on site-specific factors.
- Insufficient data exist to estimate the effectiveness (and therefore, cost-effectiveness) of grassed waterways, diversions, and stripcropping.
- Although data are presented by BMP type, it is important to realize that holistic farm management is more cost-effective than single objective BMP cost-sharing.

Based on our findings and literature review, we offer the following suggestions for programmatic direction:

- The Agricultural Cost-Share Program could place a higher priority on nutrient (and particularly, nitrogen) management. Nutrient management has been proven to be a cost-effective strategy for reducing both edge-of-field and watershed loading from agricultural lands.

- Increasing the cost-effectiveness of cost-sharing will require an Increased commitment to education and technical assistance. We have not attempted to quantify the cost-effectiveness of public education programs outside the realm of cost-sharing. However, we feel that enhanced educational efforts can be highly cost-effective and should be given high priority as a means of achieving nutrient reductions goals.

- The Nutrient Trading Program is in a positionto take a proactive approach to restoring and protecting land uses and land cover types that provide positive water quality benefits. The cost-effectiveness of this approach needs to be determined.

This report also includes detailed appendixes discussing the Chowan basin study (upon which the Phase I nutrient trading value was based) and the Chesapeake Bay Watershed Model (from which we drew both effectiveness data and loading factors for selected practices).

Table ES-1
Summary of Nutrient Reduction Cost-Effectiveness Estimates for Cost-Shared Practices in the Tar-Pamlico Basin[1]

Cost-Shared Practice	Pre-existing practice	Portion of Basin	Nutrient	Cost-Effectiveness in Reducing Nutrient Loads to Surface Runoff and Subsurface Drainage (Relative to Preexisting Practice)[2] ($ per kg of nutrient reduced)	
				Using 20-year lagoon life	Using 10-year lagoon life
Anaerobic lagoons	Undersized lagoon with land application at 2x agronomic rate	Whole basin	Nitrogen	$5 to $21	$6 to $29
			Phosphorus	$19 to $298	$26 to $395
	Undersized lagoon with land application at 3x agronomic rate	Whole basin	Nitrogen	$2 to $11	$3 to $14
			Phosphorus	$10 to $158	$13 to $209
	Undersized lagoon with land application at 4x agronomic rate	Whole basin	Nitrogen	$2 to $7	$2 to $9
			Phosphorus	$6 to $108	$9 to $142
	Direct discharge of animal wastes	Whole basin	Nitrogen	$0.02 to $4.14	$0.02 to $5.48
			Phosphorus	$0.03 to $4.00	$0.02 to $5.30
Land application	Land application at 2x agronomic rate	Whole basin	Nitrogen	$0.59 to $4.81	
			Phosphorus	$2.41 to $75.65	
	Land application at 3x agronomic rate	Whole basin	Nitrogen	$0.30 to $2.30	
			Phosphorus	$1.20 to 37.86	
	Land application at 4x agronomic rate	Whole basin	Nitrogen	$0.20 to $1.56	
			Phosphorus	$0.80 to $25.24	
	Direct discharge of lagoon effluent and sludge	Whole basin	Nitrogen	$0.04 to $0.22	
			Phosphorus	$0.05 to $0.25	
	Direct discharge of animal wastes	Whole basin	Nitrogen	$0.01 to $0.06	
			Phosphorus	$0.01 to $0.09	

Table ES-1 (continued)

Cost-Shared Practice	Pre-existing practice	Portion of basin	Nutrient	Cost-Effectiveness in Reducing Nutrient Loads to Surface Runoff and Subsurface Drainage (Relative to Preexisting Practice)[x] ($ per kg of nutrient reduced)				
				Maximum	75th Percentile	Median	25th Percentile	Minimum
Water control structures	Conventional tillage, no BMPs	Ecoregion 63 (lower basin)	Nitrogen	NA	NA	$0.90	NA	NA
		Ecoregion 63 (lower basin)	Phosphorus	NA	NA	**$75.00**	**NA**	**NA**
Conservation tillage	Conventional tillage, no BMPs	Ecoregion 63 (lower basin)	Nitrogen	$8.12	$6.74	$5.88	$5.08	$3.63
		Ecoregion 65 (upper basin)	Nitrogen	$5.61	$4.80	$4.23	$4.03	$3.82
		Ecoregion 63 (lower basin)	Phosphorus	$84.71	$65.54	$62.35	$51.20	$31.51
		Ecoregion 65 (upper basin)	Phosphorus	$56.21	$35.54	$31.63	$25.37	$24.24
Nutrient rnanagetnent	Conventional tillage, no BMPs	Ecoregion 63 (lower basin)	Nitrogen	$5.54	$2.11	$1.58	$1.21	$0.79
		Ecoregion 65 (upper basin)	Nitrogen	$2.23	$2.12	$1.92	$1.78	$1.44
		Ecoregion 63 (lower basin)	Phosphorus	$43.81	$34.26	$23.21	$18.48	$10.85
		Ecoregion 65 (upper basin)	Phosphorus	$24.24	$20.77	$18.59	$16.11	$14.29

Table ES-I (continued)

Cost-Shared Practice	Pre-existing practice	Portion of Basin	Nutrient	Cost-Effectiveness in Reducing Nutrient Loads to Surface Runoff and Subsurface Drainage (Relative to Preexisting Practice)[2] ($ per kg of nutrient reduced)				
				Maximum	75th Percentile	Median	25th Percentile	Minimum
Vegetated filter strips	Conventional tillage, no BMPs	Ecoregion 63 (lower basin)	Nitrogen	$6.97	$6.61	$5.95	$5.72	$4.88
		Ecoregion 65 (upper basin)	Nitrogen	$7.05	$5.75	$5.31	$5.14	$4.73
		Ecoregion 63 (lower basin)	Phosphorus	$101.69	$88.84	$79.47	$61 .01	$55.47
		Ecoregion 65 (upper basin)	Phosphorus	$64.22	$49.83	$45.19	$40.02	$38.13
Terraces	Conventional tillage, no BMPs	Ecoregion 63 (lower basin)	Phosphorus	$129.11	$112.80.	$100.90	$77.47	$70.43
		Ecoregion 63 (lower basin)	Phosphorus	$81.55	$ 63.27	$57.38	$50.81	$48.42

NA = Not applicable

[1] These estimates do not include a safety factor.

[2] Ranges: The spread in the cost-effectiveness ranges for animal waste management practices is primarily due to the variability in nutrient requirements of the crops that receive animal waste application. Crops that require greater levels of manure application are also prone to lose more of the waste nutrients to runoff and subsurface drainage. The less expensive end of the cost-effectiveness range represents crops with high agronomic rates of fertilization (e.g., Bermudagrass). The more expensive end of the range represents crops with tower agronomic rates of fertilization (e.g., small grains). Additionally, the phosphorus cost-effectiveness range for animal waste practices is also drtven by the variability in N:P ratios in land-applied wastes. The agronomic application rate for animal wastes is typically based only on the nitrogen content of the waste. The N:P ratio for different forms of swine and poultry wastes ranges from 1:1 to 3.9:1. This variability is incorporated into the calculations for the phosphorus cost-effectiveness range.

The cost-effectiveness ranges for non-animal waste practices are driien by the range in conventional [illegible] modeling subbasins that are in ecoregions common to the Tar [illegible]mlico basin

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Chapter 1

Introduction

1.1 Background

The Tar-Pamlico Nutrient Trading Program-was adopted in 1989 by the North Carolina Division of Environmental Management as an innovative approach to managing nutrient inputs from both point sources and nonpoint sources. The premise of the program is that cost-sharing for agricultural best management practices is more cost-effective in reducing nutrient loading than controlling nutrients from point sources. The initial phase of the program, which ended in 1994, is currently being evaluated to determine changes warranted for Phase II. During Phase I of the program, formal trading (transfer of funds) has not occurred because point source loadings have not exceeded the basinwide limits. Nevertheless, Phase I has largely been the subject of praise from governmental agencies, dischargers, and environmental groups because of the conceptual and institutional framework that it established.

1.2 Purpose

The goal of this report is to provide accurate and up-to-date info&nation on which decisions can be based concerning nutrient trading payments in Phase II of the Tar-Pamlico Nutrient Trading Program. Specifically, this report presents cost-effectiveness values ($ per kilogram of nutrient load reduced) for cost-shared BMPs in the Tar-Pamlico basin. This study has endeavored to obtain and present the most recent and geographically relevant cost and effectiveness information available. Preference was given to data collected within the Tar-Pamlico basin. When data were not available from within the basin, results were used from studies conducted in similar geographic provinces or ecoregions.

It should be emphasized that the inherent variability associated with both BMP costs and effectiveness introduces substantial uncertainty into the development and use of cost-effectiveness values. It is a goal of this report to fully document the assumptions and limitations associated with the development and use of the cost-effectiveness values that are presented. Chapter 3 discusses specific uncertainties and the resulting need for a safety factor.

1.3 Definition of Terms

In the interest of clarity, we provide the following definitions for terms used in this report:

Unit Cost — The unit cost is the yearly, per hectare (or per ft^3) cost for implementing an agricultural BMP. The unit cost accounts for both state and federal cost-share, farmer contribution, yearly operation and maintenance costs, area benefitted, and practice life expectancy. Unit costs do not account for any cost savings realized by the farmer due to increased efficiency, higher yields, or benefits realized offsite. The unit of measure for unit cost is $/hectare-year.

Effectiveness — Effectiveness is the ability of a practice to reduce nutrient loads (surface and subsurface) entering the stream. Because edge-of-stream data are usually not available, edge-of-field data are often used to estimate effectiveness. Effectiveness can be expressed as either a percent reduction or as a load reduction. For cropland BMPs, we define effectiveness relative to conventional tillage. For animal waste practices, we define effectiveness relative to excess land application or direct discharge. When expressed as a mass reduction, the units for effectiveness are kilograms reduced/hectare-year (or kilograms reduced/f?-year).

Cost-Effectiveness — Cost-effectiveness is a measure of the cost of reducing a unit of nutrient load to the stream. It is calculated by dividing *unit cost* by *effectiveness*. The units for cost-effectiveness are $/kilogram of nutrient load reduced.

Loading Factor — Loading factors are a measure of areal nutrient loading from a tract of land. In this report, loading factors are used to convert percent effectiveness values to load reduction effectiveness. The units for loading factors are kilograms of nutrient/hectare.

Agronomic *Rates* — An agronomic rate of fertilizer application is a rate calculated to meet the crop's needs without overfertilization. Agronomic rates are determined based on manure (fertilizer) analysis, soil nutrient availability, and crop needs. For animal wastes, agronomic rates are typically based only on the nitrogen content of the manure.

Chapter 2

Developing BMP Cost-Effectiveness Values for the Tar-Pamlico Basin

2.1 Interpreting Cost and Effectiveness Data

There is considerable uncertainty involved in estimating cost-effectiveness values for agricultural best management practices. Both the cost and effectiveness of a practice can vary substantially based on a variety of site-specific and management conditions. The estimates presented in the following sections are based on the best available data from studies conducted ih the Tar-Pamlico basin or similar geographic provinces. Before attempting to interpret or apply these data, it is essential to understand the key factors that introduce uncertainty into the analysis. Chapter 3, *The Need For a Safety Factor,* discusses these factors and the resulting need to incorporate a margin of safety.

2.2 Cost and Effectiveness Calculations

Evaluation of cost-effectiveness requires two key elements: cost data (dollars spent) and effectiveness data (percent or mass load reduction). Our review of projects within the Tar-Pamlico basin found that there is a subsfantial database of BMP cost data (NCDSWC, 1994). However, information on BMP effectiveness *within the basin* is available only for water control structures and animal waste management practices. Two studies that are designed to provide BMP effectiveness data on other practices (Chicod Creek and Herrings Run Marsh) will not have sufficient monitoring data for at least another one or two years (M. Cook, B. Towell, 1994). Consequently, some of the BMP effectiveness data for this analysis are drawn from studies outside the Tar-Pamlico basin

2.2.1 Sources of BMP Cost Data

The North Carolina Agricultural Cost-Share Program records BMP implementation data at the county level. The costs presented in this report represent a summary for the 14 counties that make up the basin. Because the basin boundaries do not correspond with county boundaries, some of the data reported in the summary are for land that falls outside the basin. However, it is assumed that the effect of these data on unit costs for each practice is negligible. The data represent BMPs that were implemented during the period from 1985 to 1994. Appendix 3 presents the summary cost data from the North Carolina Agricultural Cost-Share Program

The cost values given in the North Carolina Agricultural Cost-Share summaries include only funds expended by state and federal cost-share programs. Cost-share funds are generally limited to 75% of the total cost of the practice, with the remaining funds being contributed by the farmer. For the purposes of our calculations, we assume that the reported cost-share figures represent 75% of the total cost of the practice. Unless otherwise noted, the sources for cost-related data are as follows:

• Cost-share costs	NCDSW C (1994)
• Practice life expectancies	Soil Conservation Service (SCS) Technical Guide (USDA-SCS,1991)
• Operation and maintenance costs	North Carolina State University (1982)

The costs used in this analysis are the sum of dollars spent between 1985 and 1994, The figures were **not** corrected for inflation and therefore do not represent 1994 dollars. This introduces a level of error into the analysis. Correcting for inflation requires a yearly break down of total cost-share costs, by practice, in the Tar-Pamlico basin. This information is not currently available, but may become available in the future.

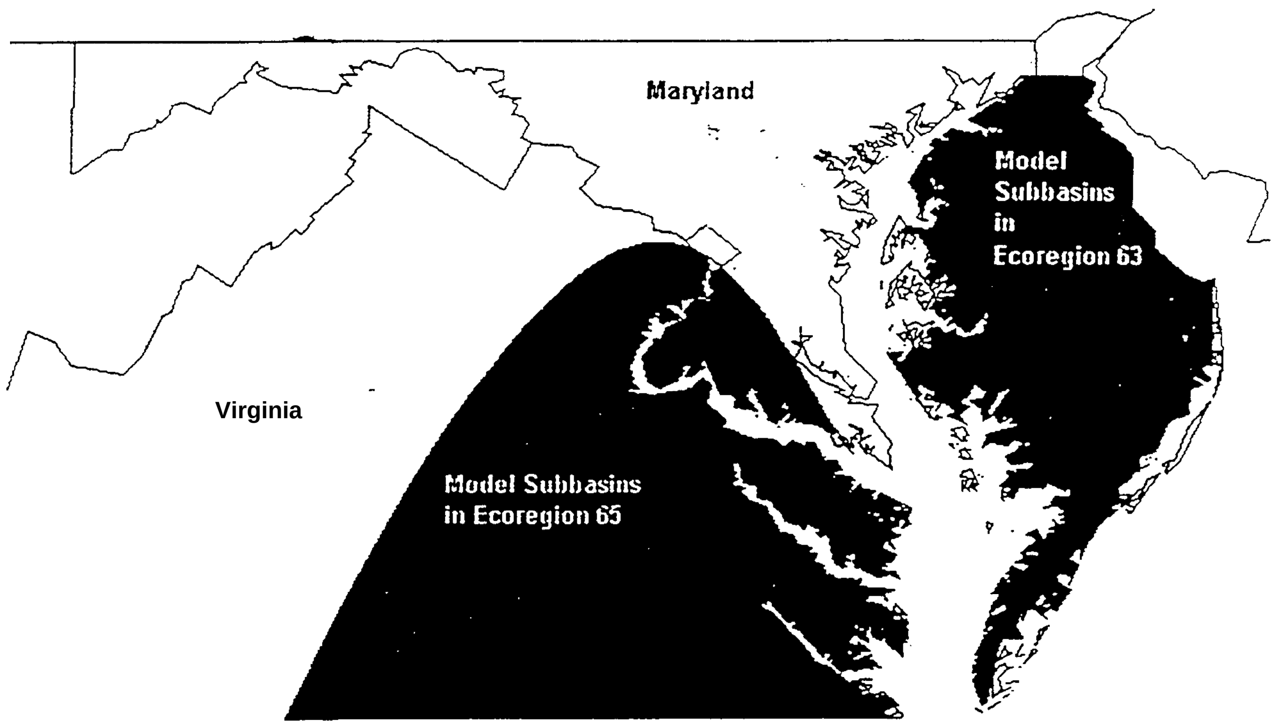
2.2.2 **Sources of BMP Effectiveness** Data

We have used various sources for estimating the nutrient reduction effectiveness of cost-shared BMPs. For each practice, the relevant literature was evaluated and the most geographically relevant and up-to-date studies were chosen as the basis for our estimates.

For **animal waste management practices** and **water control structures,** effectiveness data were available from studies conducted in (or very near to) the Tar-Pamiico basin. These studies were primarily carried out through the North Carolina State University and the North Carolina Cooperative Extension Service.

Studies on **conservation tillage** effectiveness in the Tar-Pamlico basin were not available. We therefore chose to use results from modeling studies conducted in the Chesapeake Bay basin (Casman, 1990; Camacho, 1990; Camacho, 1992). The Tar-Pamlico basin and the Chesapeake basin can differ substantially in terms of important environmental conditions, such as climate, topography and soil type. However; a portion of the Chesapeake basin contains the two ecoregions which comprise the Tar-Pamlico basin (Ecoregion 63, Middle Atlantic Coastal Plain and Ecoregion 65, Southeastern Plains). Ecoregions are areas classified as generally similar-m terms of land surface form, soils, land use, and potential natural vegetation (Omemik, 1986). Twenty-three of the 63 Chesapeake Bay modeling subbasins are located within the Middle Atlantic Coastal Plain and Southeastern Plains ecoregions.

Figure 2-1.
Chesapeake modeling subbasins in ecoregions common to the Tar-Pamlico Basin.



Consequently, we used the results from the Chesapeake Bay Watershed Model from these areas for our estimates of conservation tillage effectiveness (Figure 2-1).

The parameters used in the Chesapeake Bay Watershed Model represent a synthesis of the results of over 30 research studies conducted in the basin. However, Camacho (1990) notes thatthese were still insufficient to accurately characterize nutrient reduction efficiencies in both groundwater and surface waters for some regions of the Chesapeake basin. Consequently, the professional judgment of the modelers played a role in determining the model input parameters. (An overview of the Chesapeake Bay Watershed Model is included in Appendix 2.)

Effectiveness data for **vegetated filter strips** and **terracing** are available from numerous studies; however, only a portion of the studies are applicable to conditions in the Tar-Pamlico basin. Often, these studies reported effectiveness as the percent reduction in surface and subsurface nutrient load. To convert these percentages to mass reductions (kilograms), we used the cropland loading factors from the Chesapeake Bay Watershed Model for ecoregions 63 and 65. The box and whisker plots of effectiveness (and cost-effectiveness) that we present for certain practices are the result of multiplying an estimated effectiveness percentage by the range of loading factors. Consequently, the range of cost-effectiveness values is indicative..of the spatial variability in loading across the 23 watersheds, *but not of the variability associated with BMP cost or effectiveness.* Appendix 4 lists the loading factors for the Chesapeake subbasins in ecoregions 63 and 65. Note that the difference in spread between 'ecoregions seen in the box and whisker plots is primarily due to the different number of loading factors used for each ecoregion. For ecoregion 63, N = 16; for ecoregion 65, N = 7.

For the remaining major cost-share practices **(cropland conversion, grass waterways, diversions, and stripcropping),** the available data were either insufficient or too widely scattered to make justifiable estimates of nutrient reduction effectiveness. For these practices, only cost data are presented.

It is important to note that BMP effectiveness values for different practices are not necessarily additive. For example, a practice installed on conventional tillage may result in a 10 percent net nutrient load decrease. However, the same practice installed on *conservation tillage* will not necessarily yield 10 percent further reduction in the runoff nutrient load.

2.2.3 Cost-Effectiveness Calculation Methods

The approach that we have used to calculate the cost-effectiveness of agricultural management practices is similar in some ways to the original Chowan basin approach described in Appendix 5. The total money spent on management practices is divided by the total mass reductions in order to calculate an effectiveness ratio of the form $/kilogram of nutrient load reduced. However, our approach is also different in several ways:

- We have used the results of recent monitoring and modeling studies to estimate BMP effectiveness. These data did not exist when the Chowan study was conducted.

- While the Chowan study presented a cost-effectiveness value for management practices in general, we have calculated cost-effectiveness for several practice types (where the data permit).

- **We** have used BMP cost data specific to the Tar-Pamlico basin.
- We have accounted for operation and maintenance costs and the SCS life expectancy of each practice.
- We have evaluated effectiveness and cost-effectiveness *relative to typical preexisting practices.*

2.2.3.1 Equations **for Calculating.Cost-Effectiveness**

The following equation was used to estimate yearly per-hectare cost for a practice:

$$\frac{cost}{hectare\text{-}\ year} = \frac{\dfrac{CostShare\ \$ + Farmer\ \$}{Life\ Expectancy} + Yearly\ Operation\ and\ Maintenance\ \$}{Hectares\ Benefitted}$$

The sources for the parameters in the above equation are discussed in Section 2.2.1.

Cost-effectiveness values presented in this report are calculated according to one of two equations. If effectiveness values were reported in terms of kilograms reduced per hectare-year, then the following equation is used:

$$\frac{Cost}{kilogram\ reduced} = \frac{Cost}{hectare\text{-}year} \div \frac{kilograms\ reduced}{hectare\text{-}\ year}$$

The effectiveness of certain in-field practices (e.g., vegetated filter strips) are reported as percent reductions. In these cases, it was necessary to convert the percentages to an estimated load reduction. This process, discussed in Section 2.2.2, uses the following equation:

$$\frac{Cost}{kilogram\ reduced} = \frac{\dfrac{Cost}{hectare\text{-}\ year}}{percent\ reduction\ x\ range\ of\ loading\ factors}$$

2.3 Results

Results are presented for 16 practices that account for 97% of the cost-share funds expended on agricultural BMPs in the Tar-Pamlico basin since the inception of the program in 1985 (Table 2-l). Other BMPs were not addressed because of the historically low level of funding and the lack of adequate cost and/or effectiveness data.

Table 2-1.
Total Cost-Share Expenditures in the Tar-Pamiico Basin for the Period 1984-1994
(NCDSWC, 1994)

Practice	Dollars Spent
Animal Waste Management	$1,757,290
Land Application	$923,821
Anaerobic Lagoons	$750,194
Storage Ponds	$38,203
Composters	$36,210
Dry Stack	$8,862
Grass Waterways	$939,770
Water Control Structures	$523,845.
Field Borders	$452,975
Crop Conversion to Trees	$367,457
Crop Conversion to Grass	$347,116
Diversions	$338,842
Conservation Tillage	$149,997
Terraces	$149,369
Stripcropping	$57,624
Vegetated Filter Strips	$10,363
Nutrient Manaaement	$0

The following discussion presents the available data on cost and effectiveness of these agricultural BMPs in the Tar-Pamlico basin.. For certain practices, sufficient effectiveness studies existed to estimate the cost-effectiveness of the practice. For other practices, only cost data are presented because reliable effectiveness results either did not exist or the effectiveness was so tied to site-specific factors as to make generalizations inappropriate.

2.3.1 Animal Waste Management

In the Tar-Pamlico basin, five main types of animal waste management practices are cost-shared through the North Carolina Agricultural Cost-Share Program. They are (1) anaerobic lagoons, (2) land application of animal waste, (3) animal waste ponds, (4) mortality cornposters, and (5) dry stacking of manure. Of these practices, anaerobic lagoons and land application have received”95 percent of the cost-share funding.

Pre-Existing Conditions: ,A Baseline for Effectiveness Evaluation

To accurately estimate the effectiveness of funds spent on animal waste management, it is essential to establish a baseline condition representing the preexisting waste management practices on a farm. The effectiveness of the cost-share expenditures is the difference between the effectiveness of the baseline and new practices.

In the Tar-Pamlico basin, a majority of the cost-share money spent on animal waste management has gone to land application (Table 2-l). The North Carolina SCS office estimates that *land application at greater than agronomic rates* is the baseline condition on most (>50 percent) of the farms receiving cost-share funding for land application (R. Hansard, 1994). However, it should be noted that there are isolated cases in which the preexisting conditions may be more severe (e.g., discharge to ditches).

The situation for anaerobic lagoons is similar. Cost-sharing for anaerobic lagoons is typically done on farms where the existing lagoon is undersized or does not otherwise meet the SCS technical design specifications (R. Hansard, 1994). When a lagoon is undersized, surface waters are endangered in two ways. Storm events may exceed the lagoon’s holding capacity, causing spillage of wastes to surface waters. More frequently, the lack of sufficient storage volume forces the farmer to land apply wastes at greater than agronomic rates. Therefore, *land application at greater than agronomic rates* is the typical pre-existing management condition for operations which receive cost-share funding for lagoons.

For the purposes of this report, we have evaluated the cost-effectiveness of animal waste management practices relative to various levels of overapplication of wastes. In addition, we have presented cost-effectiveness values relative to direct discharge. Although direct discharge is believed to be a rare occurrence in the basin, the values should be useful for comparison purposes.

2.3.1 .1 Land Application

In the Tar-Pamlico basin, 53 percent of all cost-share funds expended on animal waste management has been for land application and supporting structures_ Overall, land application has accounted for approximately 18 percent of all cost-share funds expended in the basin, making it the highest funded practice (Table Z-l).

When designing an animal waste lagoon, the Soil Conservation Service requires that farm operators have sufficient land suitable for application of the effluent at agronomic rates. By applying lagoon effluent to the land, the farmer utilizes the fertilizer value of the waste while also reducing its pollution potential. The mechanisms of nutrient reduction include plant uptake, denitrification, mineralization, volatilization, and soil adhesion (phosphorus). To encourage land application, the Agricultural Cost-share Program provides incentive payments for the use of spreader trucks, honey wagons, and irrigation systems. To encourage proper nutrient management when applying wastes, all requests for payment must be accompanied by a copy of the waste analysis used to determine the application rate (USDA SCS, 1991). In addition to incentive payments, cost-share funds are also available for "solid set" sprinkler application systems and hydrants, which serve as connection points for traveling guns or other means of land application (B. Towell, 1994). Table 2-2 presents cost data for land application in the Tar-Pamlico basin. In addition to the costs directly associated with land application, the table includes the costs of dry stack structures, which serve to facilitate the proper management of a land application program (see Section 2.3.1 .5).

Table 2-2.
Land Application *Cost* Data

Area Benefited[1]	Cost-Share Total (dry stack, solid set, and hydrants)	Farmer Contribution	Incentive Payments[2]	Annual Operation & Maintenance	Life Expectancy	Unit Cost
13,961 ha	$235,025	$78,342	5688,796	10% of base cost	20 years (structures) 1 year (incentive payment)	$53 / ha-year

[1] From NC Division of Soil and Water Conservation (1994).

[2] Land application incentive payments range from $2 to $6 per 1,000 gallons of liquid or $4 per ton of solid manure. Payments decrease to $1 per 1,000 gallons for second- and third-year pumping. Incentive payments are not provided for longer than 3 years.

Effectiveness of Land Application Relative to Typical Preexisting Conditions

In the Tar-Pamlico basin, the most common baseline condition for cost-sharing of land application is *the application of wastes at greater than agronomic rates* (Section 2.3.1). The net effectiveness of cost-sharing for land application, therefore, is the reduction in surface and subsurface nutrient loading which results from reducing land application to agronomic rates. Table 2-3 summarizes the literature results for studies that have addressed nutrient runoff and sub-surface drainage from lands receiving wastes at various rates.

Of the studies summarized. in Table 2-3, the most relevant for our purposes is Evans et al. (1984), which was conducted very near to the Tar-Pamlico basin. The Evans et al. study is also the only geographically relevant study that has attempted to quantify **both** surface and subsurface losses from land-applied animal wastes. (Although other studies have examined these losses, they were not considered appropriate for this analysis due to substantial differences in either soil type, climate, or topography). Evans et al. reported that approximately 13 percent of land-applied nitrogen was lost via surface and sub-surface drainage when wastes were applied at agronomic rates. However, a mass balance was unable to account for 28% of applied nitrogen. The authors suggest that part of this deficit may have been due to N displacement "in the [soil] profile beyond the sampling zone by subsurface flow." Consequently, the actual subsurface loss may have been somewhat larger. Based on this uncertainty, we feel that a reasonable estimate of surface + subsurface N losses for wastes applied at agronomic rates would be 20% of applied N (80% effectiveness).

Table 2-3.
Surface **and Subsurface Losses from Land Application of Animal Wastes**

Land Application Rate	% Loss to Surface Runoff	% Loss to Subsurface Drainage	Surface Loss + Subsurface Drainage	Waste Type/ Crop Type/ Soil Type	Reference
Agronomic Rates	NA	5.3% NO_3 1.2% TP	NA	Swine lagoon effluent/ coastal bermudagrass/ sandy loam	Westerman et al., 1983
Agronomic Rates	1.4%[2] TN 2.5% TP	11.6%[1] TN 1.7% TP	13.2%[1] TN 4.2% TP	As above	Evans et al. 1984
Agronomic Rates	4.2% TN 2.4% TP	NA	NA	Poultry manure / fescue / red clayey soil	McLeod and Hegg, 1984
1.4X Agronomic Rates	TN 3.1% (8.7%)[2] TP 2.8% (6.4%)[2]	NA	NA	Poultry manure slurry/ fescue/ silt Loam	Edwards and Daniel, 1992
2X Agronomic Rates	NA	8.6% NO_3 0.6% TP	NA	Swine lagoon effluent/ coastal bermudagrass/ sandy loam	Westerman et al., 1983
2X Agronomic Rates	1.4%[3] TN 1.9% TP	11.0%[4] TN 0.7%[4] TP	12.4%[4] 2.6%[4]	As above	Evans et al., 1984
4X Agronomic Rates	NA	8.6% NO_3 1.2% TP	NA	As above	Westerman et al., 1983
4X Agronomic Rates	1.7%[5] TN 2.6% TP	15.6%[5] TN 1.9% TP	17.3%[5] TN 4.5% TP	As above	Evans et al, 1984
5X Agronomic Rates	TN 3.3 % (7.1 %)[2] TP 2.2% (4.9%)[2]	NA	NA	Poultry Manure Slurry/ Fescue/ Silt Loam	Edwards and Daniel. 1992

NA = Not applicable
[1] The mass balance was unable to account **for** 28 percent **of applied** N.
[2] **Low value is for rainfall intensity of 5** cm/ha; **high value is for 10 cm/ha.**
[3] **The mass balance was unable to account for 41 percent of applied N and 24 percent of applied P.**
[4] **Estimated, due to equipment failure.**
[5] **The mass balance was unable to account for 64 percent of applied N and 35 percent of applied P.**

In **all** the studies, increased application rates (greater than the agronomic rate) resulted in increased losses to surface runoff and subsurface drainage. However, when losses are measured as a percent of total applied, the values tend to fall within a fairly narrow range. For nitrogen, the range for combined losses is 12.4 percent to 17.3 percent. Based on these limited data, it is difficult to determine if the percentage loss of nitrogen and phosphorus changes significantly with increasing application rates. Because the percent loss does not appear to vary substantially by loading rate, and there is no apparent trend in the data, we will assume that percent loss is not significantly different for land application at rates up to ~~4x the~~ agronomic rate. Our estimate of combined losses at agronomic rates (20 percent N) is reasonably close to the highest reported combined N loss from excess application (17.3 percent). Given the mass balance uncertainties discussed above, we believe that 20 percent is a reasonable estimate of N loss (to runoff and sub-surface drainage) for land application at rates up to 4x the agronomic rate.

For phosphorus, Evans et al. were able to account for all of the applied P when wastes were applied at agronomic rates. At higher rates, up to 35 percent of applied P was unaccounted for in the mass balance. The range of reported combined losses for P is 2.6 percent to 4.5 percent. The surface and subsurface losses reported by the other studies are also in fairly close agreement. It is interesting to note that the surface losses (percentage) from land-applied swine waste are nearly the same as from poultry waste. This is despite the difference in N:P ratios between the two wastes. Rounding off the high end of the range, we feel that 5 percent is a reasonable estimate of surface and subsurface phosphorus losses for land application of wastes at up to 4 times the agronomic rate.

Based on the above estimates, Table 2-4 presents estimates of mass nutrient losses (to surface runoff and sub-surface drainage) which can be expected at 2, 3, and 4 times the agronomic application rate. Because the agronomic application rate varies by crop type, these values are reported as ranges that encompass the extent of agronomic rates for crops in the Tar-Pamlico basin.

Table 2-4.
Estimates of Surface and Subsurface Nutrient Loss from Land Application

Land Application Rate[1]			% Loss to Surface Runoff and Sub-Surface Drainage	Mass Loss to Surface Runoff and Sub-Surface Drainage
Agronomic Rates	Nitrogen	56 to 448 kg/ha-year	20%	11.2 to 89.6 kg / ha-year
	Phosphorus	14.3 to 448 kg/ha-year	5%	0.7 to 22.4 kg / ha-year
2x Agronomic Rate	Nitrogen	112 to 896 kg/ha-year	20%	22.4 to 179.2 kg / ha-year
	Phosphorus	29 to 896 kg/ha-year	5%	1.5 to 44.8 kg / ha-year
3x Agronomic Rate	Nitrogen	168 to 1,344 kg/ha-year	20%	33.6 to 268.8 kg / ha-year
	Phosphorus	43 to 1,344 kg/ha-year	5%	2.2 to 67.2 kg / ha-year
4x Agronomic Rate	Nitrogen	224 to 1,793 kg/ha-year	20%	44.8 to 358.6 kg / ha-year
	Phosphorus	57 to 1793 kg/ha-year	5%	2.9 to 89.7 kg / ha-year

[1] **Agronomic application rates of animal wastes are typically based on the nitrogen content of the waste (J Barker, 1994). The phosphorus content of the waste will vary by animal type and by the level of pre-treatment** (eg. **fresh vs. lagoon). The N:P ratio in the waste may also vary from** 1:1 **up to** 3.9:1 (Zublena **et al.,** 1990a, **Zublena et al.** 1990b). **The ranges presented for phosphorus account for this** variability.

Based on the mass loss estimates in Table 2-4 and the cost estimate in Table 2-2, we present estimated cost-effectiveness values for land application cost-sharing in Table 2-5.

Table 2-5.
Cost-Effectiveness of Agronomic Land Application Relative to Excess Application

Before Lagoon Cost-Sharing:[1] Land Application at:	After Lagoon Cost-Sharing:[1] Land Application at:	Mass Reduction in Surface Runoff and Subsurface Drainage[2]	Unit Cost of Land Applic.[3]	Cost-Effectiveness ($/kg reduced)
2x Agronomic Rates N loss = 22 - 179 kg/ha-yr P loss = 1.5 to 44.8 kg/ha-yr	Agronomic Rates N loss = 11 - 90 kg/ha-yr P loss = 0.71 - 22.4 kg/ha-yr	N: 11 - 90 kg/ha-yr P: 0.71 - 22.4 kg/ha-yr	$53 / ha-yr	N: $0.59 - $4.81 P: $2.37 - $74.64
3x Agronomic Rates N loss = 34 - 269 kg/ha-yr P loss = 2.2 to 67.2 kg/ha-yr		N: 23 - 179 kg/ha-yr P: 1.5 - 44.8 kg/ha-yr		N: $0.30 - $2.30 P: $1.18 - $35.33
4x Agronomic Rates N loss = 45 - 359 kg/ha-yr P loss = 2.9 to 89.7 kg/ha-yr		N: 34 - 269 kg/ha-yr P: 2.2 - 67.3 kg/ha-yr		N: $0.20 - $1.56 P: $0.79 - $24.09

[1] **From Table 2-4.**
[2] **(N losses before cost-sharing) - (N losses after cost-sharing).**
[3] **From Table 2-2.**

Effectiveness of Land Application Relative to Direct Discharge

The direct discharge of animal wastes is prohibited by North Carolina State law; and it **is unlikely that direct** discharges are occurring to any significant extent in the Tar-Pamlico basin. However, the potential water quality impact of any direct discharger may be exceedingly large. Therefore, we have also calculated cost-effectiveness values for land application relative to a preexisting condition of "direct discharge".

Table 2-6 presents effectiveness values for this scenario, which are based on a database of manure and lagoon constituent analysis maintained by the North Carolina Agricultural Extension Service (1994). As discussed in the previous section, a 20 percent loss (80 percent effectiveness) is assumed for nitrogen and a 5 percent loss (95 percent effectiveness) is assumed for phosphorus. The results are also broken down according to whether the discharged wastes were fresh or from a lagoon. Because the nutrient load (per animal) is much greater for fresh wastes, the effectiveness values for fresh manure are higher. The calculation details for this table are presented in Appendix 6.

Table 2-6.
Effectiveness of Land Application Relative to Direct Discharge

Animal Type	Effectiveness (% reduction)	Effectiveness (load reduction) if the discharged waste was *manure slurry*	Effectiveness (load reduction) if the discharged waste was *lagoon liquid + sludge*
Swine	80% N 95% P	96 kg TKN /head-year 28 kg P_2O_5 /head-year	25 kg TKN /head-year 7 kg P_2O_5 / head-year
Poultry	80% N 95% P	0.46 kg TKN / head-year 0.45 kg P_2O_5 / head-year	0.10 kg TKN / head-year 0.21 kg P_2O_5 / head-year

Using the cost figure from Table 2-2 and the effectiveness estimates from Table 2-6, we can now estimate the cost-effectiveness of land application in reducing nutrient loads (relative to direct discharge). This process **is** complicated by the fact that we have measured the effectiveness of land application in terms of nutrient reduction *per head of animal* while the cost estimates are measured per *hectare of applied waste.* The following equation was used to perform the appropriate conversions.

$$\frac{\$}{\textit{kg reduced}} = \frac{\$}{\textit{hectare of land application-year}} \div \frac{\textit{kg reduced}}{\textit{head-year}} \times \frac{\textit{hectares}}{\textit{head}}$$

The parameters for the above equation vary by animal type and by the nutrient needs of the receiving crop. Table 2-7 summarizes these parameters and presents the resulting ranges of estimated cost-effectiveness values. Calculation details are presented in Appendix 6.

Table 2-7.
Cost-Effectiveness of Land Application Relative to Direct Discharge

Animal Waste Type	Effectiveness[1]	Cost[2]	Land needed per animal[3]	Cost-Effectiveness
Swine manure slurry	96 kg TKN /head-year 87 kg P_2O_5 / head-year	$53 /ha-year	0.027 to 0.11 ha / sow	$0.01 to $0.06 / kg TKN[4] $0.02 to $0.09 / kg P_2O_5
Swine lagoon liquid + sludge	25 kg TKN / head-year 27 kg P_2O_5 / head-year			$0.05 to $0.22 / kg TKN $0.05 to $0.25 / kg P_2O_5
Poultry manure slurry	0.46 kg TKN /head-year 0.45 kg P_2O_5 / head-year		0.085 to 0.35 ha / 1000 birds	$0.01 to $0.04 / kg TKN $0.01 to $0.04 / kg P_2O_5
Poultry Lagoon liquid + sludge	0.10 kg TKN / head-year 0.21 kg P_2O_5 / head-year			$0.04 to $0.18 / kg TKN $0.02 to $0.09 / kg P_2O_5

[1] From Table 2-6.
[2] From Table 2-2.
[3] These figures cover the range of agronomic rates **for land application.** From Zublena et al. (1990a) **and** Zublena et al. (1990b).
[4] TKN = **organic nitrogen + ammonia nitrogen**

2.3.1.2 Anaerobic Lagoons

Anaerobic lagoons account for 43 percent of the cost-share money expended on animal waste management in the Tar-Pamlico basin. Lagoons use bacteria to convert an animal waste slurry into a liquid that can be spread onto the land as fertilizer or used as flush water for a recycle cleaning system. in an anaerobic lagoon, facultative and anaerobic bacteria degrade organic matter, producing organic acids, methane, and carbon dioxide (Merker, 1981). The process can greatly decrease the BOD, total and volatile solids, and nutrient concentration in the effluent. in the Tar-Pamlico basin, anaerobic lagoons are primarily used to treat swine and poultry (layer) waste (NCDSWC, 1994).

It is important to note that lagoons are only one part of an effective animal waste management system. In areas of excess rainfall (such as the Tar-Pamlico basin), lagoons will overflow unless they are regularly pumped out (Humenik et al., 1980). Although lagoons can be highly effective in reducing nutrient concentrations, the effluent nutrient concentrations are still very high and must be applied to growing crops at agronomic rates. In the Tar-Pamlico basin, anaerobic lagoons play an important role because farms often have limited land area suitable for waste application. Lagoons decrease the nutrient load in the waste, effectively decreasing the area required for land application. This, in turn, allows the farmer to raise greater numbers of animals. Table 2-8 presents the cost data for anaerobic lagoons in the Tar-Pamlico basin.

Table 2-8.
Anaerobic Lagoon Cost Data

Lagoon Volume[1]	Cost-Share Total	Farmer Contribution	Annual Operation & Maintenance	Life Expectancy	Unit Cost
18,004,662 cubic feet	$750,194	$250,065	10% of base cost	20 years	$0.00831 ft^3 - year
18,004,662 cubic feet	$750,194	$250,065	10% of base cost	10 years'	$0.01 1/ ft^3- year

[1] Based on SCS cost data of $1.50 per cubic yard.
[2] Although the SCS life expectancy for the practice is 20 years, calculations based on a 1 O-year value are presented for comparison purposes.

Lagoon Effectiveness Relative to Typical Preexisting Conditions

As discussed earlier, the key to determining the effectiveness of animal waste cost share monies is to know the preexisting animal waste 'management practice on a farm. The effectiveness of the cost-share monies is the difference between the effectiveness of the enhanced, cost-shared practice and the pre-existing practice.

In the Tar-Pamfico basin, the Soil Conservation Service estimates that most farms which receive lagoon cost-share funds typically have a lagoon which is undersized and/or not up to SCS technical design specifications (R. Hansard, 1994). When lagoons are undersized, the farmers must typically land apply their wastes at levels greater than agronomic fates. Therefore, we have used *land application at greater than agronomic rates* as the typical baseline condition for cost-shared lagoons in the Tar-Pamlico basin.

After the lagoon is installed, the operator can begin to carry out a land application program at agronomic rates.. Consequently, the pre-and post-bmp scenarios for lagoon cost-sharing are effectively the same as those used in the analysis of land application in Section 2.3.1 .1. In Section 2.3.1.1, we estimated the reduction in surface and subsurface load associated with switching from excess application (at various rates) to land application at agronomic rates. These figures were presented in Table 2-5. *Because the pre- and post-BMP scenarios for land application and lagoon cost- sharing are the same,* the mass reduction estimates in Table 2-5 also represent the nutrient reductions that can be expected from a typical cost-shared lagoon in the Tar-Pamlico basin (via the change from over-application of wastes to application at agronomic rates).

Using the lagoon cost figure in Table 2-8 and the effectiveness estimates in Table 2-4, we can now estimate the cost-effectiveness of lagoon cost-sharing relative to nutrient losses from excess application of wastes. This process is complicated by the fact that we have measured the effectiveness of lagoons in terms of nutrient reduction *per* hectare *of land application* while we have measured lagoon costs in dollars *per cubic foot of lagoon.* It is therefore necessary to convert the cost figures to be based on equivalent area of land application. The following equation was used to convert the cost figures:

$$\frac{\$}{hectare} = \frac{\$}{ft^3 \text{ of lagoon - year}} \times \frac{ft^3 \text{ of lagoon}}{\text{number of animals}} \times \frac{\text{number of animals}}{hectare}$$

As would be expected, the parameters in the above equation are different for swine and poultry. Table 2-9 summarizes these figures and the per- hectare cost results for lagoons. Calculation details are presented in Appendix 6.

Table 2-9.
Conversion of Lagoon Cost Figures

Lagoon Type	Unit Cost $ / ft^3-yr	ft^3/animal	Animals / Hectare	Cost / Hectare-Yr of applied waste
Poultry (layer)	$0.0083 [1] ($0.011)[4]	10 [2]	2,840 - 11,766 [3]	$236 - $977 ($312 to 1,294)[4]
Swine	$0.0083' GO.01 1)[4]	1417 [2]	9 - 37 [3]	$105 - $435 ($105 to $435)[4]

[1] From Table 2-8
[2] Based on J. Barker (1994) and R. Hansard (1994). For swine, based on farrow to finish operation, per sow. Includes boar and 9-pig litter.
[3] Based on Zublena et al. (1990a), Zublena et al. (1990b). This range is due to the variable nutrient requirements of crops receiving animal wastes.
[4] The numbers in parentheses are presented for comparison purposes and are based on a 10 year lagoon life span instead of the SCS estimated life span of 20 years.

Using the cost data in Table 2-9 and the effectiveness data in Table 2-4, we present estimated ranges for cost-effectiveness of anaerobic lagoon cost sharing in Table 2-l 0. Calculation details are presented in Appendix 6.

Lagoon Effectiveness Relative to Direct Discharge

The direct discharge of animal wastes is prohibited by North Carolina State law. It is unlikely that direct discharges are occurring to any significant extent in the Tar-Pamlico basin. However, the potential water quality impact of any direct discharger may be exceedingly large. Therefore, we have calculated cost-effectiveness values for anaerobic lagoons relative to a pre-existing condition of "direct discharge". It is reasonable to assume that under such a scenario, the operator will also begin to implement land application of the lagoon effluent. Therefore, the post-BMP scenario for this practice **is land application at agronomic rates.**

The effectiveness of anaerobic lagoons in reducing nutrient concentrations in wastes is directly linked to the size and loading rate of the lagoon. The SCS requires that all anaerobic swine waste lagoons in North Carolina have a minimum treatment volume of 1 ft^3 per pound of steady-state live weight (R. Hansard, 1994). The North Carolina Cooperative Extension Service maintains

Table 2-10.
Cost-Effectiveness of Anaerobic Lagoons Relative to Land Application at Excess Rates

Before Lagoon Cost-Sharing:[1] Land Application at:	After Lagoon Cost-Sharing:[1] Land Application at:	N Reduction in Surface Runoff and Subsurface Drainage[2]	Cost per Ha of Applied Waste[3]	Cost-Effectiveness ($/kg reduced)[5]
2x Agronomic Rates N loss = 22 - 179 kg/ha-yr P loss = 1.5 to 44.8 kg/ha-yr	Agronomic Rates N loss = 11 - 90 kg/ha-year P loss = 0.71 - 22 kg/ha-year	N: 11 - 90 kg/ha-yr P: 0.79- 22 kg/ha-yr	Costs range from $105 to $977/ha-yr.	N: $4.86 - 21.05 ($6.44 - $28.86)' P: $19.43- 5298.30 ($25.76 - $394.94)
3x Agronomic Rates N loss = 34 - 269 kg/ha-yr P loss = 2.2 to 67.2 kg/ha-yr		N: 22 - 179 kg/ha-yr P: 1.5 - 45 kg/ha-yr		N: $2.43 - $10.52 ($322 - $13.93) P: $9.71 - $158.20 ($12.88 - $209.40)
4x Agronomic Rates N loss = 45 - 359 kg/ha-yr P loss = 2.9 to 89.7 kg/ha-yr		N: 34 - 269 kg/ha-yr P: 22 - 67 kg/ha-yr		N: $1.62 - $7.02 ($2.14 - $9.29) P: $6.47 - $107.63 ($8.57 - 142.47)

1 From Table 2-4.
2 (losses before cost-sharing) - (losses after cost-sharing).
3 From Table 2-9.
4 The numbers in parentheses are presented for comparison purposes and are based on a 10 year life span instead of the SCS life span of 20 years.
5 See Appendix 6 for calculation details.

a database of manure, lagoon sludge, and lagoon liquid characterization from North Carolina farms (North Carolina Agricultural Extension Service, 1990). Based on these data, the averagenitrogen (TKN) reduction for a swine waste lagoon with 1 ft^3 of treatment volume per pound of live weight is 73 percent or 0.374 lb N per thousand pounds live weight per day. For total phosphate (P_2O_5), the reduction is 69 percent or 0.272 lb P_2O_5 per thousand pounds of live weight per day.

For poultry waste lagoons, the SCS guidelines require 2.5 ft^3 of anaerobic lagoon treatment volume per pound of live weight (R. Hansard, 1994). Based on the NC Extension data mentioned above, the nitrogen (TKN) reduction for this type of lagoon is 77 percent or 0.658 lb N per thousand pounds of live bird weight per day. For total phosphate (P_2O_5) the reduction is 54 or 0.392 lb N per thousand pounds of live bird weight per day.

In Section 2.3.1.2, we estimated the effectiveness (load reduction per head) associated with conversion from direct discharge of wastes to land application at agronomic rates (Table 2-6). The load reduction depended greatly upon whether that waste being discharged was manure slurry or lagoon effluent. The major difference between the two waste types is that the overall nutrient content of fresh manure (per head) is much higher than that of the lagoon liquid and sludge. Therefore, the mass reduction achieved by land application of the fresh manure is higher as well.

For the direct discharge scenario, the calculations for lagoon **effectiveness** (not cost-effectiveness) are the same as those for land application. This is because the pre- and post- scenarios are the same. Before the lagoon cost-sharing, wastes are discharged directly to the stream. After the cost-sharing, lagoon liquid and sludge are land-applied at agronomic rates. We assume that after the lagoon is cost-shared, the operator will land-apply the effluent at agronomic rates without additional cost-share funds for land application. The effectiveness values from Table 2-6 are presented again for lagoon cost-sharing in Table 2-l l .

Table 2-11.
Effectiveness of Lagoon Cost-Sharing Relative to Direct Discharge[1]

Animal Type	Effectiveness (% reduction)	Effectiveness (Load Reduction)	
		If discharged waste was manure slurry	If discharged waste was lagoon liquid + sludge
Swine	80% N reduction 95% P reduction	96 kg TKN / head-year 87 kg P_2O_5 / head-year	25 kg TKN / head-year 26 kg P_2O_5 / head-year
Poultry	80% N reduction 95% P reduction	0.46 kg TKN / head-year 0.45 kg P_2O_5 / head-year	0.10 kg TKN / head-year 0.21 kg P_2O_5 / head-year

[1] Assuming land application at agronomic rat& with no additional cost-sharing for land application. See Appendix 6 for calculation details.

Because the cost figures are based on different units than the effectiveness estimates, units had to be converted based on the range of agronomic land application rates. The following equation was used to perform the conversions:

$$\frac{\$}{\textit{kg reduced}} = \frac{\$}{\textit{hectares of land application} - \textit{year}} \div \frac{\textit{kg reduced}}{\textit{head-year}} \times \frac{\textit{hectares}}{\textit{head}}$$

The parameters for the above equation vary by animal type and by the nutrient needs of the receiving crop. Table 2-12 summarizes these parameters and presents the resulting ranges of estimated cost-effectiveness values.

Table 2-12.
Cost-Effectiveness of Anaerobic Lagoons Relative to Direct Discharge

Animal Waste Type	Effectiveness[1]	Cost[2]	Land needed per animal[3]	Cost-Effectiveness[4]
Swine manure slurry	96 kg N / head-year 87 kg P_2O_5 / head-year	Swine: $105 to $435 / ha ($140 to $577)[4]	0.027 to 0.11 ha / sow	50.03 to $1.12 / kg N ($0.03 to $1.48 / kg N) SO.03 to $1.23 / kg P_2O_5 ($0.04 to $1.63 / kg P_2O_5)
Swine lagoon liquid + sludge	25 kg N / head-year 26 kg P_2O_5 / head-year			$0.11 to $4.14 / kg N ($0.12 to $5.48 / kg N) $0.10 to $4.00 / kg P_2O_5 ($0.12 to $5.30 / kg P_2O_5)
Poultry manure slurry	0.46 kg N / head-year 0.45 kg P_2O_5 / head-year	Poultry: $236 to $977/ ha ($312 to $1,294)'	0.085 to 0.35 ha / 1000 birds	$0.02 to $0.74 I kg N ($0.02 to $0.98 / kg N) $0.02 to $0.75 /kg P_2O_5 ($0.02 to $1.00 / kg P_2O_5)
Poultry Lagoon liquid + sludge	0.10 kg N / head-year 0.21 kg P_2O_5 / head-year			$0.08 to $324 / kg N ($0.10 to $4.30 / kg N) $0.04 to $1.65 / kg P_2O_5 ($0.05 to $2.19 / kg P_2O_5)

[1] From Table 2-11.
[2] From Table 2-9.
[3] These figures cover the range of agronomic rates for land application. From Zublena et al. (1990).
[4] The numbers in parentheses are presented for comparison purposes and are based on a 1 O-year life span instead of the SCS life span of 20 years.

2.3.1.3 Animal Waste Ponds

Animal waste ponds are typically smaller than treatment lagoons and have a much higher loading rate. In contrast to treatment lagoons, the purpose of an animal waste pond is to store the waste while maintaining as *much of the nutrient content as possible* (J. Barker, 1994). Consequently, animal waste ponds are typically used on farms where the fertilizer need is greater than the nutrient load from the on-farm animal waste. Animal waste ponds also have historically received low levels of cost-share funding in the Tar-Pamlico basin (Table 2-l). It appears that, in general, animal waste ponds are not an effective or commonly used practice for decreasing nutrient loads to surface waters. We therefore will **not** present estimated cost-effectiveness values for animal waste ponds.

2.3.1.4 Composting of Poultry Mortality

Heat, disease outbreaks, and other factors routinely cause poultry mortality in large operations. Growers have the option of incinerating or burying the carcasses, sending them to a rendering plant, or composting them on site. Due to water quality and health concerns related to high water tables, growers east of I-95 are being strongly discouraged from burying their poultry carcasses (B. Towell, 1994). The composting alternative is becoming more attractive because growers can avoid the costs of rendering services and also obtain a usable nutrient source. Poultry are typically composted in bins with alternating layers of poultry fitter and a carbon source, such as peanut hulls. The composting process can eliminate the disease hazard of the carcasses and provide a valuable fertilizer as well.

In the lower half of the Tar-Pamlico basin, where poultry production is greatest, approximately 75 - 90% of the producers have their poultry mortality sent to a rendering plant (J. Parsons, 1994). The use of a rendering plant for mortality disposal results in the complete removal of the birds as potential nutrient sources. Composters would be an effective tool for reducing nutrient loads only if a less effective practice, such as pit burial, was being practiced. Even in these situations, the buried birds would be of greater concern from a public health standpoint than from a nutrient loading standpoint. Because the large majority of producers in the basin use rendering plants, there appears to be relatively few situations where mortality cornposting could provide any level of nutrient load reduction. We therefore conclude that cost-sharing for poultry mortality composters is not an effective method for reducing nutrient loads to surface waters in the Tar-Pamlico basin.

2.3.1.5 Dry Stacking

In the Tar-Pamlico basin, dry stacking is primarily used to store turkey and broiler house litter before land application. The purpose of a dry stack structure is to safely store animal wastes until they can be land-applied at agronomic rates. *In and of itself, the dty stack structure does not promote any further nutrient reductions than would occur if the waste were still lying on the floor of the confinement house* (W.F. Ritter, 1994). (The exception being in situations where manure was previously stored open to rainfall and runoff.) The primary benefit of a dry stack is that it allows the farmer the storage flexibility to carry out a proper land application/nutrient management plan. Without a storage structure, the farmer may be forced to land-apply when the risk of surface water pollution is great, such as during the dormant season or before a rainy period. However, the presence of a good storage structure does not ensure that the farmer will use proper nutrient management practices. In fact, if the farmer is not diligent, storage structures, such as dry stacks, could lead to situations with increased potential for water quality degradation. Casman (1990) gives the following examples:

(1) A storage structure may promote proper or improper timing of fertilizer application, depending on its capacity, and this, of course, will affect nutrient losses.

Manure storage capacity is typically measured by the number of days the herd's manure can be stored before the structure is filled. For instance, a structure with 90 days' capacity is filled and must be emptied four times a year. Depending on the fields available to the farmer and the crop rotations, up to 3 of these 4 loads'of manure could be wasted (not applied to the fields on which the major summer crops are grown).

(2) The presence of a properly sized storagestructure does not guarantee water quality protection.

Structures which retain solids from runoff and release liquids essentially change nonpoint sources into point sources. Also structures which are not emptied on schedule could force the farmer into spreading manure during times when the fields are likely to produce excessive runoff pollution.

Since the benefit of a dry stack structure is realized primarily when used in conjunction with a proper land application program, we will not attempt to evaluate the cost-effectiveness of the structures by themselves. Rather, we have included the costs of dry stack structures within the analysis of cost-effectiveness of land application (Section 2.3.1.2).

2.3.2 Water Control Structures

When properly designed and managed, the practice of water table management has been shown to be an effective tool for improving drainage water quality in North Carolina's eastern coastal plain (Deal et al., 1986; Evans et al., 1988; Gilliam et al. 1978). One specific water table management practice, controlled drainage, has been designated as a BMP for soils with improved drainage. Using controlled drainage, water control structures called flashboard risers are installed in drainage canals. Weirs can be placed in the riser to control the elevation of water in the canal. During the growing season, water in the canal is maintained at high levels to reduce the threat of drought stress. The water in the canals is released for planting and harvesting (Chesheir et al., 1990). Several field studies (Gilliam et al., 1978; Doty et al., 1982; and Evans et al., 1989) have shown that, under proper management, controlled drainage is highly effective in reducing pollutant outflow from agricultural fields.

A total of 421 cost-shared water control structures have been installed in the Tar-Pamlico basin at a total cost of $523,843 or $1,244 per unit (on average). The NC Cooperative Extension Service estimated that as of July 1, 1989 more than 2,500 control structures had been installed to provide controlled drainage on approximately 150,000 acres (60,704 ha) in eastern North Carolina. Based on these data; an average water control structure serves about 24 hectares. This is fairly close to the North Carolina SCS estimate of 0.013 unit/acre, which converts to 31 hectares per water control structure (USDA Soil Conservation Service, 1991). As a conservative estimate, we have used 24 hectares/unit for our calculations.

Table 2-13. Water Control Structures Cost Data

Area Benefitted[1]	Cost-Share Total	Farmer Contribution	Annual O & M	Life Expectancy	Unit Cost
10,104 ha	$ 523,845	$174,615	3% of base cost	10 years	$ 9/ha-yr

[1] Assuming 24 hectares benefitted per structure.

Evans et al. (1991) summarize the nutrient transport reductions from 14 studies of controlled drainage in eastern North Carolina. Representing 125 site-years of data, these studies give some of the most detailed effectiveness data for any practice in the Tar-Pamlico basin. From the results of these studies, it is apparent that controlled drainage can significantly reduce nutrient load to the stream. The reduction in nutrient load is attributable to decreased outflow, increased crop uptake (as evidenced through increased yields), denitrification, and sedimentation (phosphorus). Table 2-14 presents the average results of these studies, modified from Evans et al. (1991).

Table 2-14. Effectiveness and Cost-Effectiveness of Water Control Structures

Average Reduction in Drainage Outflow (kg/ha-year)		Cost-Effectiveness ($/kg)	
N	P	N	P
10.0	0.12	$ 0.90	$ 75.00

Using the cost data from Table 2-13 and the effectiveness data as summarized in Evans et al. (1991), we have calculated cost-effectiveness values for water control structures in eastern North Carolina (Table 2-14). The relatively high unit cost for phosphorusis due to the fact that controlled drainage is not highly effective for phosphorus removal. Controlled drainage tends to decrease phosphorus concentrations in systems where the majority of the outflow is via surface runoff. However, on systems where the outflow is primarily through the soil profile, controlled drainage has the opposite effect (Evans et al., 1991)

2.3.3 Grassed Waterways and Diversions

Little data exist on the nutrient reduction effectiveness of grassed waterways and diversions. Consequently, we can only present a narrative discussion of the potential nutrient reduction effectiveness for these practices. However, there is some evidence that, taken together as part of a "farm plan," these practices can afford a level of nutrient load reduction to the stream. Camacho (1990) presented a summary of literature values for **surface water reduction efficiencies.** Practices with farm plan measures showed an increased reduction efficiency over practices without farm-plan components for both nitrogen and phosphorus. However, these results were for surface water studies only and included other practices in addition to grass waterways and diversions. Due to the lack of data on both surface and subsurface losses, it was not possible to calculate cost-effectiveness values for these practices.

Grassed Waterways

Grassed waterways are used to provide a stable outlet for field water and to stabilize ditches and reduce gullying. Research studies have indicated that, under conditions of channelized flow, which are prevalent in grassed waterways, nutrient removal efficiencies will approach zero. In addition, sediment-bound nutrients deposited in the waterway tend to wash back into the water (Casman, 1990).

Diversions

Diversrons are in-field practices designed to prevent rill **and gully formation by diverting surface flow off of the field. Because diversions are essentially a method of** channeling **water off of a field, it is unlikely that they have much of an effect in reducing dissolved nutrient loads to the stream. However, by reducing length of slope and subsequent soil loss, diversions** can help retain soil-associated phosphorus on the field.

Table 2-15 presents cost data for grassed waterways and diversions in the Tar-**Pamlico basin.**

Table 2-15. Cost Data for Grassed Waterways and Diversions

Practice	Area Benefitted	Cost-Share	Farmer Contribution	Annual O & M	Life Expectancy	Unit Cost
Grassed waterway	**4766 ha'**	**$939,770**	**$313,256**	**5% of base cost**	**10 years**	$39/ha-yr
Diversions	840 ha[2]	$338,842	$112,947	5% of base cost	10 years	$81/ha-yr

[1] **Assuming 0.06 acres of waterway per acre of** cropland **(USDA SCS, 1991).**
[2] **Assuming 200' of diversion per acre of** cropland **(USDA SCS, 1991).**

2.3.4 Cropland Conversion to Trees / Grass

Crop conversion to trees or grass has historically been used to remove highly **erodible land** (HEL) from production. This practice **is** typically implemented via the Conservation Reserve Program (CRP) whereby farmers receive cost-share funds to take land out of production and plant grass or trees. By taking land out of cultivation and establishing year-round cover, soil loss from the site can be decreased dramatically. Additionally, runoff nutrient loads would also be expected to decrease due to discontinued fertilizer application and uptake of nutrients by plants. Depending on the location of the site relative to other nutrient source areas, converted land may also-effectively reduce nutrient loadings from upland areas. Table 2-16 presents cost data for cropland conversion practices in the Tar-Pamlico basin.

Table 2-16. Cost Data for Cropland Conversion

Practice	Area Converted	Cost-Share	Farmer Contribution	Annual O & M	Life Expectancy	Unit Cost
Crop conversion to grass	1,200 ha	$347.116	$115,705	3 % of base cost	10 years	$50/ha-yr
Crop conversion to trees	1,833 ha	$367,457	$ 122,486	3% of base cost	10 years	$35/ha-yr

The effectiveness of cropland conversion in reducing nutrient loads to the stream is highly dependent on a variety of site specific factors. First is the location of the converted area in relation to the cropped land. if situated at the bottom of a slope, between the cropland and the stream, the converted area can potentially function as a filter strip. However, if the converted area is upslope of the rest of the cropland, little filtering of runoff will be achieved. Size and shape of the converted area also play a role. If the converted area constitutes a substaptial portion of the cropland in a given field, nutrient reductions would obviously be higher than those where only a small portion was converted. Similarly, if the converted area is of the shape to allow dispersed overland flow, it could be much more effective as a filter strip than one that promotes concentrated flow. Another key factor is the pre-conversion use of the field. If the converted area was previously planted in row crops with high runoff and intensive fertilization, the reduction obtained by conversion could be quite high. On the other hand, if the area was previously planted in closely-planted small grains or hay, the reduction may not be as substantial. A final consideration is the type of vegetation which grows after conversion. Over a long period of time, trees are generally more effective than grass in sequestering nutrients and encouraging key processes such as denitrification (National Research Council, 1993).

Any of the factors discussed above has the ability to dramatically affect the effectiveness of cropland conversion in reducing nutrient loads to the stream. Each these factors is highly site-specific and generalizations for the Tar-Pamlico basin cannot be made with currently available data. It appears that effectiveness of cropland conversion could range from essentially zero, to perhaps 90% (the upper bound for a vegetated filter strip).

2.3.5 Conservation Tillage

Conservation tillage is defined as any tillage or planting system that leaves at least 30% of the soil surface covered with crop residue after planting. This includes practices such as no-till, and "reduced tillage" practices such as ridge-till, strip-till, coulter, chisel, and mulch-till (Casman et al., 1989). Conservation tillage has been well-accepted by the agricultural community, largely due to its success in reducing production costs, increasing'yields, conserving moisture, and maintaining the long-term productivity of soils (Heatwole, et al.. 1991).

Table 2-17 presents the cost data for conservation tillage in the Tar-Pamlico basin. The funds expended on conservation tillage via the cost-share program are incentive payments and do not reflect the actual costs of installing the practice. incentive payments are paid yearly for up to three years. it is hoped that during the three year incentive period, the farmers will see the benefits of conservation tillage and continue to use the practice on their own.

Table 2-17. Cost Data for Conservation Tillage

Area Benefitted	Incentive Payments	Life Expectancy	Unit Cost $/ha-yr
6.070 ha	149.997	1 year	$ 25

There are many site-specific factors that influence the effectiveness of conservation tillage. These include soil properties, surface slope, the previous crop, the amount of residue removed, fertilizer variables (placement, type, quantity, and timing), harvesting practice, variety of crop, planter style, orientation of contour, and the local climate. Not surprisingly, the reduction efficiencies reported for groundwater and surface water N and P loads vary tremendously (Casman, 1990). Figure 2-2 summarizes the effectiveness values taken from runs of the Chesapeake Bay Watershed Model in ecoregions 63 and 65. Note that the number of model runs is different for each ecoregion (Appendix 4). Figure 2-3 presents cost-effectiveness ranges forconservation tillage based on the Tar-Pamlico cost data and the effectiveness data from the Chesapeake Bay Watershed model runs for ecoregions 63 and 65.

Figure 2-2. Conservation tillage effectiveness.

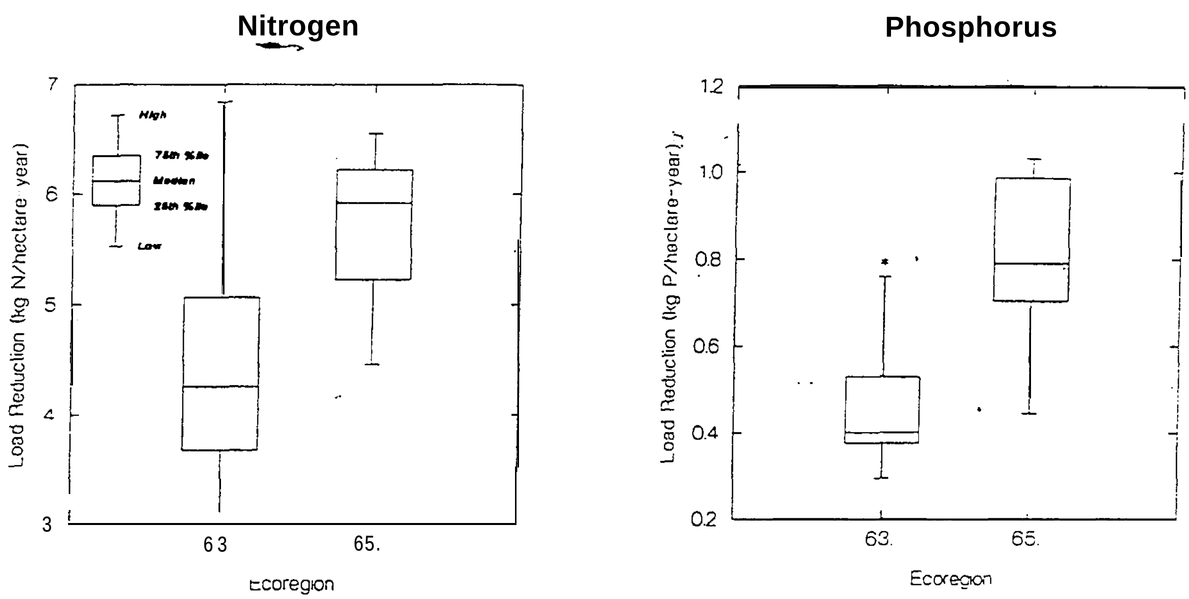
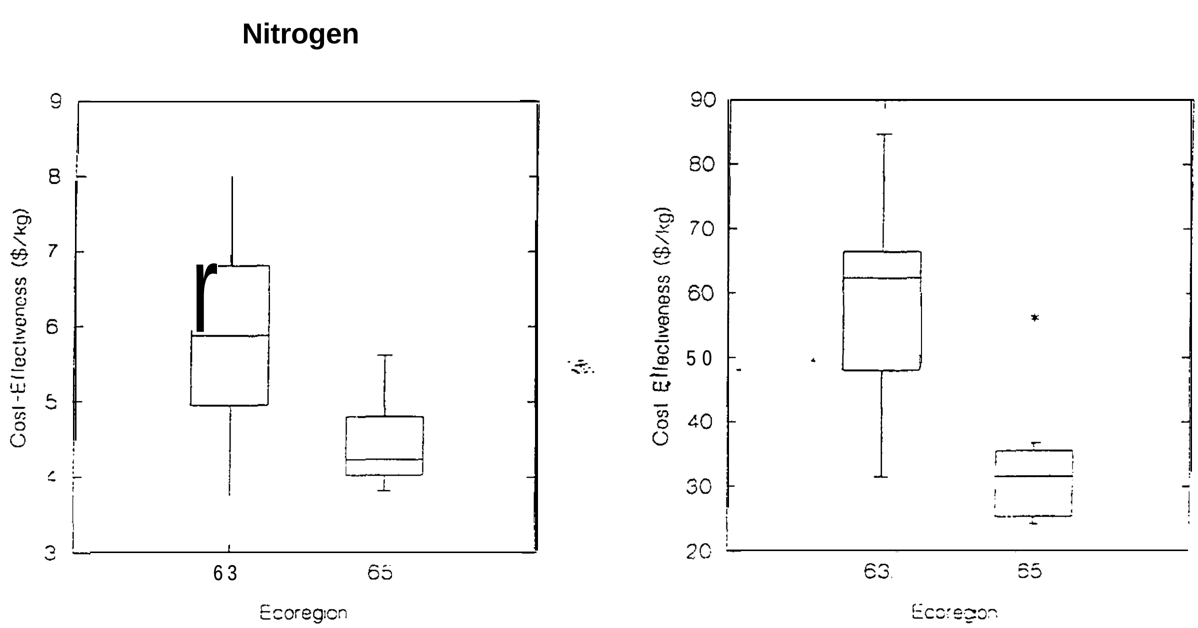


Figure 2-3. Conservation tillage cost-effectiveness.



2.3.6 Terraces

Terraces are a series of regularly spaced embankments across a slope that form a channel on the upslope side of the embankment. In North Carolina, terraces are used in concert with contour rows to act as a check for contour row failure (R. Hansard, 1994). Although they have been shown to be very effective in reducing nutrient and sediment runoff, terraces are also quite expensive to install and maintain. Also, terraces can fail during large storms if the water in a terrace overtops the embankment. A resulting cascade effect can lead to serious downslope erosion (Heatwole et al., 1991). Terraces reduce nutrient runoff by temporarily storing water, allowing sediment deposition and water infiltration. Table 2-l 8 presents cost data for terraces in the Tar-Pamlico basin.

Table 2-18. Cost Data for **Terracing**

Area Benefitted[1]	Cost-Share	Farmer Contribution	Annual O & M	Life Expectancy	Unit Cost
505 ha	$149,369	$49,709	5% of base cost	10 years	$59/ha-yr

[1] Assuming 400 ft of terrace per acre (USDA Soil Conservation Service , 1991)

Regional effectiveness data for terraces are extremely scant. Langdale et al. (1985) studied phosphorus losses from conventional till corn (winter fallow) converted to terraces in the southern Piedmont. He reported an annual total phosphorus reduction efficiency of 66 percent. However, this was for surface runoff only. Ellis et al. (1985) studied partitioning of nutrients between surface and subsurface flow. Casman (1990) used Ellis' partitioning results to estimate net (surface and subsurface) reductions based on the results of Langdale. The results are presented in Table 2-19. Casman noted that, although no studies of total nitrogen reduction by terracing were found, it **is** reasonable to assume that nitrogen is mainly shunted from surface water to subsurface water. In mineral soils with low moisture content, it would be reasonable to assume that nearly all of the subsurface nitrogen load is eventually delivered to the stream. In organic soils with sufficient moisture to create anaerobic conditions, denitrification could substantially reduce nitrogen loads (Welsch, 1991). However, in the Tar-Pamlico basin, terracing is used primarily in the Piedmont, which is dominated by mineral soils. Consequently, we assume that terracing in the Tar-Pamlico basin is not effective in reducing nitrogen loads

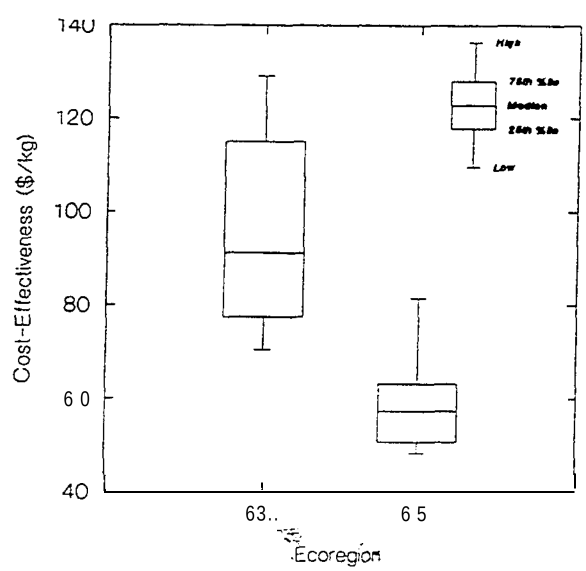
Table 2-19. Effectiveness Estimates for Terracing

Percent Reduction	
Nitrogen	Phosphorus
0 %	34 %

Using the cost and effectiveness data presented in Tables 2-l 8 and 2-19, cost-effectiveness values were estimated for terracing (with winter cover) in the Tar-Pamiico basin (Figure 2-4). Casman's percent reduction figures were converted to load reductions using conventional tillage loading factors from the Chesapeake Bay Model for ecoregions 63 arid.65

Figure 2-4. Terracing cost-effectiveness

Phosphorus



2.3.7 **Vegetated Filter Strips / Field Borders / Stripcropping**

Vegetated filter strips (VFSs) are bands of vegetation placed between pollutant source areas and receiving waters to remove suspended sediments and other pollutants from surface runoff. The major mechanisms for nutrient removal in VFSs are (1) plant uptake and (2) soil adsorption /binding. The filter strip decreases flow speed, enhancing the opportunity for infiltration, sediment deposition, and absorption of soluble nutrients. Field borders are bands of vegetation planted at the field edge. The primary difference between field borders and VFSs is that field borders are commonly used to create a stabilized pathway for equipment movement on the farm. Although no effectiveness data were found for field borders, it is likely that nutrient removal is inhibited due to soil compaction from farm equipment travel.

Stripcropping is the practice of planting a field in alternating strips of grass and crop. The area planted with the crop is equal to the area planted with grass. This practice serves several purposes. It can reduce total soil loss from a field by reducing the amount of cropland. The grass strips can slow down surface runoff, allowing for greater water infiltration, uptake of dissolved nutrients, and filtration/deposition of sediment particles. The strips also improve soil structure by providing organic matter and resisting the formation of crusts which promote runoff and erosion (R. Hansard, 1994). Although we have found little information on the effectiveness of stripcropping, it does support many of the functions provided by vegetated filter strips and may have a similar effectiveness. Table 2-20 presents cost data for VFSs, field borders, and stripcropping in the Tar-Pamlico Basin.

Table 2-20. Cost Data **for VFSs, Field Borders, and Stripcropping**

Practice	Area Benefitted	Cost-Share	Farmer Contribution	Annual O & M	Life Expectancy	Unit Cost
Vegetated Filter Strips	65 ha [1]	$ 10,363	$ 3,454	5% of base cost	7 years	$41/ha-yr
Field Borders	20,010 ha [2]	$ 452,975	$ 150,991	5% of base cost [3]	7 years	$6/ha-yr
Stripcropping	339 ha [4]	$ 57,624	$ 19,208	1% of base cost	5 years	$48/ha-yr

[1] **Assuming 175' x 25' filter strip per acre (USDA-SCS, 1991).**
[2] **Assuming 175' x 25' field border per acre (USDA-SCS, 1991).**
[3] **O&M costs were not available for field borders. We assume** O&M **costs to be** equal to those for vegetated **filter strips.**
[4] **Assuming** 50 **percent conversion to grass** (USDA-SCS, 1991).

Numerous research studies on the effectiveness of VFSs have been conducted, with highly variable results. Some of the key observations common to multiple studies include (Casman, 1990):

- Concentrated flow conditions render VFSs ineffective for sediment and nutrient removal.
- The effectiveness of a VFS decreases over time as sediment accumulates unless the vegetation within it can grow as fast as it is being buried.
- VFSs are more effective. at removing suspended solids, than nutrients.
- Numerous site-specific factors influence VFS efficiency, including filter length, depth of flow, slope, cross slope, soil type, infiuent characteristics, clogging, and hydraulic loading rate.

Much of the variabiiity in the reported values for VFS effectiveness was due to (1) inconsistent methods in defining inputs, and (2) strip design (lengths from 15 feet to 2 miles, slopes from 2 percent to 16 percent). Dillaha et al. (1988) and Magette et al. (I 987) used filter strips of sizes that are typical for cost-share practices (15 to 30 ft). Based on these two studies, Casman (1990) estimated that about 30 percent of the influent total nitrogen is retained by a VFS (surface and subsurface retention)_ For phosphorus, the only applicable study is Schwer and Clausen (1989) who presented a subsurface phosphorus retention value of 90 percent. Casman (1990) noted that this value is probably too high and that a more appropriate estimate for phosphorus removal is somewhere between 30 percent and 90 percent. Effectiveness estimates are summarized in Table 2-21.

Table 2-21. Effectiveness data for VFSs, Field Borders, and Stripcropping

Practice	Percent Reduction	
	Nitrogen	Phosphorus
Vegetated filter strips	~30%	30% - 90%
Field borders	No effectiveness data available.	
Stripcropping		

Figure 2-5. Vegetated filter strip cost-effectiveness.

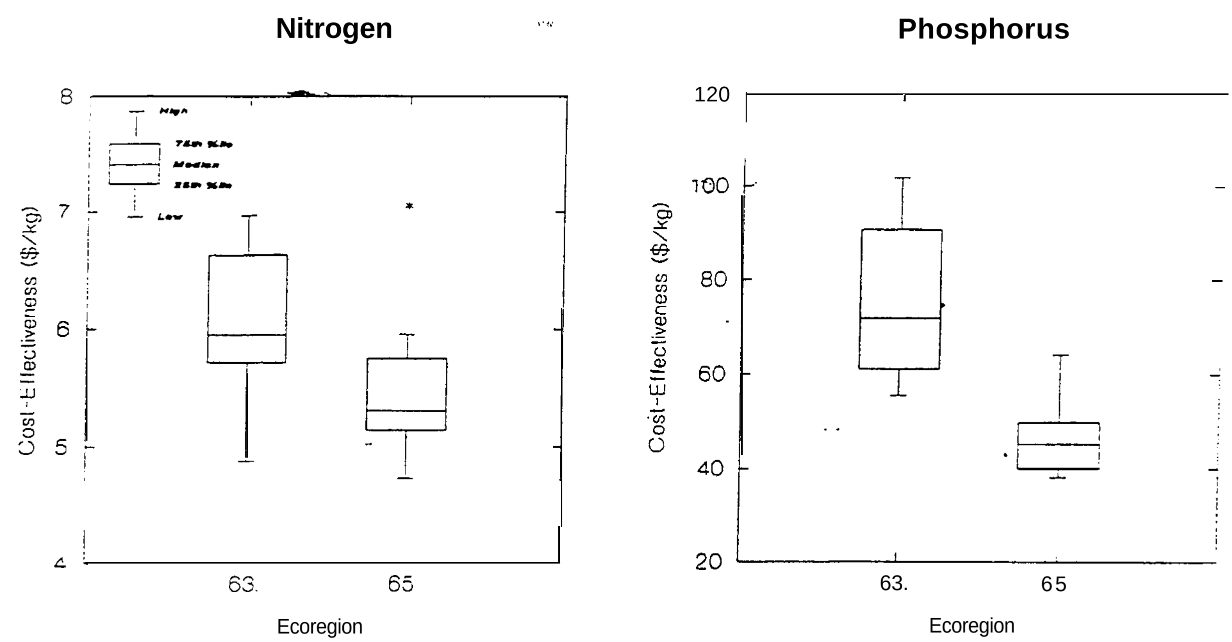


Figure 2-5 presents estimated edge-of-stream nutrient reduction Lost-effectiveness values for vegetated filter strips in the Tar-Pamlico basin. Cost-effectiveness values were calculated using loading factors from the Chesapeake Bay Watershed Model for ecoregions 63 and 65 and a 30 percent reduction effectiveness for both N and P. The lower end of the P effectiveness range was used as a conservative estimate. We believe this to be appropriate in light of research suggesting that the effectiveness of VFSs declines over time and that, without intensive maintenance, VFSs are prone to clogging and/or concentrated flow (Heatwole, 1991).

2.3.8 Nutrient Management

Nutnent management is the practice of modifying fertilizer usage based on recommendations regarding optimum rates, timing, and methods for nutrient application. These recommendations are typically based on soil and manure analysis and expected crop yields (Camacho, 1992). **The** goal is to provide only what is needed to grow the crop.

Several important observations have been made regarding the effectiveness of nutrient management practices in regard to water quality (Reckhow, 1980 as reported in Casman, 1990):

- Time of application is more important than fertilizer type.

- Incorporation of fertilizer after application reduces export by runoff.

- The type of crop (non-row versus row crops) also influences nutrient loss to runoff (row crops substantially higher).

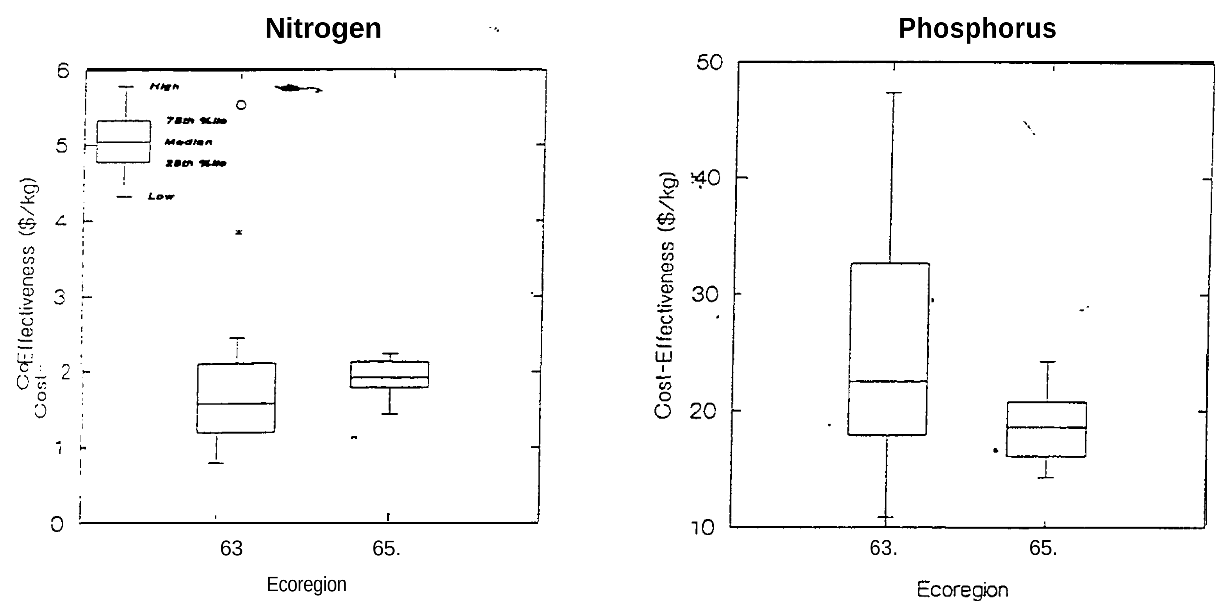
- Excessive or insufficient fertilization can increase nutrient losses to runoff .

From the standpoint of instream nuttiest load reduction, nutrient management is a highly cost-effective practice. A primary reason for its effectiveness is that nutrient management prevents the introduction of excess nutrients into the system. Most other in-field practices depend on re-routing the excess nutrients through the soil to achieve only partially effective adsorption or transformation. It should be noted, however, that nutrient management is most effective in concert with practices that increase infiltration. This allows more of the applied fertilizer to reach the root zone for plant uptake (R. Hansard, 1994). Compared to structural practices alone, nutrient management is also relatively inexpensive to implement. The primary expenses are for soil tests and the development of a nutrient management plan. Camacho's (1992) analysis of Chesapeake Bay Watershed Model results presents nutrient management as the most cost-effective practice in use.

In an analysis of the literature on nutrient management, Casman (1990) concluded that the efficiency of nutrient management depends primarily on the pre-BMP degree of over-fertilization. This efficiency is roughly equal to the percent fertilizer reduction recommended by the nutrient management plan. Consequently, the load reduction associated with nutrient management will be highly site-specific, depending on the individual farmers fertilization practices. See Chapter 4 for further discussion of nutrient management.

Nutrient management is not a formally cost-shared practice in North Carolina. Farmers who use nutrient management in the basin do so on their own, or with advice from the NCSU Cooperative Extension Service or private contractors. The Soil Conservation Service also assists in nutrient management for organic sources via the planning and writing of animal waste management plans. Since cost-sharing for nutrient management could be one of the most cost-effective means for reducing instream nutrient loads in the basin, we are presenting cost-effectiveness values for this practice. The best available effectiveness information is again from the Chesapeake Bay Watershed Model results. As we did for conservation tillage, we have taken the results of the model runs for the portion of the Chesapeake basin in ecoregions 63 and 65. Because cost data are not available, we have used cost data from the Chesapeake basin. Camacho (1992) reports $2.4/acre-year ($5.93/hectare-year) as a typical cost for nutrient management plan development. The resulting cost-effectiveness values for nutrient management on cropland are summarized in Figure 2-6.

Figure 2-6. Nutrient management cost-effectiveness.



2.4 Summary

Table 2-22 and Figure 2-7 summarize the cost-effectiveness ranges for best management practices in the Tar-Pamlico basin. As seen in the table, the effectiveness of animal waste management practices is highly variable. The major factors causing this variability are (1) the fertilization requirement of the crop receiving the waste, and (2) the preexisting waste management condition. Among cropland practices, water control structures are the most cost-effective for nitrogen removal, but are not effective for phosphorus removal. Although not cost-shared in the basin, nutrient management is also a highly cost-effective practice.

Table 2-22.
Summary of Nutrient Reduction Cost-Effectiveness Estimates for Cost-Shared Practices in the Tar-Pamlico Basin'

Cost-Shared Practice	Pre-existing practice	Portion of Basin	Nutrient	Cost-Effectiveness in Reducing Nutrient Loads to Surface Runoff and Subsurface Drainage (Relative to Preexisting Practice)[2] ($ per kg of nutrient reduced)	
				Using 20-year lagoon life span	Using 10-year lagoon life span
Anaerobic lagoons	Undersized lagoon with land application at 2x agronomic rate	Whole basin	Nitrogen	$5 to $21	$6 to $29
			Phosphorus	$19 to $298	$26 to $395
	Undersized lagoon with land application at 3x agronomic rate	Whole basin	Nitrogen	$2 to $11	$3 to $14
			Phosphorus	$10 to $158	$13 to $209
	Undersized lagoon with land application at 4x agronomic rate	Whole basin	Nitrogen	$2 to $7	$2 to $9
			Phosphorus	$6 to $108	$9 to $142
	Direct discharge of animal wastes	Whole basin	Nitrogen	$0.02 to $4.14	$0.02 to $5.48
			Phosphorus	$0.03 to $4.00	$0.02 to $5.30
Land Application	Land application at 2x agronomic rate	Whole basin	Nitrogen	$0.59 to $4.81	
			Phosphorus	$2.41 to $75.65	
	Land application al 3x agronomic rate	Whole basin	Nitrogen	$0.30 to $2.30	
			Phosphorus	$1.20 to 37.86	
	Land application at 4x agronomic rate	Whole basin	Nitrogen	$0.20 to $1.56	
			Phosphorus	$0.80 to $25.24	
	Direct discharge of lagoon effluent and sludge	Whole basin	Nitrogen	$0.04 to $0.22	
			Phosphorus	$0.05 to $0.25	
	Direct discharge of animal wastes	Whole basin	Nitrogen	$0.01 to $0.06	
			Phosphorus	$0.01 to $0.09	

Table 2-22 (continued)

Cost-Shared Practice	Pre-existing practice	Portion of Basin	Nutrient	Cost-Effectiveness in Reducing Nutrient Loads to Surface Runoff and Subsurface Drainage (Relative to Pre-Existing Practice)[2] ($ per kg of nutrient reduced)				
				Maximum	75th Percentile	Median	25th Percentile	Minimum
Water control structures	Conventional tillage, no BMPs	Ecoregion 63 (lower basin)	Nitrogen	NA	NA	$0.90	NA	NA
		Ecoregion 63 (lower basin)	Phosphorus	NA	NA	$75.00	NA	NA
Conservation tillage	Conventional tillage, no BMPs	Ecoregion 63 (lower basin)	Nitrogen	$8.12	$6.74	$5.88	$5.08	$3.63
		Ecoregion 65 (upper basin)	Nitrogen	$5.61	$4.80	$4.23	$4.03	$3.82
		Ecoregion 63 (lower basin)	Phosphorus	$84.71	$65.54	$62.35	$51.20	$31.51
		Ecoregion 65 (upper basin)	Phosphorus	$56.21	$35.54	$3163	$25.37	$24.24
Nutrient management	Conventional tillage, no BMPs	Ecoregion 63 (lower basin)	Nitrogen	$5.54	$2.11	$1.58	$1.21	$0.79
		Ecoregion 65 (upper basin)	Nitrogen	$2.23	$2.12	$1.92	$1.78	$1.44
		Ecoregion 63 (lower basin)	Phosphorus	$43.81	$34.26	$23.21	$18.48	$10.85
		Ecoregion 65 (upper basin)	Phosphorus	$24.24	$20.77	$18.59	$16.11	$14.29

Table 2-22 (continued)

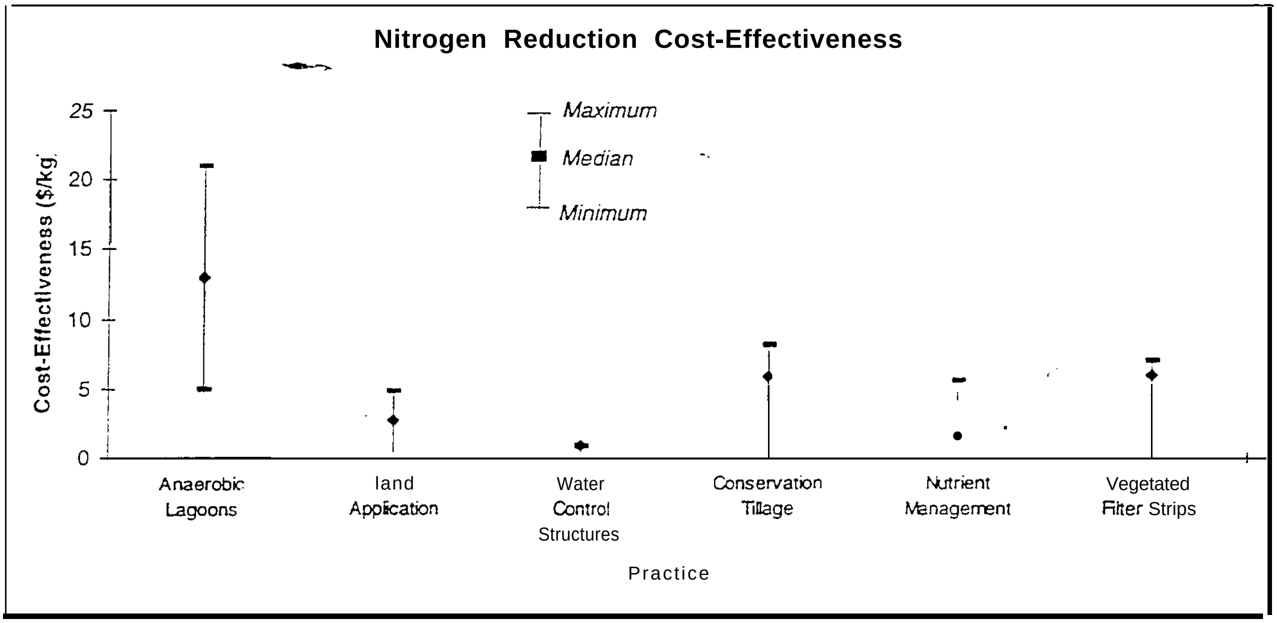
Cost-Shared Practice	Pre-existing practice	Portion of Basin	Nutrient	Cost-Effectiveness in Reducing Nutrient Loads to Surface Runoff and Subsurface Drainage (Relative to Preexisting Practice)[2] ($ per kg of nutrient reduced)[1]				
				Maximum	75th Percentile	Median	25th Percentile	Minimum
Vegetated filter strips	Conventional tillage, no BMPs	Ecoregion 63 (lower basin)	Nitrogen	$6.97	$6.61	$5.95	$5.72	$4.88
		Ecoregion 65 (upper basin)	Nitrogen	$7.05	$5.75	$5.31	$5.14	$4.73
		Ecoregion 63 (lower basin)	Phosphorus	$101.69	$88.84	$79.47	$61.01	$55.47
		Ecoregion 65 (upper basin)	Phosphorus	$64.22	$49.83	$45.19	$40.02	$38.13
Terraces	Conventional tillage, no BMPs	Ecoregion 63 (lower basin)	Phosphorus	$129.11	$112.80	$100.90	$77.47	$70.43
		Ecoregion 63 (lower basin)	Phosphorus	$81.55	$ 63.27	$57.38	$50.81	$48.42

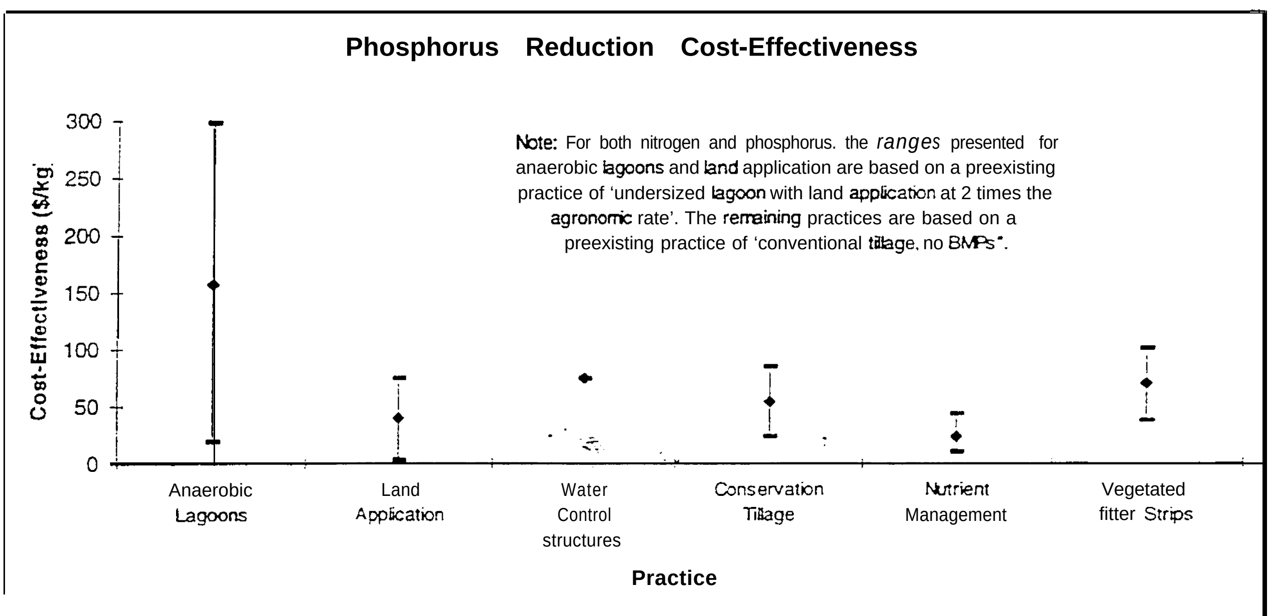
[1] These estimates do not include a safety factor.

[2] RANGES: The spread in the cost-effectiveness ranges for animal waste management practices is primarily due to the variability in nutrient requirements of the crops that receive animal waste application. Crops that require greater levels of manure application are also prone to lose more of the waste nutrients to runoff and subsurface drainage. The less expensive end of the cost-effectiveness range represents crops with high agronomic rates of fertilization (e.g., Bermudagrass). The more expensive end of the range represents crops with lower agronomic rates of fertilization (e.g., small grains). Additionally, the phosphorus cost-effectiveness range for animal waste practices is also driven by the variability in N:P ratios in land-applied wastes. The agronomic application rate for animal wastes is typically based only on the nitrogen content of the waste. The N:P ratio for different forms of swine and poultry wastes ranges from 1:1 to 3.9:1. This variability is Incorporated into the calculations for the phosphorus cost-effectiveness range.

The cost-effectiveness ranges for non-animal waste practices are driven by the range in conventional tillage loading factors for the Chesapeake Bay modeling subbasins that are in ecoregions common to the Tar-Pamlico basin.

Figure 2-7. Summary of nutrient removal cost-effectiveness ranges





Chapter 3

The Need for a Safety Factor

There are numerous factors that introduce uncertainty and variability into BMP cost-effectiveness estimates. It is important, therefore, to develop a margin of safety to ensure that these estimates are reliable indicators of the real-world reductions that can be expected from money spent on BMPs. Exactly what that safety factor should be is a subjective judgment that we will leave to the participants of the Phase 2 negotiations. The purpose of this section is to assist in the development of a safety factor by explicitly defining the elements that introduce uncertainty in to the preceding analysis.

3.1 Uncertainties in Animal Waste Management Effectiveness Estimates

There are numerous factors that influence the effectiveness of animal waste practices in reducing nutrient loads to surface waters. One of the most difficult to quantify is *local atmospheric deposition of volatilized ammonia.* One of the primary mechanisms of nitrogen reduction from stored wastes is/volatilization of ammonia. in anaerobic lagoon liquid, for example, ammonia constitutes 70 to 90 percent of the total nitrogen (Safely et al., 1992). The cost-effectiveness values presented in this report are based on the assumption that volatilized ammonia leaves the watershed. However, local deposition of ammonia is possible, particularly via rainfall. The extent to which this occurs in the Tar-Pamlico basin is unknown. If deposition is occurring at significant rates, the effectiveness values presented in this report may overestimate the actual cost-effectiveness of animal waste practices.

A second consideration is that the effectiveness values presented for land application are based on the use of proper nutrient management procedures. However, farmers do not always follow recommended nutrient management guidelines. Sometimes this is due to circumstances beyond the farmefs control, such as an unexpected rainfall following land application of wastes. Other times economic constraints, storage limitations, or lack of knowledge causes operators to apply or handle wastes improperly.

Another consideration is the difficulty in estimating the future effects of incentive payments. The SCS will provide incentive payments for up to 3 years for various types of land application methods. The purpose of the incentive payments is to help make proper land application an intrinsic part of the farmers operation. However, after the payments cease, there is no way to ensure that the farmer will continue to land-apply properly. Since we do not know how long a farmer will continue to land apply properly, our analysis counts the benefits of land application incentive payments *only for the years in which the payments are given.* This will most likely underestimate the effectiveness of land application incentive payments

3.2 Uncertainties in Cropland BMP Effectiveness Estimates

3.2.1 Loading Factor Variability

The spread in cost-effectiveness values seen in the box-and whisker plots is a result of the variability in nutrient loading factors across the 23 Chesapeake subbasins in ecoregions 63 and 65. (See section 2.2.2 for a discussion of the calculation process). However, the variability in loading factors seen in the 23 Chesapeake subbasins is not necessarily indicative of the loading variability in the Tar-Pamlico basin. The variability in the Tar-Pamiico could be either smaller or greater. Furthermore, the central tendency of the loading factors between the two basins also may not be similar. A conservative estimate would be to assume that the box-and-whisker plots represent the *minimum* variability in practice cost-effectiveness.

3.2.2 Unit Cost and Percent Effectiveness Variability

Site-specific factors can cause the costs for installing a practice/to vary substantially. In our calculations of cost-effectiveness, we have used an average cost value for practices installed in the Tar-Pamlico basin. Similarly, we have used a single value for percent effectiveness, not a range. Consequently, *the variability in unit cost and effectiveness is not incorporated into the box-and-whiskerplots of cost-effectiveness.* Despite the fact that this variability is difficult to quantify, it still greatly affects the variability of cost-effectiveness estimates and therefore must be accounted for in the safety factor.

3.2.3 Site-Specific Variability

A variety of site-specific factors can affect both the cost and effectiveness of a 'land management practice. Among these, soil type and slope are especially important. Steep slopes or highly erodible land can increase the amount of soil and/or nutrients lost from a site. In addition, the cost of constructing a BMP can be greatly increased if site conditions require labor-intensive activities such as land clearing or grading. Within the Tar-Pamlico basin, slopes and soil types vary substantially between the Piedmont and Coastal Plain. In addition, some of the BMP effectiveness data used in this report are taken from studies conducted in the Chesapeake Bay basin. Where possible, we used only results from ecoregions present in both basins. Nevertheless, conditions within a given ecoregion may also vary substantially.

3.2.4 Masking

When monitoring water quality to determine the effectiveness of a BMP in reducing nutrient loads, it is important to be able to isolate the effect of that BMP (or assemblage of BMPs) on water quality. In practice, this is a very difficult process because many activities can occur in the watershed that "mask" the effect of the practice. For example, a farmer outside the BMP project, but within the monitored watershed, may increase-his swine production while downstream monitoring is being conducted. This lack of control over external loading factors can introduce substantial uncertainty into BMP effectiveness calculations.

3.2.5 BMP Assemblages

Often, multiple BMPs are implemented on a farm as part of a "Resource Management System." The implementation process may be staggered over a period of months or years. When monitoring is conducted to evaluate effectiveness, it becomes difficult, if not impossible, to ascribe load reductions to individual BMPs. Reduction effectiveness values for a particular assemblage of practices are also difficult to use because the particular combination of practices is often tailored to the needs of the farm and will be different from the combination of practices on other farms.

3.2.6 Groundwater Loads

Research from the Chesapeake Bay Program (Camacho, 1992) has indicated that in-field BMPs that increase infiltration may route a substantial portion of the nutrient load into the groundwater. Effectiveness values based only on surface water monitoring do not account for this process. Where avaiiable, we have incorporated effectiveness values that represent the net effect of changes in both surface and groundwater loads.

3.2.7 BMP Longevity

in-field BMPs are typically assigned a "life span" indicating the time period for which the BMP is expected to function effectively, given proper maintenance. There is little information regarding the extent to which BMP effectiveness changes over the life span of the practice. If the effectiveness of a practice decreases over time, then a cost-effectiveness value based on data from a newly installed practice may overestimate the actual cost-effectiveness of the practice.

3.2.8 Short-Term Efficiencies

Some of the BMP effectiveness studies used as in this analysis focused on short-term efficiencies from single rainfall events. Extrapolation of these efficiencies to annual or long-term efficiencies is questionable due to annual hydrologic, crop, and farm activity changes (Camacho, 1992).

3.2.9 Simulated Field Conditions/Sampling Techniques

Some of the BMP effectiveness studies used in this report were carried out in highly controlled systems. For example, experimental data are frequently obtained from highly managed demonstration systems that frequently have rigorous design and control requirements (e.g., artificial rainfall, intensive site preparation, carefully controlled fertilizer application). Extrapolating such results to "the real world" requires a significant amount of subjective judgment. Furthermore, research is typically conducted on a small-field scale, and does not account for important "landscape-scale" factors, such as the location of a field in a watershed, the size of the area treated relative to the watershed size, adjacent land cover, or the distance to receiving waters.

3.2.10 Modeling Results

For certain practices, this study relies on loading factors and effectiveness data from the Chesapeake Bay Watershed Model (Appendix 2). We have not investigated in detail the assumptions and validity of the Chesapeake modeling process. However, the Chesapeake Bay Program has accepted the results of the model as the basis for developing a nonpoint source nutrient management plan in the Chesapeake basin. Nevertheless, it is important to note that modeling results are subject to uncertainty related to both the model's mechanisms and the input parameters. Where we have used the modeling results for cost-effectiveness calculations. we believe that they represent the best data available.

3.3 Economic Uncertainties

3.3.1 Decreasing Returns

As more money is spent on BMPs in the Tar-Pamlico basin, we may begin to see decreasing returns on invested dollars. Early in the program, the State may realize a high cost-effectiveness because the practices are presumably being used on farms with the worst nutrient loading problems. However, once the worst cases are addressed, it may become increasingly expensive to reduce a given level of nutrient load. It is difficult to estimate at what point decreasing returns will become a factor. However, they should be considered when developing a safety factor.

3.3.2 Cost Escalation

The costs of materials and labor directly affect the cost-effectrveness of a practice. As the costs of materials and labor increase due to inflation, the cost-effectiveness of a practice will decrease (in absolute terms, but not necessarily relative to other practices). In developing a safety factor, it is important to account for the escalation of costs due to inflation.

Chapter 4

Discussion and Recommendations

In the preceding chapters, we have presented quantitative technical information regarding the cost-effectiveness of approved agricultural management measures. The purpose of this chapter is to provide recommendations on how to target cost-sharing money to more effectively reduce nutrient loading. We recommend that several issues be considered in the process of developing a strategy for Phase II of the Nutrient Trading Program. These issues are discussed in the following sections.

4.1 **The Agricultural Cost-Share Program could place a higher-priority on nutrient (and particularly, nitrogen) management. Nutrient management has been proven to be a cost-effective strategy for reducing both edge-of-field and watershed loading from agricultural lands.** "Nutrient management" in this context refers to both the approved BMP, as well as the broader concept of focusing management attention on optimizing and managing farm systems to minimize nutrient losses to the environment. Nutrients supplied in excess of the amount needed for crop requirements results in a pool of residual nutrients in the soil. Over time, the size of the residual pool directly influences the magnitude of losses of available or mobile forms to surface water, groundwater, and the atmosphere. Nitrogen applications beyond the amount required for crop growth lead to increases in the mass residual nitrogen that **is** vulnerable to loss from the farm system. Nitrogen losses are greatest in agricultural watersheds in which inputs from synthetic fertilizer, manure, or legumes greatly exceed the amount of nitrogen taken up by the crop.

Virginia provides an example of a State with an active nutrient management program. Virginia's program is staffed with a program manager and 10 field nutrient management specialists to assist farmers. As of October 1993, they had developed over 1,000 nutrient management plans on 240,000 acres of cropland. Nutrient reductions from these activities are estimated at 5.2 million pounds of N and 4.4 million pounds of P. In addition, Virginia has 11 water quality specialists employed by the conservation districts to provide additional technical assistance. Resource management plans are required in "resource protection areas" of the coastal zone under the Chesapeake Bay Preservation Act and are also a requirement for receiving certain types of financial and technical assistance (L. Danielson, 1994)

The most important opportunity for improving nutrient management is to refine recommendations for application of synthetic fertilizers. Although nitrogen is supplied to cropping systems from many sources, including legumes and manures, most adjustments to the total nitrogen applied to cropping systems come by refining the quantity, location, and time of year that producers apply synthetic fertilizers containing nitrogen. Applications of synthetic fertilizers containing nitrogen are 'much easier to manage because the amount of nitrogen applied is known with accuracy. More important, when livestock or legumes are an important part of the farm enterprise, nitrogen additions from these sources are a fixed part of the nitrogen budget for the enterprise, and adjustments in the total amount of nitrogen applied will likely be made by adjusting the amounts of synthetic fertilizers containing nitrogen that producers apply. Legumes and applications of manure may be used to improve soil quality in addition to their value as sources of nitrogen. The single most important way to improve nitrogen management, therefore, is to reduce (or eliminate) supplemental applications of nitrogen to account for nitrogen supplied by legumes and manures.

Recommendations for application of synthetic fertilizers containing nitrogen can also be improved by setting realistic yield goals. As a crops-yield increases, the crop's need for nitrogen increases, at least initially. The dilemma for producers is that nitrogen must be applied before the crop yield is known. Nitrogen recommendations, therefore, must be based on some expectation of crop yield. For many crops, nitrogen requirements and recommendations are based on yield goals (the yield expected by the producer under optimum growing conditions). Supplying the nitrogen needed for crop growth during the period when it is most needed can be an important way to improve nitrogen management. Nitrogen is needed most during the period when the crop is actively growing. Thus, fertilizers containing nitrogen should be applied during and/or after planting whenever possible.

An increased emphasis on nutrient management relative to structural practices is essentially an investment in implementing education, technical assistance, and improved management. Implementing structural practices requires relatively less information and expertise but does not address the fundamental issue of needing to reduce system inputs.

To improve nutrient management, technical assistance is needed to establish appropriate nutrient budgets and application rates, based upon manure and soil testing. The goal is to have no excess nutrients lost into the ecosystem. Although there are associated technical assistance costs, nutrient management plans can be expected to provide cost savings to the landowner, which makes the concept attractive and enhances the potential for voluntary adoption.

In reviewing current nonpoint control efforts, we found that a systematic institutional framework that captures all aspects of nutrient management does not currently exist in the basin, although the Cooperative Extension Service has recently begun training field staff. To ensure its long-term success, the parties involved in the trading program will need to determine how improved nutrient management can be administered.

4.2 **Research indicates that cost-sharing of single-objective best-management practices is not the most cost-effective approach for soil and water quality programs at the farm level.** inherent links exist among the components of a farming system and the larger landscape. Adoption of a tillage system that increases soil cover to reduce erosion, for example, may require changes in the methods, timing, and amounts of nutrients and pesticides applied. Failure to recognize and manage these links increases the cost, slows the rate of adoption, and decreases the effectiveness of new technologies or management methods. Research and development of innovative, economically viable, sustainable, and holistic farming systems should be accelerated to meet long-term soil and water quality goals. Barriers to achieving this should be identified and removed.

4.3 **increasing the cost-effectiveness of cost-sharing will require an increased commitment to education and technical assistance. We have not attempted to quantify the costeffectiveness of public education programs outside the realm of cost-sharing. However, we feel that enhanced educational efforts can be highly cost-effective and should be given high priority as a means of achieving nutrient reductions goals.** Although the public sector certainly has an important role to play, mechanisms should be developed to augment public sector efforts to deliver technical assistance with nonpublic sector channels and to certify the quality of technical assistance provided through these channels. Crop-soil consultants, dealers who sell agricultural inputs, soil testing laboratories, farmer-to-farmer networks, and nonprofit organizations are increasingly important sources of information for producers. In many cases, these private sources of information have become more important direct sources of advice and recommendations than public sources. For example, 56 percent of farmers surveyed in five counties scattered throughout the country identified fertilizer dealers as their primary information source (National Research Council, 1993). This tendency, however, is not thought to be representative of the Tar-Pamlico basin (L. Danielson, 1994). Soil and water quality programs need to take advantage of the capacity of the private and nonprofit sectors to deliver information to producers.

4.4 **The Nutrient Trading Program is in a position to take a proactive approach to restoring and protecting land uses and land cover types that provide positive water quality benefits. The cost-effectiveness of this approach in reducing nutrient loading needs to be determined.** In particular, additional efforts are needed to encourage protection and restoration of river corridors. While we have not attempted to quantify the cost-effectiveness of wetland or riparian protection and restoration, there is a growing awareness of the importance of doing so in reducing nonpoint source loading (Dodd et al., 1993). Buffer zones can include natural riparian corridor vegetation (vegetation along waterways); simple, but strategically placed, grass strips; or sophisticated artificial wetlands. Federal, state, and local government programs to protect existing riparian vegetation, whether bordering major streams or small tributaries, lakes. or wetlands, should be promoted. The creation and protection of field and landscape-buffer zones, however, should augment efforts to improve farming systems. They should not be substitutes for such efforts.

4.5 The **lack of information specific to nonpoint source nutrient management activities in the basin is a handicap to studies of this type.** The increasingly sophisticated questions which are being asked'about the effectiveness of nonpoint source management efforts requires increasingly sophisticated information. We recommend that a greater effort be made by all participants and stakeholders to develop a focused research, monitoring, and information management strategy. For example, information needs to be made more accessible regarding fanning, forestry, and development practices being employed, proximity of operations to surface waters or vulnerable groundwater, the potential leaching or runoff for given practices, soil types, and topography, and the location of valuable forest and wetland areas that provide buffering capabilities. Obtaining this information will require an increased commitment to the process of monitoring and closer coordination by all parties involved_ Positive steps are underway in this regard, such as the implementation of the FOCS system by the Soil Conservation Service and the funding of demonstration projects to improve infonation about BMP effectiveness. However, there is still a great need for a strategic planning process to ensure that future information needs will be met.

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Appendix 1

The Tar-Pamlico Nutrient Trading Program

Background - The Tar-Pamlico Nutrient Trading Program

Over the past **20** years, a great deal of research has occurred in the Pamlico River Estuary related to the eutrophication of the system, and more specifically, to the culturally induced acceleration of the eutrophication process. Researchers have concluded that nitrogen is the limitirig factor for plant growth in the estuarine portion of the Tar-Pamlico basin (NCDEM, 1987). Ongoing research is focusing on the estuarine system's response to nutrient inputs and to the magnitude and timing of nutrient inputs from various sources. This research, along with growing public awareness, has led to the adoption of management strategies to better control eutrophication.

Nutrient budgets (estimates of the magnitude of point and nonpoint source nutrient loading) have been calculated for the Tar-Pamlico basin to aid in the development of strategies. A nutrient source budget was first prepared in 1986 by the North Carolina Division of Environmental Management However, two major events have subsequently occurred. First, the North Carolina General Assembly enacted a ban on the sale of phosphate detergents. Then, a new NPDES permit was issued to Texasgulf Industries, Inc., a phosphate mining operation whose total phosphorus loads were estimated to account for about 50 percent of the phosphorus inputs for the entire basin. After revising the budget to account for changes due to the phosphate ban and the Texasgulf permit, the North Carolina Division of Environmental Management (NCDEM) estimated that approximately 66 percent of the total phosphorus in the basin could be attributed to nonpoint sources, approximately 25 percent to point source dischargers other than Texasgulf, and the remaining 9 percent to Texasgulf Industries, inc. Eighty-three percent of the total nitrogen in the basin could be attributed to nonpoint sources, primarily agricultural runoff and, to a lesser extent, urban runoff and atmospheric deposition (NCDEM, 1989).

The development of a comprehensive strategy for the Tar-Pamlico was complicated because the phosphate ban and the new Texasgulf permit were enacted after the nutrient budgets had been calculated. NCDEM was uncertain about how much these changes would improve water quality; however, because of continuing demands on the system's assimilative capacity, they could not afford to wait until all the necessary information had been obtained through monitoring and research. Consequently, NCDEM proposed an interim strategy that required mandatory limits on nitrogen and phosphorus for new and expanding dischargers in the basin. The aim of the strategy was to halt point source increases while studies were being conducted. The NSW strategy specified effluent concentration limits of 2 mg/L for total phosphorus throughout the year, 4 mg/L for total nitrogen in the summer, and 8 mg/L for total nitrogen in the winter based on current technological capabilities.

Dischargers in the basin responded to the initial strategy with concerns regarding the high costs of new facility construction to meet the nutrient control goals. The dischargers soon formed a coalition, the Tar-Pamlico Basin Association (the Association), and began negotiations with NCDEM, the Environmental Defense Fund (NCEDF), and the Pamlico-Tar River Foundation. In 1989, these parties agreed on a new strategy that allowed for "nutrient trading" between point source dischargers and agricultural operations while meeting the overall nutrient reduction goal.

Under the agreement, the Association contributes funds for agricultural best management practices (BMPs) in order to achieve, all or part of the total nutrient reduction goals established for the member facilities. The underlying premise is that nutrient reductions via BMPs can be more cost-effective (on a per kilogram removal basis) than removing nutrients from point sources. The Association estimates that controlling one unit of nonpoint source loads with BMPs costs one-tenth as much as controlling the same load from a wastewater treatment plant (M. Green, 1993). The nutrient trading proposal was approved by the North Carolina Environmental Management Commission in December 1989. The program sets up an overall reduction goal and then allows nutrient sources to find the most cost-effective way to allocate allowable loads. Association members have the flexibility to trade reduction credits among themselves or to pay to control pollution from agricultural sources, as long as the total nutrient limit for the basin is not exceeded (NCEDF, 1993).

Determining initial Load Reduction, Goals, and Costs

NCDEM projected the 1994 flow for all the municipal Association members at 30.55 million gallons per day (mgd). Assuming no nutrient reductions from pre-strategy conditions, NCDEM estimated that total nutrient loading in 1994 would reach 625,000 kg/yr. Under the original NSW proposal, which required mandatory phosphorus and nitrogen limits for point sources, projected loadings for 1994 would decrease to an estimated 425,000 kg/yr, a reduction of 200,000 kg/yr. Subsequently, NCDEM, the Association, NCEDF, and the Pamlico-Tar River Foundation together established 200,000 kg/yr as the reduction goal for Phase I of the Nutrient Trading Program. Of this, 180,000 kg/yr is for nitrogen and 20,000 kg/yr is for phosphorus (NCDEM, 1992). The program was a popular solution because it fulfilled the States NSW reduction goals, addressed nonpoint source concerns, and reduced the economic burden of municipal dischargers.

The estimated cost of achieving the 200,000 kg/yr nutrient reduction goal using agricultural BMPs alone was $11.8 million, $10 million on the ground and $1.8 million in administration (Harding, 1990). These values were determined by multiplying the reductions by a factor of $56 per kg per year, the estimated cost for removing 1 kg of nutrient per year using BMPs. The rate was drawn from BMP funding experiencein the adjoining Chowan River basin (Appendix 2).

By addressing nonpoint sources, the trading program is more comprehensive than the original NSW strategy. Dischargers are benefitting from the increased flexibility and cost-effectiveness of the trading approach. An important reason for phasing the program was to obtain better technical information regarding the impact of nutrients on the estuary, the sources- of nutrients, and the effectiveness of management alternatives during the initial phase.

Implementing the Nutrient Trading Program

Implementation of the Nutrient Trading Program is being divided into phases. Phase 1 of the program began in 1989 and will end in 1994. During this time the trading approach is being evaluated and will be refined and re-negotiated as necessary.

Ten of the 21 major dischargers (representing over 90 percent of the permitted wasteflow) in the Tar-Pamlico basin have joined the Association. Two smaller municipal dischargers are also members. One industrial discharger is a member of the Association; however, its membership includes an exemption from weekly monitoring provisions. Membership in the Association is voluntary, but if dischargers choose not to participate in the nutrient trading program they are subject to the NSW nitrogen and phosphorus limits as previously discussed.

Under the Tar-Pamlico Nutrient Trading Program, dischargers are free to trade reduction debits and credits among themselves, as long as total loading goals for the basin are not exceeded. To date, no such trading has occured. This allows Association members to maximize the cost-effectiveness of their operations and avoids the inefficiencies and costs associated with prescribing controls for each discharger. However, NCDEM will continue to use individualized permitting and enforcement to control any localized impacts that may occur.

During the implementation phase, NCDEM has formally adopted a Basinwide Planning process. Under Basinwide Planning, a basin is viewed as the basic unit for water quality management. This **allows** NCDEM to better focus and coordinate efforts within a basin, and evaluate basinwide management efforts, such as nutrient trading. NCDEM is currently preparing a draft basin plan for the Tar-Pamlico river.

Funding BMP implementation and Maintenance

The Association payments go to the North Carolina Division of Soil and Water Conservation, which then distributes the monies to the local Soil and Water schedule for determining allowable nutrient loads and payments during Phase 1 (Table Al-l). If the total loads for the participating dischargers exceed the allowable load for that year, the Association will pay an amount equal to $56 times the excess loading (kg).

Table **A1-1. Schedule of Nutrient Loads and Payments**

(NCDEM, 1992; P. Blount, 1993)

Calendar Year	Allowable Nutrient Load (kg)	Actual Nutrient Load (kg)	Payment for Exceeding Allowable Load ($)	Minimum Payment* ($)	Report Results	Payment Due
1991	525,000	455,685	0.00	150,000	March 1, 1992	September 30, 1992
1992	500,000	422,962	0.00	250,000	March 1, 1993	September 30, 1993
1993	475,000	NA	NA	100,000	March 1, 1994	September 30, 1994
1994	425,000	NA	NA	NA	March 1,1995	September 30, 1995

* To ensure the operation of the Nutrient Trading Program.

The results of WWTP engineering evaluations indicate that a majority of the required nutrient reductions can be achieved through operational changes and minor capital improvements. Consequently, the actual level of BMP funding is likely to be substantially less than $11.8 million, which was estimated assuming all nutrient reductions would be achieved via BMPs.

The Association has agreed to make yearly minimum payments to the trading fund. These funds are in addition to the $150,000 administrative payments and will be used to fund BMP implementation. For 1991, 1992, and 1993, the Association's total loading was less than the maximum allowable load. Consequently, the Association has paid the minimum amount each year. When calculating loading payments for a given year, the. Association will receive credit **for minimum payments made in prior** years. All BMP credits will have a useful life of 10 years unless otherwise specified.

Since July 1991, Association facilities have been performing weekly effluent monitoring for total phosphorus, total nitrogen, and flow. The Association reports monitoring data to NCDEM annually. NCDEM has developed a set of guidelines for estimating flow and concentration if this information is not provided.

Currently, a pilot project is being conducted to study BMP effectiveness on Chicod Creek, a coastal plain tributary to the Tar River. Chicod Creek has been identified

Currently, a pilot project is being conducted to study BMP effectiveness on Chicod Creek, a coastal plain tributary to the Tar River. Chicod Creek has been identified by the State as severely degraded due to impacts from animal operations. The project is funded by EPA appropriations obtained through the efforts of the Association and by the minimum payments from the Association. The results of this study will provide important information on how BMP monies can be effectively targeted in other portionsof the Coastal Plain.

Over the past 30 years, an estimated 50 percent of the Pamlico's wetlands have been destroyed (NCEDF, 1993). Wetlands control excessive nutrients by trapping pollutant-laden sediment, lowering flood peaks, storing nutrients in vegetation and soil, and transforming nitrogen to gases that enter the atmosphere. For Chicod Creek, NCEDF is developing high-resolution maps to pinpoint degraded wetlands and areas of major nonpoint source pollution.

Appendix 2

The Chesapeake Bay Watershed Model
(Center for Exposure Assessment Modeling, 1994)

The Chesapeake Bay Watershed model is an application of the HSPF model (Hydrologic Simulation Program: Fortran) to the Chesapeake Bay watersheds. The model is one part of an integrated set of models designed to assess the impact of nutrient loads on the Bay system.

HSPF is a comprehensive package for simulation of watershed hydrology and water quality for both conventional and toxic organic pollutants. HSPF incorporates the watershed-scale ARM and NPS models into a basin-scale analysis framework that includes fate and transport in one-dimensional stream channels. It is the only comprehensive model of watershed hydrology and water quality that allows the integrated simulation of land and soil contaminant runoff processes within-stream hydraulic and sediment-chemical interactions.

The result of this simulation is a time history of the runoff flow rate, sediment load, and nutrient and pesticide concentrations, along with a time history of water quantity and quality at any point in a watershed. HSPF simulates three sediment types (sand, silt, and clay) in addition to a single organic chemical and transformation products of that chemical. The transfer and reaction processes included are hydrolysis, oxidation, photolysis, biodegradation, volatilization, and sorption. Sorption is modeled as a first-order kinetic process in which the user must specify a desorption rate and an equilibrium partition coefficient for each of the three solids types.

Resuspension and settling of silts and clays (cohesive solids) are defined in terms of shear stress at the sediment water interface. The capacity of the system to transport sand at a particular flow is calculated and resuspension or settling is' defined by the difference between the sand in suspension and the transport capacity. Calibration of the model requires data for each of the three solids types. Benthic exchange is modeled as sorption/desorption and deposition/scour with sut-ficial benthic sediments. Underlying sediment and pore water are not modeled.

Data Requirements

Data needs for HSPF can be extensive. HSPF is a continuous simulation program and requires continuous data to drive the simulations. At a minimum, continuous rainfall records are required to drive the runoff model and additional records of evapotranspiration, temperature, and solar intensity are desirable. A large number of model parameters can be specified although default values are provided where reasonable values are avaiiable. HSPF is a general-purpose program and special attention has been paid to cases where input parameters are omitted. In addition, option flags allow bypassing of whole sections of the program where data are not available.

output

HSPF produces a time history of the runoff -flow rate, sediment load, and nutrient and pesticide concentrations, along with a time history of water quantity and quality at any point in a watershed. Simulation results can be processed through a frequency and duration analysis routine that produces output compatible with conventional toxicological measures (e.g., 96-hour LC,,).

Assumptions and Limitations

HSPF assumes that the "Stanford Watershed Model" hydrologic model is appropriate for the area being modeled. Further, the instream model assumes the receiving water body is well-mixed with width and depth and is thus limited to well-mixed rivers and reservoirs. Application of this methodology generally requires a team effort because of its comprehensive nature.

Application History

HSPF and the earlier models from which it was developed have been extensively applied in a wide variety of hydrologic and water quality studies, including pesticide runoff model testing, aquatic fate and transport model testing, and analyses of agricultural best management practices. An application of HSPF in a screening methodology for pesticides is described by Donigian et al. In addition, HSPF has been validated with both field data and model experiments, and has been reviewed by independent experts.

Appendix 3

Summary of BMP Cost Data for Counties in the Tar-Pamlico Basin

EROSION/NUTRIENT CONTROL GRAND TOTAL

ITEM	UNITS	MSR	COST
Conserve Tillage	14,999.42	A	149,997.00
Terraces	499,579.00	F	149,369.00
Diversions	415,115.54	F	338,842.00
Crtcl Area Plan	22.76	A	17,543.00
Sod-Eased Rotatn	1,156.71	A	52,690.00
Stripcropping	838.88	A	57,624.00
Crop Cnv Grass	2,964.68	A	347,116.00
Crop Cnv Trees	4,528.98	A	367,457.00
Crop Cnv Uldlife	12.69	A	1,446.00
Crop Cnv CTrees	0.00	A	0.00
Grade Stbl Struc	101.00	N	66,835.00

SEDIMENT/NUTRIENT

ITEM	UNITS	MSR	COST
Filter Strip	16.17	A	10,363.00
Field Border	496.65	A	452,975.00
Grass Wtrway	706.64	A	939,770.00
H2O Ctrl Str	421	N	523,845.00
Heavy Use	0	u	0.00
Spring Dvlpmt	1	.U	724.00
Stock Trail	2	N	2,942.00
Stream Cross	1	N	1,525.00
Trough lank	1	N	552.00
Lvstock Excl	4,315.00	F	2,384.00
Nutr Red Mgt	0.00	A	0.00
Wells	0	N	0.00
Pesticide Load	0	N	0.00
Portable Wtr Fac	0	N	0.00

ANIMAL WASTE MANAGEMENT GRAND TOTAL

--WASTE MANAGEMENT--

ANIMALS	NUMBER	SYSTEMS	
Cows	1,610	Lagoon	56
Fish	0	cost	750,194.00
Hogs	71,843	Ponds	6
Horses	0	cost	38,203.00
Poultry	500,800	Dry Stack	1
Sheep	0	cost	8,862.00
		Pads	1
		cost	378.00
		LittrStor	0
		cost	0.00
		Composter	7
		cost	36,210.00
		Loafing Lot	0
		cost	0.00
		Abandon CAO	
		cost	415.00

--LAND APPLICATION--

ANIMALS	NUMBER		
cows	1,095	Acres	34,498.35
Fish	0	Gallons	163,436,370
Hogs	92.251	Tons	61,187.50
Horses	0	cost	688,796.00
Poultry	9,079,200	"Solid Set	18
Sheep	0	cost	210,665.00
		Wetlands	0
		cost	0.00
		Hydrants	3
		Cost	24,360.00

Conventional Tillage Cropland Loading Factors for Chesapeake Subbasins in Ecoregions 63 and 65

(Carnacho, 1992)

Ecoregion	Segment	Loading factor (lbs/acre-year) Nitrogen	Phosphorus
65	235	20.5	2.7
65	240	17.3	1.9
65	250	25.8	3.2
65	260	22	3
65	290	23.3	2.4
65	300	24.2	3.1
65	310	23	2.5
63	Bohemia	18	1.3
63	Chester	17.5	1.3
63	Chicahominy	20.2	1.4
63	Choptank	17.6	1.2
63	Elizabeth	21.1	2.2
63	Great W icomico	21.3	2
63	James	18.6	1.4
63	Nansemond	23.9	2
63	Nanticoke	25	1.7
63	Occoquan	21.4	2.1
63	Pocomoke	24.8	2
63	Potomac	20.5	1.8
63	Rappahannock	20.9	1.9
63	W icomico	20.5	1.7
63	Wye	18.3	1.4
63	York	18.5	1.3

Appendix 5

Calculation of Trading Values for Phase I and Phase II

1. Phase I Trading Value

The nutrient trading value used under Phase I of the Tar-Pamlico Nutrient Trading Agreement was $56 per kilogram of nutrient. In other words, it is assumed that, on average, the result of providing $56 in cost-share monies to a farmer would be to reduce nutrient loading by 1 kilogram. This value was derived by NCDEM based on BMP implementation data collected in the Chowan River basin from 1985 to 1989 (NCDEM, 1990). This section summarizes the approach taken in determining this value.

Essentially, the cost-per-kilogram figure was determined by dividing the total amount of cost-share money expended on BMPs in the basin by the resulting annual reduction in nutrient mass loading. The cost-share funding data were available in State records. The process used to estimate nutrient load reductions is summarized below.

Estimation of BMP Nutrient Load Reductions

For the purpose of the analysis, BMPs were grouped into three categories:

- Animal Waste Management Practices
- Cropland Practices without Water Control Structures
- Cropland Practices with Water Control Structures.

Animal Waste Management Practices

NCDEM used the following equation to estimate nitrogen and phosphorus load reductions from animal waste BMPs:

$$N_a = N_{82} \cdot A \cdot E \cdot C$$

where

N_a = 1989 total N (or P) load reduction attributable to animal BMPs
N_{82} = 1982 total N (or P) load attributable to agricultural nonpoint sources
A = Percent of N_{82} attributable to animals
E = Treatment effectiveness
C = Percent of animals treated.

The total agricultural nonpoint load, N_{82}, was estimated at 52,200 kg/yr P and 593,000 kg/yr N based on a 1982 nutrient budget conducted by DEM (NCDEM, 1982). The percent of the total load attributable to animals, A, was estimated at 69 percent for P and 57 percent for N, based on mass balance models (Craig and Kuenzler, 1983). The treatment effectiveness for both N and P was estimated at 50 percent, based on the results of a BMP effectiveness study in Fiorida (Heatvvole et al., 1986) and on values used by the state of Virginia (Virginia Water Control Board, 1985). The percent of animals treated was calculated as 20 percent, based on 1987 North Carolina Agricultural Statistics and state cost-share program records.

Cropland Management Practices

DEM used the following equation to estimate nitrogen and phosphorus load reductions from cropland BMPs:

$$P_a = (P_{82} \cdot A \cdot E_c \cdot C_c) + (P_{82} \cdot A \cdot E_{wc} \cdot C_{wc})$$

where

P_a = 1989 total P (or N) load reduction attributable to cropiand BMPs
P_{82} = 1982 total P (or N) load attributable to agricultural nonpoint sources
A = Percent of P_{82} attributable to crops
E_c = Treatment effectiveness for cropland BMPs
E_{wc} = Treatment effectiveness for acres affected for acres affected by water control structures
C_c = Percent of harvested cropland treated with cropland BMPs only
C_{wc}= Percent of harvested cropland treated with cropland BMPs and water control structures.

The values for P_{82} and A were estimated as discussed above. The treatment effectiveness for cropland BMPs was assumed to be 30 percent. Since very little BMP effectiveness monitoring had been conducted at that time, this value was based on the best professional judgment of agricultural researchers at North Carolina State University (NCDEM, 1982). The treatment effectiveness of crop management practices with water control structures was estimated at 60 percent, based on published monitoring results for urban detention basins, assuming 48 hours of settling (Schueler, 1987). The percent of land treated for each practice (C, and C_{wc}) was determined from State cost-share data.

After using the above equations to calculate load reductions, these values were divided into the total cost-share expenditures to determine cost per kilogram.

This calculation was done for each nutrient individually. and for nutrients as a whole. The results are shown in Table A2-1.

Table A5-1. Chowan Basin Cost-Effectiveness Values

Parameter	Cost per kilogram per year
Phosphorus	$409
Nitrogen	$38
Nutrients, in general	$34

Estimating a trading value

Although undocumented, it appears that the following approach was used to calculate the nutrient trading value for Phase 1 of the Tar-Pamlico Nutrient Trading Program (Danielson. 1994):

> The Tar-Pamlico Phase 1 reduction goal was 200,000 kg/yr of "nutrient". Of this total, 20,000 kg/yr was for phosphorus and 180,000 kg/yr was for nitrogen. As seen in Table A2-1, the annual cost for each kilogram was $409 for phosphorus, and $38 for nitrogen. The weighted average cost therefore is:

(20,000 kg P / 200,000 kg Nutrient) $409 - (180,000 kg N / 200.000 kg Nutrient) $38 = $75.10

> This is the estimated full cost (both government and farmer shares) of the "average" BMPs to obtain the Phase 1 target. However, since only 75 percent of the cost must be cost-shared, the value is:

$75.10 X 0.75 = $56.32

> which, when rounded is equal to the trading fee of $56/kg-yr

2. Phase II Trading Value

Based on the cost-effectiveness ranges presented in this report, the North Carolina Division of Environmental Management selected the following scenario as the basis for estimating a trading value for Phase II of the Nutrient Trading Program:

Practice:	anaerobic lagoons
Preexisting practice:	undersized lagoon with land application at 2 times the agronomic rate
Lagoon life span:	**20** years
Nutrient:	nitrogen

Our estimated cost-effectiveness range for the above scenario is from $5 to $21 per kilogram of nitrogen reduced (Table ES-l). To estimate a single trading value, NCDEM multiplied the median of this range ($13/kg N) by a safety factor of 2 and then added a 10% administrative cost. The resulting figure was $28.60, which was rounded to $29/kg N.

Appendix 6: Calculations of Effectiveness and Cost-Effectiveness for Animal Waste Practices

Calculation of Lagoon Cost Effectiveness Relative to Direct Discharge									
Animal	Waste Type	Nutrient load in waste (1)				Load Reduction (2)	Cost	Application Rate	
								Low	High
		#N/1000#-day	#N/1000#-yr	#N/#-yr	#N/head-yr	kg N/head-yr	$/ha-yr	ha/head	ha/head
TKN	Using 20 year life span and l cost range of								
Poultry	Fresh Manure	0.88	319.38	0.32	1.28	0.46	105	0.00035	0.000085
Poultry	Lagoon	0.20	72.64	0.07	0.29	0.11	105	0.00035	0.000085
Swine	Fresh Manure	0.51	187.61	0.19	264.91	96.09	105	0.11	0.027
Swine	Lagoon	0.14	50.74	0.05	71.64	25.98	105	0.11	0.027
TKN	**Using 20 year life span and high end of cost range.**								
		#N/1000#-day	#N/1000#-yr	#N/#-yr	#N/head-yr	kg N/head-yr	$/ha-yr	ha/head	ha/head
Poultry	Fresh Manure	0.88	319.38	0.32	1.28	0.46	977	0.00035	0.000085
Poultry	Lagoon	0.20	72.64	0.07	0.29	0.11	977	0.00035	0.000085
Swine	Fresh Manure	0.51	187.61	0.19	264.91	96.09	977	0.11	0.027
Swine	Lagoon	0.14	50.74	0.05	71.64	25.98	977	0.11	0.027
TKN	**Using 10 year life span** and low end of cost range.								
		#N/1000#-day	#N/1000#-yr	#N/#-yr	#N/head-yr	kg N/head-yr	$/ha-yr	ha/head	ha/head
Poultry	Fresh Manure	0.88	319.38	0.32	1.28	0.46	120	0.00035	0.000085
Poultry	Lagoon	0.20	72.64	0.07	0.29	0.11	120	0.00035	0.000085
Swine	Fresh Manure	0.51	187.61	0.19	264.91	96.09	120	0.11	0.027
Swine	Lagoon	0.14	50.74	0.05	71.64	25.98	120	0.11	0.027
TKN	**Using 10 year life span and high end of cost range.**								
		#N/1000#-day	#N/1000#-yr	#N/#-yr	#N/head-yr	kg N/head-yr	$/ha-yr	ha/head	ha/head
Poultry	Fresh Manure	0.88	319.38	0.32	1.28	0.46	1294	0.00035	0.000085
Poultry	Lagoon	0.20	72.64	0.07	0.29	0.11	1294	0.00035	0.000085

Calculation of Lagoon Cost Effectiveness Relative to Direct Discharge **(continued)**

Animal	Waste Type	Nutrient load in waste (1)				Load Reduction (2)	Cost	Application Rate		Cost-Effectiveness	
								Low	High	Low	High
		#N/1000#-day	#N/1000#-yr	#N/#-yr	#N/head-yr	kg Mead-yr	$/ha-yr	ha/head	ha/head	$/kg	$/kg
P2O5	**Using 20** year life span and low end of cost range.										
Poultry	Fresh Manure	0.72	263.17	0.26	1.05	0.45	105	0.00035	0.000085	0.00	0.02
Poultry	Lagoon	0.33	120.09	0.12	0.48	0.21	105	0.00035	0.000085	0.18	0.04
Swine	Fresh Manure	0.39	143.45	0.14	202.54	87.24	105	0.027	0.11	0.03	0.13
Swine	Lagoon	0.12	44.17	0.04	62.36	26.86	105	0.027	0.11	0.11	0.43
P2O5	**Using 20 year life span and high end of cost range.**										
		#P/1000#-day	#P/1000#-yr	#P/#-yr	#P/head-yr	kg P/head-yr	$/ha-yr	ha/head	ha/head	$/kg	$/kg
Poultry	fresh Manure	0.72	263.17	0.26	1.05	0.45	977	0.00035	0.000085	0.75	0.18
Poultry	Lagoon	0.33	120.09	0.12	0.48	0.21	977	0.00035	0.000085	1.65	0.40
Swine	Fresh Manure	0.39	143.45	0.14	202.54	87.24	977	0.027	0.11	0.30	1.23
Swine	Lagoon	0.12	44.17	0.04	62.36	26.86	977	0.027	0.11	0.98	4.00
P2O5	Using 10 year life span and low end of cost range.										
		#P/1000#-day	#P/1000#-yr	#P/#-yr	#P/head-yr	kg P/head-yr	$/ha-yr	ha/head	ha/head	$/kg	$/kg
Poultry	Fresh Manure	0.72	263.17	0.26	1.05	0.45	120	0.00035	0.000085	0.09	0.02
Poultry	Lagoon	0.33	120.09	0.12	0.48	0.21	120	0.00035	0.000085	0.20	0.05
Swine	Fresh Manure	0.39	143.45	0.14	202.64	87.24	120	0.027	0.11	0.04	0.15
Swine	Lagoon	0.12	44.17	0.04	62.36	26.86	120	0.027	0.11	0.12	0.49
P2O5	**Using 10 year life span and high end of cost range.**										
		#P/1000#-day	#P/1000#-yr	#P/#-yr	#P/head-yr	kg P/head-yr	$/ha-yr	ha/head	ha/head	$/kg	$/kg
Poultry	Fresh Manure	0.72	263.17	0.26	1.05	0.45	1294	0.00035	0.000085	1.00	0.24
Poultry	Lagoon	0.33	120.09	0.12	0.48	0.21	1294	0.00035	0.000085	2.19	0.53
Swine	Fresh Manure	0.39	143.45	0.14	202.54	87.24	1294	0.027	0.11	0.40	1.63
Swine	Lagoon	0.12	44.17	0.04	62.36	26.86	1294	0.027	0.11	1.30	5.30

Notes:

(1) Manure and lagoon nutrient content data from NC Coop. Extension (1994). Application rates from Zublena at al. (1990)

Average weight per head: poultry - 4lbs/bird, swine 1412 lbs/sow (includes boar and 9 pig litter) (Barker, 1994)

Assumptions: All N present as as TKN, all P present as total phosphate (P2O5).

(2) Load reductions based on 80% effectiveness for N and 95% effectiveness for P

Calculation of Land Application Cost Effectiveness Relative to Direct Discharge

Land Application Conversions

Animal	Waste Type	Nutrient Load in Waste (1) (2)				Load Reduction (3)	cost	Application Rate (4)		Cost-Effectiveness	
								High	Low	High	Low
		#N/1000#-day	#N/1000#-yr	#N/#-yr	#N/head-yr	kg N/head-yr	$/ha-yr	ha/head	ha/head	$/kg	$/kg
Nutrient: TKN											
Poultry	Fresh Manure	0.88	319.38	0.32	1.28	0.46	53.00	0.00035	0.000085	0.04	0.01
Poultry	Lagoon Liquid and Sludge	0.20	72.64	0.07	0.29	0.11	53.00	0.00035	0.000085	0.18	0.04
Swine	Fresh Manure	0.51	187.61	0.19	264.91	96.09	53.00	0.11	0.027	0.06	0.01
Swine	Lagoon Liquid and Sludge	0.14	50.74	0.05	71.64	25.98	53.00	0.11	0.027	0.22	0.06
Nutrient: P2O5											
		#P/1000#-day	#P/1000#-yr	#P/#-yr	#P/head-yr	kg P/head-yr	$/ha-yr	ha/head	ha/head	$/kg	$/kg
Poultry	Fresh Manure	0.72	263.17	0.26	1.05	0.45	53.00	0.00035	0.000085	0.04	0.01
Poultry	Lagoon Liquid and Sludge	0.33	120.09	0.12	0.48	0.21	53.00	0.00035	0.000085	0.09	0.02
Swine	Fresh Manure	0.39	143.45	0.14	202.54	87.24	53.00	0.027	0.11	0.02	0.07
Swine	Lagoon Liquid and Sludge	0.12	44.17	0.04	62.36	26.86	53.00	0.027	0.11	0.05	0.22

Notes:

(1) Manure and lagoon nutrient content data from NC Cooperative Extension Service (1994).

(2) Average weight per head: poultry - 4lbs, swine - 1412 lbs/sow (includes boar and 9 pig litter) (Barker, 1994)

Assumptions: All N present as TKN, all P present as total phosphate (P2O5)

(3) Load reductions based on 80% effectiveness for N and 5% effectiveness for P (See Section 2.3.1.1)

(4) Range of agronomic application rates from Zublena et al. (1990)

Calculation of Lagoon Cost-Effectiveness Relative to Excess Land Application

Animal	Baseline Practice	N loss		P loss		New Practice (Agronomic Rates) N loss		P loss		Load Reduction N		P	
		High	Low	High	Low	High	Low	High	Low	High	Low	High	Low
	Land Application at:	*kg/ha-year*				kg/ha-year				kg/ha-year			
Poultry	2x agronomic rates	179.2	22.4	44.8	1.5	89.6	11.2	22.4	0.71	89.6	11.2	22.4	0.79
Poultry	3x agronomic rates	268.6	33.6	67.2	2	69.6	11.2	22.4	0.71	179.2	22.4	44.8	1.49
Poultry	4x agronomic rates	358.6	44.0	89.7	2.9	89.6	11.2	22.4	0.71	269	33.6	67.3	2.19
Swine	2x agronomic rates	179.2	22.4	44.8	1.5	89.6	11.2	22.4	0.71	89.6	11.2	22.4	0.79
Swine	3x agronomic rates	268.8	33.6	67.2	2.2	89.6	11.2	22.4	0.71	179.2	22.4	44.8	1.4
Swine	4x agronomic rates	358.6	44.8	89.7	2.9	89.6	11.2	22.4	0.71	269	33.6	67.3	2.1

Baseline Practice	Animal	Unit Cost		Effectiveness Cost N		P	
		Low	High	Low	High	Low	High
Land Application at:		$/ha-year		$/kg reduced			
Using cost figures based on a 20 year lagoon life span							
2x agronomic rates	Poultry	236	977	21.05	10.90	298.38	43.60
3x agronomic rates	Poultry	236	977	10.52	5.45	158.20	21.80
4x agronomic rates	Poultry	236	977	7.02	3.63	107.63	14.51
2x agronomic rates	Swine	106	435	9.45	4.86	133.99	19.43
3x agronomic rates	Swine	106	435	4.73	2.43	71.04	9.71
4x agronomic rates	Swine	106	435	3.15	1.62	40.33	6.47
Using cost figures based on a 10 year lagoon life span							
2x agronomic rates	Poultry	312	1,294	27.86	14.44	394.94	57.77
3x agronomic rates	Poultry	312	1,294	13.93	7.22	209.40	28.88
4x agronomic rates	Poultry	312	1,294	9.29	4.81	142.47	19.23
2x agronomic rates	Swine	140	577	12.50	6.44	177.22	25.76
3x agronomic rates	Swine	140	577	6.25	3.22	93.96	12.88
4x agronomic rates	Swine	140	577	4.17	2.14	63.93	

Anaerobic Lagoon Cost Conversions								
Lagoon Life Span	Lagoon Cost	Lagoon Sizing Standards		Agronomic Application Rates			Cost per hectare of applied waste	
					Low	High	Low	High
years	$ / cu. ft - yr	cu. ft / lb animal	lb animal / head	cu. ft /head	animals /ha	animals /ha	$/ha-year	$ / ha-year
Poultry Waste								
10	0.011	2.5	4	10	2,640	11,766	312	1,294
20	0.0063	2.5	4	10	2,640	11,766	236	977
							3	
Swine Waste								
10	0.011	1	1417	1,417	9	37	140	577
20	0.0063	1	1,417	1,417	9	37	106	435
Lagoon sizing standards from Hansard (1994)								
Ranges of agronomic application rates from Zublena (1990)								

APPENDIX 4.9A

WATER QUALITY MANAGEMENT FOR RESERVOIRS AND TAILWATERS

by: G. Dennis Cooke and Robert H. Kennedy

PART III: PRERESERVOIR TREATMENT

Problem Addressed

Dense populations of algae, particularly blue-green algae, can create nuisance conditions in reservoirs, which can have negative impacts on project uses. These algal "blooms" are caused by a combination of favorable conditions of light, high pH, warm water, and high nutrient concentrations. A second reservoir problem involves the loss of basin volume through the deposition of silt and organic matter from the land. This process creates shallow, well-lighted, nutrient-rich areas for macrophyte growth, It also leads to impaired project use. Both of these problems are related to the transport of material from watershed to reservoir.

Nutrient concentration in the water, as described in an earlier part and in Walker (1987a), is a function of nutrient income, loss to sediments and outflow, dilution by basin volume, and release from sources inside the reservoir. When nutrient income is reduced through advanced waste treatment or land management of urban and agricultural flows, concentration in the reservoir may decline and algal blooms may decrease or be eliminated. The diversion of nutrients from Lake Washington (Edmondson 1970) is an example of this type of response. Similarly, land management to control erosion can curtail silt income and reduce the rate of basin volume loss.

However, nutrient diversion or advanced treatment and land management are often impossible to effectively accomplish in reservoir management. The drainage basin is usually very large, cutting across many political boundaries. This makes action to create lowered nutrient and sediment income very difficult or impossible for the reservoir manager. An alternative or an addition to advanced waste treatment and land management is to pretreat the water from incoming streams through the construction of structures to accelerate nutrient and silt sedimentation, or to add substances to the incoming stream water to precipitate nutrients and particulate matter.

Theory and Design

Siltation basins

A siltation basin is used to detain incoming water long enough to allow significant deposition of nutrients and particulate matter. Water that flows to the main reservoir should have greatly increased quality as a result, while materials deposited in the siltation basin can be periodically dredged. The design of a siltation basin will be site-specific in that management personnel must know the rates of water, nutrient, and silt income and then calculate the size of basin appropriate to detain water long enough to have significant deposition. In some cases, spring and early summer flows may be the only target, since this may be the water that is most stimulatory to algae or that causes, in other ways, the greatest problems in water quality. A basin to intercept summer low flows would be much smaller than a basin to handle runoff from wet seasons. The reports by Jones and Bachmann (1978); Canfield, Jones, and Bachmann (1982), and Walker (1987b) are useful in designing a basin that will allow significant sedimentation of phosphorus.

Prereservoir phosphorus removal

Since upstream treatment of runoff or effluents may be impossible or inadequate, a significant fraction of the incoming phosphorus could be precipitated in the stream or at the head of the reservoir through the addition of iron, calcium, or aluminum salts. This reservoir protection procedure has had very few published applications, and further research is needed.

Iron is added in the ferric (Fe^{+3}) form, usually as ferric chloride.. Iron in this state will precipitate as $FePO_4$, or as $Fe(OH)_3$ with inorganic phosphorus sorbed to it (Stumm and Lee 1960, Wetzel 1983) and will be carried in the streamflow and deposited in the stream and upper end of the reservoir. Phosphorus will remain bound in iron complexes as long as the redox potential in the sediments remains high. Unfortunately, the redox potential can be very low in the anoxic hypolimnia of eutrophic reservoirs. A low redox potential will reduce iron, and phosphorus will be released as iron complexes become soluble (Mortimer 1941, 1942). This internal phosphorus release may be high enough to stimulate algal growth. If dissolved oxygen is present at the sediment-water interface, or if dissolved oxygen is introduced through natural or artificial circulation, then iron should remain in an oxidized state and phosphorus will remain sorbed to it. Thus, for an iron addition to be effective, oxidizing conditions must be present continually at the sediment-water interface of the reservoir or stream site where the precipitate is deposited.

In lakes and reservoirs containing substantial amounts of alkaline earths from the solubilization of calcareous deposits in the drainage basin, a buffering action occurs based upon the equilibrium between free CO_2, calcium, bicarbonate, carbonate, and undissociated calcium carbonate. Free CO2 remains in solution after this equilibrium is reached, the amount dictating the amount of $Ca(HCO_3)_2$ also in solution. If more CO2 is added *to* this system the further solution of $CaCO_3$ will occur, producing more $Ca(HCO_3)_2$ and little pH change. If CO_2 is withdrawn, as occurs during extensive photosynthesis by algae and macrophytes, then $CaCO_3$ is precipitated. As Wetzel (1983) has pointed out, this decalcification of hard water can play a major role in regulating the reservoir's metabolism since the precipitation of $CaCO_3$ will also involve the coprecipitation of nutrients such as phosphorus and *the* sorption of labile organic matter.

This reaction of $CaCO_3$, bicarbonate, and CO_2 could be used to remove phosphorus and organic matter from incoming water, although there appears to be no case history of its use for this purpose. The effectiveness of this procedure could be altered by the quantities and types of organic matter in the stream. According to Wetzel (1983), dissolved organic matter sorbs to $CaCO_3$. In particular, fulvic acids, or low molecular weight humic acids, seem to repress $CaCO_3$ precipitation and allow phosphorus to remain in solution. This could prevent $CaCO_3$ from being effective in some streams. Effectiveness also will be altered by the lower pH-high CO2 conditions in some reservoirs. These conditions would lead to solubilization of $CaCO_3$ and release of sorbed materials.

The use of aluminum salts to precipitate phosphorus or to prevent its release from reservoir sediments is a well-known in-reservoir water quality management procedure (see Part IV). Aluminum salts could also be added to incoming stream water to precipitate phosphorus before it enters the reservoir. When aluminum sulfate (alum) or sodium aluminate are mixed with water with carbonate alkalinity, a visible floc of aluminum hydroxide is formed to which inorganic phosphorus is strongly sorbed. The formation of the floc may also trap some particulate phosphorus. The $Al(OH)_3$ floc or polymer appears to be inert to redox changes so that sorbed phosphorus will remain out of circulation (Browman, Harris, and Armstrong 1977). A hazard of this procedure is the depression of stream or reservoir pH when using alum, depending upon buffering capacity, and the subsequent appearance of a potentially toxic dissolved aluminum form (Al^{+3} or $Al(OH_2)^-$). If sodium aluminate is used, pH will increase, and at high pH (>8.5), dissolved aluminum again appears (Sung and Rezania 1984). Another hazard involves the smothering of stream invertebrates with the deposited floc. Further details of aluminum chemistry are found in Cooke et al. (1986) and Burrows (1977).

Wetlands

Wetlands, and man-made artificial wetlands or settling basins dominated by rooted plants and their epiphytes, can intercept significant amounts of nutrients and suspended solids under certain conditions. While rooted plants may absorb a comparatively small amount of nutrients, their presence creates barriers to water flow and enhances water detention time and thus. contact with the major storage compartments of a wetland, the microflora, detritus, and sediments (Howard-Williams 1985). Deposition of suspended materials will also be enhanced when water flow-through is impeded by the presence of vegetation.

In many cases, the nutrient retention capacity of a wetland is limited on a short-term basis to the growing season, and on a long-term basis to the saturation of storage compartments. High initial nutrient removal rates may be followed, in several years, by large nutrient exports. Harvesting of macrophyte biomass may prolong the wetland as a nutrient sink. Wetlands with predominantly mineral soils having a high aluminum content are far better phosphorus sinks than wetlands with peat soils. Terrestrial ecosystems, however, retain far more nutrients than wetlands (Howard-Williams 1985, Richardson 1985).

Lee, Bentley, and Amundsen (1975) list the following beneficial and adverse effects of wetlands on the quality of water discharged from them:

Beneficial effects:

- Denitrification of nitrate under anaerobic conditions permits methane formation and the degradation of certain organic compounds.
- $CaCO_3$ precipitates, along with other chemicals such as phosphorus.
- Sediments are trapped.
- Nutrients are removed during summer months, especially if flow is diffuse.

- Water storage by the marsh helps reduce fluctuation in water flow.

Adverse effects:

- Nutrients are released during periods of high spring or fall flows, necessitating nutrient interception.
- Nitrogen fixation occurs at a high rate, producing an increase in concentration of organic nitrogen in the marsh discharge.
- Water leaving a marsh may produce taste and odor problems.
- Marsh discharges may be high in organic matter and color, and low in dissolved oxygen.

Diversionary streams

Storm events often exhibit the poorest water quality and can represent a very significant fraction of'the total annual loading of nutrients, silt, and organic matter to the reservoir. In some situations, it may be possible to divert some or all of these high flows around the reservoir, especially in reservoirs used solely for recreation or as water supplies, and not for flood control or power generation. The use of this procedure has not been documented with regard to determinations of when or how to divert the water, the effectiveness of the procedure, nor the impact on downstream biotic communities. These data will vary from case to case, and a detailed budget of silt and nutrient loading will form the basis of any design to divert high flows.

Effectiveness, Costs, and Feasibility

Siltation basins

The work of Fiala and Vasata (1982) provides an example of the effectiveness of a siltation basin in removing phosphorus from incoming waters. Jesenice Reservoir, Czechoslovakia, was divided into a small (area = 76 ha; volume = 1.4×10^6 m^3) siltation basin with a detention time of 5 days. It emptied into the main reservoir (area = 670 ha; volume = 51×10^6 m^3), which had a theoretical hydraulic detention time of 180 days. Orthophosphorus fell from over 500 μg P ℓ^{-1} at the inlet of the siltation basin to 30 μg P ℓ^{-1} at its outlet. Orthophosphorus then reached about 10 μg P ℓ^{-1} at the pool behind the main dam. Phytoplankton biomass also declined. The authors note that phosphorus retention by the siltation basin increased with detention time, and they suggest a minimum of 5 days. This could be difficult to achieve on an annual basis in many situations, but could be feasible during summer low flows when symptoms of eutrophication are most extreme. Fiala and Vasata (1982) also note that maintenance of aerobic conditions in the siltation basin is essential to phosphorus removal, presumably because iron and/or calcium complexation of phosphorus may be involved and because anaerobic reservoir sediments may release phosphorus at high rates and thus reduce the efficiency of removal. In deep preimpoundment basins, maintenance of aerobic conditions might require artificial circulation.

Dry dams

In some areas, dry dams have been constructed to aid in flood control. These dams may be particularly effective in silt and nutrient control as well, since they receive and store the "first-flush" runoff, a portion of the hydrograph that can be heavily loaded with pollutants.

Prereservoir phosphorus removal

Wahnbach Reservoir (Federal Republic of Germany) was impounded in 1957. Within 10 years, treatment of the water for drinking purposes became very expensive, and organic compounds excreted into the water by algae were forming precursors for the development of trihalomethanes. Phosphorus was shown to be the limiting element, and nutrient budget studies showed that more than 50 percent of it came from diffuse or nonpoint sources on the watershed, making sufficient diversion nearly impossible. A smaller reservoir (500,000 m^3) to serve as a floodwater retention basin and a phosphorus elimination plant (PEP) were built at the upper end of the main reservoir.

After detention in the smaller reservoir, water is pumped into the PEP and treated with Fe^{+3} to precipitate phosphorus, followed by a cationic polyelectrolyte to form large floc. The water is then filtered through layers of activated carbon, hydroanthracite, and quartz sand. The plant can handle up to 5 m^3 sec^{-1}, and about 95 to 99 percent of phosphorus-containing compounds are eliminated. Output concentration to the main reservoir averaged 4 μg P ℓ^{-1} over 2 years. Also, the PEP has high removal (99 percent) of colifonn bacteria, chlorophyll, and turbidity, and lesser removal of chemical oxygen demand (77 percent) and dissolved organic carbon (58 percent). Water discharged to the reservoir approaches drinking water quality, and the trophic state of the reservoir is now nearly oligotrophic. Detailed descriptions of the PEP at Wahnbach Reservoir are provided by Bernhardt (1980, 1981). Costs have not been reported.

Prereservoir phosphorus precipitation has had few reported applications. However, results have been very encouraging, and it will probably be used with increasing frequency, Lathrop (1982) has reviewed this method.

Bannink, van der Meulen, and Peeters (1980) and Hayes et al. (1984) report on the use of iron salts to treat river water entering reservoirs in The Netherlands and England, respectively. Water quality improvements were noted, as well as reduced treatment costs of drinking water. However, Hayes et al. (1984) noted that internal phosphorus release during summer months was responsible for algal blooms, suggesting that iron-bound phosphorus may have been released under anaerobic conditions.

Harper, Wanielista, and Yousef (1983) have suggested the use of the $Al(OH)_3$ sludge,:produced during potable water treatment, as a cost-effective compound for treating incoming stream waters. No results appear to be available on their treatment of inflowing storm water to Lake Eola, Florida. Caution should be exercised in the use of the material from a potable water treatment plant since the sludge may have very large amounts of organic matter and phosphorus sorbed to it. Thus, its addition to the upper end of a reservoir could produce a pronounced oxygen demand and little phosphorus removal.

Cooke and Carlson (1986) have found that only a small dose of aluminum sulfate (1 to 5 mg Al ℓ^{-1}) was needed to precipitate all of the soluble reactive phosphorus in the Cuyahoga River just above Rockwell Reservoir, Ohio. The dose to accomplish phosphorus removal was determined with a jar test. To be certain that only insoluble aluminum hydroxide was formed, an attempt was made to keep the dose above the level that would produce a pH of 6.0 or less. Since the experiment was conducted on only a pilot scale in August and September 1985, long after substantial macrophyte and blue-green algae problems had developed in the reservoir, there was no expectation of reservoir improvement, and none occurred. The large volume of aluminum hydroxide floc that deposited in a small area due to the late summer low-flow conditions was deleterious to benthic macroinvertebrates in this area. No changes in macroinvertebrates, compared with upstream controls, were observed at stations nearer the reservoir. This apparently new and simple (compared with Wahnbach) approach to protecting a reservoir is undergoing further evaluation by Cooke and Carlson.

<u>Wetlands</u>

Case histories of the use of wetlands, marshes, or small impoundments with dense vegetation demonstrate that these systems can remove 50 percent or more of the incoming nutrients and suspended solids during the growing season (e.g., Toth 1972; Lee, Bentley, and Amundsen 1975; Spangler, Fetter, and Sloey 1977; Fetter, Sloey, and Spangler 1978; MacCrummon 1980; Sinclair and Forbes 1980; Barten 1983; Herron, LaMarra, and Adams 1984; Weidenbacher and Willenbring 1984; Willenbring 1985).

Barten (1983) describes the deterioration of Clear Lake, Minnesota (area ≅ 257 ha; drainage area = 1,518 ha) due to urban runoff. To protect the lake from further impacts, storm runoff was diverted to a 21-ha marsh, composed of peat underlain with clay loam and having a reed canary grass (*Phalaris arundinacea*) plant community. The marsh was divided into cells controlled by gates. Nutrients and suspended solids were removed by percolation through the peat. The marsh was harvested to remove nutrients and to maintain the absorption potential of the peat. During the winter, storm flows were diverted through the marsh rather than through the cells. Filtration significantly reduced nutrient concentrations, especially phosphorus (90 percent), and suspended solids (70 percent). In 1982, 897×10^6 ℓ was filtered, removing 526 kg of phosphorus. Where possible, a 5-day detention time was used.

Sinclair and Forbes (1980) examined the removal capacity of a swamp, a 16-ha reservoir dominated by waterhyacinth (*Eichhomia crassipes*), and a 0.4-ha reservoir dominated by the submergent plant *Najas* sp. The latter two systems were effective nutrient sinks, but the aerobic system (*Najas*) was most effective. The authors believe that in comparison with the swamp, the waterhyacinth- and naiad-dominated systems have the greatest potential to be nutrient sinks because they can be harvested. The systems could be used in series. Sinclair and Forbes (1980), following the suggestion of Boyd (1970), also recommended cattail systems for removal of nutrients and suspended solids due to their large standing crop, rapid growth rate, high nutrient value to cattle, and ease of harvest.

Limitations and Concerns

Many of the problems that could be encountered with the use of any of these prereservoir treatments will be site- and problem-specific. Therefore only a general listing of the most likely problems is given here.

Siltation basins and wetlands

A critical problem with the use of either of these systems is obtaining the area needed to construct them. If a portion of the upper reservoir is modified to form a smaller reservoir, then significant loss of storage capacity may occur. Both the marsh and the siltation basin may require substantial maintenance in the forms of harvesting of plants and the removal of accumulated silt through dredging.

Wetlands will discharge nutrients during high-flow, low-vegetation periods. More significantly for potable water reservoirs, they also can discharge dissolved organic molecules, which may impart taste and odor, increase the chloride demand, and perhaps contribute to trihalomethane production. Lee, Bentley, and Amundsen (1975) suggest that marsh outflows could be treated with a low dose of aluminum sulfate. Willenbring (1985) notes that channelized flow in a wetland will reduce its removal capacity.

Prereservoir phosphorus removal

An area of significant concern is the potential for adverse effects on stream biota from aluminum salts. The interested reader is referred to Part IV as well as to Burrows (1977), Kennedy and Cooke (1982), and Cooke et al. (1986) for a discussion of the chemistry of aluminum and the environmental conditions under which it can be deleterious to biota. Briefly, when aluminum sulfate is added to natural waters containing bicarbonate-carbonate alkalinity, a visible precipitate of aluminum hydroxide is formed, and pH falls. The floc is sorptive of phosphorus and organic matter, and some materials are trapped with the floc. The forms of aluminum that appear in the water are pH-dependent. Insoluble $Al(OH)_3$ predominates between pH 6 to 8, while soluble species predominate at higher ($(Al(OH)_4^-$) and lower pH ($Al(OH)_2^+$ and Al^{+3}). $Al(OH)_2^+$ and Al^{+3} are considered to be potentially toxic, and therefore pH must not fall below pH 6.0.

-Very few studies of the toxicity of aluminum to aquatic biota have been conducted. Collectively, these studies suggest that concentrations of Al^{+3} below 0.050 mg Al ℓ^{-1} are not toxic to *Daphnia*, *Tanytarsus dissimilis* (Insecta, Chirononidae) and *Salmo gairdneri* (rainbow trout). Biesinger and Christensen (1972) found that the 48-hr LC_{50} for *Daphnia magna* was 3.90 mg Al ℓ^{-1}, and a lo-percent reproductive impairment occurred at 0.32 mg Al ℓ^{-1}, when animals were reared in Lake Superior water (alkalinity 50 mg $CaCO_3$ ℓ^{-1}, pH 7.74). Lamb and Bailey (1981) report that instars of T. *dissimilis*, reared in laboratory systems at pH 7.8, were unaffected in acute tests at doses from 6.5 to 77.8 mg Al ℓ^{-1}. Dissolved aluminum (Al^{+3}) remained below 0.1 µg Al ℓ^{-1}, and the floc was used by the larvae for tube building. In chronic tests at pH 6.8, Al^{+3} remained at the same low level over the same dose range. However, mortality occurred at every dose, and pupation did not occur over the 55-day study. Narf (1978) found that there were no apparent effects to benthic insects in several lake treatments. Everhart and Freeman (1973) found that a concentration of 0.52 mg Al ℓ^{-1} produced behavioral problems in rainbow trout after several weeks of exposure, whereas a concentration of 0.052 mg Al ℓ^{-1} produced no long- or short-term effects. Buergel and Soltero (1983) found no mortality, physiological stress, gill hyperplasia or necrosis, or retardation of rainbow trout growth after a dose of 12.2 mg Al ℓ^{-1} to hard-water Medical Lake, Washington.

The deposition of floc in the stream may pose some hazard to aquatic organisms and may have an adverse appearance. High flows should displace the floc deposit to the reservoir, and offer the benefit of treating phosphorus-rich reservoir sediments with a substance that may stop internal phosphorus release at the deposition site. The negative features of the use of treatment plant-generated aluminum hydroxide sludges have already been discussed.

Summary

Prereservoir treatments are a partial substitute for watershed management and advanced waste treatment. The object is to detain or remove loads of nutrients, organic matter, and silt by settling basins, marsh filtration, or the addition of nutrient-precipitating chemicals to the stream. With the possible exception of dry dams, these methods have not been widely employed, as yet, but a review of case histories demonstrates their potential effectiveness. These methods can be costly, although reliable cost estimates are uncommon. Also, there are problems with land acquisition to build such basins, with the discharge of nutrients and organic matter from marshes during high flows, the requirement for periodic silt removal, and the potential for creating toxic conditions through the addition of aluminum salts. Table 2 is a summary of this method.

Table 2

Summary of Prereservoir Treatments

Characteristic	Description
Targets	Nutrients, organic matter, and silt income.
Modes of action	Forces deposition in siltation basin. Strips silt and nutrients from water by marsh plants. Precipitates phosphorus in incoming stream.
Effectiveness	Highly effective, depending upon method chosen, season, and maintenance frequency.
Longevity	Months to years.
Negative features	Loss of storage capacity of main reservoir if pre-impoundment basin is constructed. Maintenance requirements can be extensive. Wetlands may lose effectiveness and will discharge unwanted organics. Dissolved aluminum and/or aluminum hydroxide floc may be toxic to reservoir biota.
Costs	Unknown because of site specificity.
Applicability to reservoirs	Not as applicable to high-volume, hydropower reservoirs as to smaller recreational and potable water supply reservoirs.

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PART IV: PHOSPHORUS INACTIVATION

Problem Addressed

Many reservoirs experience intense and prolonged "blooms" of algae during summer months, particularly in the transition and lacustrine zones of the reservoir (see Part II) where the water is clearer. The occurrence and degree of these blooms can often be directly linked to a high income and a high in-reservoir concentration of nutrients, especially phosphorus (Rast and Lee 1978). A substantial reduction in nutrient loading to the reservoir, as would occur in the case of sewage diversion, will usually lead to a predictable decline in concentration in the reservoir. If the decline has been significant, then algal blooms may decrease in frequency and extent, and the degree of eutrophication or trophic state of the reservoir may shift to a far less productive condition. The dramatic improvement of Lake Washington following sewage diversion is one illustration of this type of response (Edmondson 1970).

Phosphorus release to the water column from enriched sediments, especially under conditions of low dissolved oxygen or high pH and temperature, is well known (Bostrom, Jansson, and Forsberg 1982). This "internal loading," as described in Part II, can be a very significant source of nutrients, especially phosphorus, to the water column. Internal nutrient loading may prolong the eutrophic state long after nutrient diversion. Shallow reservoirs, especially those with a prolonged history of nutrient and organic matter loading, are especially susceptible to the impact of internal nutrient release. In these reservoirs the regenerative zone or recycling zone (the sediments) is very close to the lighted, productive zone (the surface waters), and algal blooms may therefore persist after a reduction in nutrient income.

Phosphorus inactivation is a procedure to accelerate the recovery of a reservoir, following a reduction in nutrient income, in those cases where internal phosphorus release is extensive. The target of the treatment is phosphorus in reservoir sediments, and the procedure is to add an aluminum salt to the sediments to bind the phosphorus to aluminum hydroxide. The layer of aluminum hydroxide will persist, even under conditions of anoxia, and has produced a significant decrease in phosphorus concentration in the water column of natural lakes and maintained an improved trophic state in them for many years after treatment. The applicability of this procedure to reservoirs, where adequate treatment of incoming nutrients may not occur, remains open to investigation. This Part describes how the procedure works, how to apply it, and its cost and effectiveness in lakes. More reservoir treatments are needed to better define its effectiveness in this habitat.

Theory and Design

Phosphorus inactivation is carried out through the addition of aluminum sulfate or sodium aluminate (or both) to the lake or reservoir. Aluminum has been the element of choice rather than iron because the complexes and polymers that form after the addition of either of these aluminum compounds are apparently inert to changes in oxidation-reduction potential, such as would occur during the development of hypolimnetic anoxia. Phosphorus will remain bound to these complexes, whereas iron will release phosphorus as the redox potential falls.

Hayden and Rubin (1974), Burrows (1977), and Kennedy and Cooke (1982) have provided reviews of the chemistry of aluminum salts in water. A knowledge of this is essential in determining the correct dose and preventing the development of a high concentration of dissolved aluminum (Al^{+3}), an aluminum species that has been associated with toxicity to aquatic organisms. When aluminum sulfate ($Al_2(SO4)3 \cdot 18H_2O$) or sodium aluminate ($Na_2Al_2O_4 \cdot 14H_2O$) is added to water, the pH dictates the form of hydrolyzed aluminum that will predominate. Settleable, polymerized aluminum hydroxide ($Al(OH)_3$) predominates at pH 6 to 8, aluminate above this range, and dissolved aluminum (Al^{+3}) below it. $Al(OH)_3$, a visible precipitate, is very sorptive of phosphorus, particularly inorganic phosphorus, and is thus the desired form. When aluminum sulfate (alum) is added to water with carbonate alkalinity, the pH and alkalinity of the water will fall at a rate dictated by the water's initial alkalinity. Low initial alkalinity or an excessive dose would allow pH to fall below pH 6.0 and thus decrease the amount of phosphorus-sorbing $Al(OH)_3$ and increase the amount of potentially toxic dissolved aluminum (Al^{+3}).

Phosphorus inactivation is a technique in which as much aluminum sulfate or sodium aluminate as possible, within the bounds dictated by initial alkalinity, pH, and the associated formation of dissolved aluminum (Al^{+3}), is added to the sediments with the purpose of controlling phosphorus release

(Kennedy 1978, Kennedy and Cooke 1982, Cooke et al. 1986). The objective is to control phosphorus release for a period of at least several years. Another procedure, known as phosphorus removal, has been used to add small amounts of an aluminum salt to water for the purpose of removing the phosphorus in the water column rather than giving sediments a maximum dose to control phosphorus release. Phosphorus removal has been used very effectively in special situations such as the interception of nutrients released from decaying vegetation in the fall (Funk et al. 1982), and it has been used as a treatment for incoming streams water (Part III).

Fly ash, the airborne particulate matter (5 to 100 μ) that is trapped in electrostatic precipitators in coal-fired power plants, has been suggested as another type of phosphorus inactivant for lakes and reservoirs (Tenney and Echelberger 1970). Fly ashes have very large sorptive areas and are high in CaO, MgO, Na, and Al. Therefore, they would sorb phosphorus. Fly ash appears to be an attractive option for improving reservoirs because the material is produced in very large quantities and only about 20 percent of it has been used for purposes such as the manufacturing of cement (Adriano et al. 1980). Unfortunately, fly ash treatments have produced serious negative environmental impacts, due primarily to the presence of heavy metal contaminants.

Aluminum dosage to a reservoir for the purpose of removing phosphorus from the water column is determined by jar tests. Aluminum salts, usually aluminum sulfate, are added in increasing amounts to a series of continuously stirred beakers containing reservoir water and reservoir water spiked with known amounts of phosphorus. After settling, phosphorus concentration is measured, and the amount of alum required to obtain the desired phosphorus removal is used to calculate the tonnage of alum needed to treat the water column (Peterson et al. 1973, 1974; Cooke and Kennedy 1981). The amount of alum added is usually so small that large pH shifts and the appearance of dissolved aluminum do not occur. However, pH, alkalinity, and dissolved aluminum must be measured to be certain that potentially deleterious conditions do not occur.

Kennedy (1978) was the first to suggest that the most desirable lake treatment would be to add as much aluminum as possible, consistent with environmental safety, to the phosphorus-rich sediments rather than to the water column, with the purpose of inactivating this sediment store. He developed a procedure for obtaining the maximum dose for a lake by considering the relationships between alkalinity, pH, and aluminum dose. Kennedy's method is reviewed in Cooke and Kennedy (1981), Kennedy and Cooke (1982), and Cooke et al. (1986). The procedure, which is applicable for reservoirs also, is briefly outlined here.

When aluminum sulfate is added to reservoir water, pH and alkalinity fall. At pH 6 to 8, large amounts of the floc aluminum hydroxide are formed, and the -dissolved aluminum (Al^{+3}) concentration remains low.. This pH range is therefore ideal since large amounts of phosphorus will be sorbed, and toxic conditions will not be present. However, with further additions of alum, pH values below 6.0 occur and the concentration of dissolved aluminum increases rapidly (Figure 6). Therefore, the maximum amount of aluminum sulfate that can be added before the appearance of low pH and high dissolved aluminum (Al^{+3}) is dependent upon the initial alkalinity of the reservoir water. The maximum dose is therefore unique to each reservoir. General guidelines

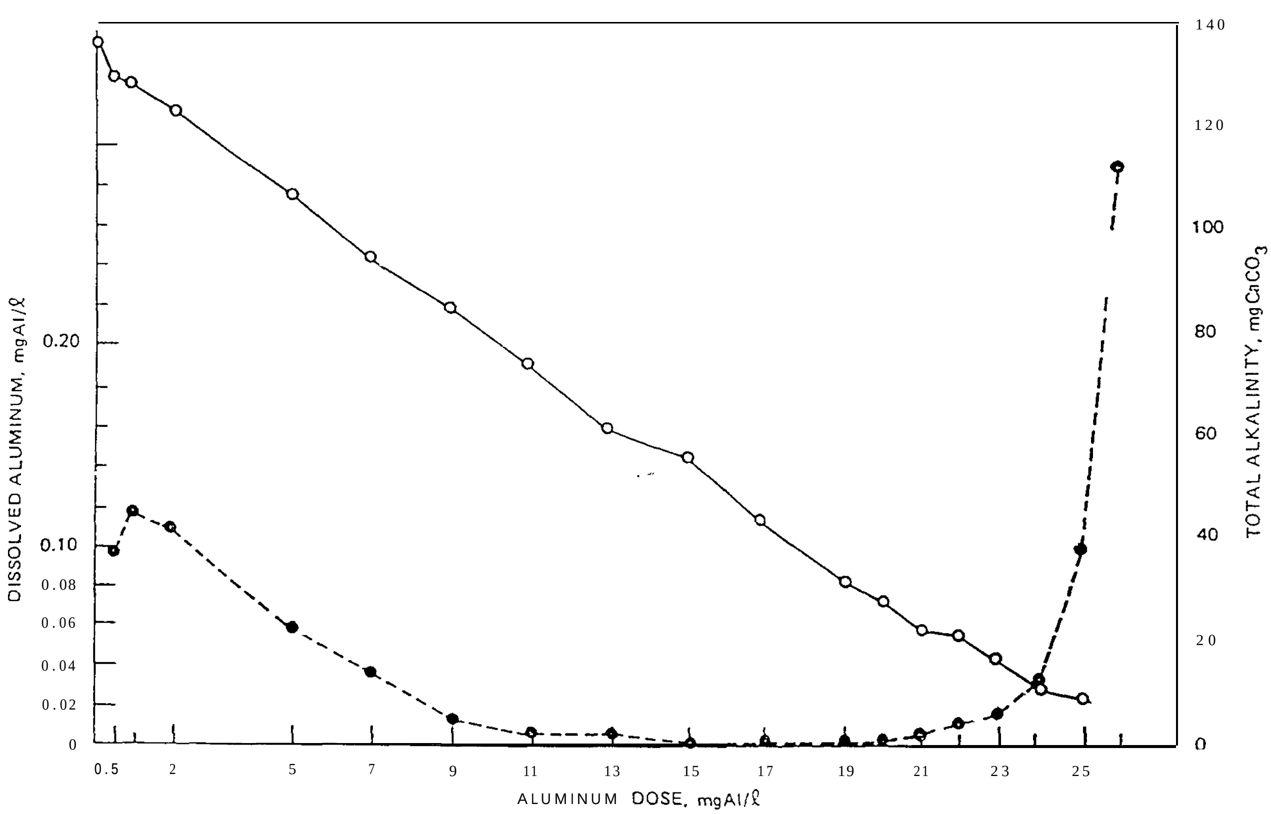

Figure 6. Changes in dissolved aluminum concentration (dashed line) and total alkalinity (solid line) for water from West Twin Lake, Ohio. treated with increasing doses of aluminum sulfate (after Cooke et al. 1978)

for dose determination can also be developed from a knowledge of pH **and** alkalinity of each stratum and use of the nomograph from Kennedy and Cooke (1982) (see Figure 7).

For dose determination, a vertical series of water samples is obtained and alkalinity is determined. Then, other water samples from the same depths are titrated with stock solutions of alum to pH 6.0. The relation between the aluminum dose and the alkalinity and pH is then used to obtain the maximum dose for any reservoir alkalinity over the range of alkalinity and pH tested. The maximum dose for each depth interval is calculated from the titration and water volume for that depth interval, and these are summed to produce the total dose for the reservoir, or section of the reservoir. Accuracy in treating the reservoir is obtained by dividing the treatment areas, or the reservoir, into zones marked by buoys. The volume and alkalinities in each of the zones are measured, and the amount of alum is then determined. By dividing the reservoir into sections, an overdose to shallow areas or an underdose to deep areas is avoided.

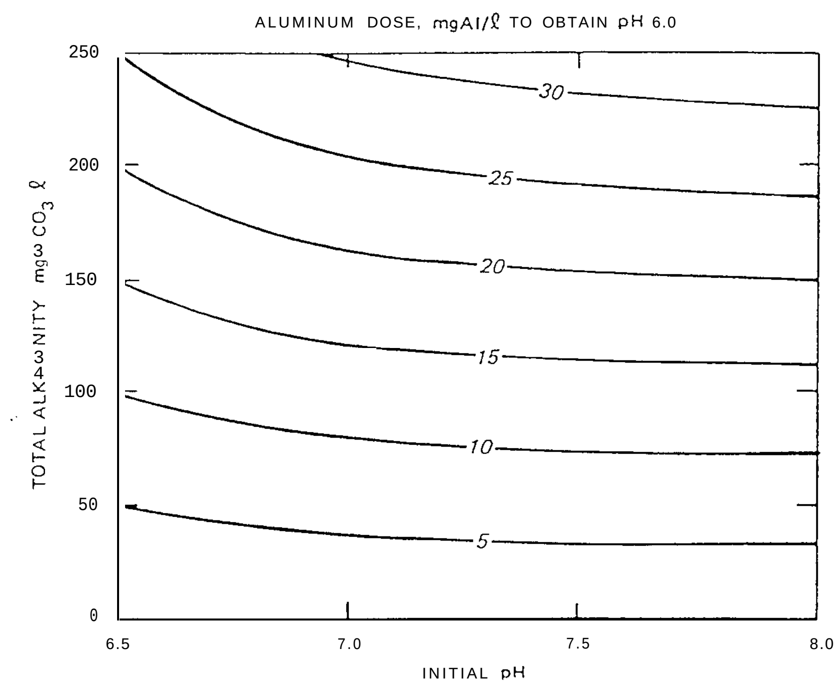

Figure 7. Estimated aluminum sulfate dose (mg Al ℓ^{-1}) required to obtain pH 6 (i.e., "maximum dose") in treated water of varying initial alkalinity and pH (from Kennedy and Cooke 1982)

In soft waters, only small amounts of aluminum sulfate can be added before the pH falls below 6.0. A. R. Gahler and C. F. Powers of the Corvallis Environmental Research Laboratory (USEPA, undated report, Corvallis, OR) suggested that sodium aluminate, which increases the pH of an aqueous solution, could be used with aluminum sulfate to maintain a pH between 6.0 and 8.0. Dominie (1978) was apparently the first to successfully use this dose approach on a large scale when Annabessacook Lake, Maine (alkalinity, 20 mg $CaCO_3\ \ell^{-1}$) was treated with this mixture in an empirically determined ratio of 1:1.6 (alum to sodium aluminate). Another alternative is to add materials such as lime or $CaCO_3$ to buffer the alum. Before attempting an alum treatment, the reader is urged to consult the primary literature, especially Kennedy and Cooke (1982) and Cooke et al. (1986), for a detailed, step-by-step outline of the dosage determination procedure.

Figure 8 illustrates the design of the application equipment used at Dollar and West Twin Lakes, Ohio (Kennedy 1978; Cooke et al. 1978, 1982; Kennedy and Cooke 1982). The delivery system was mounted on barges, and aluminum sulfate, mixed 50-50 with lake water, was pumped to an application manifold that was below the barge at the top of the hypolimnion. This allowed direct injection of the inactivant to the nutrient-rich anoxic hypolimnion and sediments without significant leakage to the littoral zone. As designed, the system added 140 m^3 of liquid aluminum sulfate in 3 days to a hypolimnetic area of 16 ha. Delivery systems similar to this have been used to treat much larger areas, but they are all labor-intensive. The development of a more rapid application system is needed. One option, where a large harvester is available, is to use the front cutter bar to attach the delivery manifold and the weed storage area to hold alum tanks. The harvester's hydraulic system can be used to operate the pumps.*

Ideally, based upon experiences with lakes, the entire area of reservoir sediments should be treated, particularly the area that becomes anoxic. Practically, this may not be possible in reservoirs due to their large size. An alternative is to determine those areas of reservoir sediments with the highest release rates of phosphorus and to treat them. This approach may

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* Personal Communication, 1986, G. N. Smith, Aquatic Control Technology, Inc., Northborough, MA.

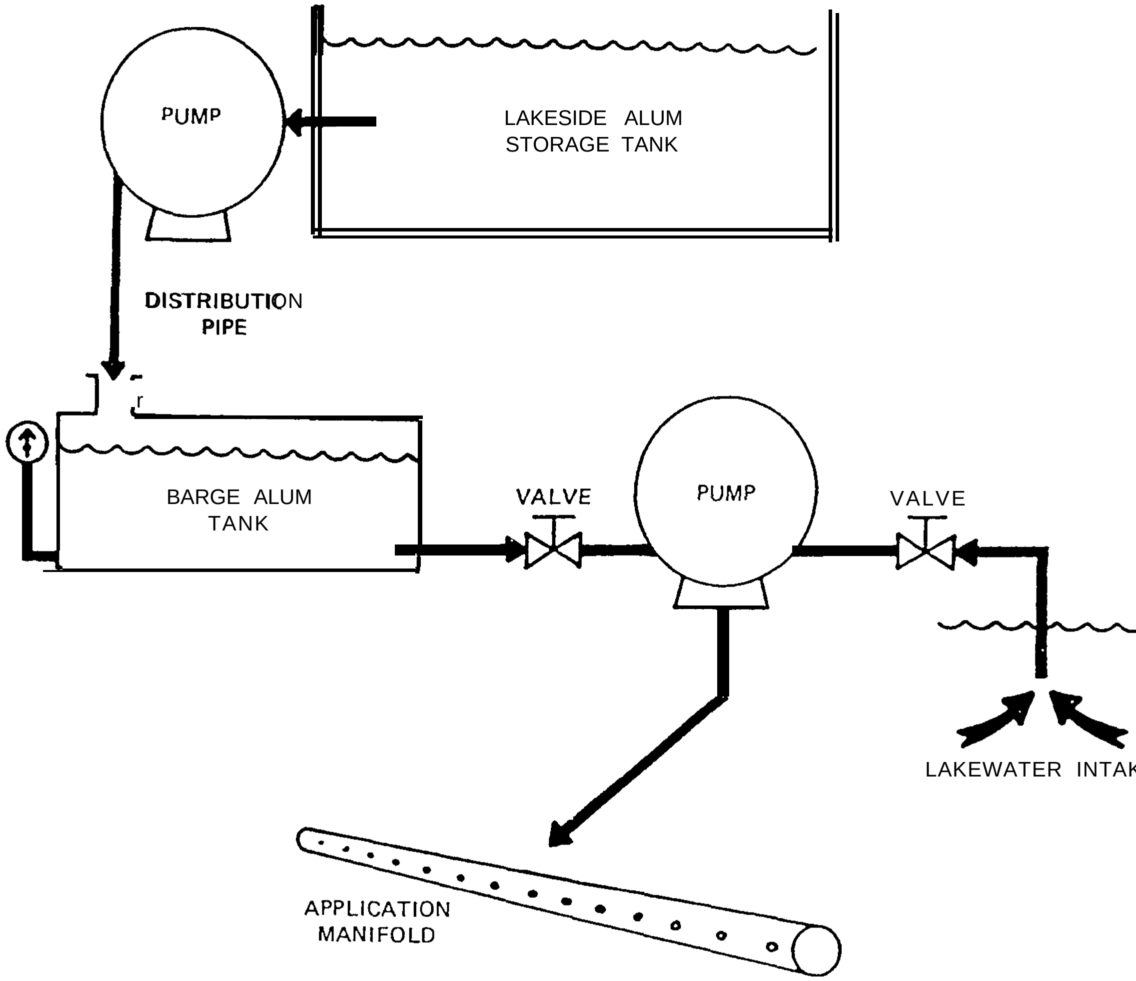

Figure 8. Generalized diagram of an alum application system (from Kennedy and Cooke 198'2)

limit the treatment area, but produces nearly the effectiveness of a treatment of the entire reservoir (Cooke and Carlson 1986).

It should be recognized that there may be extensive phosphorus release from aerobic sediments and sediments exposed to high pH, such as may occur in littoral areas during periods of high rates of photosynthesis. Much of the release of nutrients involves microbial metabolism and can be expected to be high when the water is warm. Also, it should be noted that a range of sediment types having various phosphorus sorptive or phosphorus release characteristics may be found in a reservoir. An exceptional review of this topic is provided by Bostrom, Jansson, and Forsberg (1982).

The areas of highest potential phosphorus release can be determined by studying release rates of sediment cores in the laboratory. Samples of sediments are obtained with an Ekman dredge, or a corer, from shallow and deep waters along the length of the reservoir. Sediments from inlets, macrophyte beds, and areas of anoxia should be included. Frevert (1980) and Lennox (1984) provide descriptions of laboratory procedures to evaluate the potential of a sediment sample to release phosphorus under aerobic and anaerobic conditions, high and low pH, and various temperatures. This survey will produce a map of reservoir sediments with regard to their potential to release phosphorus. At least the high-release rate areas could receive an alum application (Cooke and Carlson 1986).

Effectiveness, Costs, and Feasibility

The effectiveness of phosphorus inactivation and phosphorus removal in improving trophic state is best described through case histories that illustrate the range of conditions that have been treated. Four case histories have been chosen to describe the use of the method with aluminum salts, including the first United States treatment, a high alkalinity-high dose case, a soft water-large area application, and treatment of a shallow, nonstratified lake. A case history of fly ash application is also described.

Case histories

Horseshoe and Snake Lakes, Wisconsin. The first US lake to be treated was Horseshoe, in 1970. This small eutrophic lake received a dose of 2.6 g Al m^{-3}. In 1982, 12 years after the application, the concentration of phosphorus in the hypolimnion and the whole lake remains low, compared with pretreatment years. The lake continues to receive nutrient-rich drainage water, but the appearance of the lake has remained better than before (Garrison and Knauer 1984).

Snake Lake, a soft-water lake, was treated in 1972 with a mixture of aluminum sulfate and sodium aluminate. After application, transparency was greatly improved. Phosphorus concentration remained low through 1982 (Garrison and Knauer 1984).

These two case histories are useful to reservoir managers because they clearly demonstrate the longevity of the effect of the aluminum hydroxide floc on phosphorus concentration. Neither lake received a maximum dose, as defined by Kennedy (1978), yet their treatments were long-lasting.

Eau Galle Lake, Wisconsin. This small reservoir (area, 16 ha; mean depth, 3.2 m; drainage area, 16,600 ha) may be the first reservoir in the

United States to be treated with aluminum to control internal phosphorus release. A dose of alum equivalent to five times the average summer internal phosphorus load was added to the hypolimnion in May 1986. Deep-water phosphorus concentration, internal loading, and.blue-green algae were reduced relative to previous summers. Algal biomass remained high because a bloom of the dinoflagellate ***Ceratiwn*** occurred. External phosphorus loading remained high and may have contributed to the *Ceratiwn* bloom (Kennedy et al. 1987).

Annabessacook Lake, Maine. Annabessacook Lake-is one of the largest (575 ha) water bodies to be treated by this method. The lake supported intense blue-green algal blooms, even following nutrient diversion, due to internal phosphorus loading. Since the water is soft (20 mg $CaCO_3\ \ell^{-1}$), only small amounts of aluminum sulfate could be used before pH 6.0 was reached and dangerous levels of dissolved aluminum (Al^{+3}) appeared. A mixture of aluminum sulfate and sodium aluminate in a 1:1.6 ratio was determined through jar tests to be a dose that would maintain pH in the 6 to 7 range. Over an 18-day period, this dose was applied to the top of the hypolimnion (130 ha) with a barge upon which tank truck trailers had been driven. A concentration of 25 g Al m^{-3} was applied to the 8- to 10-m contour; 35 g Al m^{-3} was applied to the 10-m contour and deeper (Dominie 1980).

A 65-percent reduction in internal phosphorus loading occurred in summer 1979, following the 1978 application. Blue-green blooms were absent in 1979 (Dominie 1980).

Pickerel Lake, Wisconsin, and Long Lake, Washington. Application to shallow, nonstratified lakes was believed to be inappropriate because it was thought that the aluminum hydroxide would be dispersed and relocated during turbulent weather. This problem is important because many shallow, eutrophic reservoirs might also experience this problem.

This concern is supported by the results of the phosphorus removal treatment of shallow, holomictic Pickerel Lake, Wisconsin. A dose of 7.3 g Al m^{-3} was applied in April 1973, and total phosphorus was sharply reduced. After a series of mixing events, total phosphorus returned to pretreatment levels and an analysis of the sediments showed that the aluminum hydroxide floc had been redistributed to the lake's center. This left areas of the sediment free to continue phosphorus release (Knauer and Garrison 1980).

At Long Lake, Washington, however, a maximum dose of aluminum sulfate was applied. Total phosphorus declined, along with phytoplankton biomass and pH. Transparency increased. Internal phosphorus release was curtailed. This effect has lasted 4 years, and the floc was not redistributed during a winter of high winds and high flushing (Welch, Michaud, and Perkins 1982; Jacoby, Welch, and Michaud 1983; Welch et al. 1988). During the fifth summer, phosphorus levels were elevated, along with algae, and transparency declined. The floc layer apparently was dispersed to a deeper layer of sediment and also became covered with new phosphorus-rich materials (Welch, DeGasperi, and Spyridakis 1986).

Lake bottom slope, sediment chemistry, dose, and application procedure are among the factors that could have produced the disparity between the two shallow, holomictic lakes with regard to floc redistribution. It would be logical to be concerned about the problem in shallow, highly mixed reservoirs. Only further testing can provide the answers.

Lake Charles East, Indiana. Theis et al. (1979) describe the treatment of a section of Lake Charles East, Indiana, with fly ash, during summer 1975, for the purpose of sealing the sediments to prevent internal loading. There appears to be no other published report of the full-scale use of this substance for this purpose.

About 1,430 metric tons of fly ash and 275 metric tons of CaO were added to a 8.7-ha area of the lake. Some evidence of a reduction in phosphorus concentration appeared, algal blooms were reduced, transparency increased, and the phytoplankton was no longer dominated by blue-green algae. However, heavy metals, apparently from the fly ash, led to extensive mortality to fish and invertebrates.

This case history, plus the several laboratory experiments with fly ash (reviewed in Cooke 1980 and Cooke et al. 1986), illustrates the danger of using fly ash in lakes and reservoirs. Until further studies are completed, fly ashes should not be used for reservoir restoration (Cooke 1980).

costs

The principal cost of adding an aluminum salt to a reservoir is labor, and labor costs appear to be dependent upon dose. Cooke and Kennedy (1981) summarized the small amount of published data on costs, and Cooke et al. (1986) provided this equation for estimating man-days of labor from a determination of its maximum dose, based upon reported costs from six lakes:

$$Y = 0.55 + 0.1614X$$
$$r^2 = 0.9411$$

where Y represents the man-days per hectare and X is the dosage in grams of aluminum per cubic meter.

Aluminum sulfate costs vary with the market and, in recent years, a ton of liquid alum has cost about $160 to $170. Equipment costs also will vary with the size of the application. Dominie's (1980) technique of using a barge big enough to load tank truck trailers on it represents a way of reducing costs, since lakeshore storage and delivery systems would not be needed. Also, as suggested earlier, a large harvesting machine could be modified for use as an alum applicator. The cost of the equipment, as well as labor, may also vary with the depth of application. Several treatments have been directed toward hypolimnetic sediments only, and a manifold or other injection device was needed that could pump materials to the hypolimnion. A surface treatment could be accomplished with less equipment. Phosphorus inactivation is a procedure that would benefit from new designs for application.

Limitations and Concerns

The potential for serious negative impacts from low pH or the toxic effects of dissolved aluminum clearly exists with the addition of an aluminum salt to a reservoir. Aluminum sulfate, as described earlier, will produce a shift toward a low pH. At pH 5.5, dissolved aluminum Al^{+3} will begin to appear, and its concentration will increase rapidly as pH declines. Toxic conditions could be reached.

Fish mortality has not occurred during alum applications (Funk et al. 1982, Lamb and Bailey 1983). There was little or no appearance or accumulation of aluminum in the tissues of rainbow trout (*Salmo gairdneri*), as reported by Buergel and Soltero (1983), or in tissues of channel catfish (*Ictalurus punctatus*), largemouth bass (*Micropterus salmoides*), and gizzard shad (*Dorosoma cepedianum*), as reported by Berg and Burns (1985), in lakes treated with alum but maintained at a pH of 7.0 or greater. Biesinger and Christensen (1972), Peterson et al. (1973, 1974), and Lamb and Bailey (1981, 1983) have indicated that a dissolved aluminum concentration below 50 µg Al ℓ^{-1} will not bring about harmful effects to *Daphnia magna* (zooplankton), rainbow trout (S. *gairdneri*), and insect (chironomid) larvae (Tanytarsus *dissimilis*). This concentration will not be reached if pH 6.0 or more is maintained (Kennedy and Cooke 1982). Havas and Likens (1985) have found that the zooplankton *Daphnia catawba* and *Holopedium gibberum* and the insects *Chaoboms punctipennis* and *Chironomus anthracinus* were tolerant of aluminum concentrations higher than 300 μg Al ℓ^{-1}. Narf (1978) reported that there had been no damage to the invertebrate populations of four Wisconsin lakes during several years of monitoring after alum applications. A report by Moffett (1979) suggests that species diversity of planktonic microcrustacea in West Twin Lake, Ohio, was reduced for at least 3 years after an alum treatment in which dissolved aluminum never exceeded 2 μg Al ℓ^{-1} and pH and alkalinity returned promptly to normal. Gibbons et al. (1984), however, found no lasting impact to the zooplankton of Liberty Lake, Washington, after an alum application, supporting the conclusion of Moffett that predation may have produced the zooplankton shift.

Much more research, especially field studies, is needed concerning the toxicity Of aluminum to aquatic communities. However, it appears, from the laboratory and limited field data, that few risks to biota can be expected if pH 6.0 or above is maintained. It should be noted that soft-water lakes found in regions that receive extensive acid precipitation could be a future hazard after an aluminum treatment, if lake pH falls significantly below pH 6.0 in the years following aluminum treatment.

Bulson et al. (1984) have observed that the aluminum hydroxide floc is very efficient in the removal of fecal coliform and fecal streptococci bacteria during a lake treatment, suggesting that enteric species, including pathogens, might also be accumulated. Bacteria appear to die off in the floc and are not released from it. Bulson et al., however, suggest that there be a posttreatment restriction on recreational use, or a restriction of treatment to the nonpeak recreational season to allow a long die-off of bacteria. They also caution that intake of floc into a potable water treatment plant could pose a health hazard. It is likely, in many cases, that the pretreatment process with alum in potable water treatment plants should remove any floc in the raw water intake.

Aluminum sulfate and sodium aluminate applications will bring about greatly increased water clarity. This benefit of the method could produce a significant increase in the area of the reservoir that is infested with submerged macrophytes, since the outer depth limit of their growth can be light-limited. There is evidence that this has occurred in West Twin Lake, Ohio (Cooke et al. 1978), and in Long Lake, Washington (Jacoby, Welch, and Michaud 1983).

Fly ash presents a serious environmental hazard and should not be used in reservoirs (Cooke 1980). Fly ashes from bituminous coals (eastern United States) are high in sulfur, and aquatic solutions have a low pH. This environment will promote solubility of the heavy metals which they contain. Lignitic coals (western United States) produce a high pH (above pH 12) in solution and also contain heavy metals (Adriano et al. 1980). Theis and DePinto (1976) report the following negative attributes of fly ash: (a) high pH of treated waters, (b) dissolved oxygen depletion, (c) appearance of sulfide, (d) heavy metal release, and (e) physical crushing of biota or clogging of gills. Various laboratory and field studies have demonstrated the toxicity of various fly ashes to fish and invertebrates (e.g., Cairns, Dickson, and Crossman 1972; Guthrie and Cherry 1979).

Summary

Phosphorus inactivation is a technique to control the release of phosphorus from reservoir sediments, a source of "internal loading" that can maintain severe algal blooms even after diversion of nutrient income. Aluminum sulfate or sodium aluminate will produce the formation of aluminum hydroxide in water with carbonate alkalinity. This hydroxide is a visible floc or precipitate that is very sorptive of phosphorus and will not release it under conditions of low dissolved oxygen. A procedure for determining the maximum dose for a reservoir has been outlined. This dose will produce the largest amount of floc possible, consistent with environmental safety.

Case histories of the procedure have been reviewed (Cooke and Kennedy 1981, Cooke et al. 1986). This treatment has been effective for up to 12 years in controlling phosphorus release and in improving the trophic state of lakes. Large (575 ha), deep (18 m), soft-water (20 mg $CaCO_3$ ℓ^{-1}), hard-water (750 mg $CaCO_3$ ℓ^{-1}), and shallow (2 m) lakes have been successfully treated. Application procedures for very large areas, such as many reservoirs, have not been developed.

Aluminum applications pose significant risk to biota and possibly to human consumers of the water if the pH of treated water falls below pH 6.0 and dissolved aluminum (Al^{+3}) appears. The dose determination technique is designed to prevent this occurrence.

Fly ash has also been suggested as a phosphorus inactivant. This material will produce significant adverse environmental impacts and should not be added to reservoirs.

Table 3 is a summary of this method.

Table 3

Summary of Phosphorus Inactivation

Characteristic	Description
Targets	Nuisance algal blooms, low transparency, release of phosphorus from sediments.
Mode of action	Phosphorus release from reservoir sediments is sharply reduced, producing lowered phosphorus concentrations in water column.
Effectiveness	Highly effective, problem eliminated when accompanied by significant diversion of external nutrient loading.
Longevity	Up to 12 years; few long-term evaluations available.
Negative features	Use of aluminum sulfate will lower pH.
	Overdose could produce appearance of toxic dissolved aluminum.
	The floc may contain a high density of bacteria, including pathogens.
	Application is labor intensive.
Costs	Labor and chemical costs will be high but can be determined if dose is known (see text for equations).
Applicability to reservoirs	No published record of use in reservoirs with large areas.
	New methods of application may have to be developed to lower costs.
	Treatment of high phosphorus-release areas should be attempted.

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PART V: DILUTION AND FLUSHING

Problem Addressed

Reservoirs and lakes with high concentrations of nutrients may have severe blooms of nuisance algae. Algal blooms, particularly when blue-green algae are involved, interfere with the recreational use of the water and may dramatically increase potable water treatment costs. Algal excretory and decomposition products, along with other sources of dissolved organic matter, are associated with dissolved oxygen depletions, taste and odor, and with the appearance of trihalomethanes and other organohalides following chlorination of drinking water.

Dilution is a procedure in which water of low nutrient content is added for the purpose of lowering the reservoir's concentration of nutrients to a level at which algal cell growth is limited. Cell washout increases as well. Flushing, on the other hand, emphasizes cell washout through a sharp increase in the water exchange rate. The inflowing water may not necessarily have a lower nutrient concentration. The procedures become equivalent when low nutrient water is added at a rate sufficient to achieve cell washout equal to algal cell growth rate. This normally requires a large volume of scarce low-nutrient water. In practice, the procedures are differentiated because one (flushing) emphasizes what goes out of the reservoir without consideration of nutrient concentration changes and associated changes in cell growth, and the other (dilution) emphasizes a limitation on algal growth through a decrease in nutrient concentration as well as through cell washout (Welch 1981). Dilution can also improve water quality by decreasing the concentration of algal excretory and decomposition products. These procedures can be particularly effective for some reservoirs when treatment of upstream nutrient sources is not feasible. Both techniques, but particularly dilution, are limited in their applicability by the difficulty of finding an additional water source that can be diverted to the reservoir. The reader is referred to Uttormark and Hutchins (1978), Welch (1981), and Cooke et al. (1986) for reviews of these techniques.

Theory and Design

Uttormark and Hutchins (1978) were among the first to clearly describe the effect of adding low-nutrient water to a lake's inflow. According to these researchers, the following changes will occur: (a) the areal and volumetric phosphorus loading will increase with the increased income of water containing phosphorus, (b) the mean phosphorus concentration in inflowing water will decrease, and (c) the flushing rate will be increased and the sedimentation of phosphorus will decrease.

The effect of a change in loading, flushing, and sedimentation on in-lake phosphorus concentration is described by models (Vollenweider 1976, Uttormark and Hutchins 1978, Cooke et al. 1986, and Walker 1987) which assume, on a long-term basis, that lakes can be described as completely mixed reactors in which it is assumed that phosphorus income is constant, that net sedimentation is proportional to the amount of phosphorus in the lake, and that phosphorus is lost through the outlet and by sedimentation. The reader is referred to Walker (1987) for significant additional details and discussions of these models.

At steady state the lake's phosphorus concentration is described as

$$[P] = [P_o] \frac{\rho}{\sigma + \rho} \qquad (9)$$

where

$[P]$ = in-lake total P concentration, g m^{-3}

$[P_o]$ = in-flow total P concentration, g m^{-3}

ρ = flushing rate = $\frac{Q}{V}$, $year^{-1}$

Q = annual water flow rate, m^3 $year^{-1}$

V = lake volume, m^3

σ = sedimentation rate, $year^{-1}$

An alternative to Equation 9 is

$$[P] = \frac{L}{\bar{Z}(\rho + \sigma)} \qquad (10)$$

where

L = total P income, g m^{-2} $year^{-1}$

$\bar{Z}$ = mean depth, m

Larsen and Mercier (1976) and Vollenweider (1976) found that the specific phosphorus sedimentation rate, a term that is very difficult to determine empirically, can be estimated as

$$\sigma = \sqrt{\rho}$$

Equations 9 and 10 are thus rewritten as

$$[P] = \left(\frac{L}{\bar{Z}\rho}\right)\left(\frac{\rho}{\sqrt{\rho} + \rho}\right) \tag{11}$$

or

$$[P] = \left(\frac{L}{\bar{Z}\rho}\right)\left(\frac{1}{1 + \frac{1}{\sqrt{\rho}}}\right)$$

It should be noted, as first pointed out by Uttormark and Hutchins (1978), that phosphorus sedimentation is inversely related to flushing rate so that the amount of incoming phosphorus that is deposited in the reservoir bottom will decrease as the inflow is diluted with additional water. The effect of dilution is that a decrease in the concentration in the inflow may'reduce in-lake concentration, but the decrease in sedimentation will increase lake concentration.

Uttormark and Hutchins (1978) derived an expression from Equation 11 which allows comparison of predicted in-lake phosphorus concentrations, following dilution, with that before the addition of dilution water. Thus,

$$\frac{[P]'}{[P]} = \left(1 + \frac{\rho_2 [P_0]_2}{\rho_1 [P_0]_1}\right)\left(\frac{\rho_1 + \sqrt{\rho}_1}{\rho_1 + \rho_2 + \sqrt{\rho}_1 + \rho_2}\right) \tag{12}$$

where $[P]'$ is equal to the lake concentration after dilution (subscript 1 refers to conditions before dilution, and subscript 2 refers to conditions after dilution).

Figures 9 and 10 illustrate the effects of the addition of dilution water on the in-reservoir concentration. The X-axis gives the flushing. rate before dilution, and the lines on the graph show flushing rates due to dilution only, expressed as a constant proportion of undilutedflow. Thus, using Equation 12 and assuming that there is no phosphorus in the dilution water and that the dilution is equal to half the normal flow for a reservoir with a normal flushing rate of 1.0 year^{-1} ($\rho_2 = 0.5\rho_1$), theory predicts a 26-percent reduction in in-reservoir phosphorus concentration. As Uttormark and Hutchins point out, and as Figure 9 illustrates, large quantities of dilution water are needed to produce a significant change in reservoir phosphorus concentration.

Figure 10 illustrates the more realistic case wherein dilution water contains 40 percent of the phosphorus concentration found in the normal undiluted inflow. This graph clearly shows that greater and greater quantities of dilution water do not necessarily result in progressively greater

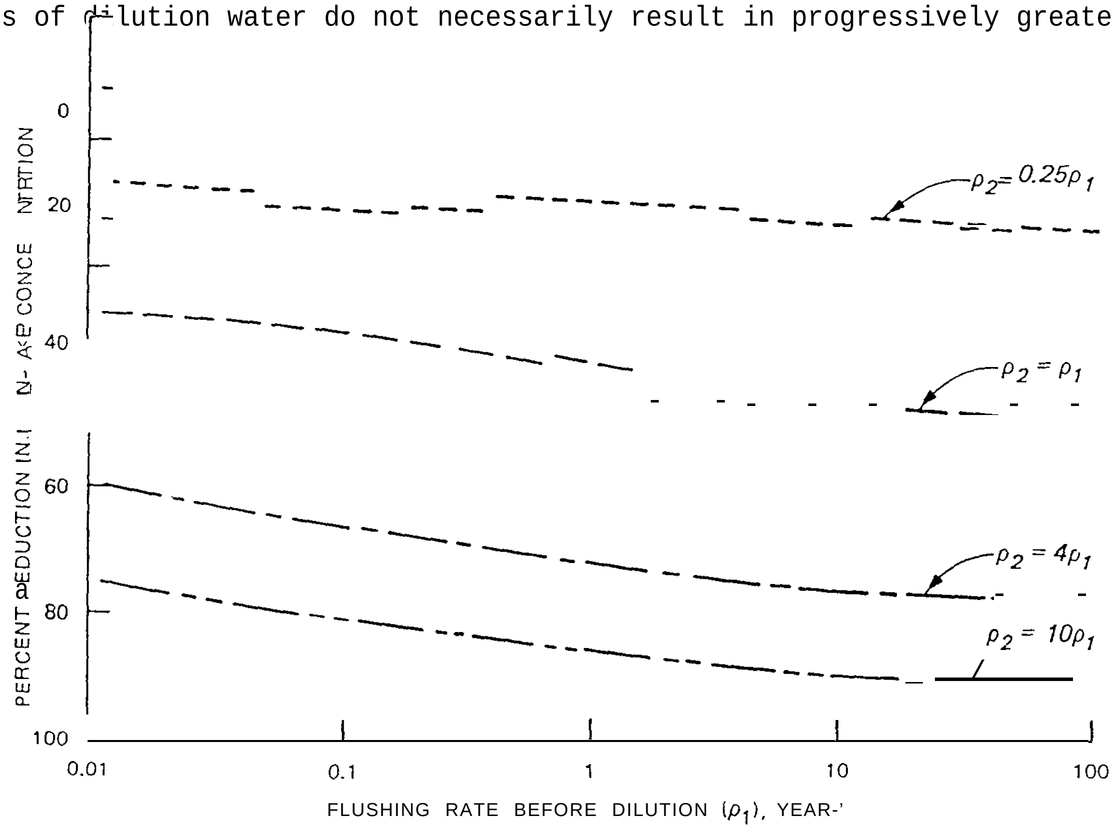

Figure 9. Percent change in in-lake phosphorus concentration following dilution with water containing no phosphorus (after Uttormark and Hutchins 1978). (See text for explanation)

reduction in in-reservoir phosphorus concentration.

is

1.0 year^{-1}, a 30-percent reduction in in-reservoir phosphorus concentration approaches the best possible reduction, even with unlimited dilution water. It can also be seen from Figure 10 that in-reservoir concentration can increase by adding dilution water when the counteracting effect of decreased loss to sediments is considered.

Uttormark and Hutchins (1978) conclude that lakes with low flushing rates are poor candidates for improvement through dilution. In these cases, in-reservoir phosphorus concentration could increase (see Figure 10, low flushing rates, <0.1 year^{-1}) unless the dilution water is essentially void of phosphorus.

The model used here does not account for internal loading. If summer internal phosphorus loading, a common phenomenon in many lakes and reservoirs, is high, then there may be less reduction in concentration than expected. Substantial empirical studies of dilution are greatly needed.

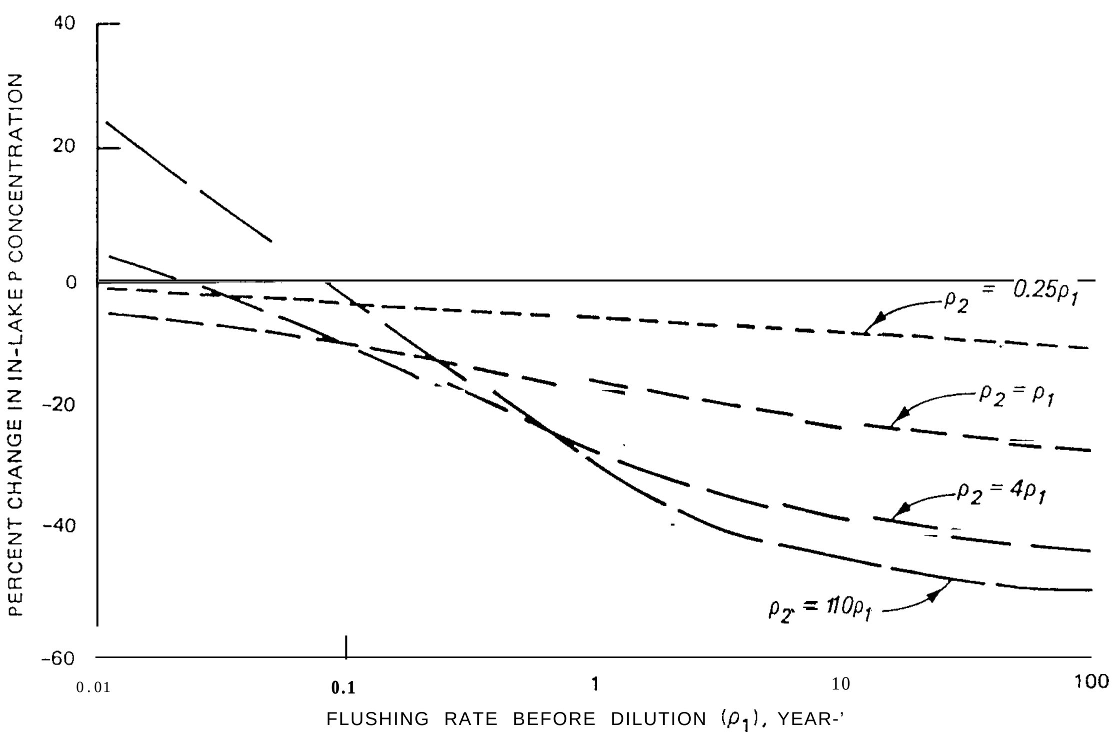

Figure 10. Percent change in in-lake phosphorus concentration following dilution with water having a phosphorus concentration that is 40 percent of the normal, undiluted inflow water (after Uttormark and Hutchins 1978). (See text for explanation)

Flushing, in contrast to dilution, does not require that the nutrient concentration in the inflowing water be less than that of the reservoir. Quantity of algal cells is controlled not by nutrient limitation but by wash-out. The flushing rate therefore must be close to the algal growth rate to be effective. Flushing rates of 10 to 15 percent per day are believed to be sufficient (Cooke et al. 1986).

Effectiveness, Costs, and Feasibility

There are only a few published case histories of the use of dilution/flushing to improve trophic state, and only two have substantial long-term documentation. These two are Moses Lake, Washington (Welch, Buckley, and Bush 1972; Welch 1979, 1981; Welch and Patmont 1979, 1980; Welch and Tomasek 1980; Welch, Brenner, and Carlson 1984; Cooke et al. 1986) and Green Lake, Washington (Sylvester and Anderson 1964; Oglesby 1968, 1969a,b; Welch 1981; Cooke et al. 1986).

The feasibility of this method for reservoir improvement is very limited since an adequate supply of low-nutrient dilution water or high flows of additional water for flushing are unlikely to be available in most instances. Further, even if there is a potential supply of water, its use for reservoir dilution/flushing may be restricted by prior usage of the water. Since this method of reservoir improvement is likely to have limited use, the results of the Moses and Green Lakes studies will only be briefly reviewed.

Moses Lake, Washington

Crab Creek, the primary water supply to this large (2,753-ha), relatively shallow (mean depth, 5.6 m) lake in eastern Washington, has very high nutrient Content. In 1977, dilution water addition to Parker Horn began, using low-nutrient Columbia River water that was diverted through Moses Lake and thence to agricultural areas for irrigation. This produced overall water exchange rates of 0.1 to 0.16 day^{-1} for Parker Horn, and 0.01 to 0.02 day^{-1} for the whole lake. In 1982, dilution water *was* pumped to previously undiluted Pelican Horn from Parker Horn.

The percent lake water in Parker Horn dropped to less than 30 percent when the dilution rate reached 0.15 day^{-1}. Dramatic improvements in lake quality occurred, not only in Parker Horn but in the entire lake. However, it was obvious that algal blooms and low transparency returned quickly if the amount of dilution water declined. This observation led Welch (1981) to conclude that continual low-rate inputs of dilution water over the entire summer were preferable to very high but irregular rates which are above the amount that can produce a decline in nutrients or a washout of cells. When input of dilution water stopped (August 1982), undiluted high-nutrient water rapidly replaced the diluted lake water.

Another effect of dilution in Moses Lake, in addition to creating nutrient limitation, was the effect of cell washout (Cooke et al. 1986). When water was pumped from Parker Horn to Pelican Horn, a sharp decrease in algal biomass occurred, particularly when the water exchange rates reached 0.09 day^{-1}. Similarly for Parker Horn; cell washout became a significant factor when the mean flushing rate was 10 percent day^{-1}. In the remainder of the lake, where flushing averaged 1.4 percent day^{-1}, cell washout was probably not a significant factor because cell growth rates, at maximum, can exceed 50 percent day^{-1}.

The cost of water for dilution at Moses Lake was zero since water already designated for downstream irrigation was simply routed through the lake. The pump for Pelican Horn cost $324,000 (1983 price), plus overhead for operations. If the water had had a cost similar to that of a typical Washington domestic supply, the 2-month cost of dilution water would have been about $2 million.

Green Lake, Washington

Dilution of Green Lake, in metropolitan Seattle, WA, began in 1962. Domestic water was added at a rate sufficient to increase the water exchange rate from an estimated 0.8 to 2.3 year^{-1}. Over the 1965-68 period, the flushing rate, based on dilution water only, ranged from 0.88 to 2.4 year^{-1} (Welch 1981, Cooke et al. 1986). Chlorophyll a, phosphorus concentration, and water transparency improved dramatically, and the fraction of algal biomass composed of blue-greens declined substantially. Water quality declined in the 1970s when dilution was reduced, and blooms of algae returned in 1982 when no dilution water was added. High costs were incurred because domestic water is expensive. It was calculated, using the mass balance models described earlier, that 7.6×10^6 m^3 year^{-1} (269×10^6 ft^3 year^{-1}) of water would be needed to reduce the mean concentration of phosphorus in this medium-sized lake (area = 104 ha, mean depth, 3.8 m) to 20 µg P ℓ^{-1}.

Limitations and Concerns

These techniques are limited by the availability of low-cost, low-nutrient water. Outlet structures of the reservoir must be capable of handling the added discharge. This may not always be the case, particularly with smaller, older impoundments. Also, the downstream impacts of significantly increased discharge must be considered. Finally, dilution/flushing water must have acceptably low concentrations of contaminants such as heavy metals or pesticides.

Summary

Dilution is a reservoir improvement technique wherein amounts of low-nutrient water are added in quantities sufficient to promote cell washout and to significantly lower in-lake nutrient concentration. The amount of reduction in concentration can be estimated, with assumptions that may not hold true for eutrophic reservoirs, from knowledge of nutrient loading, sedimentation, and flushing rate. Flushing is a procedure to wash out algal cells and does not imply dilution of nutrient concentration in the reservoir unless water with low concentration is used.

Both techniques can produce large improvements in trophic state, as illustrated by two case histories. The primary drawback to their use is the availability of the additional water and possible effects of increased reservoir discharge on downstream areas.

Table 4 summarizes this procedure.

Table 4

Summary of Dilution and Flushing

Characteristic	Description
Target	Blooms of algae.
Modes of action	Dilution water decreases in-reservoir limiting nutrient concentration and increases cell washout.
	Flushing increases cell washout.
Effectiveness	Highly effective.
Longevity	Requires continual water input during growing season.
Limitations and applicability	Dam must be structurally sound.
	Water should be free of toxic substances.
	Downstream impacts of greatly increased discharge could be significant.
	Limited due to shortage of additional and/or appropriate quality water.
Costs	Price could be very high if domestic water supply is used.

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PART VI: SEDIMENT REMOVAL

Problem Addressed

External loading of silt and organic matter, along with the deposition of partially decomposed plant biomass produced in the reservoir, can bring about loss of reservoir volume. As well, nutrient-rich sediments are sources of nutrients to the water column and, in shallow areas, provide ideal conditions for macrophytes. Shoaling may interfere with boating, and loss of storage capacity can have severe impacts in potable water supply and flood control reservoirs. Dense macrophyte infestations interfere with recreation and, along with algae, contribute organic matter to the water column. This promotes loss of dissolved oxygen in deep water, and the organic matter may interact with the chlorination step in potable water treatment to produce organohalides such as trihalomethanes. Some reservoir sediments can contain significant levels of toxic substances (e.g., heavy metals, PCBs) from upstream discharges.

Sediment removal is a highly effective method to deepen reservoirs and o remove shoals, and secondarily to remove nutrient-rich or toxic sediments and to control rooted plants. This procedure has been reviewed by Peterson (1979, 1981, 1982) (see also Cooke et al. 1986).

Theory, Design, and Costs

The object of most sediment removal projects is to regain lost'storage capacity, and secondarily to improve water quality by control of internal nutrient release. In the cases of shoaling, loss of volume, or toxic substance contamination, there is little choice except sediment removal. Internal nutrient loading may also be controlled with other methods, and macrophyte control may be better and less expensively handled through harvesting, herbicide treatments, water level drawdown, or biological controls.

Application of the method

There are two means of removing reservoir sediments. First, the reservoir may be drawn down and the sediments allowed to dewater. This is followed by the use of mechanical equipment to remove sediments. Obviously the procedure is limited to those reservoirs where significant and long-term water withdrawal is possible and where sediments will dewater to the degree needed to support heavy equipment. Born et al. (1973) describe the use of this procedure in a small reservoir, and Snow et al. (1979) describe a case history. The more common method of sediment removal is the use of dredges.

The two basic categories of dredges are mechanical and hydraulic, plus some special-purpose designs for the removal of toxic or fine-grained sediments. These are reviewed by Barnard (1978), Peterson (1979), and Cooke et al. (1986). The most common mechanical dredges are the clamshell or grab-bucket designs. Dredges of this type are limited by the requirement that discharge is either in the immediate vicinity of the dredge or into a barge or truck. The bucket types have low.productivity rates, can create uneven bottom contours, and produce significant sediment resuspension. They are highly mobile and can work in small areas. The hydraulic cutterhead dredge is the most common type. Cutterhead dredges are faster than the grab-buckets, may produce less turbidity, and are able to dredge over large distance> due to their floating pipeline discharge system. However, up to 80 percent of the removed material is water; therefore, confined disposal areas must have adequate volumes to permit the settling of suspended materials. In small reservoirs, there could be some drawdown due to the hydraulic dredging process.

There are also specialized and portable hydraulic dredges. Barnard (1978) has described these, and Clark (1983) has reviewed the operating features of 46 models of portable hydraulic dredges. Also, the Oozer and Clean-up dredges (see review in Cooke et al. 1986) have been developed for removal of contaminated sediments, though it appears that these dredges are unavailable in the United States. Several of these dredges produce very little turbidity and thus little dispersion of toxic materials. Herbich and Brahme (1983) report an average suspended solids concentration of 4.0 and 5.7 mg ℓ^{-1} at 3 and 7 m above the bottom for Clean-up and Pneuma-type dredges, versus 40 to 80 mg ℓ^{-1} for conventional hydraulic cutterhead dredges. However, unless the lake sediments are contaminated, the conventional cutterhead dredges may be used with good results.

Analysis of reservoir
sediments and sediment budget

Sediment removal for the purpose of reducing internal nutrient loading requires a predredging analysis of the sediments to determine those areas of the reservoir with the highest release rates. This analysis should also determine the depth in the sediments to which highly reactive or exchangeable forms of phosphorus and othes nutrients extend. The methods of Williams et al. (1971a,b) are recommended for phosphorus. Methods for nitrogen can be found in Chen, Keeney, and Sikora (1979). Release rates should be measured either in situ (Sonzogni et *al. 1977)* or with sediment cores in the laboratory (Lennox 1984). The study should be conducted in a manner that will produce a map of the reservoir which indicates the areas with high release rates. Appropriate statistically based sampling techniques should be applied to ensure that release rates obtained are representative of the areas examined. For a discussion of considerations required in reservoir sampling and monitoring, see Waide (1986). Sediment removal should be to depths resulting in a significant decrease in nutrient release. There is little value to superficial dredging that leaves nutrient-rich layers exposed.

It is important to know how fast the dredged areas of the reservoir will refill with silt and organic matter. If silt loading is high, it may not be cost-effective to carry out sediment removal. Establishment of appropriate land use management techniques or the construction of prereservoir sedimentation basins might be necessary before dredging. Or, use of dredged materials as top soil could reduce costs (see Stout and Barcelona 1983). Evans and Rigler (1980) and Ritchie and McHenry (1985) describe measurement methods for determining sedimentation rates. Or, direct measurement of the net suspended solids income, particularly during storm runoff events, can be made..:

Reservoir sediments in agricultural and industrialized areas may contain PCBs, chlorinated hydrocarbon pesticides, oil and grease, heavy metals, and coliform bacteria. Dredging can release these materials to the water column in association with suspended particulates, and thus the presence of contaminants must be known before initiating operations. Mutagenic substances have been found in reservoir sediments. Allen, Noll, and Nelson (1983) and Lower et al. (1985) describe methods of sediment analysis for mutagenic and toxic materials. Also, an elutriate test (Palermo 1986a,b; 1988) has been devised to evaluate the short-term potential of sediments and disposal area effluents to release hazardous substances into the water column.

Dredge selection

S. A. Peterson's definitive review of sediment removal (in Cooke et al. 1986) provides detailed criteria for the selection of dredge equipment. Additional data are available in Pierce (1970). Since this selection can be highly site specific, the reader is urged to consult these reports.

Containment area design

One of the most common problems with the use of sediment removal is inadequate design of the containment area. Detailed summaries of the procedures for containment area design are found in Palermo, Montgomery, and Poindexter (1978); Montgomery (1978, 1980, 1982, 1984); Averett, Palermo, and Wade (1988); and US Army Corps of Engineers (1987).

The volume of sediments to be removed and the sediment characteristics, such as water content, Atterburg limits, organic content, specific gravity, bulking, grain size, consolidation, and shear strength, must be known. Montgomery (1978) and Averett, Palermo, and Wade (1988) describe the flocculent settling test, which is used to ensure solids retention. These data allow the design of a confined disposal area that will have sufficient volume and area to accommodate continuous hydraulic dredging, and is large and deep enough to allow settling to occur so that the effluent meets suspended solids requirements. Reservoir sediments can be very flocculent, with a low specific gravity (Walsh, Bemben, and Carranza 1984), and the water detention time of the disposal area must be sufficient to allow these materials to settle. If the suspended solids requirement is not met, the project may have to be temporarily stopped or the discharge chemically treated to improve suspended solids removal. In either case, project costs will escalate. Therefore, disposal area design criteria are meant for end-of-project efficiency and not some average or estimated discharge requirements over the entire project period. It is important to note that there is a wide range of settling velocities for sediments so that the use of averages or literature values may produce poorly designed containment areas. The design of a containment area is site specific and should be based. on the laboratory settling test (Averett, Palermo, and Wade 1988).

Determination of sediment removal depth

One objective of sediment removal can be the control of internal nutrient release through the removal of nutrient-rich sediments. Reservoir sediments may have a sharp gradient of nutrient concentrations with depth into the sediments, or a horizontal gradient over the reservoir. A map of this vertical and horizontal gradient should be made, as described in earlier paragraphs. Lake Trummen, Sweden, is an example. It was found (Bjork 1972) that 40 cm of silt had accumulated between 1940 and 1965, an interval during which effluents from a flax mill and a wastewater treatment plant discharged to the lake. Sediments below this layer, under both aerobic and anaerobic conditions, had distinctly lower phosphorus release rates. Thus, the depth of sediment removal was judged to be 40 cm.

Stefan and Hanson (1979, 1980) described another method for determining depth of sediment removal to control internal phosphorus release. They observed that in shallow Minnesota lakes, brief periods of summer thermal stratification produced a sharp loss of dissolved oxygen in the hypolimnion, followed by a high rate of phosphorus release from the anoxic sediments. Like shallow, polymictic reservoirs (see Gaugush 1984 for case history), summer wind storms disrupted the thermal stratification, mixed the lakes, and introduced nutrient-rich water to the whole water column. An algal bloom then occurred. Stefan and Hanson calculated the depth that was required for the lake to remain stratified for the entire summer season. This depth became the target depth for sediment removal. This approach, however, requires a massive volume of sediment removal. Cooke et al. (1986) recommend the approach used at Lake Trummen.

There is a direct relationship between transparency and the maximum depth of colonization by submersed macrophytes. While each plant species may have different light requirements and thus different depths to which it can grow, it is possible to estimate the depth to which a reservoir would have to be dredged in order to control nuisance submersed macrophyte growth through light limitation. Canfield et al. (1985) provide the following equations to determine the maximum depth (in meters) of submersed macrophyte colonization (MDC) for Florida and Wisconsin lakes:

State	N	Equation	Coefficient of Determination
Florida	26	log MDC = 0.42 log SD + 0.41	0.71
Wisconsin	55	log MDC = 0.79 log SD + 0.25	0.57

where SD is the Secchi disc depth in meters.

Thus, a Wisconsin lake with a mean Secchi disc depth of 6.6 ft (2.0 m) should have few submersed macrophytes beyond a depth of 9.8 ft (3.0 m), suggesting that sediment removal in shallow, macrophyte-infested areas to this depth might produce significant relief from these plants. In the Florida lakes, a depth of 11.5 ft (3.5 m) might have to be achieved for submersed macrophyte control.

Effectiveness and Costs

Sediment removal is one of the most effective and commonly used methods of improving reservoirs. In most situations where increased depth or storage capacity is desired, or where toxic materials must be removed, sediment removal is the method of choice. In smaller reservoirs, it may be economically and environmentally feasible to dredge the entire reservoir. As the volume of material to be removed increases, so does cost and, more significantly in many cases, so do problems of disposal. Environmental impacts are often short-lived, or can be minimized, assuming that the method is used properly and that adequate containment areas and discharge treatment are available. Negative environmental impacts are most often associated with disposal, and feasibility for any situation may turn on this issue. Case histories of dredging projects are described in Peterson (1981) and Cooke et al. (1986).

Sediment removal has been carefully examined for costs, and detailed reviews are found in Peterson (1982) and Cooke et al. (1986). Cooke et al. (1986) list six factors that influence dredging costs: (a) type of equipment used, (b) volume of material to be removed, (c) availability of a containment site, (d) density of material to be removed, (e) distance to containment area, and (f) ultimate use of removed materials. Saucier et al. (1978) have indicated that costs are also reflected in the price of land for disposal sites, and the value the dredged material may have as a landfill, wildlife site, or future recreation area. Peterson (1981) reports a cost range for 64 US :ojects of \$0.24 to \$14.00 m^{-3}, with a frequent range for hydraulic dredging projects of \$1.25-\$1.75 m^{-3}. Costs may be reduced through productive or beneficial use of the dredged material (Patin 1981).

Two of the most effective means of controlling internal loading are sediment removal and phosphorus inactivation. Cooke et al. (1986) have compared the cost-effectiveness of these methods, and preliminary evidence suggests that they may be similar when amortized over the effective life of the treatment.

Limitations and Concerns

Sediment removal has high potential for both short- and long-term negative impacts, both at the dredging site and the containment area. Most of these problems are of short duration and can have minimal negative impacts following project completion when containment area design has been proper. Sediments contaminated with toxic materials involve special precautions.

Several possible deleterious actions can occur at the dredging site. ihese include creation of plumes of turbid water, liberation of nutrients (Churchill, Brashier, and Limmer 1975), destruction of benthic organisms (Carline and Bryneldson 1977), and the release of toxic substances (Murakami and Takeishi 1977). At the disposal site, whether in-reservoir or upland, some of these same problems could occur. In addition, in-reservoir disposal may result in burial of organisms and the creation of new and less desirable substrates. Upland disposal can create nuisance conditions 'for nearby residents, contaminate ground water, and discharge toxics in the drainage water. Detailed descriptions of these problems are found in Chen et al. (1978); Gambrell, Kincaid, and Patrick (1978); Saucier et al. (1978); and Peterson (1981). In general, these reports indicate that sediment removal and disposal seldom generate significant negative impacts in the short term, except where toxics such as mercury, cadmium, and chlorinated hydrocarbons are involved. Little is known about long-term impacts. A reader contemplating a sediment removal project is urged to consult these reports, especially Gambrell, Kincaid, and Patrick (1978) and Francingues et al. (1985). A brief review of tential environmental problems and some steps to prevent them follows.

Sediment removal itself will create at least a temporary problem with turbidity, nutrient release, and transport of contaminated particles. Normally, particulates settle rapidly. In some situations, turbidity or the transmission of particulate matter to other reservoir areas is undesirable. In these cases, specialized dredges are available (Cooke et al. 1986), or a silt curtain can be installed (Barnard 1978). Montgomery (1984) describes specialized dredging equipment and procedures that can be used to minimize hazards of sediment resuspension while removing contaminated sediments. Disturbance of nutrient-rich sediments may release significant amounts of nutrients, leading to algal blooms. Nutrient levels should return to normal or even lowered concentrations after dredging. Gibbons and Funk (1983) point out errors in hydraulic dredge operation that can produce reservoir problems. In the case of Liberty Lake, Washington, the paths of the cutterhead did not overlap, resulting in mounds and trenches that later merged through slumping. As a result, nutrient-rich sediments still covered the lake bottom, and neither nutrient release nor macrophyte coverage was improved.

Disposal methods and sites are a very important part of the process of minimizing the environmental impacts of sediment removal, and guidelines for their construction for this purpose are available (US Army Corps of Engineers 1987). Upland containment areas are commonly used. Sediment removal for a reservoir improvement project would be defeated by in-reservoir disposal unless the sediments could be placed in very deep water (25 to 30 m) where currents are minimal, or unless the sediments are placed in a containment area used to create an island acceptable to reservoir users. Unconfined disposal in shallow water means that problems may simply be displaced (i.e., creation of new shoals or creation of another site of nutrient release or macrophyte infestation) or that the undesired sediments will be dispersed by'currents.

Prior to selection of a disposal method, some preliminary data must be obtained. The short-term pollution potential of nutrients, heavy metals, and organics should be estimated with an elutriate test (Palermo 1986a,b; 1988). While most dredged material poses little risk from release of toxic contaminants, the level of such contamination must be known. In the event that the target sediments are contaminated, environmental risks can be minimized. The reader is referred to the reports referenced above for guidelines and methods to control these factors and risks.

Summary

Sediment removal is used for deepening, and secondarily to remove nutrient-rich or contaminated sediments and to control macrophyte infestations. Sediment removal projects require careful planning, design, and construction since the costs may be considerable and there is potential for negative environmental impacts. Planning will include dredge selection, sediment analysis, and containment area design. A summary of the method is given in Table 5.

Table'5

Summary of Sediment Removal

Characteristic	Description
Targets	Shoaled areas. Nutrient-rich or contaminated sediments. Nuisance macrophytes.
Mode of action	Sediments are removed.
Effectiveness	Highly effective; problems eliminated.
Longevity	Years, if dredged deeply and/or sediment income controlled.
Negative features	Temporary turbidity and nutrient release. Improper disposal design may lead to release of toxics, or discharge of turbid water with high turbidity, nutrient content, and oxygen demand.
Costs	High (\$0.24 to \$14.00 m^{-3} for uncontaminated sediments).
Applicability to reservoirs	Highly applicable.

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PART IX: WATER LEVEL DRAWDOWN

Problem Addressed

Water level drawdown is a multipurpose reservoir improvement technique. It is used to control some nuisance plants, to provide access to dams, docks, and shorelines for repair and installation purposes, for fish management, for sediment consolidation and removal, and for installation of sediment covers. Reviews of this procedure are found in Cooke (1980), Culver, Triplett, and Waterfield (1980), Triplett, Culver, and Waterfield (1980), Ploskey (1982), and Cooke et al. (1986).

Theory and Design

Water level drawdown for control of nuisance macrophytes has been used successfully against susceptible species in certain climates of the United States. The objective is to expose the plant to freezing-desiccation or heat-desiccation for a period sufficient to destroy the thallus, roots, and rhizomes, and perhaps some reproductive structures.

Water may be withdrawn for several other simultaneous purposes, including access to structures for repairs and installations or for sediment consolidation through drying or sediment removal with dredges or earth-moving equipment. As described in detail in Part XIII, exposed sediments may also be conveniently covered with screens to eliminate rooted plants.

Ploskey (1983) and Ploskey, Aggus, and Nestler (1984) provide detailed reviews of water level changes and their use in fish management. Actions to benefit fisheries can also produce improvement in the trophic state of eutrophic reservoirs, except in instances when the management of the fish community includes stocking of zooplanktivorous fish, such as gizzard shad, for game fish forage. As described in Part XI, elimination of algae-grazing zooplankton may result from this practice, and the reservoir may experience continued algal blooms. Readers interested in the use of water level manipulations for fisheries should consult the above reports.

Prior to the use of a water level drawdown to control nuisance aquatic plants, a survey of the kinds of plants in the reservoir is necessary because this procedure is species specific. Some plants are eliminated, some unaffected, and others may flourish when water levels are restored. The survey will also identify the area of the plant infestation, thus giving the depth to which water must be withdrawn to achieve partial or complete exposure of macrophytes, Procedures for conducting the plant survey are outlined in Forsberg (1959) and Nichols (1982).

As illustrated in the case histories, most drawdowns for the control of macrophytes in northern states have taken place during winter months. Many plants (Table 10; refer to Cooke 1980 and Cooke *et* al. 1986 for list of responses of 74 plants species) are susceptible to prolonged periods (3 to 4 weeks) of freezing and dewatering. Soils that remain moist or that are not frozen will protect roots and rhizomes, and regrowth will occur.

Water level drawdown will provide access to areas in need of maintenance and repair. Shoreline erosion, a significant source of turbidity in some reservoirs, *is* a problem that can be treated during water withdrawal. Shoreline erosion may be caused by one or several of these factors: waves, abrupt water level fluctuations, erodible and bare soils, ground-water seepage, bluff slumping, and surface runoff erosion. In one of the few studies of its kind, Wilson (1979) found that 82 percent of the total solids income to a small Ohio reservoir was from shoreline erosion. Shoreline stabilization may be brought about through construction of protective structures, planting of vegetation, development of drainage controls from the land, and by altering bluff slopes. Details of these procedures are available through the Soil Conservation Service (SCS) and in several reports (US Army Corps Engineers 1973; Clemens, undated).

Drawdown also provides access for the installation of sediment covers in areas such as beaches and docks. Normally, these materials are applied during summer months using SCUBA (see Part XII). A less expensive and more effective method is to fasten them to frozen reservoir soils.

With regard to eutrophic reservoirs and their improvement or restoration, water level drawdown is used primarily as a procedure to control rooted macrophytes. Further discussion of this procedure will be confined to this purpose.

Table 10

A Summary of the Response of Some Common Nuisance Macrophytes to Drawdown (Modified from Cooke et al. 1986)

A. Species That Usually Increase
   1. *Alternanthera philoxeroides* (alligatorweed)
   2. *Hydrilla verticillata* (hydrilla)
   3. *Najas flexilis* (bushy pond weed)

B. Species That Usually Decrease
   1. *Ceratophyllum demersum* (coontail)
   2. *Elodea (=Egeria) densa* (Brazilian elodea)
   3. *Myriophyllum* spp. (milfoil)
   4. *Najas guadalupensis* (southern naiad)

C. Species That Are Unaffected or Whose Response is Variable
   1. *Eichhomia crassipes* (waterhyacinth)
   2. *Elodea canadensis* (elodea)

Cooke et al. (1986) describe and review case histories for several climatic regions of the United States and provide a detailed summary of the published responses of 74 species of aquatic plants to summer, winter, and annual exposure to water level drawdown. Table 10 is a summary of the responses of some of the most common plants. Some of these case histories are summarized herein to indicate the effectiveness of this method.

Eurasian watermilfoil (*Myriophyllum spicatum*) has become a nuisance in Tennessee Valley Authority reservoirs, particularly those with small (0.2 to 1.0 m) annual water level fluctuations. The herbicide 2,4-D is effective against this plant in cove areas, but winter water level drawdown has been found to be the most effective means of control. The normal 3-m drawdown at Watts Bar and Chickamauga Reservoirs kills all of the plants along well-drained shorelines. Because these reservoirs are multipurpose, the use of drawdowns is sometimes limited (Smith 1971). At Melton Hill Reservoir, 2,4-D

and **high-frequency**, short-duration winter drawdowns are also used for control of *M. spicatum* (Goldsby, Bates, and Stanley 1978).

Water level manipulation was one of the primary methods of plant control in Louisiana, principally because herbicides were too costly, and harvesting tended to spread the infestation (Richardson 1975). Lantz et al. (1964) and Lantz (1974) have described the use of drawdown for plant control in several Louisiana reservoirs. A mid-summer to mid-October drawdown at Anacoco Reservoir opened it to recreation by eliminating water shield (*Brasenia schreberi*) and by controlling parrot feather (*Myriophyllum brasiliense*) and water lily (*Nuphar odorata*). However, *Chara vulgaris* (muskgrass) increased. An infestation of pondweed (*Potamogeton* sp.) and naiads (*Najas guadalupensis*) was reduced from 285 to 16 ha by a winter drawdown at Bussey Reservoir.

The winter drawdown of Lake Ocklawaha (Rodman Reservoir) in central Florida was probably a failure. Some nuisance plants were controlled, such as coontail (*Ceratophyllum demersum*) and Brazilian elodea (*Elodea densa*), but hydrilla (*Hydrilla verticillata*), waterhyacinth (*Eichhornia crassipes*), and alligatorweed (*Alternanthera philoxeroides*) were not. Hestand and Carter (1975) attribute at least some of this response to a mild winter.

Beard (1973) described the successful use of a winter drawdown of Murphy Flowage, Wisconsin. The pondweeds *Potamogeton robbinsii* and *P. amplifolius*, coontail, and milfoil were controlled, and 80 percent of the reservoir was opened to fishing.

However, Geiger (1983) found that the mild, wet winter of the Pacific Northwest (Oregon) was inappropriate for using drawdown to control milfoil, and a herbicide application was finally required to produce the desired control.

Effectiveness, Costs, and Feasibility

Alligatorweed and hydrilla are serious nuisances in some southern reservoirs, and drawdown apparently does not control them (Table LO), while milfoil, coontail, Brazilian elodea, and southern naiad are controlled. The prospective user of this procedure should be aware that responses to drawdown are species-specific and that successful control of some species may mean that resistant ones will proliferate. This problem may be solved by the use of drawdown followed by 1 to 2 years of no drawdown so that natural competitive interactions between drawdown-sensitive and insensitive species remain and the less sensitive do not become dominant. Winter drawdowns are the most successful for northern reservoirs, interfere least with other reservoir uses, and should refill promptly with spring runoff. In general, the long-term effects of drawdown on aquatic plants are poorly understood.

The feasibility of this method for a particular reservoir is dictated largely by its use. Long-term drawdown could not be used in a hydropower reservoir, for example. Many of these reservoirs, however, have few macrophyte problems due to the absence of shallow areas and to the large water level fluctuations over short periods. The level of mainstream reservoirs is dominated by riverflow and the amount of water level manipulation possible is sometimes limited (Ploskey, Aggus, and Nestler 1984). High-frequency, short-duration withdrawals, as used by Goldsby, Bates, and Stanley (197-8) at Melton Hill, could be used for this type of reservoir. Flood control impoundments are good candidates for water level drawdowns, particularly during winter months.

Comparatively very low costs are associated with this procedure, and it is possible that the implementation of other techniques during the drawdown could produce some cost savings in overall reservoir management.

Limitations and Concerns

Ploskey (1983) lists several ways in which drawdown can interfere with reservoir uses, including interference with navigation, access for boaters and swimmers, fishing, and fish management. Most, if not all, of the problems are averted by winter drawdowns and spring refills. Algal blooms after reflooding were reported by Hulsey (1958) and Beard (1973), although the causes of such blooms are poorly understood. High external loading, release of nutrients from sediments, and elimination of competitive effects of higher plants with algae may all be involved. Spring and summer drawdowns can have several negative effects on fishing (Ploskey 1983). These include destruction of littoral food organisms, elimination of cover, and interference with spawning. Also, low dissolved oxygen in the remaining pool can produce a fish kill, or summer drawdowns can eliminate thermal stratification and introduce anoxic waters to the entire reservoir (Geagan 1960, Richardson 1975, Shaw 1983). One very significant problem is the failure to refill due to an unexpected drought or to poor timing of the drawdown relative to expected rainfall. Winter drawdowns for flood control projects have a low probability of refill problems due to the usually high volume of spring runoff. Summer drawdowns can remain low if autumn is dry. An absence of refill problems, minimal negative impact on reservoir users and the fishery, and the best plant control are most likely to be achieved with the use of winter drawdowns, particularly in cold climates.

Summary

Water level drawdown is an effective procedure for the control of certain species of nuisance aquatic macrophytes. Control is achieved through drying and freezing over a period of at least 3 to 4 weeks for projects located in northern areas. Somewhat longer periods may be required for southern projects. Plants that remain in moist soil or in shallow water can be expected to survive.

Drawdown can also be used to implement other procedures, including repair or installation of structures for control of shoreline erosion or for gaining access to dams, docks, and piers. Also, exposed, consolidated sediments can be more easily removed with earth-moving equipment, assuming sufficient consolidation to bear weight, than by removal during normal water level with a hydraulic dredge. Water level manipulation can also be used for fish management and to facilitate the installation of sediment covers.

Negative aspects of this method primarily involve problems of access to water by reservoir users, failure to refill, and possible effects on fisheries. Most of these are avoided at many projects by use of winter drawdowns. This procedure cannot be used at all reservoirs since only some types of operations will permit long-term winter drawdowns. Water level drawdown is summarized in Table 11.

Table 11

Summary of Water Level Drawdown

Characteristic	Description
Targets	Nuisance aquatic macrophytes. Unconsolidated sediments.
Modes of action	Desiccation and freezing of thallus, roots, rhizomes, and other reproductive structures. Sediment consolidation and oxidation.
Effectiveness	Winter drawdown highly effective against some species.
Longevity	Usually effective for at least 1 year.
Negative features	Proliferation of resistant species. Limited access to water during withdrawal. Reduced storage volume.
Costs	Minimal.
Applicability to reservoirs	Highly applicable for reservoirs where operation allows drawdown.

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PART X: HARVESTING

Problem Addressed

Shallow areas of reservoirs often have extensive growths of macrophytes ("weeds"). These plants may inhibit recreation, clog intake structures, release nutrients, and contribute to dissolved oxygen depletions when their tissues, and the dissolved organic matter they may release, are oxidized. Figure 4 illustrates the major interactions of these plants in aquatic systems.

Theory and Design

Harvesting is a procedure by which aquatic plants are cut, collected, and removed from the water. This technique can bring about some control of plant regrowth, open the infested areas to recreation, lower the amount of organic matter in the water column or deposited on sediments, and may contribute to improvement in water quality through the removal of nutrients and organic matter.

Most harvesters in use today are single-stage machines that cut the vegetation with one horizontal and two vertical sickle-blade cutter bars, store the collected plants onboard via a conveyor from the cutterhead to a storage Compartment, and unload the plants at shore via additional conveyors aft of the storage compartment. Machine storage capacities vary from about 6 to 23 m^3 of cut vegetation, depending upon model and manufacturer. Cutter depth is usually limited to a maximum of 2 m, since hydraulic drag is considerable if cutting operations are carried out at greater depths. The operator controls the depth of the cutting bar. Some manufacturers also sell shore conveyor units-that convey vegetation from the harvester unit onto trucks. Also available are transporter units that are loaded by the harvester at the harvesting site and transport the cut vegetation to shore. Most machines are built on pontoons and driven by diesel-powered paddle wheels.

In practice, the cutter bar should be lowered until it is just in. the mud and both root crowns and stems are cut. Most operators attempt to work parallel to shore. Docks, trees, boulders, and other obstacles will hinder operations or damage equipment.

Effectiveness, Costs, and Feasibility

Cooke et al. (1986) have listed the advantages and disadvantages of harvesting for managing nuisance aquatic vegetation. These are:

a. Advantages.

(1) Most harvesting is not regulated by laws, nor is there a waiting period for water use.

(2) Nutrients and organic matter are removed.

(3) Harvesting may facilitate other treatments, such as grass carp or herbivorous insect introductions.

(4) Little impact occurs to nontarget areas.

(5) Costs compare favorably with herbicides in the midwestern United States, but not in southern areas where there are dense infestations of exotic plants such as waterhyacinth (*Eichhornia crassipes*).

b. Disadvantages.

(1) Harvesting is labor intensive.

(2) Relatively small areas can be treated per day.

(3) Fragmentation and spread of nuisance plants may occur.

(4) Harvesting and unloading sites may be separated by great distances.

(5) Operating depths are limited.

(6) Favorable weather is needed.

(7) High initial capital costs occur, and there may be substantial downtime for repairs.

(8) Possible problems of access may occur.

(9) Harvesting is of limited applicability when the growing season is long, regrowth rates high, and infestations very dense'.

The effectiveness of harvesting in producing control of vegetation appears to be related to the number of harvests per season, when harvesting occurs, the types of plants present, the amount of vegetation per unit area, and how the machine is used. Most reservoirs are too large to obtain complete control of nuisance vegetation by harvesting. Therefore, selected harvesting has to be planned. Harvesters can be used effectively in more restricted areas such as marinas, swimming areas, docks, and water intakes, if a machine of proper size is used.

Nichols and Cottam (1972) compared the effectiveness of single and multiple harvests in controlling biomass and next-season regrowth of Eurasian watermilfoil in Wisconsin. A single harvest reduced biomass to 10 to 25 percent of the original level; three harvests 1 month apart essentially eliminated all plant material. Reduced growth in the following year was most apparent in plots harvested three times in the preceding summer, and was apparently due to cutting of root crowns. Wile et al. (1977), Wile, Hitchin, and Beggs (1979), and Conyers and Cooke (1983) also found excellent control in northern lakes if the cutter bar was operated in the sediments so as to cut root crowns. Cooke and Carlson (1986) investigated the effects of harvesting frequency and technique on regrowth of a dense infestation of *Myriophyllum spicatwn*. They found that season-long control of this plant could be achieved with one harvest if the plant root crowns were also harvested. By way of contrast, Anderson (1984) followed the regrowth of Eurasian watermilfoil, in another area of the same reservoir (LaDue Reservoir, Ohio) as Cooke and Carlson (1986), after the plants had been harvested with the traditional method in which 2- to 5-cm stumps are left and few if any root crowns are removed. Anderson found that the milfoil biomass in the harvested area equaled the original biomass and the biomass of a control area within 21 days. While the technique of cutting and removing root crowns may be more time-consuming and can produce damage to cutter blades, the harvested area may not require a second harvest. Further testing of this approach is needed.

Some macrophyte species are more affected by harvesting than others. Nicholson (1981) has suggested that harvesting promoted milfoil growth in Chautauqua Lake (New York) because this plant can spread and become established from fragments. Other species, such as the pondweed *Potamogeton,* which can be a *severe* nuisance in reservoirs, are susceptible to harvesting because they emphasize sexual reproduction and regenerate poorly from fragments. Pondweeds therefore might be replaced by milfoil in harvested reservoirs where both species are present.

Efficiency is related to the density of plants, the size of the area to be harvested, the number of harvesters available, the presence of obstacles in the water, and the distance of disposal sites from harvesting areas. For example, some lakes in British Columbia (Newroth,* Cooke et al. 1986) have very narrow bands of dense milfoil beds along their length. Harvesting was

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* Personal Couununication, 1986, P. R. Newroth, British Columbia Ministry of Environment, Vancouver, BC.

slow and inefficient there since the machine had to travel very long distances to the shore disposal sites. Obviously, efficiency will be increased if disposal sites are located near the harvesting site, or if a separate transporter unit is used with the harvester.

In southern waters, particularly Florida, harvesting is seldom applicable. Here, plant densities may often be over 62 tons ha^{-1} and as much as 370 tons ha^{-1} in dense waterhyacinth infestations, and regrowth rates can be several hectares per day of new vegetation. In these situations, harvesting rates will be slow, disposal costs will increase, harvests will have to be repeated very frequently, and exotic plant growth rates will exceed the rate at which they can be harvested during some periods. In these cases, harvesting is not an effective management option, and reservoir managers may have to rely on other procedures, such as herbicides, or an introduction of biological controls following a harvest or herbicide application.

Harvesting could have an additional restorative or improvement effect for a reservoir when large amounts of nutrients are removed as plant biomass. Removal of nutrients would have to be sufficient to significantly lower the net external nutrient loading or to significantly interfere with the nutrient release that occurs during the autumnal dieback of plants. Nutrient removal to this extent is unlikely in many large reservoirs. Burton, King, and Ervin (1979) have listed the conditions that must be met to accomplish sufficient nutrient removal: (a) macrophyte densities must be high, (b) phosphorus loading to the reservoir must be less than 1.0 g m^{-2} $year^{-1}$, (c) most of the reservoir surface must be covered with plants, and (d) macrophytes must regrow every year. They provide a useful nomograph to estimate the macrophyte harvest required to equal net phosphorus income as a function of percent coverage by plants. While many reservoirs have nutrient loading in excess of the above value, few have complete macrophyte coverage, and the harvest season may be short. As well, complete macrophyte removal could be detrimental to a sports fishery, could contribute to increased turbidity through erosion of littoral sediments on windswept shores, or may stimulate an algal bloom. Harvesting is therefore unlikely to be a factor in improving reservoir trophic state through nutrient removal alone.

Harvesting may lead to improvement in dissolved oxygen conditions through the removal of particulate and dissolved organic matter, which is continually produced by sloughing of plant tissue and by plant decay at summer's

end. Evidence regarding the significance of this removal to reservoir water quality is insufficient. Experiments with enclosures (Landers and Lottes 1983) and calculations from shallow Lake Wingra, Wisconsin (Carpenter 1980), suggest that macrophyte decay is a significant factor to trophic state.

Carpenter (1983) also points out that macrophytes can close a positive feedback loop (see Figure 4) that enhances sediment accumulation and thus macrophyte growth. This occurs through stimulation of algal growth by release of nutrients and organic matter. Sedimentation of dead algal cells and macrophyte tissues adds to sediment accumulation. Since these plants are limited by light penetration, any factor that promotes a decrease in depth, as increased sedimentation would do, will ultimately promote an increase in the area of coverage of macrophytes in the reservoir. Harvesting of plants may be one factor in disrupting this positive feedback loop.

The costs of harvesting are related not only to high purchase price and problems with efficiency, but also with machine breakdowns and the number of reharvests per year. Downtime may increase sharply when an undersized machine is employed or where the equipment is heavily stressed or not operated properly. The British Columbia lakes were an example of high stress on the equipment. A typical operating year there consisted of 2,764 hr of work, of which 44 percent was downtime (Cooke et al. 1986). It is strongly recommended by manufacturers that the machine purchased be of a size appropriate to the area to be harvested, as well as to plant density. Or, the reservoir manager may wish to employ one of the several contract harvesting companies and, in this way, test harvesting as a solution of that particular reservoir's weed problems.

Costs for harvesting vary regionally and reflect differences'in the density of plant infestations, their regrowth rate, and other factors affecting operations. Table 12 is a comparison of harvesting and herbicide costs for the midwest and Florida. Literature cost values have been converted to 1987 dollars by using changes in the consumer price index.* In the Midwest, expenditures for harvesting and herbicides are clearly comparable, but in Florida, harvesting is significantly more expensive (and less effective). Cost comparisons are difficult to make due to wide variances in reporting of

* Personal Communication, 1988, Dr. Thomas Lough, Kent State University, Kent, OH.

Table 12

Comparison Between Midwest and Florida Cost Ranges (1987 Dollars) for Harvesting and Herbicide Treatments of Lakes and Reservoirs

Procedure	Cost Range (per hectare)
Harvesting	
Midwest	\$333-918
Florida	\$734-35,531*
	\$734-2,900**
Herbicides	
Midwest	\$467-905
Florida	\$434-863

Note: Based on data from Koegel, Livermore, and Bruhn 1974, 1977; Culpepper and Decell 1978; Dunst and Nichols 1979; McGehee 1979; Smith 1979; Cannellos 1981; Sassic 1982; Shireman 1982; Shireman et al. 1982; Conyers and Cooke 1983; Sabol and Hutto 1984; Cooke et al. 1986; and Thayer and Ramey 1986.

* Larger number refers to cost for dense waterhyacinth population.

** Larger number refers to cost for dense hydrilla population.

factors such as overhead, labor, transportation, and disposal, and most investigators do not report costs at all so that data are scarce. Further, costs vary with the type of plant infestation. For example, Thayer and Ramey (1986) report a harvesting cost range for *Hydrilla* to be \$1,230 to \$2,900 ha^{-1}, but for waterhyacinth, the cost range is \$10,960 to \$35,531 ha^{-1}.

Good estimates of effort (manpower)', time, and cost of a proposed mechanical harvesting operation can be obtained using the US Army Corps of Engineers' computer model HARVEST (Sabol and Hutto 1984). Input requirements of the model are generally straightforward and easily measured or estimated. Naturally, the more precise the input variables provided by the user, the more precise the results will be. Output includes estimated time and costs, given specific machinery configurations, and density of vegetation growth. Compared with actual operations performed, these time/cost estimates are accurate enough to give a "best-case" estimate of the proposed effort to the planner. The HARVEST program, which runs on a personal computer, can be obtained by contacting the Program Manager, Aquatic Plant Control Research Program, Environmental Laboratory, US Army Engineer Waterways Experiment Station, Vicksburg, MS 39181-0631.

Limitations and Concerns

Carpenter and Adams (1977) have reviewed the environmental impacts of harvesting. Macrophyte removal constitutes habitat removal for organisms such as snails, insects, and young fish, and the abundances of these animals can be sharply reduced. The evidence of a negative impact on fish is conflicting. Haller, Shireman, and DuRant (1980) found that harvesting removed about 85 kg of fish per hectare (76 lb per acre), primarily sunfish. Whereas Wile (1978), Storch and Winter (1983), and Mikol (1984) found that fish populations of lakes remained stable during harvesting. At Saratoga Lake (Mikol 1984) and Chautauqua Lake (Storch and Winter 1983), New York, small sunfish, perch, and bullheads were the dominant fish removed by the harvester. This suggests another possible benefit of harvesting. Removal of sunfish and perch also means removal of the organisms that can have a significant role in size-selective predation on the zooplankton species that graze algae. Ultimately, a significant removal of these small fish may mean lake improvement with regard to algae as well, since herbivorous zooplankton may have increased population density when fish predation on them is lowered.

Some investigators have been concerned about the initiation of algal blooms following extensive plant removal. This has been found in some lakes and reservoirs (Neel, Peterson, and Smith 1973; Nichols 1973; Anderson 1984; Cooke and Carlson 1986), but not in others (Wile, Hitchin, and Beggs 1979). The causes of this phenomenon are very poorly understood. Cooke and Carlson (1986) found much higher phosphorus concentrations in a bay after harvesting, although there is no evidence that this caused the algal bloom. There have been many suggestions of an antagonistic effect between algae and macrophytes (e.g., Shireman et al. 1983) so that removal of macrophytes "releases" algae from these restraints, and blooms occur. The inhibition postulated by these investigators could include shading, release of an inhibitory substance, or removal of cover for algae-grazing zooplankton species. Normally this problem will not occur on large reservoirs where complete vegetation removal is neither desirable nor feasible, and could therefore be an environmental problem only in some coves or bays. Another more significant impact of macrophyte harvesting (or any other procedure that produces plant elimination) may be the increased erosion of the littoral zone (Burton, King, and Ervin 1979). Rooted plants are important in preventing wave-generated erosion and turbidity. Windswept shores and coves should probably not have 100 percent plant removal.

Summary

Harvesting of aquatic plants is a procedure that can quickly improve portions of a reservoir for recreation (such as marinas and swimming areas), and at the same time may interfere with the release of nutrients and organic matter from these plants to the open water of the reservoir. Harvesting in which the cutter bar also cuts root crowns appears to be the best technique in retarding regrowth. Costs can be estimated with the US Army Corps of Engineers' model HARVEST (Sabol and Hutto 1984). The environmental impacts of harvesting are normally minor.

Table 13 is a summary of this technique.

Table 13

Summary of Harvesting

Characteristic	Description
Targets	Aquatic macrophytes.
Mode of action	Cut, collect, and remove plants.
Effectiveness	Moderately effective; symptoms may remain; nutrients and organic matter removed.
Longevity	Weeks to months.
Negative features	Habitat removal. Turbidity. Occasional algal blooms.
Costs	High initial equipment costs. Operator, storage, and maintenance costs may be high. Costs can be estimated through computer simulation (HARVEST);
Applicability to reservoirs	Could not manage infestation over large area unless several machines were available. Poor applicability in some southern waters with dense, rapidly growing infestations of exotic plants. Excellent for coves, marinas, and beach areas.

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PART XI: BIOLOGICAL CONTROLS

Problem Addressed

The biological control of nuisance macrophytes and algae is potentially the most effective method in terms of costs and long-term control. However, our knowledge of how to use biological controls is not well developed.

Biological controls are employed to obtain an acceptable level of plant biomass through the introduction of species that graze or parasitize specific plants, or by the manipulation or elimination of endemic animal species that directly or indirectly control plant growth. The use of biological controls can produce a slow, gradual response that may be long-lasting, in contrast to other methods.

This part describes biological control techniques that are currently available or are undergoing testing and provides general guidelines for their use. Emphasis is placed on phytophagous fish and insects and upon the control of nuisance algae through manipulation of fish populations. Portions of these topics have been reviewed by Schuytema (1977) and Cooke et al. (1986).

Phytophagous Fish

The grass carp (white amur, *Ctenopharyngodon idella* Val.) was introduced to the United States in 1963. Earlier, it was exported to Europe, Japan, and Mexico, primarily as a food fish and later as a controller of macrophytes. The fish's native habitat is in eastern China. The animal has some characteristics which make it desirable as a biological control agent for aquatic plants, including tolerance to low dissolved oxygen and to a wide range of temperatures (Opuszynski 1972). In other ways, this fish is far from ideal, since its tolerances are so high that it cannot be restricted in range and it is not easily controlled or eliminated if it escapes to a nontarget lake or reservoir.

The following paragraphs discuss the biology of grass carp. Information on their reproduction (including the development of sterile triploids), feeding preferences, stocking rates, effectiveness, costs, and associated problems is included. Because grass carp have the potential to drastically alter the biota of a reservoir, and in some states they cannot be used at all, managers are urged to consult with the appropriate state agency before stocking this fish.

Reproduction

A significant question about the use of grass carp in reservoirs, where escape to downstream ecosystems could easily occur, is whether they will infest or reproduce in sites where vegetation is desirable. A marsh or lake that is important for nesting or migrating waterfowl would be an example of this kind of habitat. It was originally believed by early importers of grass carp that the stringent conditions for reproduction would not be found in the United States. Spawning occurs in the deep channels of large rivers following a sharp rise in water level, temperature above 17" C, and a current velocity above 0.6 to 0.8 m sec^{-1}. Survival of fry depends upon a downstream quiescent area where plankton are abundant and predation low (Smith and Shireman 1983, Pauley and Thomas 1987). However, direct evidence of reproduction in Arkansas and Louisiana has been reported (Connor, Gallagher, and Chatry 1980), leading investigators to search for a means of obtaining sterile fish which are as effective in consuming vegetation as the fertile, diploid (both members of each pair of chromosomes in each cell) fish originally introduced to the Nation's waters.

Early attempts to eliminate the possibility of reproduction involved the use of monosex populations and surgically sterilized animals. Shortcomings of this approach included the possibility of an unwanted introduction of animals of the opposite sex and the regeneration of gonads, A second approach involved the intergeneric cross of female grass carp and male bighead carp (*Hypophthalmicthys mobilis*) to produce sterile, hybrid offspring. Two major problems with the hybridization approach were the production of some subvital but fertile diploids and a comparatively low feeding efficiency (Allen and Wattendorf 1987; Bonar, Thomas, and Pauley 1987).

A solution to the problems of the sterile hybrid involved the production of pure (unhybridized) triploid (three members of each chromosome in cells) grass carp. Hydrostatic pressure or high temperature techniques are used to produce nearly loo-percent triploids (Cassani and Caton 1986). Since no known procedure can produce loo-percent triploidy consistently, and because the diploids and triploids cannot be accurately separated by looking at them, fish producers needed a technique to verify that every fish sold is triploid.

Currently, the best technique is to use a Coulter Counter to examine a dropof blood taken from each fish. Triploid red blood cells are larger than diploids, and the Counter easily verifies cell size. Three workers can examine 2,000 to 3,000 fish per day. Triploids are apparently functionally sterile, and there is an extremely low probability that triploids could be a source of a large population of reproducing diploids (Allen and Wattendorf 1987; Allen, Thiery, and Hagstrum 1986).

The production and verification of loo-percent sterile fish has prompted several more states to allow their introduction. As of September 1987, 18 states prohibit grass carp, 12 have no constraints on the use of fertile diploids, 16 allow only triploids, and'4 are studying the triploid prior to release (Allen and Wattendorf 1987).

Food preferences

Grass carp (diploid and triploid) are voracious consumers of aquatic plants but exhibit distinct feeding preferences that seem to vary from region to region in the United States. Table 14 is a summary of these preferences. The data for Florida are extensive, reflecting the longer history of the species' introduction and the number of investigators.

The regional differences in grass carp food preferences could have important management implications. *Elodea densa* is a preferred species in Florida but is not preferred or may not be eaten when plants grown in Oregon-Washington are offered to fish. *Ceratophyllum demersum* is a preferred plant in Florida, variably eaten in Oregon-Washington trials, but not eaten by Illinois grass carp. Feeding trials in Illinois were with diploid fish. The question remains whether palatability of plants varies from region to region, whether there is some genetic basis to carp feeding behavior or whether further studies will demonstrate that these apparent geographical differences are produced by the design of experiments. It appears that when preferred plants such as *Hydrilla*, many species of *Potamogeton*, *Chara*, or native *Elodea* are present, and stocking rates are moderate to low, nonpreferred or noneaten nuisance species that are also present could predominate after several years of grass carp feeding.

It should be noted that major nuisance exotic species, including waterhyacinth, alligatorweed, and Eurasian watermilfoil, are not eaten or may be nonpreferred plants. Much additional research is needed with regard to grass carp feeding preferences and their management implications.

Table 14

Feeding Preference List, in Approximate Order of Preference, for Triploid Grass Carp in Florida, Illinois, and Oregon-Washington Studies

Florida	Illinois*	Oregon-Washington
	Preferred Plants	
Hydrilla verticillata (hydrilla)	*Najas flexilis* (brittle naiad)	*Potamogeton crisps* (curly-leafed pondweed)
Potamogeton illinoiensis (Illinois pondweed)	*Najas minor* (naiad)	*Potamogeton pectinatus* (sage pondweed)
Potamogeton spp. (pondweeds)	*Chara* (muskgrass)	*Potamogeton zosteriformis* (flat-stemmed pondweed)
Najas guadalupensis (southern naiad)	*Potamogeton foliosus* (pondweed)	*Elodea canadensis* (elodea)
Elodea densa (Brazilian elodea)	*Elodea canadensis* (elodea)	*Vallisneria* sp. (tapegrass)
Elodea canadensis (elodea)	*Potamogeton peetinatus* (sago pondweed)	
Chara spp. (muskgrass)		
Lemna spp. (duckweed)		
Nitella spp. (stonewort)		
Ceratophyllum demersum (coontail)		
Eleocharis acicularis (needle rush)		
Pontederia lanceolata (pickerelweed)		
Wolffiella spp. (bog mat)		

(Continued)

Note: Data based on Hestand and Carter 1978; Osborne 1978; Nall and Schardt 1980; Van Dyke, Leslie, and Nall 1984; Sutton and Van Diver 1986; Bowers, Pauley, and Thomas 1987; and Leslie et al. 1987.

* Diploid carp.

(Sheet 1 of 3)

Table 14 (Continued)

Florida	Illinois	Oregon-Washington
	Preferred Plants (Continued)	
Wolffia spp. (watermeal)		
Azolla spp. (azolla)		
Spirodela (duckweed)		
	Variable Preference - May Eat	
Myriophyllum spicatum (Eurasian watermilfoil)	*Potamogeton crispus* (curly-leafed pondweed)	*Myriophyllum spicatum* (Eurasian watermilfoil)
Bacopa spp. (bacopa)		*Ceratophyllum demersum* (coontail)
Polygonwn spp. (smartweed)		*Utricularia vulgaris* (bladderwort)
Utricularia spp. (bladderwort)		*Polygonum amphibium* (amphibious smartweed)
omba spp. (fanwort)		
Fuirena spp. (umbrellagrass)		
Nymphaea spp. (waterlillies)		
Brasenia schreberi (watershield)		
Hydrocotyl spp. (pennywort)		
Panicum spp. (torpedograss)		
Stratiotes aloides (water aloe)		

(Continued)

(Sheet 2 of 3)

Table 14 (Concluded)

Florida	Illinois	Oregon-Washington
	Nonpreferred - Does Not Eat	
Nuphar luteum (spatterdock)	*Ceratophyllum demersum* (coontail)	*Potamogeton natans* (floating leaf pondweed)
Vallisneria americana (tapegrass)	*Myriophyllum* spp.	*Brasenia schreberi* (watershield)
Myriophyllum brasiliense (parrotfeather)		*Elodea densa* (Brazilian elodea)
Eichhornia crassipes (waterhyacinth)		
Altemanthera philoxeroides (alligatorweed)		
Nymphoides spp. (floating heart)		
Pistia stratiodes (waterlettuce)		
Phragmites spp. (reed)		
Carex spp. (sedge)		
Scirpus spp. (bulrush)		
Ludwigia octovalis (water primrose)		
Colocasia esculentum (elephant-ear)		

(Sheet 3 of 3)

The existence of these feeding Preferences suggests the possibility that ass carp could allow nonpreferred plants to become abundant, particularly when understocking or fish escape occurs and only the most palatable species are then consumed. Fowler and Robson (1978) and Fowler (1985) report a shift in dominance from *Potcmogeton* to *M. spicatwn* in an understocked lake. In Deer Point Lake, Florida (Van Dyke, Leslie, and Nall 1984; Leslie et al. 1987; Van Dyke*), a large reservoir stocked in 1976, *M. spicatwn* returned as a problem. Carp have either escaped or have been removed by predation from this impoundment, and the remaining low density of animals has been unable to control the milfoil. This lake has been restocked with grass carp.

Grass carp exhibit a metabolic strategy unlike most fish (Wiley and Wike 1986). Their aerobic metabolism rate is about half that of other fish, but their average consumption rate (at 21" C or higher) is about 50 to 60 percent of body weight per day, which is 2 to 3 times that of carnivorous fish. These two factors offset their very low assimilation efficiency, about one third that of carnivorous fish. About 50 percent of ingestion, on the average, is egested. Triploids have a growth rate of about 9 g day^{-1} (for a l-kg carp fed 420 g of *Potcmogeton crispus* per day). An energy budget for triploid carp is iley and Wike 1986):

$$100I = 13M + 74E + 13G \tag{13}$$

where

I = ingestion

M = metabolism

E = egestion

G = growth

Average daily growth rates from Florida lakes range from 10.0 to 10.4 g day^{-1}. The largest grass carp (a diploid) recorded (Lake Wales, Florida) in the United States weighed 32.7 kg (72 lb).**

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* Personal Communication, 1987, J. M. Van Dyke, Florida Department of Natural Resources, Tallahassee, FL.

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Stocking rates

An appropriate stocking rate is critical to successful aquatic plant management with grass carp. Some managers may choose to stock at a rate that will produce plant eradication. This strategy is achievable with grass carp but is also associated with water quality changes and is unlikely to be compatible with sport fishing. Stocking rates are directly affected by the length of the growing season, water temperature, palatability of the plants, size of fish stocked, present and desired plant coverage, and preapplication plant management procedures. Fewer fish are needed when the feeding season is long, the nuisance plants are palatable, the water body is only partially plant-covered (perhaps due to harvesting or chemical treatment), and the management plan calls for some vegetation to remain. Also, serial stocking, where fish are added at intervals, usually requires fewer fish than batch stocking and also allows better management of the rate of plant reduction.

Aquatic plant infestations vary in coverage, biomass of plants per unit area, and species. Grass carp stocking rates such as "rule-of-thumb" formulas (e-g., 10 fish per acre of reservoir), which do 'not take these factors into account, can produce a density of grass carp that is too low when there is a dense, widespread infestation of a nonpreferred or unpalatable plant (e.g., *Myriophyllum spicatwn)* or too high when opposite conditions are found. In one case, plant control may not be achieved. In the other case, eradication can occur.

Stocking rate guidelines, based upon reservoir-specific variables, have been developed (Bonar, Thomas, and Pauley 1987; Leslie et al. 1987; Wiley, Tazik, and Sobaski 1987), and these should be consulted. The most accurate stocking rates are developed from a program of data acquisition and the use of one of the computer models. Models in use now include (a) the US Army Engineer Waterways Experiment Station model (Miller and Decell 1984), (b) the Illinois Herbivorous Fish Stocking Simulation System (Wiley and Gorden 1985), and (c) the Colorado model (Swanson and Bergersen 1986). A model for the Pacific Northwest is under development (Pauley and Thomas 1987). Reservoir managers planning to introduce *grass* carp are strongly urged to consult a model appropriate to their area or to examine reports by Leslie et al. (1987) and Wiley, Tazik, and Sobaski (1987).

The effectiveness of grass carp is also related to sources of carp mortality. Fish less than 450 mm total length are susceptible to largemouth bass predation in Florida (Shireman, Colle, and Rothman 1978). Also, grass carp may escape in large numbers from some reservoirs, and barriers may have to be constructed.

Case histories

Detailed case histories of grass carp additions to large impoundments are not abundant, and most are from southern states because problems are most severe there and carp have been under study for years. Additional reports will become available for midwestern and far western lakes and reservoirs in the near future.

Lake Conroe, an 8,100-ha impoundment near Houston, TX, was filled in 1973 and is used for recreation and potable water supply. By 1981, *Hydrilla* had infested about 44 percent of its area, along with Eurasian watermilfoil and coontail. In 1981, triploid grass carp were stocked at a rate of 75 per vegetated hectare (about 23 kg ha^{-1}), a rate considered to be "overstocked." Two years later, nuisance plants had been eliminated. There was also a 40-percent reduction in water clarity due to a phytoplankton bloom in the year following plant eradication (Martyn et al. 1986; Noble, Bettoli, and Betsill 1986). The treatment was a success with regard to protecting lakeshore property values and enhancing recreation. Impacts on drinking water quality have not been reported.

Lake Conway, a 729-ha multipool impoundment in Orlando, FL, was stocked with diploid grass carp at a rate of about 15 fish per vegetated hectare (about 6 kg per vegetated hectare). In 2 years, *Hydrilla* and Nite *lla were* eliminated in all pools, although *Vallisneria* was unaffected. The treatment is considered to be a success. There was minimal negative impact, although blue-green algae increased and waterfowl population decreased. Fishing for largemouth bass improved dramatically (Miller and King 1984).

There are several useful case histories of grass carp introduction to small impoundments (e.g., Mitzner 1978) and urban recreational lakes (e.g., Van Dyke, Leslie, and Nall 1984; Leslie *et* al. 1987).

costs

Grass carp may be the least expensive means of achieving long-term control/eradication of some nuisance aquatic plant species (see Table 14 for lists of preferred and nonpreferred plants). In southern waters, in particular, plant harvesting is often either ineffective or too expensive, leaving herbicides and biological controls as primary alternatives. Shireman (i982) and Shireman et al. (1985) provide the following case history which demonstrates the comparative low cost-high effectiveness of grass carp.

At Lake Baldwin, an 80-ha lake in Orlando, FL, about 80 percent of the area was infested with *Hydrilla*. About $100,000 was spent over 3 years to control the plants with HYDOUT (an endothall salt), at an average annual cost of $33,333 for 64 ha ($520 ha^{-1}). Treatments provided control but had to be repeated annually. In 1975 and 1979, grass carp were added at a total stocking rate of 35 fish per vegetated hectare and a cost of $8,499 ($106 ha^{-1}). There was no appreciable growth of aquatic plants between 1980 and 1985. When costs are amortized over the expected duration of effectiveness (10 years or more*), they fall to $11 ha^{-1} (1975 dollars) or $22 ha^{-1} in 1987 dollars. Over that same lo-year interval, had herbicide treatments continued with no inflation of chemical and labor costs, about $333,333 would have been spent.

Environmental Impacts

The positive aspects of the use of grass carp to control aquatic plants are easily identified. These include elimination of the target plants (and often all plants); very low initial and long-term costs; low maintenance costs (unless fish removal is required); greatly enhanced boating, water skiing, and sometimes fishing; and long-term effectiveness. These animals can approach the "ideal" biocontrol agents when the system to be treated is large (making other management procedures very costly) and when the nuisance plant is a preferred species such as *Hydrilla*.

The negative effects are less easily identified, in part because so little is known about the long-term consequences of the elimination of all macrophytes from a lake or reservoir. Eradication of submersed and emergent plants is not uncommon when stocking densities are high; it has so far proven difficult to stock just enough carp to allow a certain desired percentage of plant cover to remain, but stocking models (e.g., Wiley, Tazik, and Sobaski 1987) may correct this problem. With techniques such as harvesting or herbicides, far more precise control of plant densities is possible. In many cases, distinct water quality changes accompany macrophyte control or eradication by grass carp. These changes almost always include increased

* Personal Communication, 1987, J. M. Van Dyke, Florida Department of Natural Resources, Tallahassee, FL.

turbidity and shoreline erosion, perhaps because macrophytes previously damped wave action. Erosion in some carp lakes has been extensive enough to uproot trees along the shore. In some cases, there have been increases in algae, particularly nuisance blue-green algae, and a replacement of phytophilic invertebrate species with bottom dwellers. A major concern had been the potential impacts of the high rate of egestion of organic and inorganic matter. There appears to have been no report of a "pulse" of nutrients following carp introduction, in contrast to procedures such as herbicide treatments where very large amounts of plant biomass may be left to decay. Instead, there appears to be a continuous transformation of plant matter to fish tissue and *to* egested materials so that some nutrients in plant biomass are lost to a "sink" (fish tissue) and the rest to decomposing organic matter.

There have been concerns about negative impacts to other fish species. Some'investigators have reported enhanced fishing success following elimination of vegetation (e.g. Bailey 1978, Miller and King 1984), while others have found interferences with spawning and growth (e.g., Ware et al. 1975, Forester and Lawrence 1978). Further studies in this area are required. There is some speculation that a slow release of nutrients from decomposing carp fecal material can be a subsidy to the planktonic food web and in this way contribute to enhanced energy flow to game fish.

Overstocking of grass carp can present other problems in addition to the direct effects of plant eradication. For example, waterfowl are dependent on aquatic plants for food. Lakes and reservoirs on significant flyways may be important resting/feeding sites and could be avoided if plants were eradicated. Further, if significant numbers of carp escape to downstream sloughs and marshes, further damage to desirable communities could occur.

Once stocked, management of fish density has proven to be very difficult in Florida waters.* Carp apparently avoid nets and traps and can escape electroshockers. It appears that once a lake or reservoir is stocked, users of the reservoir are committed to their decision until the fish die (10 years or so).

In summary, triploid grass carp are clearly an effective, low-cost, long-term agent for the eradication of some nuisance aquatic plants. Their use in the management of macrophytes to achieve a desired coverage has prover.

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to be more difficult, and significant environmental changes can occur after their stocking. A nationwide research program is now under way to establish regional food preferences and stocking rates and to assess the effectiveness of carp introductions.

Tilapia zillii, another phytophagous fish, has been used successfully to control rooted plants and filamentous algae, but there are few literature reports concerning this fish. A distinct characteristic of this species is its intolerance to prolonged exposure to temperatures below 10" C. This limits the use of this species for northern lakes or reservoirs. Childers and Bennett (1967) report that a *Tilapia* density of 1,000 acre^{-1} (2,500 ha-l) was sufficient to control *Chara, Potamogeton foliosus,* and filamentous algae in an Illinois pond. This fish was used to control *Utricularia vulgaris* (bladderwort) in a North Carolina cooling water impoundment. A stocking density of 50 acre^{-1} (124 ha^{-1}) eliminated the problem with bladderwort, but blue-green algae and rooted plants then invaded the lake (Schiller 1984). *Tilapia* has also been used successfully to control *Elodea densa* in Hyco Reservoir, North Carolina (Schiller 1984).

Insects and Plant Pathogens

The history of the deployment of insects and plant pathogens as biocontrol agents of nuisance exotic vegetation is in striking contrast to that of the grass carp. Alligatorweed (*Alternanthera philoxeroides)* and waterhyacinth *(Eichhomia crassipes)* are exotic plants that have become nuisances of great economic significance in southern waters of the United States. Several cooperative investigations, involving scientists from several universities, the US Department of Agriculture, and the US Army Corps of Engineers (e.g., Coulson 1977, Cofrancesco 1984, Balciunas 1986), revealed species of insects in the native habitats and ranges of these plants that might be imported to the United States and released as biocontrol agents. These insects were kept under strict quarantine while studies were conducted to assess their specificity to the target plant and the possibility that their introduction might also introduce parasites and diseases.

Six species of insects have been imported and released for plant control. *Neochetina eichhomiae, Neochetina bruchi* (Coleoptera:Curculionidae), and *Sameodes albiguttalis* (Lepidoptera:Pyralidae) were imported from

Argentina, a native habitat of waterhyacinth. *Neochetina eichhomiae* is ligatorily monophagous on hyacinth, and *N. bruchi* is found on plants of only wo genera. Its distribution does not exceed that of waterhyacinth, and its life cycle can be completed only on the hyacinth (Perkins and Maddox 1976, Center 1981). The first insect to be released in the United States following importation, quarantine, and testing was the alligatorweed flea beetle *Agasicles hygrophila* (Coleoptera:Chrysomelidae). Two other species have been introduced for alligatorweed control, *Vogtia malloi* (Lepidoptera:Pyralidae) and *Amynothrips andersoni* (Thysanoptera:Phlaeothripeae). These species are also confined in their feeding or life cycle to the target plant (Coulson 1977).

Insects have been successful in exerting control of alligator-weed and waterhyacinth in several states. The primary control of alligatorweed in Florida, Louisiana, Mississippi, and south Alabama is by the alligatorweed flea beetle *A. hygrophila* and the alligatorweed stem borer *V. malloi*. Both have contributed to control in Georgia, but their impact is limited in North Carolina because the colder weather inhibits or completely prevents over-wintering of the insects (Cofrancesco 1984). The alligatorweed thrips *mynothrips andersoni)* over-winter better and thus stress the plants earlier in the season. They also attack the terrestrial form of this plant, while the flea beetle and stem borer require the aquatic form (Cofrancesco 1984, McGehee 1984). *Neochetina eichhorniae* and *N. bruchi* have been released in Florida, Georgia, Louisiana, Mississippi, Texas, and California for control of water-hyacinth, and effective control has been reported from Florida, Louisiana, and Texas (Center and Durden 1984). *Sameodes albiguttalis*, the waterhyacinth moth, has been found to be most effective against small, luxuriantly growing plants, such as might occur following a chemical or mechanical treatment (Center and Durden 1984).

The impact-of the two weevils on waterhyacinth in Louisiana serves as an example of insect effectiveness (Perfetti 1983, Goyer and Stark 1984, Sanders 1984). The distribution of waterhyacinth reached a statewide peak of 690,000 ha in 1975. Ey 1980, the beetles had reached swarming levels. and dry weather had concentrated the plants. Plant distribution declined to 122,000 ha in 1980; the insects, along with the drought and increased efficiency of herbicide spraying, are believed to have been the major factors.

Goyer and Stark (1984) reported that beetle densities as low as one individual per plant had an adverse effect on plant height and number of reproductives.

Plant harvesting or herbicide application can sharply reduce the effectiveness of an established population of *Neochetina* on waterhyacinth. After such treatments, insects disperse and the plants regrow in log phase without control. It may then take several years for insect densities to reach effective levels (Center and Durden 1984, 1986). Haag (1985) has found that diquat, 2,4-D, and glyphosate were nontoxic to the weevils at low doses and that adult insects will migrate to nearby untreated plants. Integrated methods of aquatic plant management may ultimately prove to be among our most successful approaches. Haag (1986) describes an experiment in which insects and a herbicide were used together to control a waterhyacinth infestation in a pond. About 75 percent of the weed mat was sprayed with 2,4-D, in increments of 20 percent, and both species of waterhyacinth weevils were "herded" into an unsprayed area where they overwintered, increased sharply in density, and exerted complete control of the weeds in the next spring. Additional experiments with this approach are in progress in larger lakes and reservoirs.

Some native insect species are known to damage aquatic plants. Work is now in progress to identify these species and to determine their usefulness in plant control (Buckingham, Haag, and Habeck 1986; Haag, Habeck and Buckingham 1986; Habeck, Haag, and Buckingham 1986).

Plant pathogens should be ideal for control of rooted aquatic plants because they are numerous and diverse, usually are host specific, easily disseminated and self-maintaining, exert a limiting influence on target plants without eradication, and normally are not dangerous to man and domestic animals (Zetter and Freeman 1972; Freeman, Charudattan, and Conway 1975; Freeman 1977). Only one plant pathogen has been significant in plant control, the fungus *Cercospora rodmanii*, which was isolated from waterhyacinth in Rodman Reservoir, Florida. Pilot field tests were held in Louisiana in 1977-80, and a commercial formulation has been developed by Abbott Laboratories (Freeman et al. 1981; Perfetti 1983). The effectiveness of the fungus in Louisiana cannot be separated from the impacts of the beetles, but it is believed that its effect on waterhyacinth has been less than expected (Sanders 1984). Another species, *Cercospora piaropi,* was generally believed to produce only moderate plant damage. Martyn (1985), however, has documented a situation at Lake Conroe, Texas, where this fungus has apparently caused widespread damage to waterhyacinth. There are other fungal pathogens, including *Fusarium roseum* for control of *H. verticillata* (Freeman, Charudattan, and Cullen 1980) and *Fusarium sporotrichoides* for control of milfoil, *M. spicatum* (Andrews and Hecht 1981). Neither is operational.

The application of insects and/or plant pathogens may produce a reduction of plant biomass slowly, and control may be long-lasting. Few negative side effects are expected. Research is continuing with this type of biological control.

Biomanipulation

Biomanipulation is a term coined by Shapiro, La Marra, and Lynch (1975) and Shapiro (1978), although one of the first observations that algal blooms were absent in ponds with a certain type of fish community was made by Caird (1945). Biomanipulation includes some potentially effective but currently experimental procedures to control algal biomass. Among them is the manipulation or management of food webs to control fish species that recycle nutrients during browsing and feeding, or that promote algal growth through thkir predatory activities on microscopic animals (zooplankton) which graze on algae. These procedures may be difficult to implement in large reservoirs without continual management due to the frequent introduction of undesirable fish species, and because fish management is difficult in such habitats. However, it appears that food web manipulation can improve eutrophic systems. When biomanipulation is combined with control of nutrient loading, major declines in algal abundance can be expected (Benndorf 1987). Fish stocking programs should be initiated with regard for problems of algal biomass, as well as sports fishing. This means that the addition of gizzard or threadfin shad as forage for game fish, for example, should be undertaken with great caution. These fish preferentially consume the species of zooplankton which are the major grazers of some species of algae. Reservoir manipulations such as an occasional extreme drawdown can provide an ideal opportunity to restructure the fish community.

Figure 14 illustrates the open-water food chain or web (Shapiro et al. 1982). Briefly, zooplanktivorous fish (e.g., gizzard shad, alewives, perch, and small sunfish) graze on the largest species of zooplankton. These zooplankton are the most efficient grazers of algae, and their absence eliminates

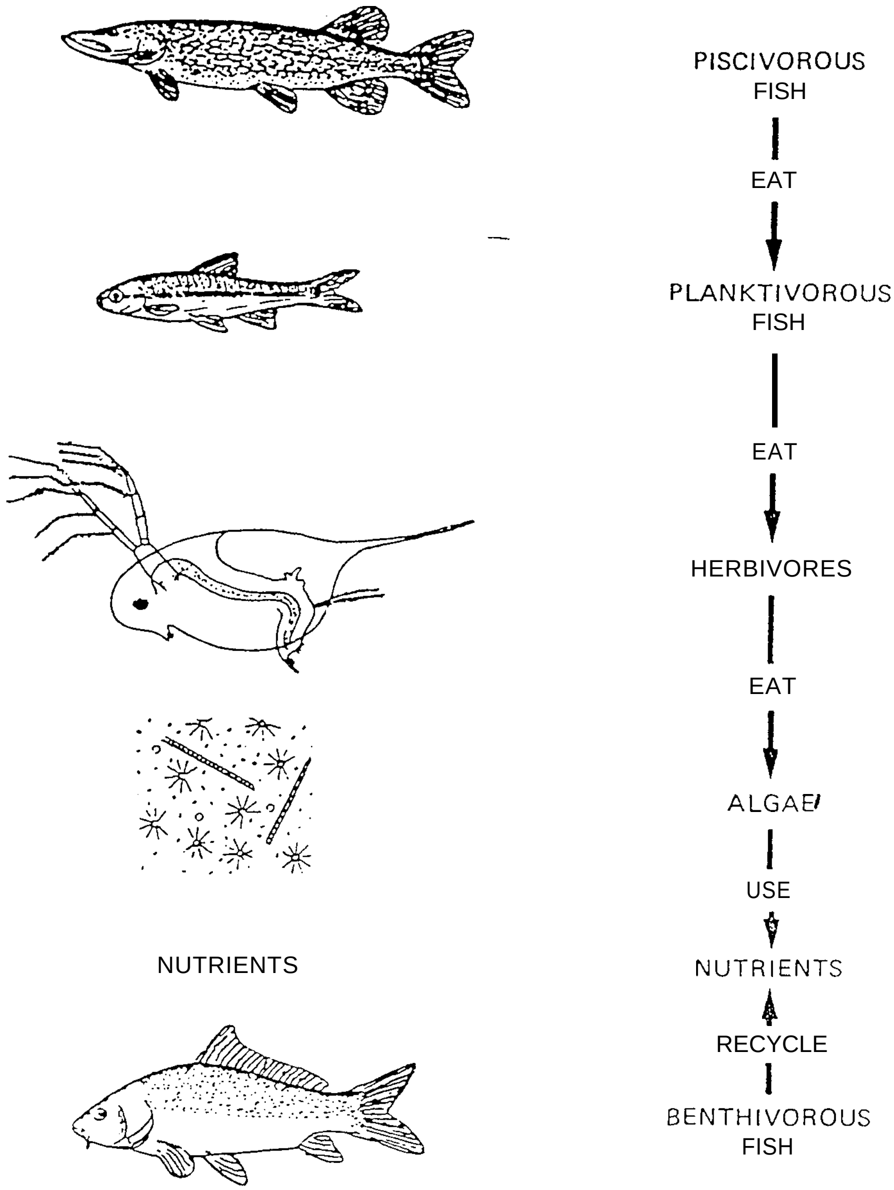

Figure 14. The aquatic food chain, indicating interactions between the components of the biomanipulation model (after Shapiro et al. 1982). See text for an explanation of the biomanipulation model

a significant source of mortality to algae. In a number of experiments with enclosures or small lakes, fish have been eliminated either by winter fish kill or by rotenone application (e.g., Lynch and Shapiro 1981, 1982; Shapiro and Wright 1984; Stenson et al. 1978). In these experimental systems, blooms of some algal species have been eliminated through zooplankton grazing alone, even under conditions of nutrient enrichment. In controls, where fish predation has eliminated the most effective algal grazers, leaving a system dominated by rotifers and small. Cladocera, algal blooms and low water transparency continue.

Similar effects have occurred in macrophyte-dominated lakes to which grass carp have been introduced. When macrophytes are eliminated, the zooplankton community may shift from one dominated by large-bodied Cladocera, or Copepoda-Cladocera, to one dominated by Rotifera and the small-bodied Cladocera, Bosmina and *Ceriodaphinia*. The algal blooms that sometimes accompany the elimination of macrophytes (whether through grass carp or through mechanical or chemical methods) are often attributed to increased nutrient availability. They may in fact be due to the elimination of cover for zooplankton, followed by intense predation by fish and the elimination of effective algal grazers. Or, algal blooms may in part be explained not by the low grazing rate of small-bodied zooplankton, but by the high rates of nutrient cycling associated with small-bodied zooplankton (Henry 1985).

A reduction in zooplankton grazing on algae may also occur when dissolved oxygen in deep water is low or absent. Zooplankton migrate to the cooler, darker waters of the metalimnion and hypolimnion during the day to escape predation by sight-feeding fish, and then return to surface waters to graze at night when sight-feeding by fish is less (Vuorinen, Rajasilta, and Solo 1983). Low dissolved oxygen will restrict this daily migration and permit effective fish feeding on zooplankton during daylight. As discussed in another section, one of the benefits of artificial circulation and hypolimnetic aeration may be to provide an oxygenated refuge for zooplankton from the sight-feeding of fish.

Any factor that produces significant and prolonged zooplankton mortality may bring about persistent algal blooms, low transparency, and associated problems with dissolved oxygen and quality of potable water. Shapiro (1979) points out that agricultural runoff can have significant concentrations of pesticides that are lethal to zooplankton in trace amounts. He lists 10 commonly used Insecticides, including malathion, diazinon, and Baytex, for which the 48-hr LC50 concentration to zooplankton is less than 1.0 $\mu g\ \ell^{-1}$. Copper sulfate, the most commonly used algicide, is very toxic to zooplankton (DeMayo, Taylor, and Taylor 1982). Winner and Farrell (1976) and Winner et al. (1977) found significant mortality to species of *Daphnia*, an effective algal grazer, at a copper concentration of about 40 µg Cu ℓ^{-1}, a Concentration well below that normally used for algal control.. Chelated copper algicides are somewhat less toxic to *Daphnia* (Biesinger, Andrew, and Arthur 1974). The often observed "rebound" of algal biomass shortly after a copper treatment may be due to mortality of herbivorous zooplankton.

Fish have been shown to be very significant in the regeneration of nutrients from sediments to the water column. The common carp (*Cyprinus carpio*) can add phosphorus to the water column of lakes at rates similar to the external loading (La Marra 1975). Similar findings have been made about the brown bullhead (*Ictalurus nebulosus*) by Keen and Gagliardi (1981). Control or removal of these rough fish can improve water clarity by elimination of their bottom-browsing activities and by lowering the rate at which phosphorus is recycled from sediments to water and then to algae.

Several fish management techniques can produce an improvement in the trophic state of the reservoir. The first step is to evaluate the reservoir, as outlined in an earlier section, to establish the types of fish and zooplankton and thus the likelihood that improvement through fish manipulation can occur. The addition of piscivorous fish to control planktivorous fish may meet with limited success, if any (Bennett 1970), but this idea has had little study as far as trophic state improvement is concerned. Rough fish removal by seining has also met with little success, primarily because the technique is labor intensive and very inefficient.

The use of rotenone, a fish poison, to eliminate all fish may be the only feasible procedure to correct fish problems, as illustrated by the work of Shapiro and Wright (1984). Round Lake, Minnesota, a small, shallow, natural lake, was treated with rotenone to eliminate a fish community dominated by planktivorous bluegill (*Lepomis macrochirus*), black crappie *(Pomoxis nigromaculatus)*, and the benthivorous black bullhead (*Ictalurus melas*). Following treatment, the piscivorous largemouth bass (*Micropterus salmoides*) and walleye (*Stizostedium vitreum*) were added, along with channel catfish (*Ictalurus pumetalus*) to control the reestablishment of the black bullhead, a fish that can increase internal nutrient loading. A marked improvement in transparency occurred. Prior to rotenone application, the mean summer Secchi disc transparency was 2.1 m; in the two subsequent summers, it averaged 4.8 m and 4.7 m, respectively. *Daphnia pulex,* a large-bodied herbivorous zooplankton species that was rare before elimination of the planktivores, became abundant.

The use of rotenone to eliminate fish in a reservoir may not be feasible due to the large area to be treated, the possibility of damage to downstream communities, the possibility of reinvasion from upstream, and the use of water for drinking. According to Bennett (1970), a dose of 1.0 mg ℓ^{-1} will produce a-complete fish kill if applied when the weather is warm (water temperature 20" C or warmer). No toxicity to fish should remain after 7 days, although this must be checked by placing caged fish in the reservoir. Rotenone may not be used in potable waters.

Water level drawdown can be very effective in restructuring fish communities, as well as in eliminating certain species of macrophytes (see Part IX). Pierce, Frey, and Yaun (1963) and Lantz (1974) describe the enhancement of piscivorous fish populations through drawdown. Small fish are trapped in vegetation and killed. Forage-size bluegills, which are among the planktivores, decline conspicuously, apparently due to bass predation in the remaining pool (Herman, Campbell, and Redmond 196.9; Bennett 1970). Lantz et al. (1964) report that winter drawdown can remove gizzard shad and sunfish and that summer drawdown can prevent their spawning. Drawdown when spring water temperature reaches 10° to 15" C has been used effectively to kill common carp eggs (Shields 1958). Hulsey (1958) recommended that new reservoirs have a deep channel or harvest basin that will support fish during drawdown. Then, seining or use of rotenone could take place without endangering downstream fish communities.

Restocking of a reservoir must be done with the objectives of the reservoir users in mind. Highly productive water bodies are usually ideal for fishing, but often have low transparency, high algal biomass, and problems associated with dense algae. Fish stocking in a fishery reservoir should be directed toward that activity. However, in those cases in which other types of recreation are important, a drinking water supply is involved, or an oxygen-free hypolimnion poses a problem, fish stocking to enhance zooplankton grazing and stocking to reduce or eliminate nutrient recycling by fish such as common carp or bullheads would be an important consideration. The use of planktivores such as shad as forage for game fish would be inconsistent with an objective of high water transparency or potable water free of taste and odor.

Summary

Biological control of nuisance algae and macrophytes is a rapidly developing area of lake and reservoir management. A number of insects and plant pathogens are operational and have been proven to be successful in control of waterhyacinth and alligatorweed. Restructuring of fish communities offers great promise for algal control but may be impractical for large reservoirs.

The introduction of phytophagous fish (primarily sterile, triploid grass carp) for control, or more likely elimination, of macrophytes has proven to be successful and inexpensive. Significant problems can be associated with their use. Grass carp are operational in several states.

Table 15 is a summary of biological control techniques.

Table 15

Summary of Biological Controls

Characteristic	Description
Targets	Macrophytes and algae.
Modes of action	Grazing by fish, insects, and zooplankton. Infection with pathogens.
Effectiveness	Variable, dependent on biological control agent and target plant.
Longevity	Short- to long-term.
Limitations	Possible replacement of one noxious plant species by another. Few proven biological agents are currently available. Environmental conditions can control distribution and effectiveness of control agents.
Costs	Potentially less expensive than typical mechanical or chemical controls.
Applicability to reservoirs	Questionable due to open nature of some reservoirs.

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PART XII: SEDIMENT COVERS

Problem Addressed

Sediment covers are used=to control or eliminate nuisance rooted plant growth, particularly in limited areas of reservoirs such as coves, marinas, swim beaches, and boat docks. The procedure can provide'reservoir users with immediate and long-term access to areas previously choked with plants. This method of plant control has been reviewed by Cooke (1980).

Theory and Design

Sediment covers control the growth of rooted aquatic plants in several ways. Some covering materials are actually screenlike in nature so that gas bubbles do not accumulate under them. The screens appear to operate by physical limitation of the plants, since light can penetrate the screen (Perkins, Boston, and Curren 1980). Other materials are completely opaque and may exert control by light limitation also.

Engel (1982) lists these advantages of the use of sediment covers for rooted plant control:

- Use is confined to specific areas.
- Covers are usually out of sight.
- Covers can be installed where harvesters or sprayers cannot gain access.
- No toxic substances are used.
- Permits or licenses are usually not required.
- They are easily installed over small areas.

Engel (1982) cites these disadvantages to the use of sediment covers:

- They do not address the cause of the problem of excessive rooted plant growth.
- They are expensive.
- They are difficult to apply over large areas or over areas with obstructions.
- They may slip on steep grades or float to the surface if gases accumulate beneath them.
- They can be difficult to remove or relocate.

- They may rip during application.
- Some materials are degraded by sunlight.

The successful use of sediment covers to control rooted aquatic vegetation is directly related to the technique of application, as well as to the selection of materials. Mater-ials that have been used as sediment covers in reservoirs and lakes include polyethylene, polyvinyl chloride, polypropylene, nylon, fiberglass, synthetic rubber, and burlap. Some of these materials are highly unsuitable; some are very effective.

The cover should be applied directly on the sediment surface without "ballooning" or bare spots. In many cases, especially with soft, muck sediments, this will be very difficult' since stakes may not hold, and bricks or other weights will anchor only the site at which they are placed. When covers are installed over heavy vegetation, as is often the case, there will be great difficulty in getting the material flush with the sediment surface.

Several steps, if included, can help ensure a successful installation. Use SCUBA to survey the reservoir sediments. Locate and mark obstructions such as stumps or boulders. Test the sediments for their ability to hold a stake. If the sediments are highly fluid or muck-like, then either very long stakes will be needed. or anchors such as bricks will have to be used. Stakes cannot be used in hard sediments such as gravel. If the sediments are unsatisfactory for steel stakes, examine the possibility of sewing link chain into the edge of the fabric. Obviously, some sediment covering materials cannot be used for this type of holdfast.

The most commonly used stake has been made of 1/8- or 1/4-in.-diam steel wire. Bend the stake so that one end is L-shaped. Sharpen the other end to ease the penetration of the fabric and the mud. The length of the stake needed will vary directly with the degree of softness of the mud.

In nearly every situation, SCUBA will be needed for an effective application. A successful procedure in many applications has been to build a reel in the stern of a boat. Most of the materials are sold in rolls of 7- to 8-ft width and 100-ft length. Two applicators, one on each side of the boat, unroll the screen from the reel. The screen is pressed and smoothed to the sediment surface and fastened. Stakes, bricks, or other fasteners will be needed every 1 to 2 m, depending upon the density of the vegetation. In deeper waters, divers will be needed to apply the covers. Be certain that a backup, fully suited diver is prepared to assist the applicators.

The screen is laid down out to the edge of the zone of rooted plants, if needed. Be certain to overlap the rolls so that there are no bare spots. The application should be examined in about 2 to 3 weeks to reposition stakes and smooth the screen to the sediment surface. A poor application will have missed areas, and extensive "ballooning" or even lift-off of the cover could occur. This will be a nuisance to boaters and swimmers, and any screen with a specific gravity less-than 1.0 poses a hazard to reservoir operation if substantial amounts of it break free and float toward intake structures.

Application is best accomplished during water level drawdown. Plants are absent, obstructions can be avoided or removed, and the applicators can see their work. Otherwise, early summer before dense vegetation develops is optimum. As Perkins, Boston, and Curren (1980) and Engel (1982) have shown, screens can be removed after 2 months and placed elsewhere. The area treated first (May-June) should remain essentially free of weeds during the remainder of the summer, and the screen can be used for July-hugust control elsewhere. Harvesting prior to application should also make the job easier and the cover--age more complete.

Effectiveness, Costs, and Feasibility

Three materials, Aquascreen (fiberglass), Dartek (nylon), and Typar (polypropylene), have received extensive testing and have been shown to be effective in controlling rooted aquatic plants. The use of these materials is best described by presenting brief case histories of their use. The other materials will also be briefly described. The reader should consult Cooke et al. (1986) for other case histories and descriptions of techniques.

Aquascreen

Aquascreen is a flexible, heavy (specific gravity = 2.54) fiberglass screen coated with polyvinyl chloride. It resembles window screen. It is sold in rolls of 7 by 100 ft, with a mesh size of 62 apertures cm^{-2} at \$0.20 ft^{-2} or \$21,000 ha^{-1} (1984 prices; Menardi-Southern Division of US Filter Corporation, Augusta, GA), plus installation charges.

Aquascreen has been shown, in many investigations, to be completely effective in controlling rooted aquatic plants, at least for the season of application. Mayer (1978) first described the use of this product following extensive experiments in Lake Chatauqua, New York_. He reported that

Myriophyllum spicatum (Eurasian watermilfoil) and *Potamogeton crispus* (curly-leafed pondweed) infestations decayed in 2 to 3 weeks following application. The -deposition of sediment on the screen was significant after 2 to 3 years of placement, and this permitted regrowth of the plants. Mayer found that autumnal removal of the screens and repositioning in the spring maintained 95-percent control of nuisance weeds.

Perkins, Boston, and Curren (1980) examined the relationship between coverage time by Aquascreen and control of Eurasian watermilfoil. Panels that measured 30 by 80 ft (9 by 24 m) were set out in shallow (0.5 to 2.0 m) and deep (2 to 3 m) areas of Lake Washington, and then removed at 1-, 2-, and 3-month intervals. Compared with control areas, panels in place for 1 month produced 25- and 35-percent decreases of biomass in shallow and deep plots, and plant regrowth was small. Two months of coverage produced 78- and 56-percent decreases in standing crop of plants and little regrowth. Three months of coverage essentially eliminated the plants. Both Perkins, Boston, and Curren (1980) and Engel (1982, 1984) found that Aquascreen was most effective when applied tightly to the sediment surface. Growth will occur under loosely applied screen. Engel emphasizes that screens left in position for a second season will have plant growth on them. The screens should be removed, cleaned, and repositioned. Similar conclusions were made about Aquascreen by Lewis, Wile, and Painter (1983), following experiments in an Ontario lake. Newroth and Trvelson (1984) reported an average period of plant control in British Columbia lakes to be 2 years. The screen was found to allow some plants to grow through it, while fragments of some plants were able to root through it.

Newroth and Trvelson (1984) have found that vinyl-coated window screen is a satisfactory and less expensive substitute for Aquascreen.

Dartek

Dartek is a black-pigmented, impermeable nylon sheeting material (DuPont, Canada) with a specific gravity greater than 1.0. It is sold in 2.5- by 30.5-m rolls (about $8,000 ha^{-1}) (1983 prices). Perkins (1984) evaluated the effectiveness of Dartek in Green Lake, Washington, where one roll was placed over soft unconsolidated organic muck that supported a population of *Elodea canadensis*. Four panels of Dartek were placed in Lake Washington over sand-gravel sediments. Sheeting without gas-venting slits and those with diagonal slits lifted off the sediments. Panels that had 12 cross-hatch slits per square meter lifted less than 20 cm. After 35 days, plant decomposition beneath the panels was nearly complete. Perkins concluded that Dartek was "highly effective." Newroth and Trvelson (1984) report that Dartek, on the average, was effective for 2 years.

Polypropylene

Polypropylene is a spun-bonded, woven, semipermeable sheeting with positive buoyancy. It is sold under various trade names, such as Typar (DuPont) and Terratrack (Terratrack Ltd., Rexdale, Ontario). Engel (1982) reports that Typar (1984 price) costs about $8,000 ha^{-1}, plus installation. Terratrack (Lewis, Wile, and Painter 1983) costs about $6,254 ha^{-1}.

Cooke and Gorman (1980) found that Typar, anchored with cement blocks, was completely effective in eliminating rooted plant biomass in the treatment area for one season. No evaluation in subsequent seasons was made. Filamentous algae grew profusely on the surface of the Typar in shallow water. Engel (1984) also found that Typar completely controlled vegetation during the first year. Removal and repositioning were difficult, and plants regrew on Typar after sediments accumulated on the panels. In contrast, Terratrack, anchored with concrete blocks, did not allow Eurasian watermilfoil to reroot on it over three summers, even though fragments were found (Lewis, Boston, and Curren 1983). Newroth and Trvelson (1984) report that the use of sand to anchor Typar provided a substrate for rerooting of plants.

Other materials

The experiences with polyethylene sheeting have not been satisfactory, largely because polyethylene is impermeable and buoyant (Armour, Brown, and Marsden 1979; Nichols 1974; Petersen, Born, and Dunst 1974; Engel 1982; Engel and Nichols 1984). Since polyethylene floats, it is very difficult to apply, and the anchoring must be very secure so that the sheets do not move under the influence of waves. Sand has been used as an anchor, but plants grow on it. Also, large slits must be cut to allow the escape of gases, the material deteriorates in sunlight, and it will slip down steep inclines (Armour, Brown, and Marsden 1979). It is not very expensive, however. Armour, Brown, and Marsden (1979) report a price of about $4,600 ha^{-1} (1984 prices), plus installation.

Hypalon, a synthetic rubber, is effective due to its strength and weight but is impermeable. It is also very costly. In 1984, the inflation-corrected price was $59,270 ha^{-1} (Armour, Brown, and Marsden 1979). Polyvinyl was also found to be effective, in part because it is negatively buoyant. It is imper meable and thus requires venting, and 'it is expensive (inflation-corrected to 1984, the price was $22,194 ha-l). Polyvinyl also tended to crumple and was easily dislodged by waves (Armour, Brown, and Marsden 1979).

Burlap (10 oz yd^{-1}) was found by Jones and Cooke (1984) to be effective for one season in an Ohio reservoir, but the material rotted even when treated with a preservative. Despite the permeability of burlap, it ballooned, suggesting that the pores may have been clogged with organic matter and the associated microbial community. Newroth and Trvelson (1984) report control of Eurasian watermilfoil with burlap for two or three growing seasons in British Columbia lakes. Despite its negative buoyancy, burlap must be securely anchored. Cost of burlap according to Jones and Cooke (1984) is about $3,400 ha^{-1}.

Sediment covers, while very effective in eliminating problems with nuisance rooted aquatic plants, cannot usually be used over large areas due to costs. A 30- to 40-ha treatment with Dartek would cost over $400,000 plus installation. However, smaller areas, such as docks and marinas, could be treated at a much lower cost. In most cases, this would be an appropriate use of sediment covers.

Reservoirs often receive substantial silt loads, and waves may further increase the quantity of suspended solids that could settle on the covers. This problem could limit the longevity of control by sediment covers since plant fragments may root in the deposited silt. Newroth and Trvelson (1984) report that a fabric called Texel, a negatively buoyant, needle-punched, polyester fabric, may be particularly applicable to this situation because fragments of plants appear to have difficulty in attaching to the upper surface and in penetrating the fabric. In many cases, the reservoir manager will have to be aware that sediment screens will require periodic cleaning and repositioning to prevent new plant growth.

Several candidate materials are available, as reviewed earlier. Feasibility in reservoirs depends not only costs and the possibility of plant regrowth on deposited sediments, but also the effectiveness and ease of application. An onsite evaluation of this should be conducted prior to full-scale application to a particular area. Ease of application, longevity of materials, and duration of control should be eval.uated for several types of sediment covers.

Limitations and Concerns

While little study has been made of their impacts, a few negative impacts of the use of sediment covers have been reported to date. Engel (1982, 1984) found that macroinvertebrates were eliminated by Aquascreen in Cox Hollow Lake, Wisconsin, apparently due to low dissolved oxygen. Boston and Perkins (1982), however, reported that plant death following Aquascreen application was slow enough that they believed deoxygenation would be a minor problem. Additional studies are needed.

Applications of sediment covers to eliminate rooted plants at swimming beaches, while completely successful in meeting this objective, have produced complaints by swimmers about walking on the screens.

Summary

Table 16 is a summary of the properties, costs, and effectiveness of seven materials that have been used as sediment covers. Table 17 is a summary of this reservoir management technique.

Table 16

Summary of Features of Sediment Covering Materials

Material	Specific Gravity	Weatherability	cost $/ha	Application Difficulty	Permeability to Gases	Comments
Black polyethylene	0.95	Poor	4,600	High	Impermeable	Easily dislodged, vents permit plant growth, "balloons"
Hypalon	>1.0	Excellent	59,270	Moderate	Impermeable	Strong, effective, "balloons"
Polyvinyl	1.2-1.5	Good	22,194	Moderate	Impermeable	Easily dislodged, crumples, effective,
Polypropyl (Typar)	0.90	Good	8,000	Low	Permeable	Effective
Aquascreen	2.54	Good	21,500	Low	Permeable	Highly effective, easily removed and repositioned
Burlap	>1.0	Poor	3,400	Low	Permeable	Effective, "balloons," rots when placed on highly organic muds
Dartek	>1.0	Fair	8,000	Moderate	Impermeable	Effective, "balloons"

Table 17

A Summary of Sediment Covers

Characteristic	Description
Target	Nuisance rooted plants.
Mode of action	Prevents plant growth by a physical barrier over reservoir sediments.
Effectiveness	Effective; problem often eliminated.
Longevity	Months to years.
Negative features	Elimination of habitat. Installation may be difficult and costly. May float to surface and clog intake structures. May annoy waders at swim beaches.
Costs	Ranges from \$3,400 to \$60,000 ha^{-1}, plus installation, depending upon material selected.
Applicability to reservoirs	Suitable for eliminating plants in selected areas such as marinas, swimming areas, and docks.

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PART XIII: HERBICIDES AND ALGICIDES

Problems Addressed

Excessive biomass of rooted aquatic plants can interfere with recreational uses of the water and may produce organic substances in the water, which lead to dissolved-oxygen depletions or to problems-with taste, odor, and production of trihalomethanes in potable water plants. Nuisance "blooms" of algae, often blue-green algae, can also produce the same problems.

Herbicide and/or algicide applications are used to kill the plants, and together represent one of the most commonly used means of managing the symptoms of eutrophication. There are few modern scientific reviews of their use. However, the reader will find the following useful: Brooker and Edwards (1975); Kearney and Kaufman (1975); Robson and Barrett (1977); Brown (1978); Janik, Taylor, and Barko (1980); Hanson and Stefan (1984); Westerdahl and Getsinger (1988).

Theory and Design

Herbicides and algicides are applied either as liquids or granules to nuisance plant populations at concentrations known to be sufficient to produce elimination or control of biomass, usually through interference with plant cell metabolism. In some instances, chemicals are used together (e.g., diquat and copper) for controlling macrophyte and algal problems simultaneously, or chemicals may be used in conjunction with other procedures such as harvesting, grass carp, or insects.

Until the last 15 years, the use of chemicals to alleviate the symptoms of eutrophication was essentially the only effective method available to improve or open reservoirs and waterways. Today, other methods that are restorative or protective in nature, or with less environmental impact, have often been substituted for them, especially where longer term control is desired. These methods have been described in earlier parts of the report and in Cooke et al. (1986). In many instances, however, particularly in densely infested southern waters, herbicides remain as one of the more cost-effective management options. When the causes of excessive algae or macrophyte growth :nnot be remedied, or while nutrient control or other restorative procedures are being implemented, there'may be **no** other option than the use of algicidal or herbicidal chemicals.

The chemical and physical properties of herbicides are described by Beste (1983), including a list of manufacturers who will provide data on application, dose, target species, and user precautions. A handbook entitled "Aquatic Weeds: Their Identification and Methods of Control" has been published by the Illinois Department of Conservation, Springfield, IL. This and similar reports prepared by other states can be useful in calculating doses and in determining the identity of nuisance plants. Reservoir managers should be aware that a commercial applicator's license, along with liability insurance, is required when the work is contracted. Prospective users of these chemicals should consult the appropriate state regulatory agency to obtain a list of approved chemicals for that state. Assistance in determining which chemicals are approved for use cannot be obtained from the Office of Pesticide Programs of the US Environmental Protection Agency at this time.

Effectiveness and Feasibility

The following paragraphs briefly review the most commonly used chemicals with regard to effectiveness, persistence, toxicity to nontarget organisms, and other factors.

Copper-containing compounds

Copper has been used as an algicide throughout this century to produce short-term relief from nuisance algal blooms. More recently, chelated copper compounds have been used to keep the copper concentration in the water column high for longer periods. Usually a dose of 1 to 2 ppm (0.8 mg Cu r^{-1}) provides significant algal control, apparently through inhibition of photosynthesis and nitrogen fixation, for a period ranging up to 10 days.

Unless nutrient input and water column nutrient concentration of a eutrophic reservoir are significantly reduced, or zooplankton grazing on algae sharply enhanced, problems with nuisance algal blooms will likely continue. The use of copper or other algicides such as simazine may be the only feasible choice for control of an algal bloom if nutrient reduction does nor occur.

The use of copper results in several significant environmental impacts. A massive algal kill, such as could occur if too great an area is treated per day, may produce a large decline in epilimnetic dissolved oxygen as the cells decay. Hypolimnetic dissolved oxygen may fall sharply. Copper is a highly toxic metal to animal groups, especially in soft water of low organic content Iodson, Borgmann, and Shear 1979). Some fish, particularly salmonids and walleye, are very sensitive to copper, and effects will appear at concentrations between 10 and 20 μg Cu ℓ^{-1}. Bluegills (*Lepomis macrochirus*) will exhibit sublethal effects at 40 to 160 μg Cu ℓ^{-1}, and perch (*Perca flavescens*) at 40 μg Cu ℓ^{-1}. Significant tissue accumulation of copper occurs in bluegills exposed to between 40 and 160 μg Cu ℓ^{-1} in soft water (45 mg ℓ^{-1} as $CaCO_3$). Persistent use of copper could produce extinctions of fish species in a particular reservoir due to effects on reproduction and larval stages. Some fish food organisms, including *Daphnia* (Cladocera), *Gammarus* (Amphipoda), *Physa* (Mollusca), and some insects, are also extremely sensitive to copper (Benoit 1975; Birge and Black 1979; Hodson, Borgmann, and Shear 1979; Ingersoll and Winner 1982; Harrison, Knezovich, and Rice 1984; Collvin 1985; Blaylock, Frank, and McCarthy 1985). The elimination or reduction in population density of algae-grazing zooplankton, such as *Daphnia pulex*, may account for the rebound in phytoplankton density that can occur after a copper application. It should be noted that the use of chelated copper in combination :th the herbicide diquat, a treatment technique to attempt to control both rooted plants and algae at once, greatly enhances the chemicals' toxicity to trout (Simonin and Skea 1977).

Ingestion of copper in potable water apparently will not produce copper toxicosis in humans (Scheinberg 1979).

Diauat

Diquat, a bipyridilium salt, is known to be an effective herbicide, but is quickly deactivated in turbid water through sorption to inorganic particles. Even a rainstorm that introduces turbid water shortly after application may reduce effectiveness. Diquat persistence of up to 160 days in the mud has been noted by Frank and Comes (1967), and Birmingham and Colman (1983) have found that chronic applications could lead to sufficient desorption of "loosely bound" diquat from sediments to produce phytotoxic conditions in the water. The duration of its effectiveness ranges from weeks to an entire summer (Blackburn and Weldon 1964, Schenk and Jarolimek 1966). Apparently its mode of action is to inhibit photosynthesis and stimulate respiration (Funderburk and Lawrence 1964). The recommended dose ranges from 0.5 ppm for

Potamogeton to 1.0 ppm for *Myriophyllum spicatum*, *Ceratophyllum demersum*, and *Najas flexilis*.

As noted earlier, the combined use of diquat and copper is particularly toxic to brown trout (*Salmo trutta*) and also **to** some invertebrate animals of the fish food web (May, Hestand, and Van Dyke 1973; Simonin and Skea 1977). Crustacea, a group of animals of great significance to fish diets, are extremely sensitive to diquat at levels well below those needed for plant treatment (Wilson and Bond 1969; Nicholson and Clerman 1974; Storch and Winter 1983; Williams, Mather, and Carter 1984). Other invertebrate animals appear to survive doses well in excess of those that would be encountered during reservoir treatment (Naqvi, Leung, and Naqvi 1980; Marshall 1984). Bluegills tolerate high levels of diquat (Surber and Pickering 1962), but perch and rainbow trout (*Salmo gairdneri*) exhibit sublethal effects that would interfere with successful spawning (Bimber, Boening, and Sharma 1976; Dodson and Mayfield 1979). The latter study has shown that rainbow trout accumulate significant amounts of diquat in their tissues, although the study did not specify whether these were edible or nonedible tissues.

Endothall

Endothall is produced in several formulations, including the liquid (Aquathol K) and granular (Aquathol) dipotassium salt and the di(N,N-dimethylalkylamine) salt (Hydrothol 191) in liquid and granular forms. Endothall acts on plant tissues to produce abnormal permeability, loss of water, and wilting (Keckemet 1969). Susceptible plants may be controlled for weeks to months. For example, Rodgers, Reinert, and Henman (1984) found that it took 2 months before *M. spicatum* began to reinvade after treatment, Doses of endothall to target macrophytes range from 1 to 3 ppm, depending on target species. The use of endothall has been reviewed by Armstrong (1974).

Both the mono and di(N,N-dimethylalkylamine) salts have been found to be very toxic to some fish at concentrations below those needed for plant control (Walker 1963, 1964; Finlayson 1980). Armstrong (1974) reports that rainbow trout, chinook salmon (*Oncorhynchus tshauytscha*), mud minnows (*Pimephales promelas*), largemouth bass (*Micropterus salmoides*), and bluegills will tolerate disodium endothall concentrations from 10 to 100 times the concentration recommended for plant treatment. The disodium and potassium salts apparently persist in water for periods ranging from 2 to 46 days (Hiltibran 1962, Yeo 1970, Simsiman and Chesters 1975, Holmberg and Lee 1976, Serns 1977).

2,4-D

The herbicide 2,4-D is a phenoxyacetic acid. It is available as the acid or in salt or ester form. The herbicide must be absorbed by plant tissue to be effective. The sodium and potassium salts penetrate poorly, the ammonia and amine Salts somewhat better, and the hydrolyzed ester (e.g., butoxyethanol ester, BEE) is readily absorbed and translocated. The action of 2,4-D in the plant is unclear except that it behaves like an auxin (Loos 1975, Westerdahl and Hall 1983).

The herbicide 2,4-D is particularly effective against *Myriophyllum spicatum*. Gangstad (1982) considers 2,4-D to be the most effective and economical treatment of this plant. Doses of 20 to 40 kg acid equivalent/ha, normally of 2,4-D DMA (dimethylamine salt) or 2,4-D BEE, are usual. Goldsby, Bates, and Stanley (1978) have reported that 2,4-D is effective against milfoil, particularly when combined with water level drawdown. Adams (1983) cautions that root contact is essential for long-term milfoil control and that the use of granular formulations can be ineffective if the pellets become trapped and suspended in the foliage. If an early season application is not possible, then the use of a harvester prior to application will allow the herbicide good contact with the roots. Adams (1983) also recommends that the normal dose of 22 kg active ingredient/ha of the granular formulation of 2,4-D BEE (Aqua-Kleen) be doubled for dense infestations in deep water (>2.4 m) where there is a high water turnover rate. Otherwise, dilution will be too great to produce control.

Depending on the dose and degree of infestation prior to treatment, 2,4-D remains effective against milfoil for at least the season of application, and often longer (Smith 1971; Aiken, Newroth, and Wile 1979; Getsinger, Davis, and Brinson 1982; Adams 1983). Other plants may not be adequately controlled. Controlled-release formulations may provide even longer control of milfoil (Van, Steward, and Jones 1986). Pierce (1960) found that species of *Potamogeton* returned in 1 month and grew heavily, while *Utricularia* was unaffected at doses up to 6 ppm. Adams (1983) also reported an invasion of Potamogeton after elimination of milfoil.

Although 2,4-D appears to have a short persistence in the water column, it can be detected in mud samples for months (Faust and Aly 1961;; Smith and

Isom 1967; Cope, Wood, and Wallen 1970; Adams 1983; Birmingham and Colman 1985). Degradation is far slower in anaerobic sediments than in aerobic (DeLaune and Salinas 1985).

At the concentrations achieved with the usual dose of 2,4-D, there is little evidence of toxicity to fish and invertebrates, with some significant exceptions (Smith and Isom 1967, Vardia and Durve 1981, Couch and Nelson 1982, Adams 1983). Low doses of 2,4-D BEE are toxic to developmental and juvenile stages of sockeye salmon (*Oncorynchus nerka*), chinook salmon, and rainbow trout, according to McBride, Dye, and Donaldson (1981) and Finlayson and Verrue (1985). Cope, Wood, and Wallen (1970) found that the propylene glycol butyl ether ester produced lesions and liver, blood, and central nervous system abnormalies in bluegills. It apparently does not bioaccumulate to significant levels in tissues of bluegills, channel catfish, or largemouth bass (Schultz 1973). Areas within 0.5 mile (0.8 km) of a potable water intake cannot be treated with 2,4-D. Moreover, it can be used only for Eurasian watermilfoil control in Tennessee Valley Authority reservoirs unless specifically approved by EPA under Section 18 or 24C of FIFRA.

Mullison (1981) concluded that there is a wide margin of safety for humans with respect to the use of 2,4-D. However, a case study (Colton 1986, Hoar et al. 1986) provides evidence for increased risk of non-Hodgkin's lymphoma among men exposed to 2,4-D for more than 20 days per year. This finding suggests that 2,4-D should not be used in potable water and that applicators should use every precaution to avoid exposure.

Fluridone

Fluridone is registered under the trade name SONAR and is sold as an aquatic suspension or as pellets. A review of its mode of action, effectiveness, dose, and environmental impacts has been provided by Schmitz (1986). Fluridone is a slow-acting, rapidly degradable herbicide that is very effective against a broad spectrum of submersed and emergent aquatic plants (Table 18). A more complete list of plants controlled by fluridone can be found in Schmitz (1986) and Westerdahl and Getsinger (1988). Its action is to inhibit synthesis of plant pigments which protect chlorophyll from photodegradation. The normal dose for reservoirs with a mean depth greater than 4 m is 2.2 to 4.5 kg (active ingredient)/ha, and treatments are most effective when applied prior to rapid plant growth. Treatments when plants are visible

Table 18

Common Aquatic Weed Species and Their Responses to Herbicides*

Plant	Diquat	Endothall	2,4-D	Glyphosate	Fluridone
Emergent species					
Alternanthera philoxeroides (alligatorweed)			YES	YES	YES
Dianthera americana (water willow)			YES		
Glyceria borealis (mannagrass)	YES	NO	NO		
Phragmites spp. (reed)				YES	
Ranunculus spp. (buttercup)			YES	YES	
Sagittaria sp. (arrowhead)	NO	NO	YES		YES
Scirpus spp. (bulrush)	NO	NO	YES	YES	YES
Typha spp. (cattail)	YES	NO	YES	YES	YES
Submersed species					
Ceratophyllum demersum (coontail)	YES	YES	YES	NO	YES
Chara spp. (stonewort)	NO(1)	NO(1)	NO(1)	NO(1)	NO(1)
Elodea spp. (elodea)	YES	?	NO	NO	YES
Hydrilla verticillata (hydrilla)	YES(2)	YES	NO	NO	YES
Myriophyllum spicatum (milfoil)	YES	YES	YES	NO	YES
Najas flexilis (naiad)	YES	YES	NO	NO	YES
Najas guadalupensis (southern naiad)	YES	YES	?	NO	YES
Potamogeton amplifolius (large-leafed pondweed)	?	YES	NO	NO	
P. crispus (curly-leafed pondweed)	YES	YES	NO	NO	
P. diversifolius (waterthread)	?	YES	NO	NO	
P. natans (floating leaf pondweed)	YES	YES	YES	NO	YES
P. pectinatus (sago pondweed)	YES	YES	NO	NO	YES

(Continued)

Note: Data based on Arnold 1979, McCowen et al. 1979, Pennwalt Corporation 1984, Monsanto Company 1985, Nichols 1986, Schmitz 1986, Westerdahl and Getsinger 1988.

* Abbreviations are defined as follows: YES = controlled, NO = not controlled, (1) = controlled by copper sulfate, (2) = plus chelated copper sulfate, and ? = possible control. No entry indicates data unavailable.

Table 18 (Concluded)

Plant	Diquat	Endothall	2,4-D	Glyphosate	Fluridone
Submersed species (Continued)					
P. illinoiensis (Illinois pondweed)				NO	YES
Floating species					
Brasenia schreberi (watershield)	NO		YES	NO	NO
Eichhomiae crassipes (waterhyacinth)	YES(2)		YES	YES	NO
Lemna minor (duckweed)	YES	NO	YES	NO	YES
Ne lwnbo lutea (American lotus)	NO	?	YES	YES	NO
Nuphar spp. (cowlily)	NO	?	YES	YES	?
Nymphaea spp. (waterlily)	NO	?	YES	YES	3

in spring-summer are also effective. Fluridone acts slowly, and 30 to 90 days may be required to establish plant control under optimum conditions.

Fluridone appears to have a very low toxicity to fish and invertebrates and does not accumulate in animal tissues. Because it is slow acting, dramatic changes in physicochemical variables, such as dissolved oxygen, are unlikely. Fluridone cannot be applied within 0.25 mile (0.4 km) of a potable water intake. There is no waiting period following application.

Glyphosate

Glyphosate, registered under the trade name RODEO, is used for treatment of emergent vegetation. It is ineffective against submersed plants. Table 18 lists some of the plants known to be controlled. Glyphosate is formulated as a liquid combined with a surfactant, and appears to affect amino acid metabolism in treated plants. Glyphosate is new, and limited data are available in the scientific literature regarding toxicity, bioaccumulation, and persistence. Studies performed by Monsanto Company (1985) indicate that RODEO biodegrades, does not bioaccumulate, and has very low invertebrate and mammalian toxicity. A study by Servisi, Gordon, and Martens (1967) concludes that glyphosate antagonizes the toxicity of its surfactant MONO 818 and that the surfactant is more toxic than the herbicide. Glyphosate cannot be applied within 0.5 mile (0.8 km) of potable water intakes. There is no waiting period for water use after application.

Table 18 summarizes the effectiveness of commonly used herbicides against some aquatic plants. Westerdahl and Getsinger (1988) have developed a thorough guide to the types of aquatic plants that may appear in reservoirs and have listed the herbicides that are effective against them. Readers contemplating a herbicide application should examine this report.

Costs

Costs of herbicide treatments range widely. Plant density, area to be treated, types of plants, and other factors will influence cost greatly. Table 12 (Part X) is a comparison of cost ranges for harvesting and herbicides. In the Midwest, cost ranges are clearly comparable (about $\$350-\$900\ \mathrm{ha}^{-1}$) for these two techniques and will be affected by some of the local factors listed above. In Florida (and other areas with dense, rapidly growing populations of exotic plants such as waterhyacinth), herbicide treatments are usually less costly than harvesting. In some southern waters, harvesting cannot keep up with plant growth and becomes a continuous operation, whereas herbicide applications may be sufficient to manage the problem in one or a few applications.

Limitations and Concerns

Brooker and Edwards (1985) point out that most discussions of the negative effects of algicides and herbicides deal only with direct toxic effects to selected species. Some of these effects have been briefly outlined in a previous section. While these reports provide some useful information, they are not very informative about the effects of the addition of toxic materials to reservoirs. Reservoirs are complex units of interacting biological, chemical, and physical components called ecosystems. The ecosystem is the actual level of biological organization to which herbicides and algicides are applied, not the species level. There are very few studies about the effects of these chemicals on reservoir or lake ecosystems. Some of these studies are briefly outlined here, based upon the list of concerns in Conyers and Cooke (1982, 1983) and Cooke (1983).

Hanson and Stefan (1984), in one of the few long-range ecosystem-level studies of the impact of chemical treatments on an aquatic system, found both short- and long-term impacts. Copper sulfate applications over a period of 58 years were effective in providing temporary, short-term (days) control of algae, but produced oxygen depletions and increased phosphorus cycling and occasional fish kills from copper toxicity and low dissolved oxygen. The longer term effects over the years included copper accumulation in sediments, increased tolerance by some algae to copper sulfate, a shift in species composition from green to noxious blue-green algae and from game fish to rough fish, disappearance of macrophytes, and reductions in benthic macroinvertebrates. Hanson and Stefan (1984) conclude that the short-term gain of expedient and brief algal control is essentially traded for long-term degradation of the lake. However, the use of chelated copper compounds, in place of copper sulfate, alone and in combination with other herbicides is EPA-approved. Moreover, previous problems with copper toxicity attributable to copper sulfate are not apparent with the chelated copper compounds.

When plants are killed with chemicals, their biomass, including the plant nutrients contained therein, is left in place to decompose. Many authors, among them Walker (1964), Nichols and Keeney (1973), Carter and Hestand (1977), Carpenter and Adams (1978), Hestand and Carter (1978), Myers (1979), Peverly and Johnson (1979), Morris and Jarman (1981), Wingfield and Johnson (1981), Getsinger, Davis, and Brinson (1982), James (1984), and Goldsborough and Robinson (1985), have observed a pulse of nutrients, and in some cases a loss of dissolved oxygen and a phytoplankton bloom, following a herbicide treatment. Thus, one problem, excessive macrophytes, is replaced by one or more other severe problems (oxygen depletion, algal bloom). However, these side effects can often be lessened or avoided through careful planning and proper use of chemicals. Ways to prevent these effects include chemical application in spring before biomass develops, staged applications so that limited areas are treated each day until the entire target area receives an application, use of a harvester for plant biomass removal prior to application, or installation of aeration/circulation devices.

A common occurrence in herbicide use is the replacement of the nuisance target plant with another species that is unaffected by the chemical. The alga *Chara* and the rooted plant *Potamogeton* are often replacement species. Reiser (1976), Conyers and Cooke (1982, 1983), Conyers (1983), Cooke (1983), and Richard, Small, and Osborne (1984) describe case histories of this response to herbicides. Nichols (1986) provides a useful list Of common quatic nuisance plants and their responses to endothall, diquat, 2,4-D, and water level drawdown.

Another potential negative effect of herbicide use is the waiting period (3 to 10 days, depending upon chemical) between application and water use. Diquat, endothall, and 2,4-D all have waiting periods.

Summary

Herbicide and algicide applications for the control of nuisance macrophyte and algal problems are widely used methods of reservoir management. These chemicals provide an expedient and often highly effective means of producing at least short-term control of problem species. However, when one nuisance species is controlled, another species may take its place.

It is important to note that the use of registered herbicides may be the only feasible solution in many instances.

Table 19 is a summary of the use of herbicides and algicides.

Table 19

Summary of Algicides and Herbicides

Characteristic	Description
Target	Excessive algal and/or macrophyte biomass.
Mode of action	Toxic to plants.
Effectiveness	Highly effective against susceptible species.
Longevity	Algicides are effective for days.
	Some herbicides provide at least seasonal control.
Negative features	Decay of plants may produce a temporary dissolved oxygen depletion, and release of nutrients may stimulate an algal bloom.
	Some chemicals are toxic to fish and fish food organisms.
	Long-term effects on ecosystems remain unclear.
Costs	Costs range from about \$430 to \$900 ha^{-1} (\$175 to \$370 $acre^{-1}$).
Applicability to reservoirs	Their use may be the only feasible option; however, use in potable waters is restricted, except for copper, or restricted to use at some specified distance from potable water intakes. For some herbicides, a waiting period is designated following treatment.

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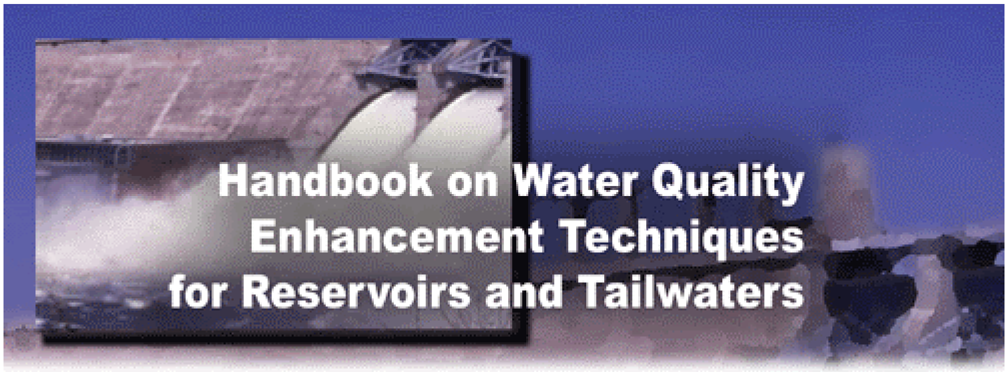

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