Meeting Minutes
Columbia River Comprehensive Impact Assessment
Weekly Management Meeting
February 20, 1996
Bechtel Building, Room 1B40, 9:30 - 12:30
ETB Building, Columbia River Room, 1:30 - 4:30

Attendees(morning@, afternoon*)/Distribution(#):

Dick Biggerstaff, BHI@*#
Michael Blanton, PNNL*#
Charlie Brandt, PNNL#
Amoret Bunn, Dames & Moore@*#
Sandra Cannon, PNNL@*#
Paul Danielson, NPT*#
Greg deBruler, HAB#
Kevin Clarke, RL#
Roger Dirkes, PNNL@*#
Sue Finch, PNNL@*#
Larry Gadbois, EPA*#
Richard Gilbert, PNNL@*#
Stuart Harris, CTUIR#
Dave Holland, Ecology@*#
Jay McConnaughey, WDFW#
Terri Miley, PNNL*#
Dick Moos, BHI#
Nancy Myers, BHI*#
Bruce Napier, PNNL*#
Lino G. Nicolli, YIN@*#
Tara O'Neil, PNNL*#
Roger Ovink, BHI#
Doug Palenshus, Ecology#
Ralph Patt, Oregon#
Stan Sobczyk, NPT#
Bob Stewart, RL@*#
Dan Tano, RL*#
Mike Thompson, RL#
JR Wilkinson, CTUIR#
Thomas W. Woods, YIN@*#
Jerry Yokel, Ecology#
Admin Records-CRCIA#

Summary of Discussions:

Comprehensive Section - Thomas Woods

Discussed the minimum attendance needed to hold the morning meetings on the comprehensive section. In addition to DOE and its contractors, there are seven organizations that need to be represented, including EPA, Ecology, Yakama Indian Nation, Confederated Tribes of the Umatilla Indian Reservation, Nez Perce Tribe, State of Oregon, and the Hanford Advisory Board. A rule needs to be set that, if after 30 minutes from the start of the meeting, a defined number of the seven organizations is not present, the meeting will be canceled.

The goal of today's meeting is to develop a statement of purpose and half of the study requirements.

Tom led a facilitated discussion that resulted in the following purpose statement for the Columbia River Comprehensive Impact Assessment:

> The purpose of the Columbia River study is
> - to conduct and
> - to provide a protocol and mechanism for
>
> a comprehensive assessment of the current and future impact on human health, the environment, and cultures of Hanford Site activities on the river, so that
> - projected effects can be assessed relative to regulatory standards, sound health physics, sociological, and biological standards
> - adequacy of pollution prevention and facility pollution control can be assessed
> - public health, environmental, and cultural impact of Hanford Site operations can be assessed
> - information for environmental management of the Hanford Site can be provided

The following requirements were also developed during the facilitated session:

- Temporal: current and future
  - Period during which degradation may occur from exposure to radiological and chemical materials

- Who is exposed: Socioeconomic? Ethnic? To be determined prior to conduct of the study.

- What (ecosystems)

- Impact compared to: (comparative basis)
  - Non-Hanford conditions as defined by the present conditions up river, e.g., between Hanford and Priest Rapids Dam.

- Section of river: Hanford through the mouth's sediment fan.

A compilation of the parking lot items from the "Comprehensive Section" of the weekly management meeting minutes was handed out and is attached to these minutes.

Status of Exposure Model Evaluation - Michael Blanton

A decision has been made to use a spreadsheet model for nonradiological chemical exposure to biota. Radiological chemical exposure will be modeled using CRITR2 computer code. One of the primary sources of algorithms used in the spreadsheet model will be adopted from those developed by Dr. Robert Thomann of Manhattan College. The algorithms used in Dr. Thomann's model are used as a basis for many of the "computer code" models we have previously discussed. This model fits the needs of being user friendly for other users and works on a personal computer. It has been through the rigor of peer review and published in numerous journals. An action was assigned to Michael to bring documentation on the model to the next meeting. Copies of a book chapter that Thomann wrote were handed out.

Public Involvement Issue Paper - Nancy Myers

At the 2/13/96 weekly management meeting, a Public Involvement Issue Paper was handed out with a request that comments be provided on 2/20/96. Comments were received from EPA and the Yakama Tribe; the Nez Perce representative indicated that they would be providing comments.

A meeting was held with DOE, Bechtel, and Ecology to discuss the public involvement plan and the focus sheet. The focus sheet will be drafted and forwarded to the project management team for review and comment. The plan is to distribute the focus sheet to announce the public comment period on the next three CRCIA documents to be issued and the schedule for public comment on the final document. The focus sheet will be mailed to the 1500 highly interested mailing list. The public involvement plan is being drafted and will incorporate comments received on the issue paper.

Data Selection - Terri Miley

A presentation on data selection was made by Terri Miley. A copy of the viewgraphs is attached. A summary of discussions is provided below.

- A meeting was held last week to discuss the CRCIA data assessment. This was a follow-up to last week's formation of a sub-management team on data issues.

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- We have collected all data with 3 exceptions: 1) the BHI chromium special studies, 2) data from Washington Department of Health, and 3) additional data for the McNary Pool area. The chromium data is expected soon. The Washington Department of Health will not be able to provide the data until February 28 or 29. We currently have McNary Pool data from the SESP project and the Department of Health data will include McNary Pool data. All leads provided for obtaining additional McNary Pool area data have been followed up. Some additional contacts were received from the Corps of Engineers and those are being pursued. The Pasco USGS office was not aware of any sampling below Richland but referred us to the Portland office who would know for sure. We have calls into the USGS in Portland.

- Processing data - Where sufficient data exists, a representative value will be determined over time for each location in the segment. A data distribution will be applied to the representative values for each location in a segment. Where there are insufficient data points, we will apply a data distribution to observed values.

- A decision is needed for determining how to choose a representative value for a single location over time. Where there is an obvious increasing or decreasing trend, the most recent value will be used. The team agreed with this concept. These values will be part of the distribution and will be represented on a lognormal chart. However, there was much discussion with no resolution on how to determine a representative value where no trend exists.

- There were discussions about using filtered versus unfiltered samples and the divergence of the results. It was noted that when the wells are first drilled, the unfiltered samples will initially show higher results which is from the drilling process. It was also noted that in some cases the unfiltered samples became higher in more recent years. Because of criticisms that sampling was taking to much time, the sampling methodology was changed. The rate of purging was increased which increased the sediment that is generated and collected in the samples. An artifact of this sampling process is higher values for unfiltered samples.

- Objections will be raised where data is sparse and a distribution is used across too large of a segment. If there is no data, then a segment needs to be represented as having no data. Data should not be extrapolated beyond what is real.

- The information contained in the "100-D Area Groundwater Chromium Levels" charts was provided by Bechtel groundwater staff. This data has been corrected for known errors, e.g., data not in the same units, problems with the names, etc.

- Several team members recommended that unfiltered data be used. In cases where there is a difference between filtered and unfiltered samples, a decision is needed as to which data to use.

- The decision on using filtered or unfiltered should be based on what you are going to do with the data. For the human health scoping level risk assessment, groundwater will be used as a surrogate for seep data. For this reason, some team members felt that filtered data should be used. It was noted that when filtered data is used, the data would only include chromium plus 6 and not chromium plus 3. Technically, you can argue for both filtered or unfiltered. Legally, there is not a clear signal as to which should be used.

- A session to review data plots using a fast graphics data package was proposed. Larry Gadbois volunteered to work on this for a full day. After much discussion, it was realized that the volume of graphs could not be looked at in one day and professional judgement would be needed on each graph to determine a representative value. This would also result in reentering all of the data by hand. This option was dismissed.

- A suggestion was made to go with eigen values rather than visually looking at each plot. After some discussion, this method was not accepted.

- A suggestion was made to use the median for representative values where there is no trend.

- An action was assigned to Larry Gadbois to write a heuristic for selecting representative data values. This process would identify outliers and would be automated. The methodology will be reviewed by Jerry Yokel, Paul Danielson, Amoret Bunn, Bruce Napier, Dick Gilbert, and Dick Biggerstaff. The team agreed that Terri can move forward with a process agreed to with the subteam just identified.

- The viewgraph "Strontium-90 at D Spring" was discussed. This graph portrays all the seeps and springs data for this segment. It contains data from Bechtel and the SESP samples. Question: do you throw all the data together? It was suggested that the best you can do is use the maximum. You can't look at trends. Seeps and springs are underwater most of the time. If the maximum is used, it would be a conservative value.

- On the viewgraph "Chromium in Seeps at 100D", the sixth line, identified as BOBMK2, is an outlier. BOBMK2 is a filtered sample. The companion unfiltered sample, BOBMK3, has a much lower value which is in line with other samples. The decision rule that Larry Gadbois is writing will screen out the outlier.

- Groundwater will be used as a surrogate for seeps and spring data. Where seeps and spring data exists, we can compare at isolated points, and superimpose the risk. The seeps and springs data is very limited.

- A lognormal distribution will be used. We will need to propose methods where the assumption for lognormal is questionable. Lognormal is a reasonable assumption as it gives a long tail towards the high side and is more conservative. For sampling water, lognormal is standard practice. A subset of plots can be reviewed to look at trends. A concern was voiced that while we may have time to check for it, there may not be time to go back and do anything about it. It will become a data validation exercise.

- Where there is not much data, use the maximum instead of applying a distribution to all data.

- Terri Miley was assigned the action to prepare a succinct summary of the data decision rules and present them at the 2/27/96 weekly management meeting. This summary should include the heuristic that Larry Gadbois is preparing.

Publication Update

Sandra Cannon handed out three draft packages of information: 1) document title page, table of documents in initial phase of the CRCIA, and outline for the Scoping Level Risk Assessment Report, 2) report distribution list, and 3) document review process flow chart (all attached). It was requested that the team members take the information back, review it, and provide any comments. If there are any substantial comments, an agenda item can be added to an upcoming meeting. The distribution list identifies who will receive the reports. Changes received to date have been incorporated.

The document review process identifies the steps for incorporating comments into the documents. For the contaminants of concern report, a total of 387 comments were received with 263 of those comments from the technical peer reviewers. Later this week, EPA, Ecology and DOE will identify key comments and reach agreement on proposed resolution. Next week, the CRCIA project management team will review the comments and proposed resolutions. Following team input, PNNL will complete resolution of the comments. Copies of the comment resolution will be distributed to the directors of the technical peer reviewers and other reviewers. On the back of the flow chart is a sample of the review comments form. Electronic copies were distributed to team members via ccmail and diskettes after last week's meeting. Where possible, it is recommended that the electronic form be used to submit comments. This will ease the comment consolidation process and make it very clear what is a comment. When the comments are part of a letter, it's hard to tell if the introductory paragraph is intended to be a comment.

Publication Questions for the Comprehensive Section - Thomas W. Woods

The team needs to decide by next meeting who will be the writer for the comprehensive section. Bob Stewart indicated that Larry Gadbois and Dave Holland had initially offered to help with the draft. The writer needs to attend all of the weekly morning meetings. Everything needs to be captured on one computer and we may want to record the meetings. Tom Woods agreed to take the lead to prepare an outline for the comprehensive section with Paul Danielson, Dave Holland, Larry Gadbois, and Amoret Bunn providing assistance. Team members need to review information that Tom presented at the 2/13/96 weekly meeting and provide comments to Tom. This presentation will be the beginning for the outline.

The issue of document clearance was raised. There no longer is a requirement for a document clearance.

This section will be a part of the scoping level risk assessment document. How do we want to receive and handle comments on this section. It was agreed to write the section as a team and also receive the comments as part of the whole document. The comment resolution process will also be a team effort.

Comprehensive Chapter:

- None identified at this meeting.

Agreements:

- Where there is a clear upward or downward trend, a representative value will be chosen from the most resent data point.

- The data task can move forward using a heuristic to determine representative values that Larry Gadbois will develop. Jerry Yokel, Paul Danielson, Amoret Bunn, Bruce Napier, Dick Gilbert, and Dick Biggerstaff will review the heuristic and provide comments and concurrence.

- It was agreed to write the comprehensive section as a team and also receive the comments as part of the whole document. The comment resolution process will also be a team effort.

Action Items:

Action Description	Assigned To	Due Date
Bring Thomann model documentation to next weekly meeting.	Michael Blanton	2/27/96
Write a heuristic for selecting representative data values	Larry Gadbois	2/27/96
Prepare summary of data rules, including the heuristic that Larry Gadbois is preparing	Terri Miley	2/27/96
Review publication update handouts and provide comments to Sandra Cannon	Project Team	ASAP
Determine who will be the writer for the comprehensive section	Project Team	2/27/96

Date/Location of Next 2 Scheduled Meetings:

- Tuesday, February 27, 9:00 - 12:00, Bechtel Building, Room 2A01; 1:00 - 4:00, EESB Building, Snoqualmie Room
- Tuesday, March 5, 9:00 - 12:00, Bechtel Building, Room 2A01; 1:00 - 4:00, EESB Building, Snoqualmie Room

Attachments:

- 2/20/96 meeting agenda
- Purpose of Columbia River Comprehensive Impact Assessment
- Parking lot items from the CRCIA weekly management meetings
- Book chapter written by Robert Thomann
- Presentation by Terri Miley, "Data Selection"
- Document title page and table of contents for the draft final report
- Report distribution list
- Document review process flow chart

Prepared by SM Finch on 2/26/96

AGENDA
Columbia River Comprehensive Impact Assessment
Weekly Project Management Team

Scheduled from 9:00 - 12:30 p.m., February 20, 1996
BHI Building, 3350 George Washington Way, 1B40 Conference Room

Scheduled from 1:30 - 4:30 p.m., February 20, 1996
Battelle's ETB Building, Columbia River Room

Morning Session

1. 9:00 - Bob Stewart - Introduction

2. 9:10 - Thomas W. Woods - Comprehensive Section
   - Facilitated whiteboard discussion
   - Statement of purpose
   - Uses and users of information
   - Discussion of requirements for 2/27/96 meeting

Afternoon Session

1. 1:30 - Bob Stewart - Introduction

2. 1:35 - Michael Blanton - Status of Exposure Model Evaluation

3. 1:45 - Nancy Myers - Project team to turn in comments on "Public Involvement Issue Paper"

4. 1:50 - Terri Miley - Data Update
   - Method for choosing representative data
   - Example data sets
   - Concentration data distributions

5. 3:00 - Break

6. 3:10 - Sandra Cannon - Publication Update
   - Report distribution list
   - SLRA document outline
   - Review process for FY96 documents

7. 3:30 - Thomas W. Woods - Publication Questions of Comprehensive Section
   - Identify writing team and lead writer for comprehensive section
   - Clearance issues/requirements for comprehensive section
   - Process for incorporating comments on the comprehensive section

8. 3:50 - Thomas W. Woods - Summary of Morning Session on the Comprehensive Section

9. 4:20 - Review of Upcoming Meetings

   2/27/96 - EESB Snoqualmie Room
   - Terri Miley - Data Update
   - Bruce Napier - Team input on comment resolution for the Contaminants of Concern report
   - Thomas W. Woods - Continued effort on comprehensive section

   3/5/96 - EESB Snoqualmie Room
   - Thomas W. Woods - Continued effort on comprehensive section

2/20/96

Purpose of Columbia River Comprehensive Impact Assessment

The purpose of the Columbia River study is
- to conduct and
- to provide a protocol and mechanism for

a comprehensive assessment of the current and future impact on human health, the environment, and cultures of Hanford Site activities on the river, so that

- projected effects can be assessed relative to regulatory standards, sound health physics, sociological, and biological standards

- adequacy of pollution prevention and facility pollution control can be assessed

- public health, environmental, and cultural impact of Hanford Site operations can be assessed

- information for environmental management of the Hanford Site can be provided

Parking lot items from the CRCIA weekly management meetings

- Vadose zone. (11/7/95)
- Affect on the river from the various remediation efforts ongoing around the site or are projected to go on around the site, including solid waste burial grounds. (11/7/95)
- Synergistic and combined effects. (11/7/95)
- Include impacts to the river from cleanup operations themselves. (11/21/95)
- Expansion of the contaminants of concern list from cleanup operations. (11/21/95)
- Contaminants below McNary. (11/21/95)
- Vadose zone. (12/5/95)
- Travel time from tanks to river. (12/5/95)
- Development of scenarios based on future use of river and shoreline. (12/12/95)
- Identify work done/ongoing on the Hanford Site that this team does not know about. (12/19/95)
- Flag data gaps for inclusion in the remaining work beyond FY96. (1/9/96)
- Species of concern that are not on the FY96 list that is limited to 30 species (1/16/96)
- Identify additional data needed to raise the percent of confidence over the work done in FY96 with existing data. (1/17/96)
- Implement the DQO/DQA process for defining remaining work to be done beyond FY96. (1/17/96)
- Look at models beyond steady state models. (1/23/96)
- Cultural risk assessment models. (1/23/96)
- Not a random collection of ideas; parking lot must be well organized proposal. (1/23/96)
- More samples in drive points. (1/30/96)
- Future impacts from groundwater. (1/30/96)
- Travel time of groundwater. (1/30/96)
- Groundwater data where out of high well density areas near the shoreline. (1/30/96)
- More comprehensive data. (1/30/96)
- Develop finer segments for the river. (2/6/96)
- Further development of tribal subsistence scenario. (2/6/96)
- Areas where uncertainty for FY96 work is unacceptable. (2/13/96)

Last update on 2/16/96

2/20/96

(2)

Chapter 10

Deterministic and Statistical Models of Chemical Fate in Aquatic Systems[1]

Robert V. Thomann[2]

This chapter has several purposes, among them: to summarize the basic models of the steady-state transport and fate of chemicals in aquatic systems including uptake and distribution in the aquatic food chain; to illustrate the deterministic time variable behavior of chemical fate models with several applications to the Great Lakes; to develop some statistical models of chemical variability in aquatic organisms, specifically, fish.

The ability to analyze and predict the transport of potentially toxic chemicals is one of the central requirements of risk assessment and subsequent risk management. Steady-state models can be of specific value in the early stages of chemical screening for generic problem contexts and to elucidate basic principles of chemical fate and uptake into the food chain. Time variable models are particularly useful for predicting recovery times of aquatic systems following some abatement program of chemical control. These steady-state and time variable models essentially estimate the average or deterministically varying chemical exposure concentration to aquatic organisms. Risk assessment also requires some evaluation of the stochastic behavior of chemicals both in the water and in fish. This chapter is therefore divided into four parts: 1) the basic theory and associated equations; 2) steady-state simplifications; 3) deterministic time variable models; 4) analytical and numerical models of statistical behavior of chemicals in fish.

[1] This article also appears in "Applied Mathematical Ecology," edited by S.A. Levin et al., Springer-Verlag, Berlin, Heidelberg, New York, 1989.

[2] Manhattan College, Environmental Engineering and Science Program, Bronx, New York 10471

10.1 Theory

10.1.1 Physical-Chemical Fate and Transport Model

The principal components of the physical-chemical fate and transport model framework are reviewed by Thomann and Mueller (1987), Delos et al. (1984), Thomann and Di Toro (1983), and Di Toro et al. (1981) among others.

The development can begin by considering a simple one-dimensional river as shown in Figure 10.1. The chemical in the water column is transported by the flow Q. Losses of chemical may occur as a result of microbial degradation, volatilization, or other pathways. The sediment however in all of the models discussed in this paper is not considered to be moving. There is a transfer of chemical from the sediment to the water column and vice versa via settling and resuspension of particulate chemical forms and sediment diffusion of dissolved chemical.

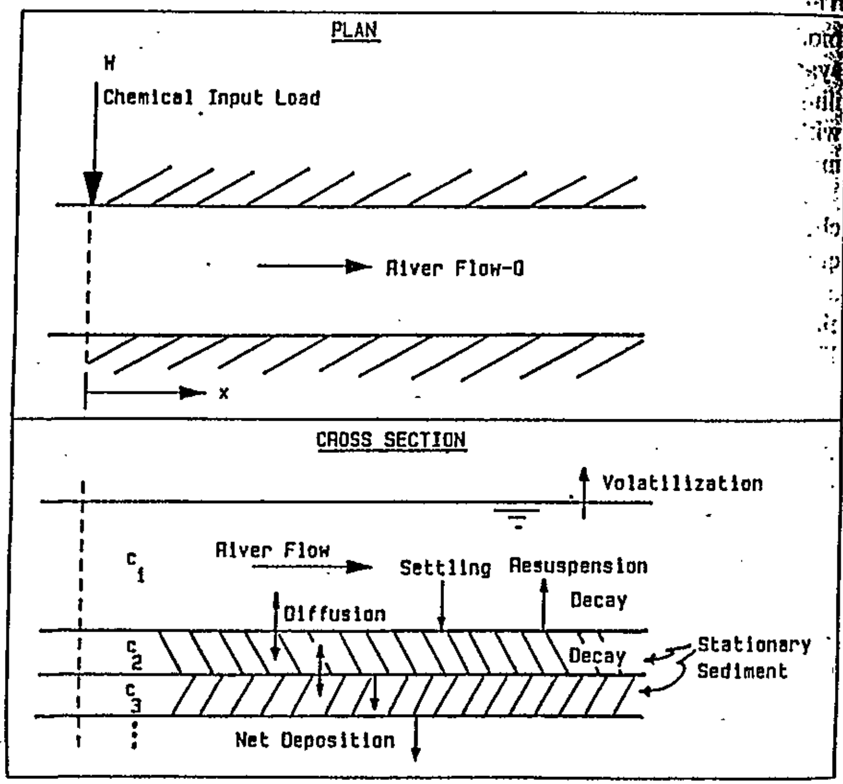

Figure 10.1. Notation for physico-chemical fate model in streams.

The one-dimensional mass balance equation for any form of the chemical (dissolved or particulate) is for the water column

$$\frac{\partial c_1}{\partial t} = -\frac{1}{A}\frac{\partial}{\partial x}(Qc_1) + \frac{1}{A}\left[\frac{\partial}{\partial x} EA \frac{\partial c_1}{\partial x}\right] + sources - sinks \quad (10.1)$$

and for the surface sediment

$$\frac{\partial c_2}{\partial t} = sources - sinks \quad (10.2)$$

where c_1 and c_2 are the chemical concentrations in the water column and sediment (m_T/l^3; m_T = mass of toxicant, l^3 = bulk volume of solids plus water); Q is the river flow (l^3/t); A is the cross-sectional area (l^2); E is the longitudinal dispersion coefficient (l^2/t); x is distance downstream and t is time.

The chemical in the models discussed herein is assumed to be composed of two forms: 1. the dissolved form, c'_d (m_T/l^3_w; l^3_w = volume of water); and 2. the particulate form, c_p (m_T/l^3), i.e., the toxicant sorbed onto particulate matter in the water column or sediment. The total chemical concentration is then

$$c_T = c_p + \phi c'_d \quad (10.3)$$

where ϕ is the porosity [l^3_w/l^3].

Equation 10.3 is

$$c_T = c_p + c_d \quad (10.4)$$

where

$$c_d = \phi c'_d \quad (10.5)$$

for c_d (m_T/l^3) as the porosity corrected dissolved concentration.

With the general framework described, the detailed equations for the various forms of the chemical can be presented.

Dissolved Chemical

An explicit finite differencing of Equation 10.1 together with sources and sinks of the dissolved chemical in a temporally constant control volume (V_1) of the water column is given by

$$V_1 \frac{dc_{d1}}{dt} = [(Qc^+_{d1}) - Q_1 c_{d1} + E'((c_{d1})^+ - c_{d1})$$

$$+ E'((c_{d1})^- - c_{d1})] \quad \text{(Transport)}$$

$$+ k_{d1} V_1 c_{p1} - k_{u1} m_1 V_1 c_{d1} \quad \text{(Sorption-desorption)}$$

$$+ K_{f12} A (c'_{d2} - c'_{d1}) \quad \text{(Sediment diffusive exchange)}$$

$$- K_{d1} V_1 c_{d1} \quad \text{(Decay \& losses)}$$

$$- k_{l1} A(c_g/H_e - c'_{d1}) \quad \text{(Volatilization)}$$

$$+ W_{d1} \quad \text{(Input)} \tag{10.6}$$

The group of terms in brackets represents the transport and dispersion of the dissolved toxicant. Superscript + indicates the upstream direction and superscript − indicates the downstream direction. The net transport flows, Q, are written in an equivalent backward difference approximation to the underlying partial differential equation (Equation 10.1). The dispersion or mixing between segments of length Δx is given by the bulk dispersion coefficient which in turn is related to the dispersion coefficient by

$$E' = \frac{EA}{\Delta x} \tag{10.7}$$

The second group (sorption-desorption) of Equation 10.6 is the balance between the desorption of the chemical in the particulate phase ($k_{d1} V_1 c_{p1}$) which increases the dissolved form (the desorption rate is k_{d1} [$1/t$]), and the adsorption from the dissolved phase onto the particulates given by $k_{u1} m_1 V_1 c_{d1}$. The sorption rate is $k_{u1}(l^3/m_s-d)$ and the solids concentration is m_1 (m_s/l^3). (m_s = mass of solids.) Note that this latter term depends on the mass of solids available for sorption from the dissolved phase.

The third group of Equation 10.6 represents the diffusive exchange between the sediment dissolved chemical concentration c'_{d2} in the interstitial water and the dissolved chemical concentration in the water column, c'_{d1}. The sediment–water diffusive transfer coefficient, $K_{f12}(l/t)$ can be considered as an overall interfacial transfer coefficient relating to the diffusion of the toxicant across the sediment–water interface.

Decay and loss mechanisms such as biodegradation, photolysis etc. of the dissolved form are included in the fourth group of the equation. Therefore, $K_{d1}(1/t)$ represents the sum of individual rates, some of which in turn may represent rather complex mechanisms. Note that for this model all the loss rates are assumed to be first order.

Volatilization of the dissolved toxicant is given by the fifth group of Equation 10.6 where c_g represents the gas phase of the chemical (m_T/l_g^3; l_g^3 = volume of gas) which may or may not be zero, and H_e is the Henry's constant for the chemical ($m_T/l_g^3 \div m_T/l_w^3$).

The last line represents all external sources or inputs of dissolved chemical, $W_{d1}(m_T/t)$ from point direct discharge sources as well as non-point and tributary inputs.

An equation similar to Equation 10.6 can be written for the dissolved chemical in the sediment layer underneath the typical water column segment 1. This layer is designated with the subscript 2. Thus,

$$V_2 \frac{dc_{d2}}{dt} = k_{d2} V_2 c_{p2} - k_{u2} m_2 V_2 c_{d2}$$

$$+ K_{f12} A(c'_{d1} - c'_{d2})$$

$$- K_{d2} V_2 c_{d2}$$

$$- v_{d2} A c_{d2}$$

$$+ K_{f23} A(c'_{d3} - c'_{d2}) \tag{10.8}$$

The first three lines of the right side of Equation 10.8 have already been discussed relative to the water column. The fourth line of Equation 10.8 expresses the "burial" or transfer down into the sediment of the dissolved toxicant due to net sedimentation or build-up of the sediment layer at a net sedimentation rate of $v_{d2}(l/t)$. The last line of Equation 10.8 is the diffusive exchange of dissolved toxicant between the first and second sediment layers under the water column. Similar equations can be written for each successive sediment layer. Note that there are no dissolved transport terms for the sediment thereby indicating that the sediment is assumed to be stationary in the horizontal direction. Also, mechanical mixing of sediment layers (due, for example, to bioturbation) is not included, but is readily added with an additional mixing term.

Particulate Chemical

The mass balance equation for the chemical sorbed onto the particulates in the water column segment 1 is given by

$$V_1 = \frac{dc_{p1}}{dt} = [(Qc_{p1})^+ - Q_1 c_{p1} + E'((c_{p1})^+ - c_{p1})$$

$$+ E'((c_{p1})^- - c_{p1})] \quad \text{(Transport)}$$

$$- k_{d1} V_1 c_{p1} + k_{u1} m_1 V_1 c_{d1} \quad \text{(Desorption-sorption)}$$

$$- v_s A c_{p1} \quad \text{(Particulate settling)}$$

$$+ v_u A c_{p2} \quad \text{(Particulate resuspension)}$$

$$- K_{p1} V_1 c_{p1} \quad \text{(Decay)}$$

$$+ W_{p1} \quad \text{(Input)} \tag{10.9}$$

The first group of this equation is the transport of the particulate chemical due to net advection (Q) and dispersion (E'). The particulate chemical is assumed to be transported in the same manner as the dissolved form. The second group is the desorption-sorption mechanism discussed above and as can be noted for the particulate form, sorption is a source and desorption is a sink of toxicant. The third and fourth groups are respectively the particulate settling of the chemical from the water column and the resuspension of particulate chemical from the sediment into the water column. The settling velocity, $v_S(l/t)$ and the resuspension velocity $v_u(l/t)$ are functions of particle type (sand, silt, organics) and the hydrodynamics of the water–sediment interface. The fifth group represents any decay mechanisms (e.g. bacterial degradation) of the chemical on/in the particulates at a rate $K_{p1}[l/t]$ and the last line is the external mass input of particulate toxicant, $W_{p1}[m_T/t]$.

The particulate chemical in the sediment is given by an equation similar to Equation 10.9 except that, as noted, the sediment is assumed to be stationary in the horizontal direction. That is, bed load transport or sediment movement horizontally throughout the water body is not considered.

The particulate chemical equation for the sediment segment underlying the water column segment 1 is then given by

$$\begin{aligned} V_2 \frac{dc_{p2}}{dt} &= -k_{d2} V_2 c_{p2} + k_{u2} m_2 V_2 c_{d2} \\ &+ v_s A c_{p1} - v_u A c_{p2} \\ &- K_{p2} V_2 c_{p2} \\ &- v_d A c_{p2} \end{aligned} \tag{10.10}$$

The first three lines of this equation parallel the equivalent mechanisms in the water column (sorption-desorption, settling-resuspension, and decay; at rate K_{p2}). The fourth line represents the net downward flux of sediment particulate toxicant due to the net sedimentation velocity v_d. Again, mixing of the sediment due to factors such as bioturbation or deep sediment mixing is not included, but can be added as an additional mixing term.

10.1.2 Local Equilibrium Equations

Equations 10.6 and 10.8 for the dissolved component and Equations 10.9 and 10.10 for the particulate component in the water column segments and sediment segments, respectively, represent a set of interactive, differential equations, one for each control volume of the finite difference grid. Note that the coupling of the dissolved and particulate components is through the reaction kinetics of sorption and desorption. For some chemicals, these reaction kinetics tend to be "fast" (i.e. completion times on the order of hours) compared to the kinetics inherent in other mechanisms of the problem. These latter mechanisms include bacterial decay, net loss

rates to the sediment, and sedimentation rates that have reaction times on the order of days to years.

The "fast" kinetics of sorption-desorption indicate that for time scales of days to years, there will be a virtually continuous equilibration of the dissolved and particulate forms depending on the local solids concentration. This partitioning between the two components permits the specification of the fraction of dissolved and particulate chemical to the total. The dissolved and particulate chemical are therefore assumed to be always in a "local equilibrium" with each other. Assuming that the kinetics are reversible and that the sorption/desorption kinetics are linear, then a partition coefficient $\P[m_T/m_S \div m_T/l_w^3]$ can be defined as follows:

$$\P = r/c_d' \tag{10.11}$$

or since $c_d' = c_d/\phi$

$$\P' = \P/\phi = r/c_d \tag{10.12}$$

for $\P'$ as $[m_T/m_s \div m_t/l^3]$ and r as the chemical concentration on a solids basis $[m_T/m_s]$.

The particulate toxicant concentration relative to the bulk volume is given by (for m as the solids concentration, m_s/l^3)

$$c_p = rm \tag{10.13}$$

The fraction of the total that is dissolved, f_d, is given by

$$f_d = (1 + \P'm)^{-1} \tag{10.14}$$

and the particulate chemical as a fraction of total chemical (f_p) is given by

$$f_p = \frac{\P'm}{1 + \P'm} \tag{10.15}$$

The local equilibrium assumption therefore permits specification at all times and places of the fraction of the total toxicant in the dissolved and particulate form. It should be stressed again here that this local equilibrium assumption assumes complete reversibility between the solid and liquid phases. There is evidence (e.g., Di Toro et al. 1982a; Di Toro 1985) that this is not the case for certain chemicals.

Also in these relationships it is assumed that the partition coefficient does not depend on the concentration of the sorbing solids. There is considerable evidence, however, as given by O'Connor and Connolly (1980) and Di Toro (1985) who indicate that the partition coefficient does apparently depend on the concentration of solids. The development continues here on the assumption of a constant partition coefficient.

With this assumption, attention can then be focused solely on the mass balance equation for the total chemical. The total chemical in the water column or sediment is given by Equation 10.3. Adding the water column equations for dissolved chemical (Equation 10.6) and particulate chemical (Equation 10.9) and using Equations 10.14 and 10.15 gives

$$V_1 \frac{dc_{T1}}{dt} = [Qc_{T1}^+ - Q_1 c_{T1} + E'(c_{T1}^+ - c_{T1}) + E'(c_{T1}^- - c_{T1})$$
$$+ k_f A(f_{d2}c_{T2}/\phi_2 - f_{d1}c_{T1}) - (K_1)V_1 c_{T1}$$
$$+ k_{l1}A[(c_g/H_e) - f_{d1}c_{T1}] - v_s A f_{p1} c_{T1} + v_u A f_{p2} c_{T2}$$
$$+ W_{T1} \tag{10.16}$$

where $K_1 = K_{d1} + K_{p1}$

Note that the kinetics of sorption-desorption do not appear in this equation because it represents a mass balance of the total. The net loss rates and exchanges that are dependent on the form of the toxicant do however, remain.

A total chemical equation for the sediment segment (subscript 2) can be obtained in a similar manner. Thus adding Equations 10.8 and 10.10 gives:

$$V_2 \frac{dc_{T2}}{dt} = -K_f A(f_{d2}c_{T2}/\phi_2 - f_{d1}c_{T1}) - (K_2)V_2 c_{T2}$$
$$+ v_s A f_{p1} c_{T1} - v_u A f_{p2} c_{T2} - v_d A f p_2 c_{T2}$$
$$+ K_f A(f_{d3}c_{T3}/\phi_3 - f_{d2}c_{T2}/\phi_2) \tag{10.17}$$

where $K_2 = K_{d2} + K_{p2}$

Equations 10.16 and 10.17 are the fundamental equations used in the succeeding analyses. These equations are coupled parametically to the suspended solids and sediment solids concentrations (see Equations 10.14 and 10.15). These concentrations can be specified externally as an input or the mechanisms of solids settling, resuspension, and deposition can be explicitly modeled. In addition, an independent tracer can be used to calibrate these parameters; see Thomann and Di Toro (1983) for the use of plutonium-239,240 as a tracer.

10.1.3 Food Chain Model

The transfer of a chemical in the aquatic food chain occurs through two principal routes:

1. direct uptake from the water.
2. accumulation due to consumption of contaminated prey.

The uptake of a chemical directly from water through transfer across the gills as in fish or through surface sorption and subsequent cellular incorporation as in phytoplankton is an important route for transfer of chemicals. This uptake is often measured by laboratory experiments dur-

ing which test organisms are placed in aquaria with known (and fixed) water concentrations of the chemical. The accumulation of the chemical over time is then measured and the resulting equilibrium concentration in the organism divided by the water concentration is termed the bioconcentration factor (BCF). A simple representation of this mechanism is given by a mass balance equation around a given organism. Thus,

$$\frac{d\nu'}{dt} = k_u w c - K\nu' \tag{10.18}$$

where ν' is the whole body burden of the chemical (m_T); k_u is the uptake sorption and/or transfer rate ($l^3/t \cdot m(w)$; $m(w)$ = mass of organism, wet weight; w is the weight of the organism, $m(w)$; c is the dissolved water concentration (m_T/l_w^3); K is the desorption and excretion rate ($1/t$) and t is time. This equation indicates that the mass input ($\mu g/d$) of toxicant given by $k_u wc$ is offset by the depuration mass loss rate ($\mu g/d$) given by $K\nu'$. The whole body burden ν' is given by

$$\nu' = \nu w \tag{10.19}$$

where ν is the concentration of the chemical ($m_T/m(w)$). Substitution of Equation 10.19 into Equation 10.18 gives, after simplification

$$\frac{d\nu}{dt} = k_u c - K'\nu \tag{10.20}$$

where $K' = K + G$

for $G(1/t)$ as the net growth rate of the weight of the organism. At equilibrium or steady state,

$$\nu = \frac{k_u c}{K'} \tag{10.21}$$

and the BCF is given by

$$N_w = \frac{\nu}{c} = \frac{k_u}{K + G} \tag{10.22}$$

The ratio N_w, the bioconcentration factor, is in units $m_T/m(w) \div m_T/l^3$, e.g., μg/kg $\div$ μg/l (= l/kg).

For organic chemicals, the BCF is conveniently defined on a lipid-normalized basis, i.e., $m_T/m(\text{lip}) \div m_T/l^3$, e.g., μg/kg(lipid) $\div$ μg/l. The lipid normalization assumes that the lipid compartment of the organism is the principal receptor of the hydrophobic organic chemical.

The octanol-water partition coefficient (K_{ow}) of a chemical is a useful ordering parameter to express the tendency of organic chemicals to partition into the lipid pool.

At equilibrium then for organic chemical BCF, to first approximation,

$$N_w = K_{ow} \tag{10.23}$$

for the laboratory case of no organism growth and N_w now defined as the lipid normalized BCF. Thomann (1987, 1988) suggests the following expression for the field BCF, as a function of K_{ow}

$$N_w = K_{ow}\left[1 + \frac{10^{-6}K_{ow}}{E(K_{ow})}\right]^{-1} \tag{10.24}$$

where $E(K_{ow})$ is an efficiency of chemical transfer across the gills as a function of K_{ow} and for fish can be approximately expressed as (Thomann 1987, 1988)

$$\begin{aligned} \log E &= -1.5 + 0.4 \log K_{ow} && \text{for } \log K_{ow} = 2\text{–}3 \\ E &= 0.5 && \text{for } \log K_{ow} = 3\text{–}6 \\ \log E &= 1.2 - 0.25 \log K_{ow} && \text{for } \log K_{ow} = 6\text{–}10 \end{aligned} \tag{10.25}$$

Age-Dependent Model

The general age-dependent model utilizes a mass balance of chemical around a defined compartment of the aquatic ecosystem. In the most general case, a compartment is defined as a specified age class of specified organism or in steady-state simplification, a compartment is considered as an "average" age class or range of ages for a given organism. Figure 10.2 schematically shows the compartments. As indicated in Figure 10.2a, each age class of a given trophic level is considered as a compartment and a mass balance equation can be written around each such age class. The zero trophic level is considered to be the phytoplankton–detritus component representing one of the principal sorption mechanisms for incorporating toxicants into the food chain.

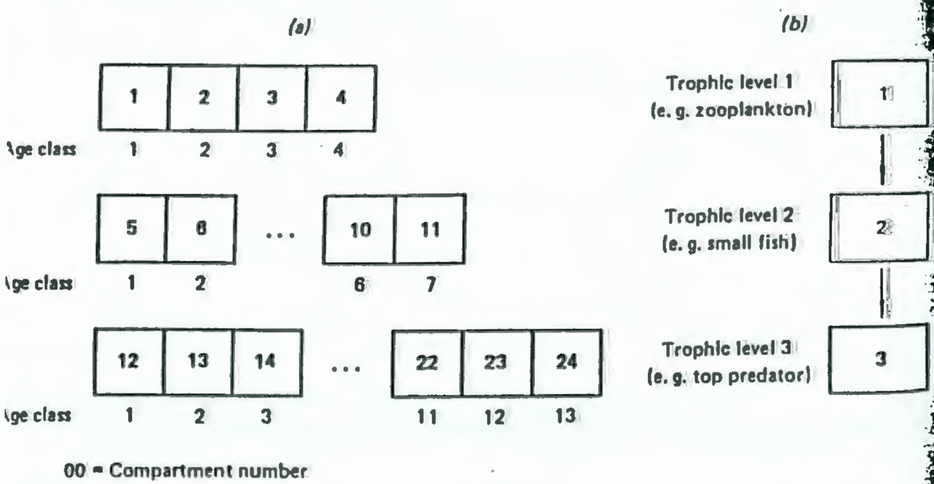

Figure 10.2. Schematic of compartment definition for (*a*) age dependent model, and (*b*) simplified steady-state model.

Consider then the phytoplankton, detrital organic material, and other organisms, all of size approximately $< 100\ \mu m$ as the base of the food chain. An equation for this compartment is given by a simple reversible sorption-desorption linear equation as:

$$\frac{d\nu_0}{dt} = k_{u0}c - K_0\nu_0 \tag{10.26}$$

where all terms have been defined, the subscript zero refers to the base of the food chain, and t is real time.

For a compartment above the phytoplankton/detritus level, the mass input of the toxicant due to ingestion of contaminated food must be included. This mass input will depend on: toxicant concentration in the food; rate of consumption of food; and the degree to which the ingested toxicant in the food is actually assimilated into the tissues of the organisms.

The general mass balance equation for the whole body burden for a given compartment, i, is then similar to Equation 10.18 for water uptake but with the additional mass input due to feeding. Therefore,

$$\frac{d\nu_i'}{dt} = \frac{d(\nu w)_i}{dt} = \frac{w_i d\nu_i}{dt} + \frac{\nu_i dw_i}{dt} = k_{ui}w_i c - K_i\nu_i' + \sum_j p_{ij}\alpha_{ij}C_i\nu_j w_i \qquad i = 1 \ldots m \tag{10.27}$$

where α_{ij} is the chemical assimilation efficiency (m_T absorbed/m_T ingested); C_i is the weight-specific consumption of organism i, $m(w)$ predator/$m(w)$ prey-d; p_{ij} is the food preference of i on j; and t is real time (days). Consider now a simple case of a sequential food chain where predation is only on the next lowest trophic level.

An equation for the individual organism weight is

$$\frac{dw_i}{dt} = (a_{i,i-1}C_i - r_i)w_i, \qquad i = 1 \ldots m \tag{10.28}$$

where $a_{i,i-1}$ is the biomass assimilation efficiency, $m(w)$ predator/$m(w)$ prey, and r_i is the respiratory weight loss ($1/t$) due to routine metabolism, swimming, and other activities. The weight change is therefore

$$G_i = \frac{dw_i}{dt}\Big/ w_i = (a_{i,i-1}C_i - r_i) \tag{10.29}$$

Equation 10.27, for a food chain in contrast to a food web, can then be written as

$$\frac{d\nu_i}{dt} = k_{ui}c + \alpha_{i,i-1}C_i\nu_{i-1} - K_i'\nu_i, \qquad i = 1 \ldots m \tag{10.30}$$

where i is the predator and $i - 1$ is the prey.

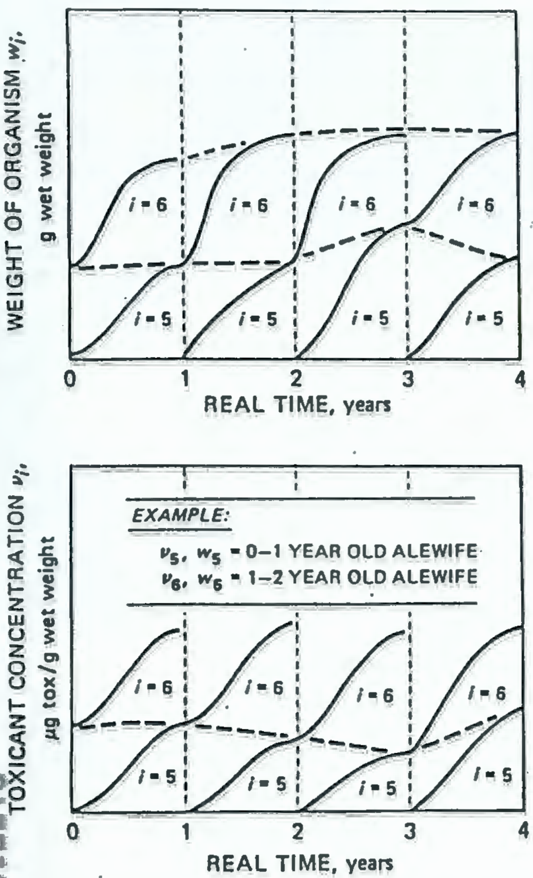

Figure 10.3. Illustration of meaning of $w_i(t)$ and $\nu_i(t)$ showing as an example a 0–1-year-old and 1–2-year-old alewife.

The interpretation of w_i and ν_i in Equations 10.28 and 10.30 is further explained in Figure 10.3. The variation of the weight (and chemical organism concentration) of a given compartment (i.e., a given age class of a given organism) is shown with real time. If as an example, w_5 is a 0–1-year-old alewife then it is seen that the weight of this age class may vary from year to year. Similarly, the distribution of the chemical may change from year to year for a given compartment depending on, for example, the variation in the water column toxicant concentration. The specification of the boundaries of a given compartment depends on the life cycle of the organism.

It should be recognized therefore that several of the biological and chemical parameters of the weight change equation 10.29 and the food chain equation 10.30 are functions of organism weight within the time interval defined by the compartment. See Thomann and Connolly (1984) for an application of the age-dependent model to PCB accumulation in the lake trout of Lake Michigan.

10.2 Steady-State Simplification

10.2.1 River Physico-Chemical Fate

In spite of the complexity of Equations 10.16 and 10.17, it can be shown (see e.g., Thomann and Mueller 1987) that under steady-state and constant spatial parameters that for a single source to first approximation, the maximum concentration c_o is at the outfall. Thus,

$$c_o = (Q_u c_u + Q_e c_e)/Q \tag{10.31}$$

where Q_u and c_u are the upstream flow and concentration, respectively; Q_e and c_e are the effluent flow and concentration, respectively; and Q is the total river flow.

Since the maximum generally occurs at the outfall, there are generally two situations to be considered in estimating the downstream fate of a discharged chemical:

1. There is a critical water use point downstream of a single discharge and the concentration at the point of use (e.g., a water supply withdrawal) needs to be estimated.
2. There are several inputs of the same chemical along the length of the river and the total concentration must be estimated.

The downstream fate of the chemical or mixture depends on:

1. The properties of the river such as the depth, velocity, and dilution downstream due to groundwater infiltration or tributary inflow.
2. The chemical properties, such as volatilization, biodegradation, or partitioning onto the solids.

Thomann and Mueller (1987) discuss these factors in some detail. A simplified summary from Thomann and Salas (1986) is given here. An important point for the computation of the downstream fate of a chemical or chemical mixture is that the calculation is very similar, indeed for preliminary analyses, the computation is identical to classical stream water quality calculations. The basic equation under steady-state conditions is given by

$$c_T = c_{T0}\, exp\left[-\left(\frac{v_T}{HU} + q\right) x\right]$$

$$= c_{T0}\, exp[-K_T + q')t^*] \tag{10.32}$$

where c_T is the concentration as a function of distance downstream; K_T is the overall net loss rate ($1/t$); v_T is the net loss of chemical expressed as velocity, l/t; t^* is the time of travel ($= x/U$); q is the slope of the natural logarithm of river flow with distance (the flow dilution rate); H is river depth; U is river velocity; and q' is the river flow slope on a travel time basis.

The calculation of the downstream fate of a toxic substance or toxicity of an effluent or effluents depends then on the estimation of the dilution of the river and the loss rate of the chemical. The discharge of a conservative substance with no dilution would result in a constant concentration in the downstream direction, i.e., from Equation 10.32 with K_T and $q' = 0$, $c = c_o$.

The determination of whether there is a downstream infiltration of groundwater or overland drainage can be made by examining the downstream distribution of a known conservative substance such as chloride or total dissolved solids or other tracer.

If dilution exists then a conservative substance may exhibit apparent non-conservative behavior. Then for $K_T = 0$, but $q' \neq 0$, Equation 10.32 is

$$c = c_o \, exp(-qt^*) \qquad (10.33)$$

which indicates an exponential decline in the conservative substance due to distributed downstream dilution.

Finally, if the toxic substance is non-conservative, i.e., the chemical undergoes biodegradation or volatilization or other losses and dilution is also occurring, then the net loss rate of the chemical, K_T must be estimated.

Table 10.1 provides some guidelines for a preliminary assessment of downstream fate. Toxicity in this table is whole-effluent toxicity of the chemical mixture to standard organisms and is expressed on a toxic-unit basis. A toxic unit is defined (USEPA, 1985)

$$\text{Tu} = 100/\text{LC}_{50} \text{ or NOEL} \qquad (10.34)$$

where LC_{50} is the lethal concentration to 50% of the organisms and NOEL is the no observed effect level. As indicated, both the heavy metals and the toxicity measure are assumed to be a conservative variable for first approximations. Thus, only dilution needs to be considered for these variables. For the organic chemicals a somewhat arbitrary division has been made based on the water solubility of the chemical.

Table 10.1. Guideline for estimating downstream loss rate of chemicals and toxicity

Group	Guideline[a]
Heavy metals	Conservative ($K_T = 0$) and additive
Toxicity	Conservative ($K_T = 0$) and additive
Organic chemicals, water solubility > 1 μg/l	Conservative ($K_T = 0$) and additive
Organic chemicals, water solubility > 1 μg/l	Estimate loss rate (Eq. 10.34)

[a] In all cases, dilution in the downstream direction must be included. Source: Thomann and Salas 1986.

The rationale is that at solubilities less than about 1 μg/l, the chemical will partition onto the solids because of a relatively high partition coefficient (about 10^4–10^6 l/kg). Also, the general tendency will be for such chemicals to biodegrade and volatilize to a lesser degree than the more soluble chemicals. For first approximations, then, such low solubility chemicals may be assumed conservative.

For organic chemicals with solubilities greater than about 1 μg/l, the loss rate must be estimated. For first approximations, the net loss velocity v_T ($=K_TH$) can be estimated from

$$v_T = K_T H = (K_d H + k_l) f_d + v_n f_p \qquad (10.35)$$

where $K_d[T^{-1}]$ is the decay rate of the chemical due to processes such as biodegradation or photolysis; k_l is the loss due to volatilization; v_n is the net loss velocity of the solids in the river, l/t; and f_d and f_p are the dissolved and particulate fractions of the total (see Equations 10.14 and 10.15).

For rivers, the net loss of solids often, although not always, can be assumed equal to zero. Thus, $v_n = 0$ and the chemical loss rate depends only on the degradation rate, volatilization rate, and fraction dissolved.

Table 10.2 provides some guidelines for estimating the fraction of the chemical that is in the dissolved form. These guidelines employ a more complex interaction of chemical partitioning and solids concentration as given in Di Toro (1985) and discussed in Thomann and Mueller (1987). The f_d can also be approximated with Equation 10.14. Figure 10.4 shows

Table 10.2. Approximate fraction of total chemical in dissolved form and on particulates

Organic chemicals	(f_d) fraction of chemical in dissolved form[a]		Ratio of particulate conc. to total conc. (r/c_T [μg/g ÷ μg/l])[b]	
Chemical solubility (μg/l)	Range	Approx. mean	Range	Approx. mean
>100	0.5–1.0	0.7	0–50	10
10–100	0.3–0.9	0.5	0.1–70	50
<10	0.3–0.8	0.4	0.1–70	60
Heavy metals				
	0.6–1.0	0.8	0.4–16	2.5

[a] Approximated from solids dependent partition coefficient relationships of Di Toro (1985), solids range 10 → 1000 mg/l

[b] $r/c_T = (1 - f_d)/(.01 \rightarrow 1.0)$

Source: Thomann and Salas 1986.

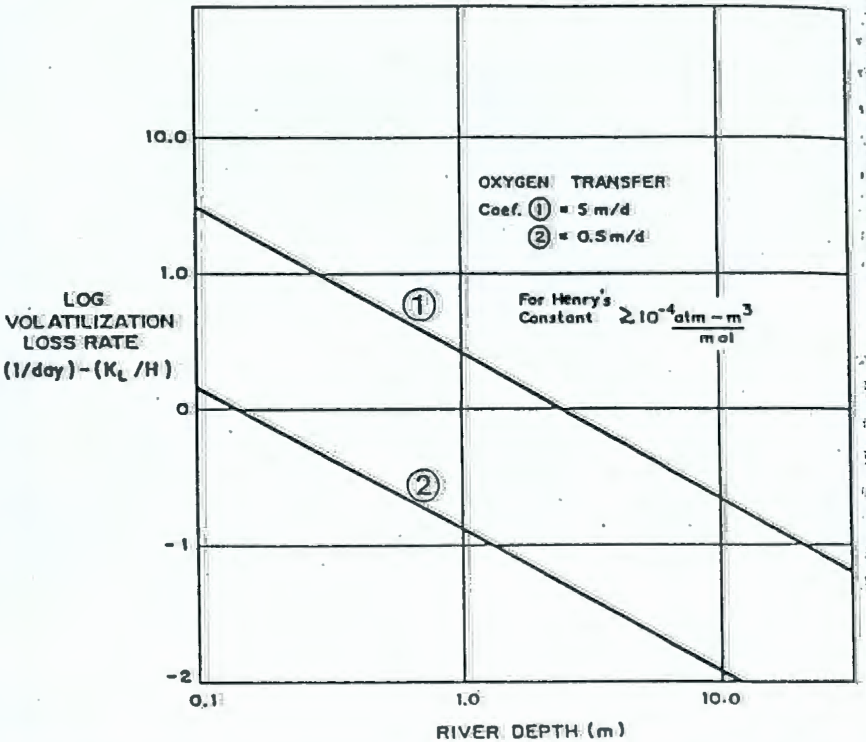

Figure 10.4. Range of volatilization loss rate as a function of river depth for different oxygen transfer rates.

the range of the volatilization loss rate as a function of the river depth and reaeration characteristics. This figure is for substances with Henry's constant > 10^{-4} atm-m³/mol for which the volatilization rate is estimated from the oxygen transfer rate K_L. Thus,

$$k_l \approx \left(\frac{32}{M}\right)^{1/4} K_L \qquad (10.36)$$

where $K_L = \left(D_L \dfrac{U}{H}\right)^{1/2}$

for D = *oxygen diffusivity* (.000181 m^2/d)

The loss rate, K_d, is generally site-specific and chemical specific and no general simplification is available.

Table 10.2 also gives the approximate ratio of the chemical adsorbed to the suspended particulates to the total water concentration. Therefore,

$$\Pi = r/c_T \qquad (10.37)$$

for Π in μg/g(d) ÷ μg/l, r in μg/g(d) and c_T in μg/l, g(d) = grams dry weight.

It can also be shown that for $\P_2 = \P_1$ and $K_{d2} = 0$, the sediment particulate concentration is equal to the water column particulate concentration, i.e.,

$$r_2 = r_1 \qquad (10.38)$$

where r_2 is the sediment particulate concentration and r_1 is the water column particulate concentration both in μg/g(d).

In summary, it is seen that the calculation of the fate of the chemical or mixture is similar to the procedure for conventional water quality variables. For many chemicals including the toxicity measure, the assumption that the chemical is conservative is appropriate for first approximations. If such an assumption cannot be made, then an estimate of the downstream loss rate must be made using the preliminary guidelines discussed here.

10.3 Deterministic Time Variable Models

10.3.1 Application of Time Variable Model to Benzo(a)pyrene and Cadmium in the Great Lakes

In this section, the fully time-variable model (Equations 10.16 and 10.17) is applied to two chemicals: 1. benzo(a)pyrene, a polycyclic aromatic hydrocarbon (PAH), and 2. cadmium, a representative metal. The model uses the Great Lakes segmentation of Thomann and Di Toro (1983) shown in Figure 10.5 and details are in Thomann and Di Toro (1984).

Benz(a)pyrene

The distribution of this chemical, one of the PAH compounds resulting from incomplete combustion of organic materials has been widely studied (e.g., Neff 1979) because of its potential carcinogenicity. The fate of benzo(a)pyrene (BaP) in the Great Lakes has been evaluated in a series of papers by (Eadie 1983; Eadie et al. 1982, 1983). In that work, data are presented for the range of concentration of BaP in the water column and surficial sediments as well as preliminary data on the BaP concentration in the pore water of the sediment. It is those data (together with estimates of loading) that can be used as an application of the physico-chemical model.

BaP is sparingly soluble in water 0.172 μg/l (Neff 1979) and as such would be expected to have an affinity for solids. The partitioning onto particulates and in addition, the extent of volatilization of the BaP must be

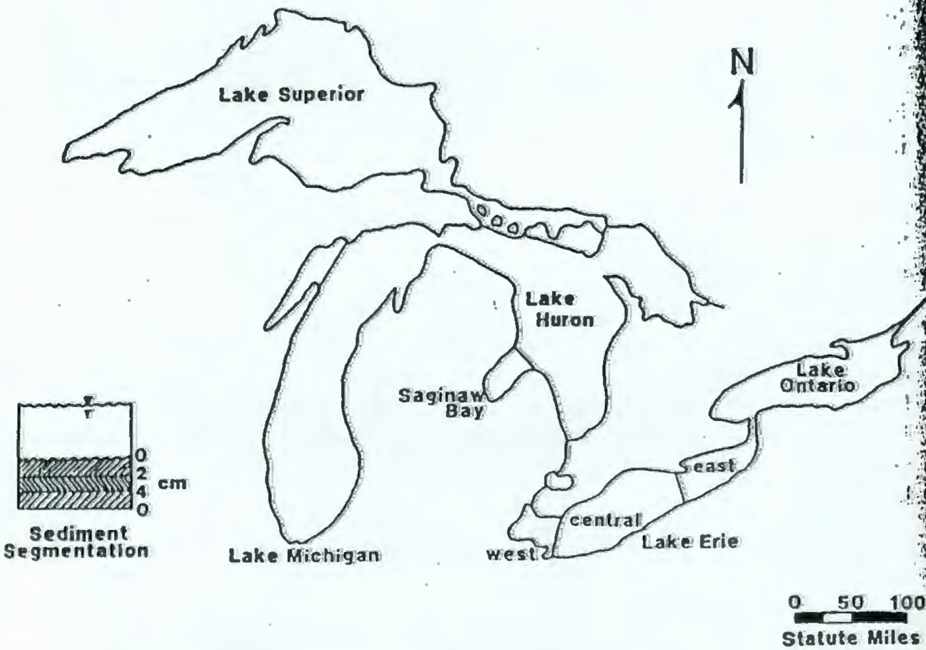

Figure 10.5. Great Lakes and Saginaw Bay segmentation used in model.

estimated. Although BaP is known to undergo photolysis (Neff 1979) this pathway is not considered in this application.

The estimated atmospheric loading on an area basis is about 95 g BaP/km^2-yr across all of the lakes but as noted by Eadie (1983), all BaP load estimates are based on quite limited data and therefore may vary as additional information becomes available. From the data given in Eadie (1983) for Lake Michigan, and other data, the model for BaP was run for ¶ from 10,000 to 100,000 l/kg, thereby providing a range of one order of magnitude in the solids partition coefficient.

The time variable calculation using a steady loading for 20 years is shown in Table 10.3 and in Figures 10.6 and 10.7. As shown in Table 10.3, the lake-to-lake variation in BaP concentrations either in the water column or the sediment differs by less than about a factor of two. The highest concentration of BaP in the surface sediments (¶ = 100,000) is in Saginaw Bay. It can also be noted in Table 10.3 and Figure 10.6 that at ¶ = 10,000 the calculated surface sediment concentration for Lakes Michigan and Erie is about 45 ng/g(d) or about one order of magnitude lower than the observed data. Figure 10.7 shows the calculated time history under the two partition coefficients. For ¶ = 100,000 l/kg, the water column and sediment are at about 80 and 60% of steady state, respectively, while for ¶ = 100,000 l/kg, the water column and sediment are about 20% of steady

Table 10.3. Comparison of calculated and observed BaP for Great Lakes under different solids partition assumptions

	Calculated range of BaP across all lakes[a]			
	¶ = 10,000 l/kg	¶ = 100,000 l/kg	Observed mean of BaP	Reference
Total water conc. (ng/l)	5–6	1–2	12 ± 8[b]	Eadie et al. 1983
Surficial sediment conc. (ng/g(d))	38–60	46–133	Michigan 480 ± 246(7)[c] Erie 255 ± 152(3) Superior 28(1) Huron 294(1) Ontario 306(1)	Eadie 1983
Sediment pore water conc. (ng/l)	3–5	0.5–1.3	850 ± 1260	Eadie et al. 1983
Particulate conc. in water column (ng/g(d))	46–64	46–165	Michigan 200–400	Eadie 1983

[a] After 20 years of loading
[b] Mean ± SD
[c] ( ) = no. of samples

Figure 10.6 and Table 10.3 indicate that a more favorable (but not totally desirable) comparison to observed data is obtained at the higher BaP partition coefficient of 100,000 l/kg. The results also indicate the need to determine the partition coefficient for BaP, as a representative PAH for Great Lakes solids concentrations. On the basis of this application of the physico-chemical model to BaP in the Great Lakes, it is concluded that:

1. The estimate of the BaP partition coefficient, obtained from published empirical relationships, is probably low by about an order of magnitude for the Great Lakes system.
2. With an increased BaP partition coefficient and assuming loss due to volatilization, the physico-chemical toxic substances model of the Great Lakes approximate observed BaP water column and sediment data only to order of magnitude.
3. The model confirms that on a lake-wide scale, the principal external source of BaP is the atmosphere.
4. For larger lakes such as Lake Michigan, the 50% response time of the lake to external loads is about 6–10 years for the water column–sediment system while for Lake Erie the response time is about 2 years.
5. Lake-to-lake variations in BaP water column and sediment concentrations are less than a factor of two.

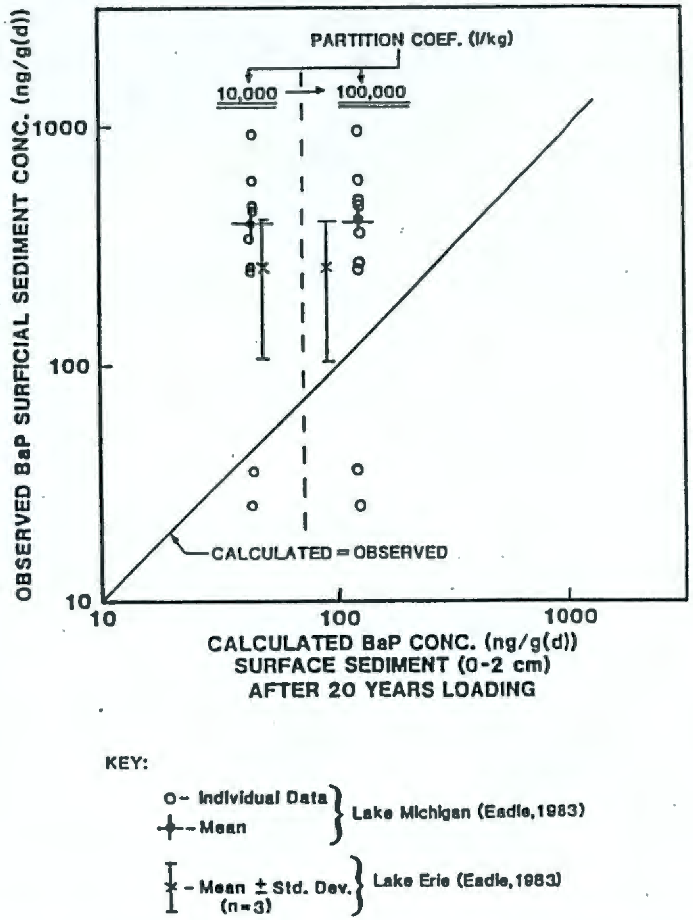

Figure 10.6. Comparison between calculated surface sediment BaP concentration after 20 years and observed concentration.

Cadmium

Time variable model calculations for cadmium were made using the low and high load estimates of Table 10.4 and two assumptions on the cadmium partition coefficient: a. variable partition with solids concentration as given in HydroQual (1982) by

$$\P = (3.52 \cdot 10^6)m^{-0.92} \tag{10.39}$$

and b. a constant partition coefficient of $2 \cdot 10^5$ l/kg. Cadmium was assumed to be conservative. For all calculations, zero initial conditions were assumed and the loads were input as constant over time. It became apparent from initial runs that the time to steady state especially for the

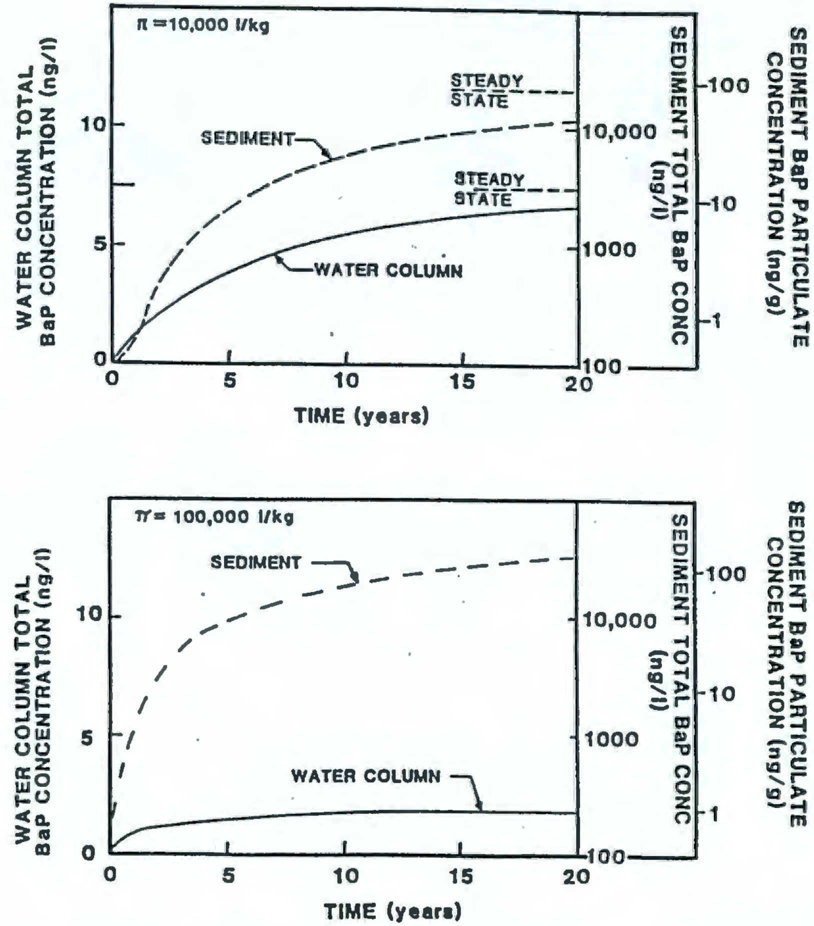

Figure 10.7. Time variable benzo(a)pyrene (BaP) response in Lake Michigan under two partition coefficients with 20 year constant loading.

upper lakes is long so that the computation was carried out for a 100-year period. The computation therefore represents the response of the Great Lakes system to a constant loading and such loading can be viewed as a loading in addition to the background loading and cadmium concentrations shown in Table 10.5. If increased loading of cadmium is assumed to have begun in approximately the 1920s, then the output from the model calculation at $t = 50$ years would be representative of the 1970s. This is the period for which some reliable data are available.

Figure 10.8 shows the comparison of calculated and observed surface sediment cadmium data for $t = 50$ years. As shown, the calculation is reasonable to order magnitude. Figure 10.9 (high-load estimate) shows the full 100-year calculation for Lake Michigan and central Lake Erie as

Table 10.4. Summary of contemporary external cadmium loads (not including upstream loads)

Lake/Region	Atmospheric[a] mt/yr	Tributary[c] mt/yr	Mun. + Ind.[d] (mt/yr)	Total mt/yr	Total g/km²-yr
iperior	41–108	13–126	0–0.3	54–234	651–2823
ichigan	12–120	9–92	0.4–4	21–216	363–3739
uron	23–57	14–136	0.1–1	37–194	644–3378
Saginaw Bay[b]	3	2	0	5	1184
rie West	1–18	2–20	1–8	4–46	1321–15202
Central	3–94	2–16	0.2–2	5–112	318–7126
East	1–38	1–12	0.1–0.5	2–51	320–8155
ntario	10–44	7–66	0.6–6	18–116	924–5953

[a] From Allen and Halley 1980.
[b] From Dolan and Bierman 1982.
[c] Assuming tributary flow at total cadmium conc. of 0.2–1.0 μg/l.
[d] At 0.5–5.0 μg/l for direct municipal point sources.

illustrations. The sensitivity of the calculation to the assumptions in the partition coefficient is shown. As indicated, the effect of the solids-dependent partition coefficient is to greatly increase the time to steady state as a result of the diffusive flux of cadmium from the sediment due to the lower partition coefficient. For Lake Michigan, the surface sediment concentration decreases with the variable partition coefficient but for Lake Erie the

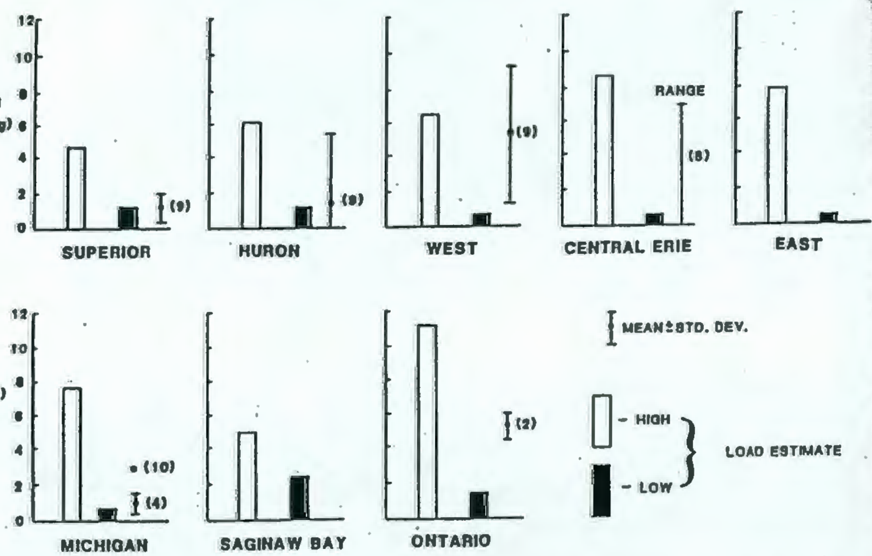

Figure 10.8. Calculated surface sediment concentration, μg/g(d), at t = 50 years with partition coefficient as a function of solids concentration.

Table 10.5. Estimate of approximate background cadmium concentration for Great Lakes

Lake/Region	Range in total background water col. conc. (ng/l)[a]	Background conc. used in load estimate (ng/l)	Background total load (mt/yr)[b]	External background load (mt/yr)[c]	External background load g/km²-yr	Ratio of contemporary load to background load
Superior	2.8–5.6	4.0	6.1	6.1	37	18–76
Michigan	2.8–5.6	4.0	3.1	3.1	54	7–69
Huron	2.8–5.6	4.0	5.2	4.8	83	8–41
Saginaw Bay	6.5–13.0	10.0	0.6	0.4	105	11
Erie West	12.5–25.0	20.0	10.3	8.6	283	5–54
Central	5.0–10.0	8.0	22.0	10.0	635	0.5–11
East	5.0–10.0	8.0	12.0	4.7	754	0.4–11
Ontario	2.8–5.6	4.0	4.0	2.6	132	7–45

[a] At assumed background sediment conc. of 05–1.0 μg Cd/g(d); $r_2 = r_1$ and ¶ = 200,000 l/kg.
[b] Includes any upstream or exchange loads.
[c] Not including any upstream or exchange loads.

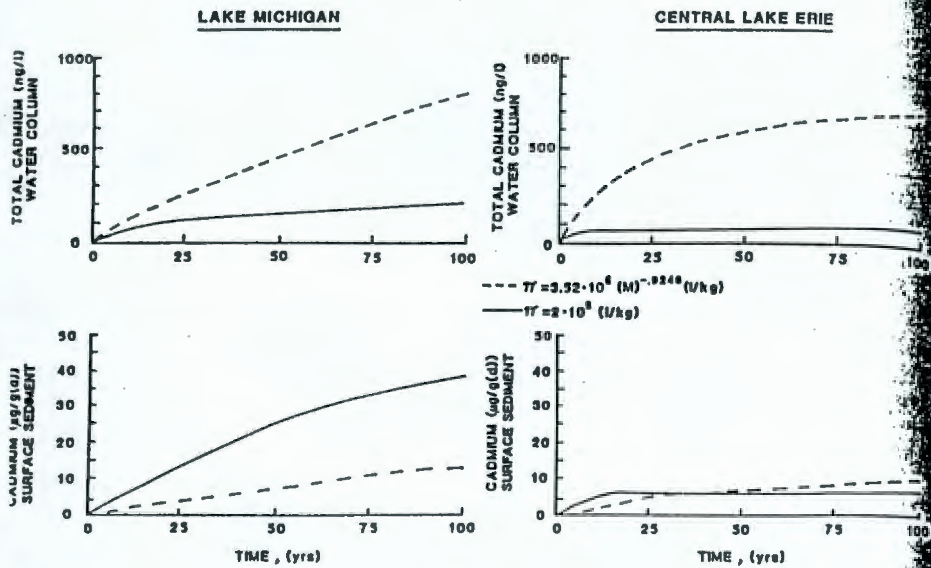

Figure 10.9. Comparison of calculated cadmium concentration under two assumptions on partition coefficient. High load estimate.

surface sediment concentration increases. The continual increase in concentration for central Lake Erie reflects the non-equilibrium condition of the upstream lakes.

If a constant partition coefficient is assumed (as in Muhlbaier and Tisu 1981), then it is seen that for the water column a steady state is reached after about 25 years for Lake Michigan and after about 10 years for Lake Erie. A calculation then for Lake Michigan that attempts to calibrate to a mean concentration of 27 ng/l (Muhlbaier and Tisue 1981) then is simply a matter of estimating the approximate average load and may not reflect a non-steady-state condition as concluded by Mulhbaier and Tisu (1981). However under a solids-dependent partition coefficient for cadmium, the Great Lakes are not in equilibrium with the external load and for all practical purposes never reach a steady-state condition. Clearly then, under this model construct, it is important to determine the solids dependence of cadmium for the range of solids encountered in the Great lakes water column and sediments (i.e., 0.5–240,000 mg/l). If however it is assumed that a solids-dependent partition coefficient is applicable to the Great Lakes, then the system is not in equilibrium with respect to the external load.

It is concluded from this application of the time variable physico-chemical model to cadmium in the Great Lakes that

1. The degree of any dependence of the cadmium partition coefficient with solids has a marked effect on time to steady state and interstitial

2. Under a solids-dependent cadmium partition assumption, the Great Lakes, especially the upper Lakes, do not reach a steady-state condition after 100 years of constant loading.
3. Under a constant partition coefficient for cadmium, the Great Lakes do reach an equilibrium condition varying from about 25 years for Lake Michigan to 10 years for Lake Erie.
4. The concentration of cadmium in the Lakes would be expected to increase by about 60% over the next 50 years if the average cadmium loading for the preceding 50 years continues.

10.4 Statistical Variation in Fish

The statistical behavior of a chemical in a fish population is of interest for at least three reasons:

1. U.S. Food and Drug Administration action limits for fish have resulted in the closing of commercial fisheries because of excessive concentrations of chemicals in predators such as the striped bass in the Hudson estuary and surrounding waters. The ability to predict not only the mean value of a chemical in a fish but also the variance of the concentration is of importance in control strategies needed to reopen a fishery.
2. The relationships between variable exposure concentrations in the water column and resulting variability in fish and other organisms is also related to the subsequent acute and chronic toxicity effects on the organism, and, as such, a framework for predicting organism chemical variance is of value in elucidating toxic effects on aquatic animals.
3. The integration of the physico-chemical modeling framework (which includes external inputs of the chemical to the water column) and the biological modeling framework in a time variable sense to predict statistical properties in the water and fish is of particular value in a generalized risk assessment determination.

The variability of chemicals in water over time is large. Figure 10.10 from the Mississippi River is an example. Recognizing this variability in the water chemical concentration of the food chain model of Section 10.1.3, analytical models of expected response in the fish can be developed.

10.4.1 Analytical Model of Statistical Variation of an Organic Chemical in Fish–Water Uptake Only

Consider the case where the water concentration to which the fish is exposed is represented by a first-order autoregressive process. Thus, the

$$\rho_{ck} = exp(-ak);\ k = 0, 1, 2, \ldots m \tag{10.39}$$

where ρ_{ck} is the normalized autocovariance; k is the lag number; m is the maximum number of lags; and a is the exponential decay rate of the autocorrelation.

Now

$$\rho_{ck} = \rho_{cl}^{k} \tag{10.40}$$

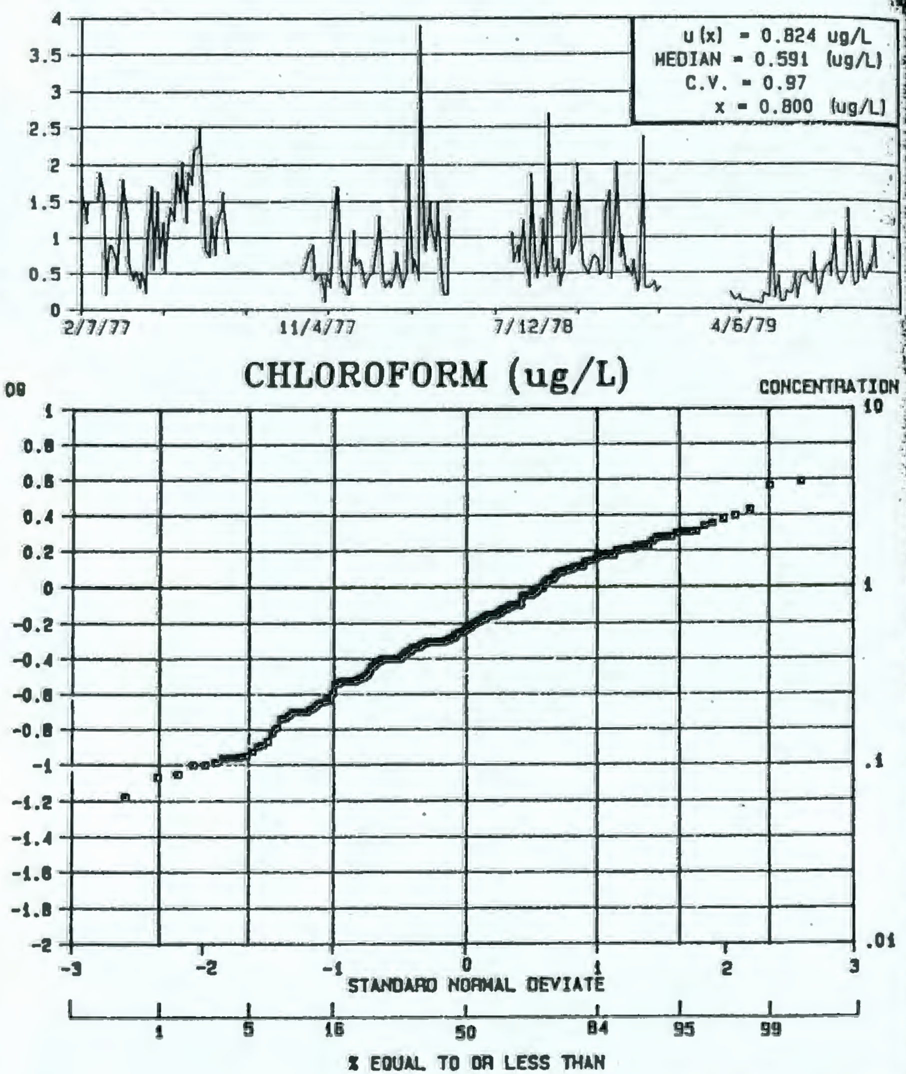

Figure 10.10. Organic chemical data from Mississippi River, Jefferson Parish station. (*Top*) Chloroform time series; (*bottom*) cumulative frequency distribution. (Compiled from data of USEPA 1986.)

and then from Equations 10.39 and 10.40,

$$\rho_{c1} = exp(-a) \tag{10.41}$$

For the synthetic generation of a time series x_t with a first-order autoregressive input, Bras and Iturbe (1985) give

$$(x_t - \mu) = \rho_1(x_{t-1} - \mu) + \sigma_x(1 - \rho_1^2)^{1/2}Z(t) \tag{10.42}$$

for mean μ, variance σ_x^2 and $Z(t)$ a standard normal deviate (mean = 0; variance = 1).

Thus if $a \gg 0$, i.e., the autocorrelation function drops rapidly to zero at about lag one, $\rho_1 \rightarrow 0$ and Equation 10.42 gives

$$x_t - \mu = \sigma_x Z(t)$$

or a normally distributed uncorrelated random variable, the "white noise" case.

For the autocorrelation given by Equation 10.39, Bendat and Piersol (1971) show that the spectrum for frequency f is

$$G_c^N(f) = \frac{4a}{a^2 + 4\pi^2 f^2} \tag{10.43}$$

where $G_c^N(f)$ is a normalized spectrum. Therefore,

$$G_c(f) = \sigma^2 G_c^N(f) \tag{10.44}$$

The variance of the fish concentration can then be shown to be (Thomann 1987)

$$\sigma_\nu^2 = \sigma_c^2 k_u^2 4a \int_0^\infty (K'^2 + 4\pi^2 f^2)^{-1/2}(a^2 + 4\pi^2 f^2)^{-1/2}\, df \tag{10.45}$$

Completing the integration, the ratio of the coefficients of variation between the fish and the water is

$$r = \sqrt{\frac{K'}{a + K'}} \tag{10.46}$$

where $r = \dfrac{v_\nu}{v_c}$ and $K' = K + G$,

for v_ν and v_c as the coefficients of variation of the chemical in the fish and in the water, respectively. This remarkably simple result indicates the significance of the depuration rate plus growth rate, K', as the principal controlling factors in generating relative variability in the fish concentration.

In Equation 10.46, r depends primarily on K' because at low excretion rate, pulses in water concentration tend to be retained by the fish over a longer period of time than at high excretion rates. Similarly, as a fish increases in weight, concentration variability will tend to shift into the low frequency end of the variance spectrum.

Figure 10.11 shows the behavior of r from Equation 10.46 and indicates that correlated water concentrations increases the variability of the fish concentration relative to water. This is a consequence of correlated inputs introducing more "low frequency" variations in water concentration which are not dampened by the fish. Thus additional variance propagates through the fish and is reflected in the increase in the coefficient of variation.

Laboratory data for evaluating this analytical development come from two papers of Oliver and Niimi (1983, 1985). In this work, rainbow trout were exposed to chlorinated and brominated organic chemicals for up to about 100 days. In Oliver and Niimi (1983), the data reported include the individual fish concentrations and associated water concentrations and the accompanying statistics of mean and standard deviation. For Oliver and Niimi (1985) only the BCF are reported. To use the data in this research, the fish concentrations were calculated from the individual BCF values and the mean water concentration. These data are therefore only an approximation to the actual fish concentration.

As noted frequently by the authors, several of the chemicals did not reach equilibrium during the test. The preceding statistical development

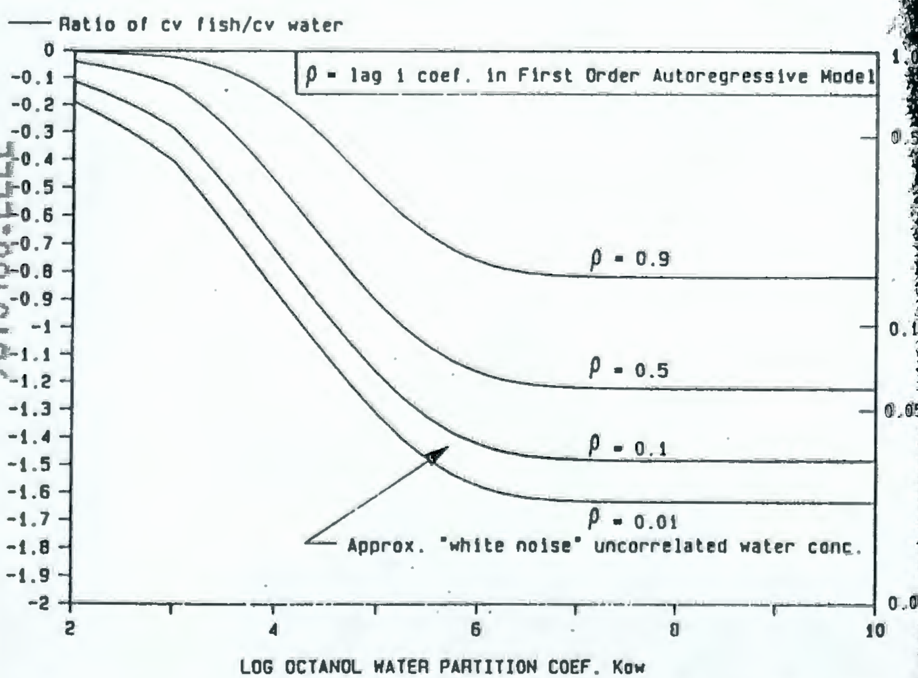

Figure 10.11. The effect of the first order autocorrelation parameter of the water concentration on the relationship between r and K_{ow}. Growth rate variable, wt = 1000 g.

assumes a dynamic equilibrium has been reached. The computation of the variance would be severely biased by the increase in concentration to a dynamic equilibrium. With a few exceptions, those chemicals excluded by the authors have also been excluded herein.

Figure 10.12 (top) shows the results of compiling these data and calculating the r ratio: v_f/v_c. As a function of log K_{ow}, it is seen that there is a general downward trend to the data from initial values of >1 at low log K_{ow}. The four circled points at the high log K_{ow} value of >6 are exceptions. The chemicals are pentabromotoluene, pentabromoethylbenzene, hexabromobenzene, and octachloronapthalene. These points were also noted by Oliver and Niimi (1985) in their evaluation of the BCF data. No ready explanation is available for why these chemicals exhibit less relative variability than expected, although there are no data in this log K_{ow} range where r values are low. Indeed, the data indicate a trend towards decreasing r values to a minimum at about log K_{ow} of about 5.5 and then an increase to higher r values with increasing log K_{ow}. This is not consistent with the previously developed theory as shown in Figure 10.12 (bottom).

This figure shows the calculation of the theoretical r ratio under three different assumptions. For these calculations the growth rate of the rainbow trout was calculated from the reported data at an average level of about 0.01/d. Also, the excretion rate is given in Equations 10.22 and 10.23 and the uptake rate as a function of the efficiency of transfer E. Thus,

$$K = \frac{10^3 w^{-0.2} E/p}{K_{ow}} \tag{10.47}$$

where p is the fraction lipid and E is given by Equation 10.25.

The first calculation assumed the water concentration was statistically "white noise," i.e., uncorrelated in time. As noted, this calculation underestimates r by a significant amount.

The second calculation using Equation 10.46 assumed a correlated first order process (a = 0.9) for the water concentration. Now, the lower bound of the r ratios is captured although at log K_{ow} >6 the model deviates from the data.

The third calculation assumes some metabolism at a high rate of $1.0/d$. This is essentially an additive factor to the excretion rate and as seen, the calculation now approximates the data somewhat better. As noted however, this calculation is simply an hypothesized mechanism for increasing the loss rate of the chemical. In the theoretical development, the only parameter that influences the shape of the r function is the excretion (plus metabolism) rate. To capture the four circled points with the theory would require an increase in the effective excretion plus metabolism rates to levels of >1.0/day.

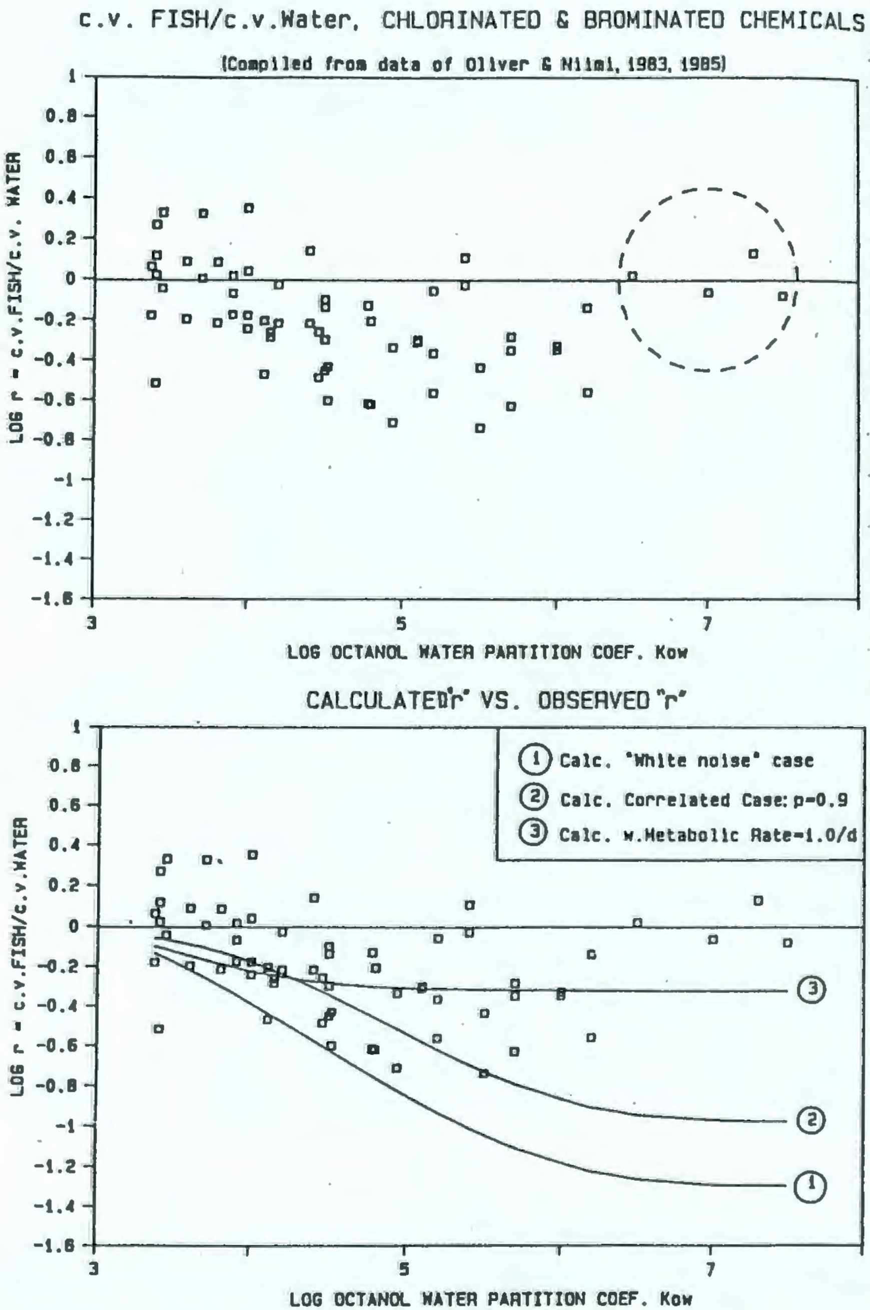

Figure 10.12. (*Top*) Compiled laboratory r values from experiments of Oliver and Niimi (1983, 1984); (*bottom*) comparison of calculated and observed r values under assumptions on water correlation structure and metabolic rate.

In summary, the laboratory data of Oliver and Niimi show a general downward trend of the r ratio with log K_{ow}, but with several notable exceptions at high log K_{ow}. The theory approximately duplicates the observed data only with a high level of first order correlated water input to about log K_{ow} of about 5.5–6.0. At higher log K_{ow}, increased metabolism and/or excretion would be necessary to reproduce the observed data.

10.5 Conclusions

The physio-chemical and food chain model structures discussed herein provide a basis for understanding and predicting the fate and transport of chemicals in surface water systems. While the fully time variable equations appear formidable, steady-state simplifications permit rapid assessment of chemical fate, both in the water column and in the food chain. Such initial screening models have the same structure as "classical" water quality models; notably a linear response to external inputs of chemicals.

The interaction of the sediment and water column plays a significant role and again under some reasonable assumptions, rapid estimates can be made of chemical concentrations in the sediment.

In a similar setting, the age-dependent food chain models can be simplified at steady state and rapid screening assessments of chemical bioconcentration can be made. For organic chemicals, the octanol water partition coefficient is a useful ordering parameter. Long-term time variable deterministic calculations for the Great Lakes indicate the significance of the sediment as a reservoir of the chemical. Response times to changes in external load are long (e.g., years) and are significantly increased when sediment partition coefficients are assumed to be different from water column partition coefficients.

Stochastic variability in chemical concentration is high. Analytical models of the resulting variability of chemical concentration in fish indicates the importance of the excretion rate of the chemical. However, additional research is necessary to more fully describe the observed variance in laboratory experiments, especially for chemicals with log octanol water partition coefficients greater than about 6.5.

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Data Selection

Terri Miley

February 20, 1996

Processing Data

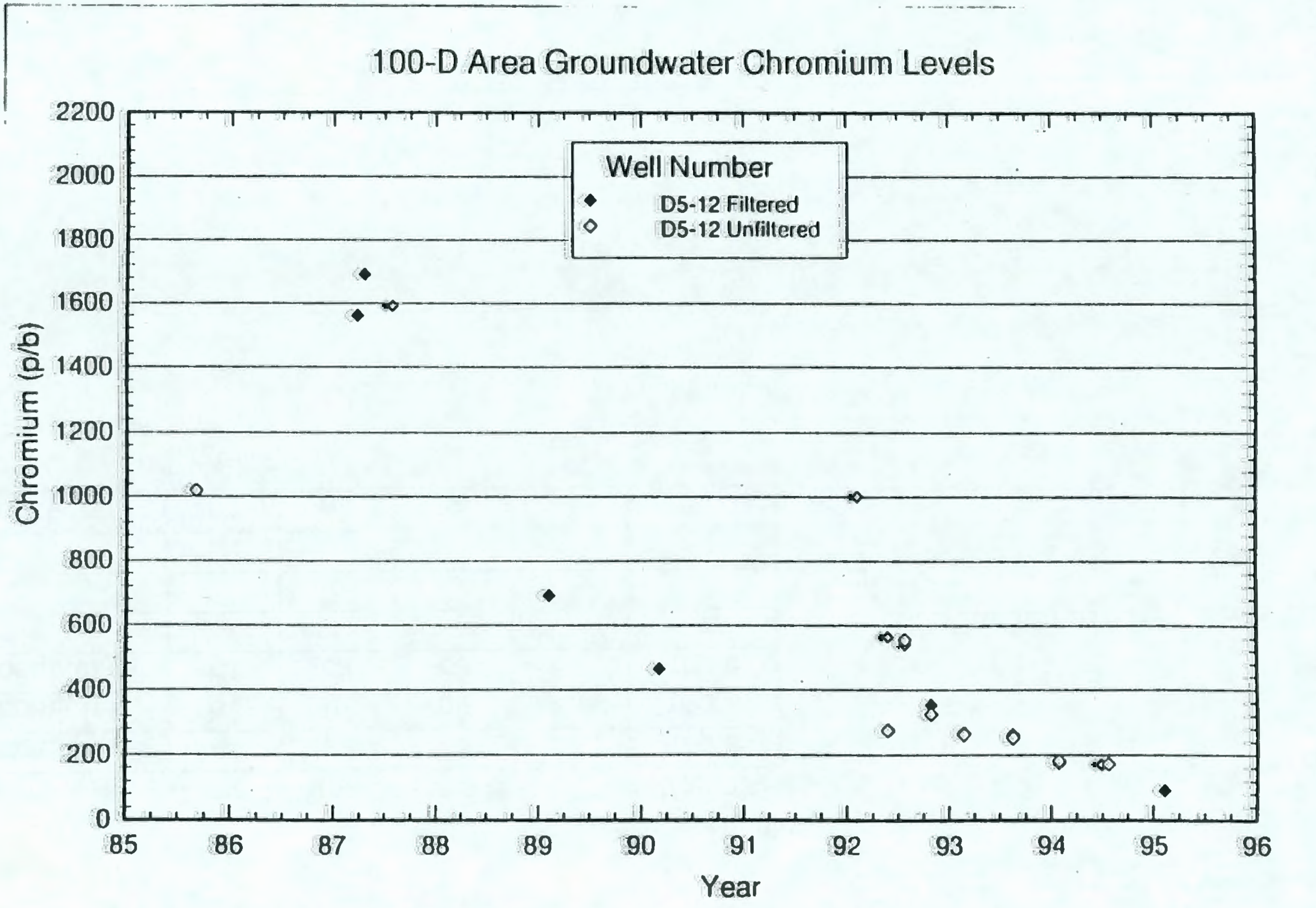
- If Sufficient Data Points
  - By sampling location, identify representative concentration over time
  - Apply data distribution to representative values
- If Insufficient Data Points
  - Apply a data distribution to observed values

Groundwater at a single segment for a single contaminant

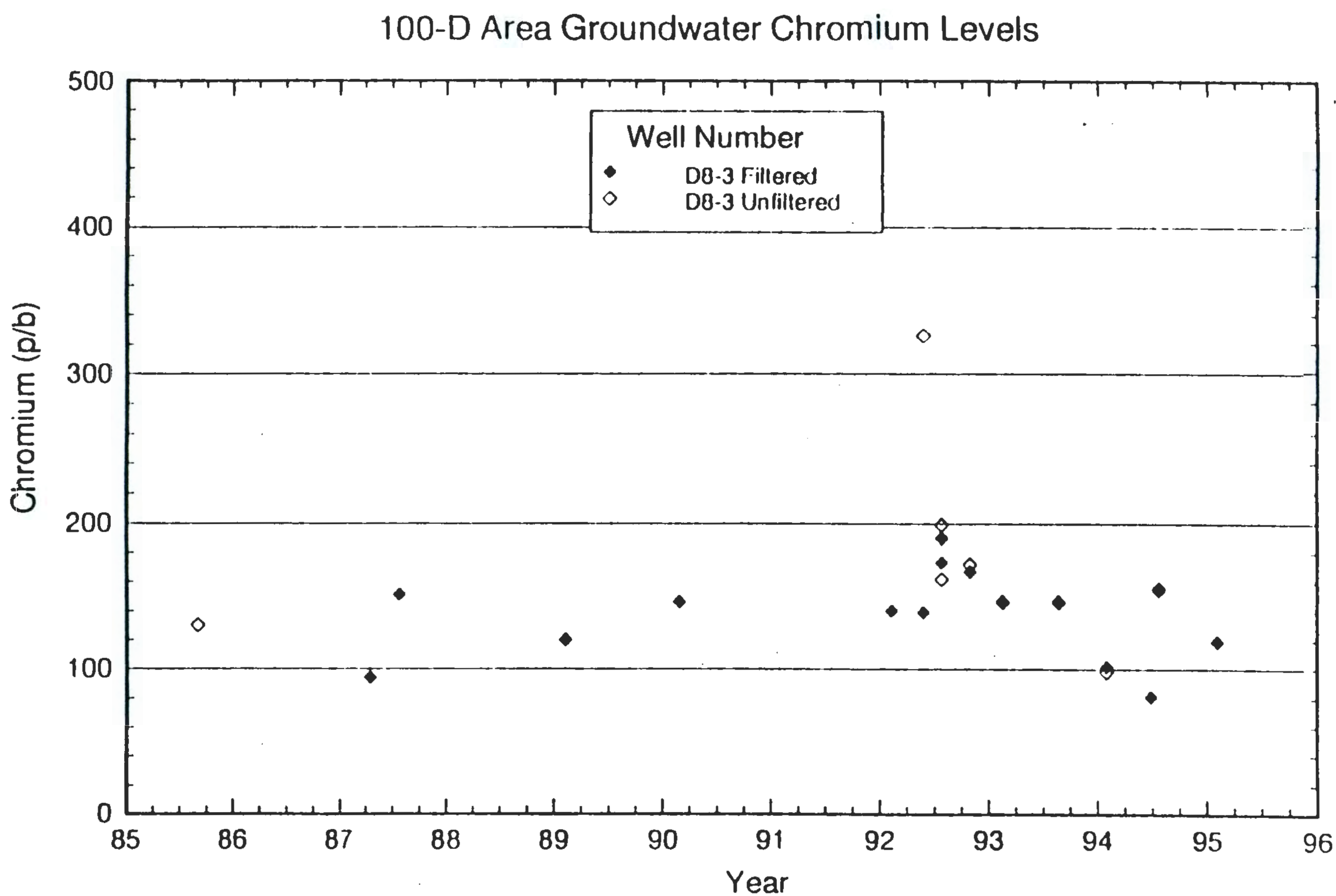
	Time 1	Time 2	Time 3	...	Representative Values
Location a	a1	a2	a3	...	rep a
Location b	b1	b2	b3	...	rep b
Location c	c1	c2	c3	...	rep c
.					.
.					.
.					.
Location n	n1	n2	n3	...	rep n

Max(rep a, rep b, ..., rep n)

Distribution



100-D Area Groundwater Chromium Levels
Well Number
D5-12 Filtered
D5-12 Unfiltered
Chromium (p/b)
2200
2000
1800
1600
1400
1200
1000
800
600
400
200
0
85
86
87
88
89
90
91
92
93
94
95
96
Year



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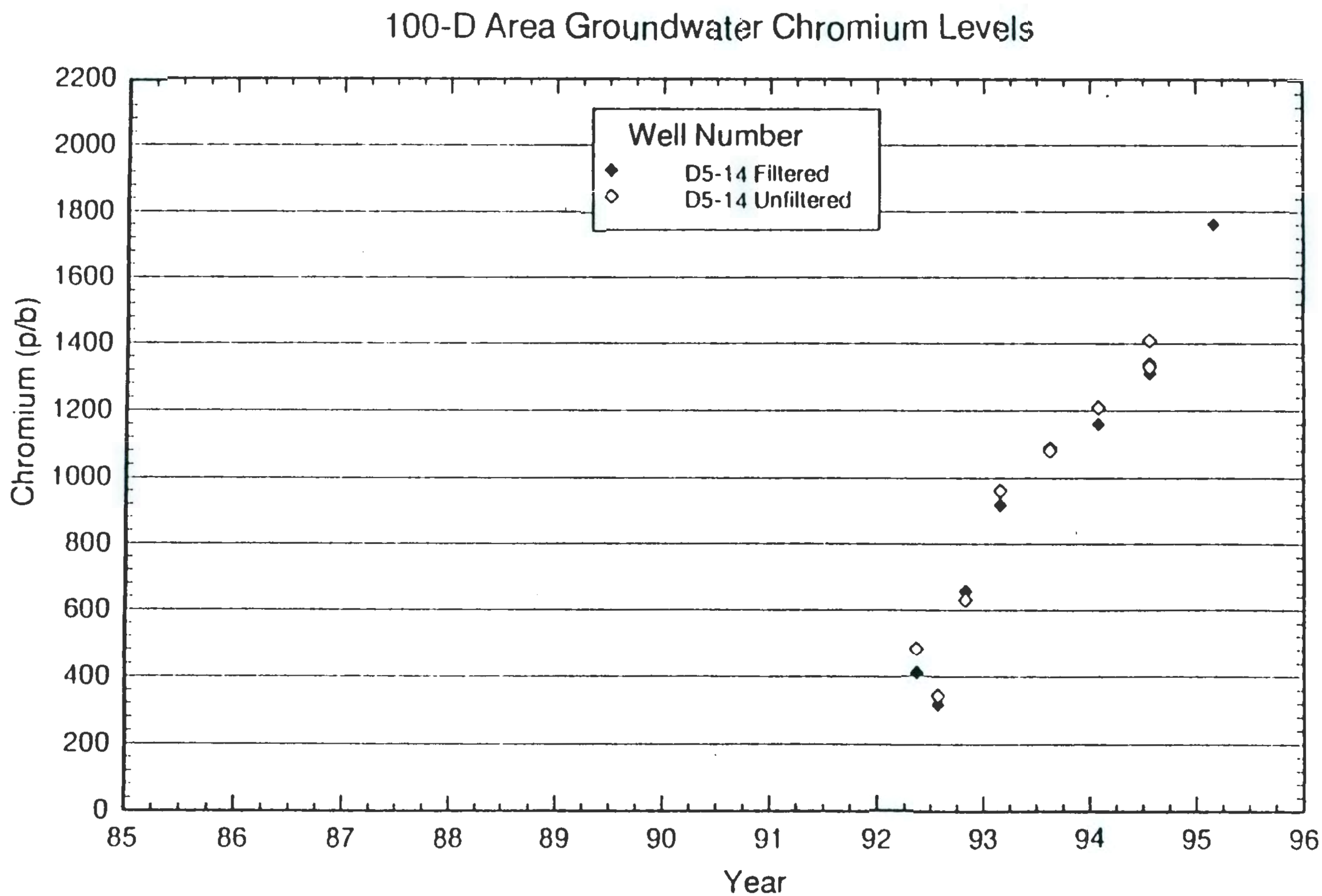
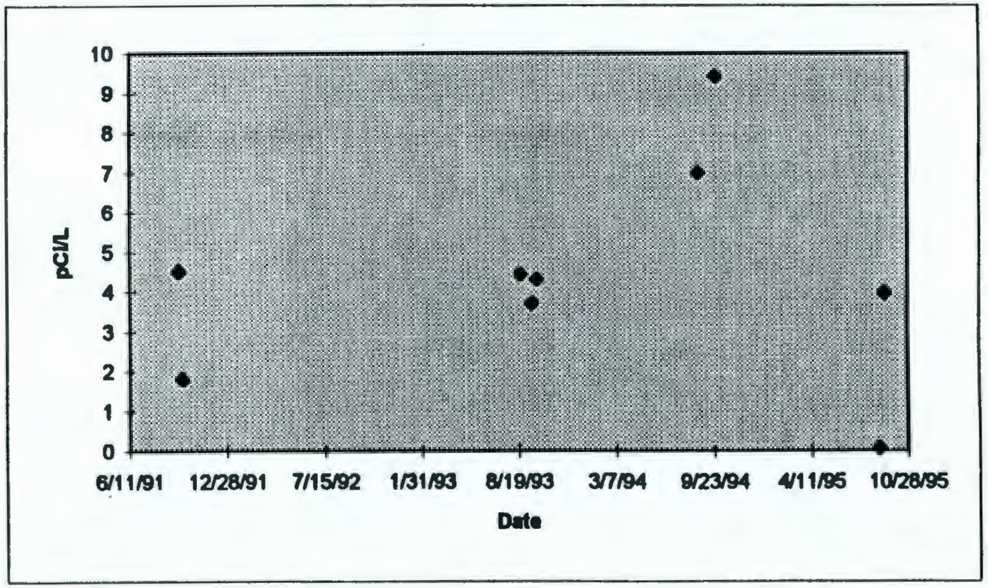


100-D Area Groundwater Chromium Levels
Well Number
D5-14 Filtered
D5-14 Unfiltered
Chromium (p/b)
2200
2000
1800
1600
1400
1200
1000
800
600
400
200
0
85
86
87
88
89
90
91
92
93
94
95
96
Year

Strontium-90 at D Spring

SW	B06KX3	SEEP 110-1	9/19/91	10098-97-2	SR-90	4.5		pCi/L
SW	B06KX6	SEEP 110-2	9/26/91	10098-97-2	SR-90	1.8		pCi/L
SW	E124514	100-D SPRING	8/23/93	10098-97-2	SR-90	4.44		pCi/L
SW	B091M3	SEEP 110-1	9/15/93	10098-97-2	SR-90	3.7		pCi/L
SW	E124666	100-D SPRING	9/26/93	10098-97-2	SR-90	4.31		pCi/L
SW	B0C5F5	100-D SPRING	8/22/94	10098-97-2	SR-90	7		pCi/L
SW	B0C5F6	100-D SPRING	9/26/94	10098-97-2	SR-90	9.41		pCi/L
SW	B0G8B7	100-D SPRING	8/29/95	10098-97-2	SR-90	0.0694	U	pCi/L
SW	B0GKD5	100-D SPRING	9/11/95	10098-97-2	SR-90	3.96		pCi/L



Chromium in Seeps at 100D

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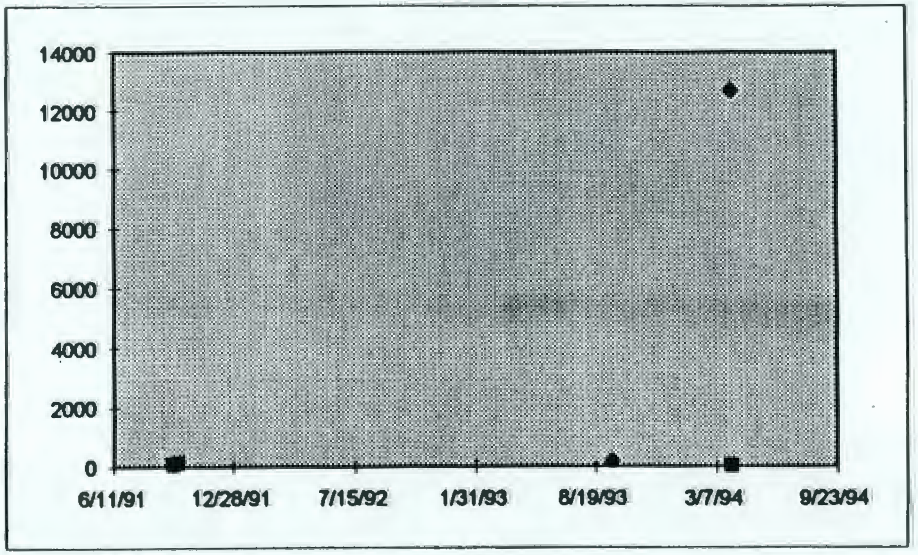
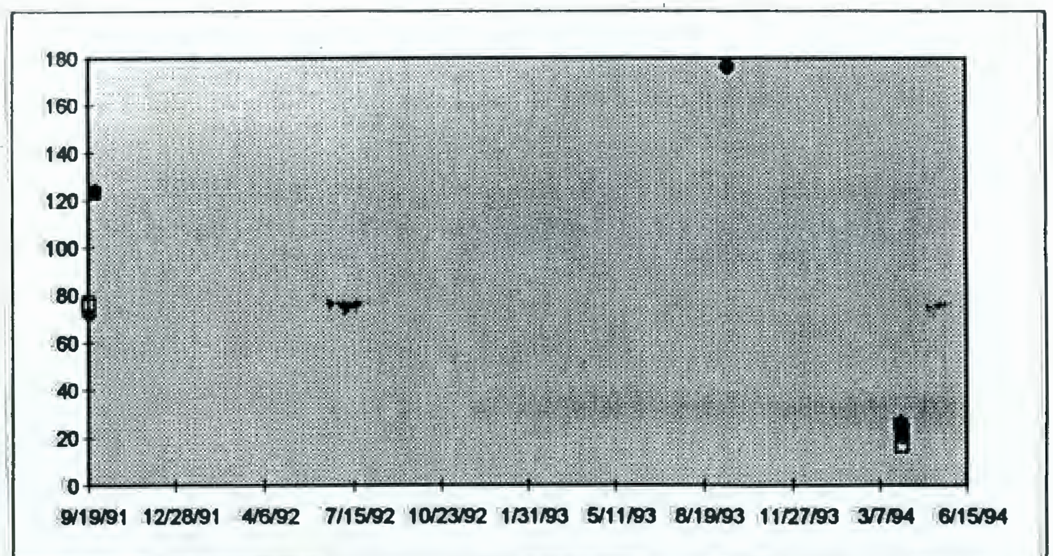
SEEP 110-1	B091M4	9/15/93	N	Chromium	7440-47-3	177	ug/L	
SEEP 110-1	B091M3	9/15/93	N	Chromium	7440-47-3	176	ug/L	
SEEP 110-1	B0BMJ6	3/31/94	N	Chromium	7440-47-3	26.4	ug/L	
SEEP 110-1	B0BMJ9	4/1/94	N	Chromium	7440-47-3	24.6	ug/L	
SEEP 110-1	B0BMK2	4/1/94	N	Chromium	7440-47-3	12700	ug/L	
SEEP 110-1	B0BMK5	4/1/94	N	Chromium	7440-47-3	19.8	ug/L	
SEEP 110-2	B06KX6	9/26/91	N	Chromium	7440-47-3	124	ug/L	
SEEP 110-1	B06KX4	9/19/91	Y	Chromium	7440-47-3	75.8	ug/L	
SEEP 110-1	B0BMJ7	3/31/94	Y	Chromium	7440-47-3	24.8	ug/L	
SEEP 110-1	B0BMK0	4/1/94	Y	Chromium	7440-47-3	16	ug/L	
SEEP 110-1	B0BMK3	4/1/94	Y	Chromium	7440-47-3	21.7	ug/L	
SEEP 110-1	B0BMK6	4/1/94	Y	Chromium	7440-47-3	20.8	ug/L	
SEEP 110-2	B06KX7	9/26/91	Y	Chromium	7440-47-3	123	ug/L	



Chromium in Seeps at 100D

SEEP 110-1	B06KX3	9/19/91	N	Chromium	7440-47-3	71.7	ug/L	
SEEP 110-1	B091M4	9/15/93	N	Chromium	7440-47-3	177	ug/L	
SEEP 110-1	B091M3	9/15/93	N	Chromium	7440-47-3	176	ug/L	
SEEP 110-1	B0BMJ6	3/31/94	N	Chromium	7440-47-3	26.4	ug/L	
SEEP 110-1	B0BMJ9	4/1/94	N	Chromium	7440-47-3	24.6	ug/L	
SEEP 110-1	B0BMK5	4/1/94	N	Chromium	7440-47-3	19.8	ug/L	
SEEP 110-2	B06KX6	9/26/91	N	Chromium	7440-47-3	124	ug/L	
SEEP 110-1	B06KX4	9/19/91	Y	Chromium	7440-47-3	75.8	ug/L	
SEEP 110-1	B0BMJ7	3/31/94	Y	Chromium	7440-47-3	24.8	ug/L	
SEEP 110-1	B0BMK0	4/1/94	Y	Chromium	7440-47-3	16	ug/L	
SEEP 110-1	B0BMK3	4/1/94	Y	Chromium	7440-47-3	21.7	ug/L	
SEEP 110-1	B0BMK6	4/1/94	Y	Chromium	7440-47-3	20.8	ug/L	
SEEP 110-2	B06KX7	9/26/91	Y	Chromium	7440-47-3	123	ug/L	



Data Decisions Needed Today

- Choice of Representative Sample
- Concentration Distributions

DOE/RL-96-16-X
Revision 0
UC-630
Draft

[TITLE GOES HERE]

Columbia River Comprehensive Impact Assessment

[Authors]

[Month] 1996

Prepared by the Pacific Northwest National Laboratory
In Consultation with

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Table P.1. Documents in Initial Phase of Columbia River Comprehensive Impact Assessment

Title	Document No.	Publication Date	Status
Data Compendium for the Columbia River Comprehensive Impact Assessment (Eslinger et al. 1994)	PNL-9785	April 1994	Final publication
List of Currently Classified Documents Relative to Hanford Operations and of Potential Use in the Columbia River Comprehensive Impact Assessment January 1, 1973 - June 20, 1994 (Miley and Huesties 1995)	PNL-10459	February 1995	Final publication
Identification of Contaminants of Concern (Napier et al. 1995)	PNL-10400	January 1995	Draft - Issued first in January 1995 for review, then again in January 1996; comments from both review periods will be addressed and report will be a section in the final scoping level risk assessment report
[Scenarios]	DOE/RL-96-16-a Rev.0	March 20, 1996 (draft)	To be published as a draft - Then comments will be addressed and report will be a section in the final scoping level risk assessment report
[Ecological Receptors - formerly Species of Concern]	DOE/RL-96-16-b Rev. 0	March 26, 1996 (draft)	To be published as a draft - Then comments will be addressed and report will be a section in the final scoping level risk assessment report
[Data}	DOE/RL-96-16-c Rev.0	April 2, 1996 (draft)	To be published as a draft - Then comments will be addressed and report will be a section in the final scoping level risk assessment report
[Scoping Level Risk Assessment]	DOE/RL-96-16 Rev.0	August 1, 1996 (draft)	Will incorporate all previous draft publications (not those published as final) plus sections on site characterization, scoping level risk assessment, and CRCIA Team statement of work to be done after the initial phase
[Scoping Level Risk Assessment]	DOE/RL-96-16 Rev.1	October 1996 (final)	Will incorporate responses to comments and minority opinions should any comments not be reconciled

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DRAFT

Outline for Scoping Level Risk Assessment Report
Columbia River Comprehensive Impact Assessment
February 20, 1996

Title Page (p. I): Scoping Level Risk Assessment of Impact to Columbia River (Priest Rapids Dam to McNary Dam)[?]

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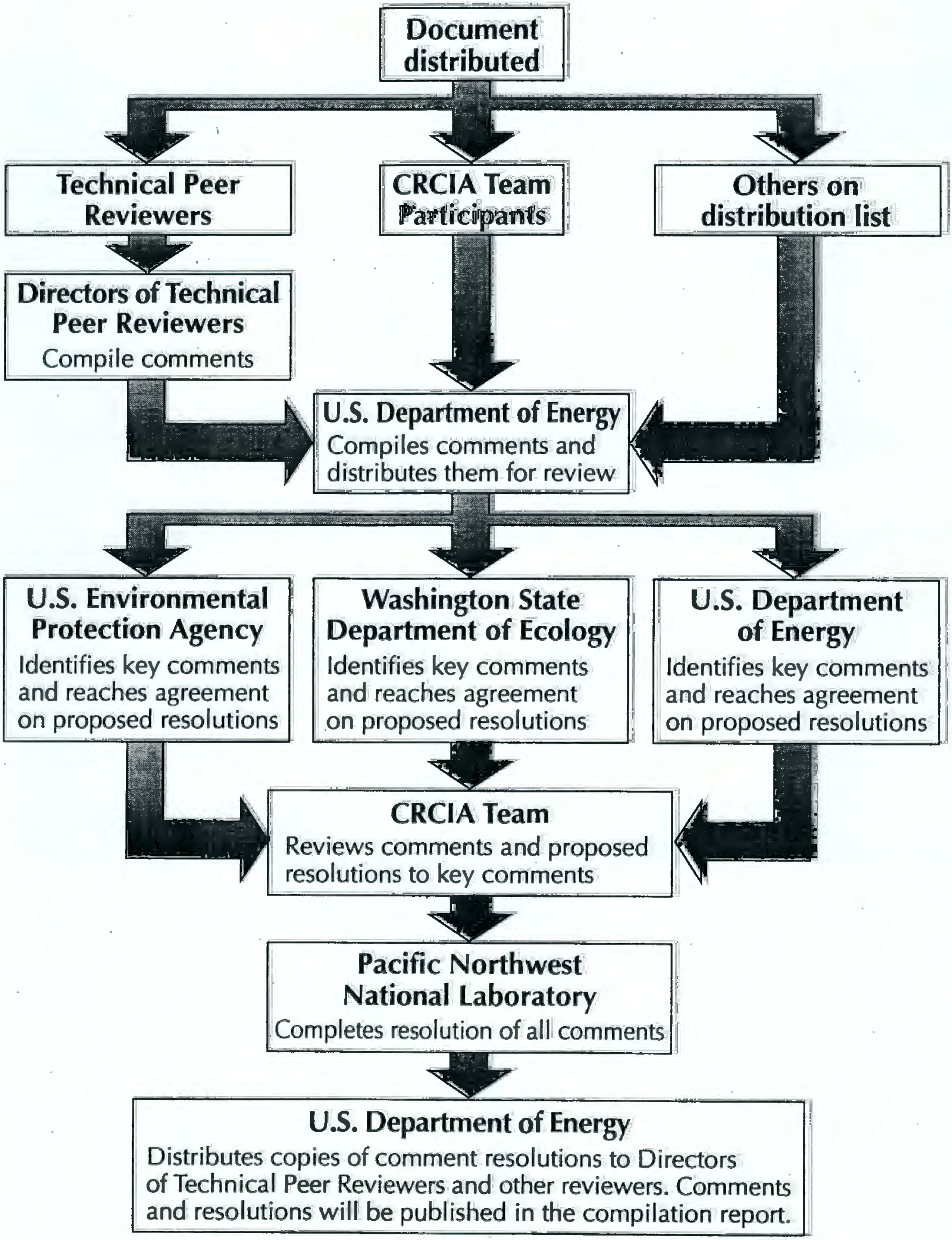

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