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ANALYSIS OF ENVIRONMENTAL ISSUES RELATED TO SMALL-SCALE HYDROELECTRIC DEVELOPMENT I. DREDGING

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ENVIRONMENTAL SCIENCES DIVISION
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ABSTRACT

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The small hydroelectric potential ($\lesssim$15-MW capacity) at existing dams in the United States has been estimated to be approximately 5000 MW. Development of this resource by retrofitting these dams for hydroelectric generation may require dredging in order to (1) reclaim reservoir storage capacity lost as a result of sediment accumulation; (2) clear intake structures; and/or (3) construct/repair powerhouses, tailraces, and headraces. Dredging and disposal of dredged material at small-scale hydro sites may result in several potential environmental impacts, and their magnitude will depend upon many site-specific factors. The physical and chemical effects of dredging and disposal, their causes, and the biological effects engendered by these physical and chemical changes are discussed. Factors that could affect the severity (magnitude) of these effects (impacts) are emphasized, with the intent of providing guidance to developers of potential sites rather than simply preparing an exhaustive review of the literature on the subject. Consequently, a discussion of environmental contraints and mitigation, as well as guidelines for the early evaluation of the environmental feasibility of dredging, are included.

In addition to the review and evaluation of environmental effects, the report includes a general introduction on dredging equipment and disposal practices, with emphasis on those practices that would be

applicable to small reservoirs. Applicable regulations related to dredged material disposal and wetlands protection are also discussed, and a preliminary analysis of the economic costs associated with dredging and disposal is presented.

Adequate mitigation capability exists for most of the environmental impacts of dredging, but the cost of this mitigation may place significant economic constraints on project development. How the sediments are dredged and disposed of will greatly affect both the nature and magnitude of potential environmental impacts. At the majority of the small hydro sites, hydraulic cutterhead dredges and confined upland disposal will be employed. Most difficult to mitigate is the impact on threatened or endangered species at the site, especially endemic mussel populations downstream of the dam. Establishing a dialogue between the developers and appropriate personnel at the local, state, and federal levels in early stages of project planning and development may be the most effective method for designing a dredging operation and assessing the magnitude of potential environmental constraints.

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PREFACE

The Small-Scale Hydroelectric Power Development Program of the United States Department of Energy (DOE) was organized to promote the use of small hydropower, a readily available renewable energy resource. By providing financial and technical assistance to potential developers in both the private and public sectors, DOE hopes to encourage and accelerate the redevelopment of existing dams for hydroelectric generation with a potential capacity of $\leq$15 MW (U.S. Department of Energy 1979a). The amount of developable small hydroelectric potential that actually exists (after considering both environmental and economic constraints) is not completely known. Preliminary assessments suggest that its development should be encouraged because this source of energy could supply a meaningful portion of regional energy demands (O'Brien et al. 1979). Between 1940 and 1976, more than 575,000 kW of hydropower capacity have been retired (U.S. Army Corps of Engineers 1977). The small hydroelectric potential at existing dams in the United States amenable to redevelopment has been estimated to be approximately 5000 MW (U.S. Army Corps of Engineers 1979a). The development of this resource, however, may involve several potential environmental impacts (U.S. Department of Energy 1979a). The extent to which one of these impacts, that associated with dredging and dredged material disposal, is indeed a constraint and can, or should be, minimized/mitigated is assessed in this report.

This evaluation is limited to a consideration of the dredging that may be required to (1) increase the storage capacity of reservoirs;

(2) clear intakes/penstocks; and (3) repair/construct powerhouses, headraces, and/or tailraces. Dredging required for new dam construction or the extensive rehabilitation or repair of existing dams is not specifically considered. However, the analyses of environmental effects discussed in this report may prove useful in evaluating the impacts associated with these activities. Furthermore, any indirect effects of dredging such as water-level fluctuations in the reservoir or alterations in downstream flows are not discussed, since these issues will be the subject of future reports in this series. Finally, this document is not intended to be an exhaustive treatise on the environmental effects of dredging and dredged material disposal. [Numerous literature reviews on this subject already exist (see, for example, Darnell 1976; Hirsch et al. 1978; Morton 1977; Windom 1975).] Rather, the major environmental issues of dredging and dredged material disposal are discussed (with appropriate references), and the reader is directed to other sources should additional information on a particular issue be needed.

The report is divided into three major sections. Following a brief introduction (Section 1), the types of dredging and disposal that might take place at a small hydro site, should either an increase in reservoir storage or clearance of intakes be necessary, are described in Section 2. In the discussion of the environmental impacts of dredging and dredged material disposal that follows (Section 3), each impact (issue) is addressed by first identifying the reasons for concern (why it is an issue) and then discussing those factors that affect its significance. The final section (Section 4) focuses on

(1) the extent to which dredging for the purpose of increasing reservoir storage or intake clearance places a constraint on the development of our hydropower resources and (2) guidelines for evaluating and, in some cases, minimizing the impact. In addition to environmental constraints, consideration is also given to the economic aspects (costs) of dredging at these sites. Such an evaluation of the potential impacts of dredging and dredged material disposal at small hydro sites represents an attempt to provide some guidance to developers who may need to assess the magnitude of these potential impacts during the very early stages of project development (i.e., during the feasibility study).

1. INTRODUCTION

1.1 OPERATION OF SMALL ($\lesssim$15-MW) HYDROELECTRIC FACILITIES

Hydroelectric generation includes two general types of facilities. Base load plants operate at or near full capacity 24 h/d to supply power for meeting the base load demand, whereas peaking plants may generate for periods less than 24 h/d, depending upon the quantity of water available and the demand. Small hydroelectric facilities may either be operated as run-of-river facilities to meet base load demands or, if storage (pondage) is available, as store and release plants to meet daily peaking demands (U.S. Department of Energy 1979b). Approximately 75% of the 49 potential small hydroelectric projects evaluated in the DOE cost-shared feasibility studies would operate facilities in a run-of-river mode (U.S. Department of Energy 1979c).

Small hydro projects with very small or no storage must depend upon rather uniform inflows for optimal power generation. Natural flows in streams and rivers, however, may exhibit considerable seasonal variability, especially in the far Northwest. As a result, questions regarding the amount of energy that can be produced over a given period of time are critical. The lack of available pondage coupled with wide variations in stream flows, especially the occurrence of low-flow periods when no generation might be possible, could result in a near-zero dependent energy capacity. An increase in the storage capacity of the reservoir could result in a concomitant increase in the amount of firm energy (or dependable capacity) that would be available. By having the capacity for daily storage at a site, a greater range of river flows could be utilized. Thus, increased utilization of the hydropower resource could be accomplished by dredging the reservoir to increase pondage.

1.2 SEDIMENT ACCUMULATION IN RESERVOIRS

Streams and rivers can transport large volumes of sediment, especially during storm events when soil erosion is accelerated and the sediment-carrying capacity of rivers is greatly increased. By impounding a river, a gradient of decreasing velocity is established from the upstream reaches to the deeper regions of the reservoir near the dam. Thus, sediments, especially silt and larger particulates, accumulate in the reservoir and gradually reduce the storage capacity. This problem is particularly apparent in those areas, such as the Midwest, where agriculture is the dominant form of land use in the watershed. Although the implementation of soil conservation programs has reduced sedimentation by 43 to 92%, the small water supply reservoirs in Illinois are nevertheless losing their storage capacities at an average rate of 0.6% per year (Roberts 1976). It is likely that many potential small hydro sites in the East and Midwest are located below impoundments that have lost considerable storage due to siltation. For example, the average age of the dams reported for those projects included in the feasibility assessments performed under DOE's Program Research and Development Announcement (PRDA) ET-78-D-07-1706 was 58 years (U.S. Department of Energy 1979c). Not surprisingly, the oldest dams were located in the Northeast (average age = 70 years; range 24-134 years).

2. DREDGING EQUIPMENT AND DREDGED MATERIAL DISPOSAL PRACTICES

2.1 TYPES OF DREDGES

Dredges currently in use can be classified into two general categories - mechanical and hydraulic, each of which, in turn, consists of several different types (Fig. 1). Of the three major classes of mechanical dredges, the bucket dredge, particularly the clamshell, is the type most commonly used. The bucket dredge utilizes an open jaw-like bucket in contrast to the dipper dredge which is a heavy-duty excavator very similar to an ordinary power shovel (U.S. Comptroller General 1972). Although the ladder dredge is used extensively throughout the world, none are used in the United States (except as part of the mining plant) (Mohr 1976).

Of the two major methods of dredging, hydraulic dredging is the most common practice, having been used to excavate and transport approximately 96% of the bottom material dredged each year (Lee 1976a). Most hydraulic dredging, in turn, is performed with a pipeline dredge, of which the cutterhead dredge is the most widely used type in the United States and is the basic tool of the private dredging industry. Pipeline dredges either suck the material directly off the bottom or employ pressurized streams of water or rotating blades (cutters) to loosen the material before pumping it through the discharge line to the disposal area. Hopper dredges, on the other hand, are generally large, self-propelled vessels that employ drag arms to pump material into hoppers or bins on the ship for transport to the disposal area (Gren 1976). The third class of hydraulic dredges

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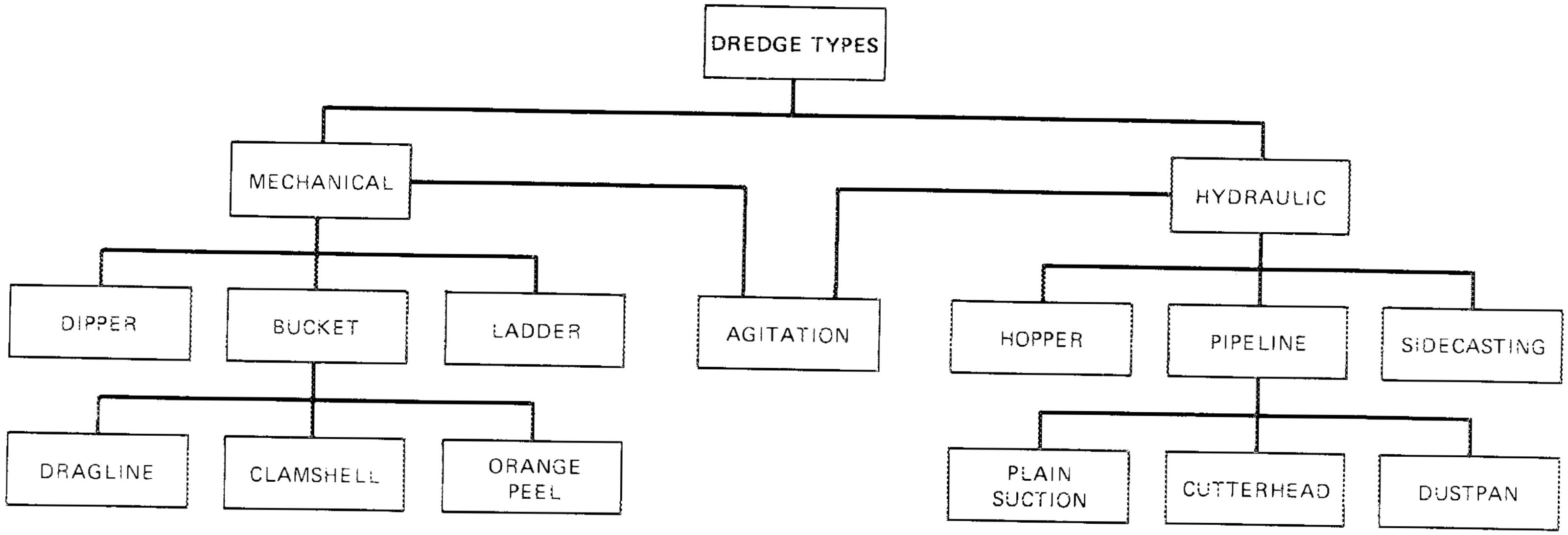

Fig. 1. Classification of the major types of dredges in use today.

includes the sidecasters. These are similar to hopper dredges except that the dredged material is deposited (or cast) a short distance from the dredging site.

Agitation dredging, one of the oldest forms of dredging, simply involves agitating or disturbing the bottom material and allowing the currents to carry it away. Because of the turbidity and uncontrolled settling of the dredged material associated with agitation dredging, the use of this method has declined substantially over the years (Mohr 1976).

Because not all of the bucket dredges or pipeline dredges shown in Fig. 1 are likely to be used on small rivers or reservoirs, more detailed information on these dredges has not been presented. Those dredges that have been used previously on small lakes and impoundments or that might be used at small hydro sites are discussed in Section 2.4.

2.2 DREDGED MATERIAL DISPOSAL PRACTICES

Three general methods of disposal are available. Open-water or subaqueous disposal (Gambrell et al. 1978) could include disposal areas in either very deep water or relatively shallow regions of a water body. Disposal of dredged material in shallow water (e.g., coves, sloughs) or wetlands is referred to as intertidal disposal by Gambrell et al. (1978). In many cases, the disposal area is diked and the effluent, following a period of settling, is returned to the water body. The third method of disposal, referred to here as upland disposal, occurs on land that may be some distance from the dredging site. The necessity of confining upland disposal areas is often determined by the nature of the dredged material (e.g., type of material, degree of contamination).

Prior to about 1970, bottom materials that were dredged to maintain navigation channels were disposed of in the most economical manner, either in nearby waterways or on land (Lee 1976b). In small reservoirs, however, open-water disposal using either mechanical or hydraulic dredges would not be employed. The disposal of dredged sediments in the open water of small reservoirs would likely alter the distribution of storage capacity in the impoundment and, therefore, could have an adverse economic effect on power generation. Moreover, the deepest region of these reservoirs is near the dam, and unless a region deeper than the bottom elevation of the intake canal or penstock existed, no gain in storage capacity would result from open-water or intertidal disposal. In all likelihood, the sediments would have to be removed from the reservoir if the purpose of dredging is to increase storage capacity. Because open-water disposal does not appear to be a viable option for the disposal of dredged material, the biological effects associated with this type of disposal have not been addressed in this report. Information on the biological effects of open-water disposal can be found in Wright (1978) and Hirsch et al. (1978).

The confined or unconfined disposal of dredged material in a small cove of the reservoir would also be unlikely because (1) the increase in storage capacity from the dredging and removal of accumulated sediments in the reservoir would be offset by the loss of capacity from filling of the cove, and (2) the impact on aquatic resources could be significant. Coves or embayments are usually shallow, and these inshore areas may be the most productive regions of the reservoir. If the cove is a wetland area, the magnitude of the impacts could be even greater.

The only remaining method of disposal, confined or unconfined upland disposal, is the one most likely to be employed at small hydro sites, especially if the purpose of dredging is to reclaim lost storage capacity in the impoundment. Unconfined disposal should be considered only if the sediments are uncontaminated. However, because hydraulically dredged material is transported to the disposal area as a slurry, and hence contains considerable quantities of water, the environmental impacts caused by erosion and runoff from an unconfined disposal area could be significant. Although appropriate mitigation could minimize these impacts (see Section 4.1), the number of projects where unconfined disposal would be employed is expected to be minimal. From an environmental standpoint, the best alternative for the disposal of dredged material is likely to be confined upland disposal.

Because the practice of confined upland disposal of dredged material has not been widely practiced until recently, literature on the environmental effects is limited. Upland disposal may lead potentially to problems of surface and groundwater contamination (see review and summary by Chen et al. 1978). In addition, improperly managed land disposal can be associated with other adverse effects, such as odor emission, mosquito breeding, and proliferation of undesirable wildlife. On the other hand, many benefits may accrue from properly planned and managed disposal operations, including the production of fertile land, filling of strip mines, and creation of beneficial wetlands and more diversified habitats for wildlife.

2.3 COMPARISON OF HYDRAULIC VS MECHANICAL DREDGES

The type of dredge selected will directly affect both environmental and economic aspects of the project. Consequently, the basic differences between hydraulic and mechanical dredges should be recognized and considered during the early stages of project development. Since the bucket and hydraulic pipeline dredges are the most commonly used mechanical and hydraulic dredges, respectively, the discussion that follows will focus on these. Other mechanical (e.g., dipper) and hydraulic (e.g., hopper, sidecasting) dredges are not considered because it is unlikely that these large pieces of equipment would be used on small impoundments.

For the comparison that follows, we have assumed that mechanically dredged material would be (1) deposited directly into a truck and transported to the disposal site, or (2) if the dredging site were located some distance fron shore, deposited into a small barge or scow, carried to shore, and then transferred to a truck using another clamshell dredge. If, however, the mechanically dredged material is reslurried from the barge to the disposal site by a rehandling dredge, then the environmental effects from such an operation would be similar to those described for hydraulic dredges. How the dredged material is transported to the disposal area will depend upon (1) the proximity of the disposal area to the dredging site and (2) the physical nature of the substratum (i.e., sediments that are stiff or more compact could best be transported by truck).

The most significant difference between the mechanical and hydraulic methods of dredging is the amount of water associated with the dredged material. With hydraulically operated dredges, dilution water is added to the dredged material to form a slurry. Sediments are typically slurried at a 1:4 or 1:5 sediment-water ratio (Lee 1976b). Mohr (1976) estimates that, as a general rule, hydraulic dredges add between one and three times the amount of diluting water to the bottom material. Mechanical dredges, on the other hand, pick up bottom material near its in-place density, and in the case of coarse material, drain off most of the water (Mohr 1975). As a result, a much more compact sediment is obtained.

Differences in the amount of dilution water required for hydraulic vs mechanical dredging have important environmental and economic implications regarding disposal of the dredged material (Table 1). Because of their greater volume as a result of the addition of dilution water, hydraulically dredged sediments must be placed in larger confined disposal areas than mechanically dredged material. The area required for disposal may represent important terrestrial or aquatic habitat, so the environmental impact could be potentially greater. The additional acreage required would also increase costs associated with disposal. In addition, the higher water content of the hydraulically dredged material means that the volume of the overflow to the nearby waters will be greater than the overflow from the disposal of mechanically dredged material. To the extent that this effluent would be transporting contaminants and very fine particulates (thus increasing turbidity), the environmental impacts to the receiving waters could also be greater. Mechanical dredges, however, collect and maintain bottom materials near their in-place density. As a result,

Table 1. Comparison of the economic and environmental parameters associated with hydraulic vs mechanical dredging and disposal operations (+=greater/higher)

Parameter	Hydraulic	Mechanical[a]
Economic		
Production rate (m^3/h)	+	
Cost		
Disposal area[b]	+	
Transportation[c]		+
Environmental		
Dredging site		
Suspended solids/turbidity[d]		+
Decrease in dissolved oxygen		+
Release of nutrients, toxic substances		+
Downstream siltation		+
Substrate removal (inhibition of repopulation)[e]	+	
Upland disposal site		
Surface runoff	+	
Suspended solids/turbidity	+	
Release of nutrients, toxic substances	+	
Downstream siltation	+	
Groundwater contamination	+	
Loss of habitat	+	
Uptake by terrestrial biota		+

[a]Assumes dredged material is transferred from the barge(s) to trucks by a second mechanical dredge (i.e., it is not reslurried to the disposal site).

[b]Includes costs for land acquisition and dike construction.

[c]Includes cost of transporting material from dredging site to disposal site.

[d]Assumes proper rate of hydraulic dredge head transfer.

[e]Assumes exposed substrate is suitable for habitation.

the mixture contains less water and initially occupies less space, and consequently, the overflow from confined disposal areas containing mechanically dredged material would be nominal (Mohr 1975). By comparison, Mohr (1976) states that "only in very dilute bottom materials, under ideal conditions, and with special equipment is it possible for hydraulic dredges to handle bottom material at in-place density."

At the dredging site, however, hydraulic dredging using a pipeline cutterhead dredge generally produces less turbidity than does mechanical dredging, although the magnitude of the effect depends not only on the type of dredging but also on the nature of the sediments. Particle size distribution and the cohesiveness of the material being dredged determine both the ease of resuspension and the rate of deposition of bottom sediments. Although the larger, heavier particles, such as sand and clumps of cohesive mud, settle rapidly out of suspension, the fine silts and clays remain suspended for longer periods. These smaller particles may be transported away from the dredging site by local currents or out of an upland disposal basin used to clarify hydraulic dredge slurry.

Finally, it should be noted that the degree to which bottom sediments are disturbed by either mechanical or hydraulic dredges is dependent, to some extent, on how the equipment is operated. Present payment methods encourage striving toward the highest material flow rate. As a result, hydraulic dredges are frequently moved through the sediments at a rate exceeding their pick-up ability, thus frequently disturbing and displacing the lighter bottom materials (Mohr 1976).

Another basic difference between hydraulic and mechanical dredges, and one that has important economic implications, is the operating mode of the two types of dredges. Mechanical dredging usually involves lifting the material from the bottom and placing it into a conveyance, usually a barge, for transport and disposal. As a general rule, mechanical dredges do not transport and dispose of the dredged material (Mohr 1976). In hydraulic dredging, however, all three steps (dredging, transporting, and disposing) are combined into a single operation. Thus, cost comparisons must be made on the basis of the total operation. If increasing emphasis is placed on selecting disposal areas that will result in minimal impact on the environment, then many of these areas could be located at increasingly greater distances from the dredging site. Transport costs could be expected to rise accordingly. Mechanical dredge production, on the other hand, is independent of transport distance because a change in distance is adjusted for by a change in the number and size of the barges. With hydraulic dredges, production decreases with transport distance which is limited by the size of dredge pump and pressure (Mohr 1976). However, with the aid of booster pumps in the discharge line, material can be pumped to sites located great distances from the dredging site (Gren 1976).

2.4 USE OF DREDGES IN SMALL RESERVOIRS

"An astute observer can easily see
The following based on man's history
Since there are two dredging schemes today,
It is rather easy to predict and say
That people will forever ponder and sob
Which dredge to use for a particular job."

A. W. Mohr (1976)

The type of dredging operation selected at a particular small hydro site should be based on both environmental and economic considerations. As discussed previously, the range of possible physical, chemical, and biological effects depends on the type of dredge used and the method of disposal, as well as many site-specific factors such as the nature of the sediments (e.g., particle size, chemical composition), reservoir morphometry (depth, degree of thermal stratification), amount of bottom area that would be disturbed, and volume of sediments to be dredged.

At those small hydro sites where extensive dredging is required, the most probable choice would be the hydraulic cutterhead dredge with upland disposal in a confined area near the lake or impoundment. These dredges have been used extensively to restore nutrient-rich lakes (Pierce 1970), restore or enhance fisheries through habitat alteration (Carline and Brynildson 1977), or reclaim lost storage capacity in water supply reservoirs (Roberts 1976). Hydraulic cutterhead dredging has also been utilized as a method of lake reclamation in the U.S. Environmental Protection Agency's Clean Lakes Program (Peterson 1979). Information on the equipment used to dredge a wide variety of small lakes and impoundments is summarized in Table 2.

According to Pierce (1970), the hydraulic cutterhead dredge is the most practical and economic tool for removing lake sediments from all areas except near the shoreline. The inability of some conventional cutterhead dredges to excavate in shallow waters (< 1 m for 30-cm dredge) can be overcome by using draglines, either operated from shore or from a barge, to move sediments to deeper water where it can be

Table 2. Characteristics of various dredging projects on small lakes and reservoirs. N/A = information not available from reference.

Site	Surface area of lake/reservoir (ha)	Volume dredged (m^3)	Dredge characteristics: Type	Diameter of discharge pipe (cm)	Production rate (m^3/h)	Comments	Reference
Lake Carlinville (Illinois)	N/A	29,640[a]	Hydraulic cutterhead	20	227[b]	Slurry had solids content of 10-15% by weight	Roberts (1976)
Oakland (Illinois)	10	254,954[c]	Mud Cat	15	454[b]	Average cutting speed is 2.4-3.6 m/min; maximum width and depth of cut are 2.4 and 0.4 m, respectively	Roberts (1976)
Collins Lake (New York)	22	100,000	Mud Cat	N/A	N/A	Approximately 40,000 m^3 of sediment have been removed since July 1977	Snow et al. (1979)
Trout Lake (Florida)	42	N/A	Mud Cat	15,20	227[b]	Width of cut = 3 m; Maximum dredging depth = 3.4 m; only 7 ha of bottom area were dredged	Crumpton and Wilbur (1974)
Krause Springs (Wisconsin)	<1	4,610	Hydraulic cutterhead	15	N/A	Maximum and average depths prior to dredging were 1.07 and 0.34 m, respectively	Carline and Brynildson (1977)
Sunshine Springs (Wisconsin)	<1	5,280	Hydraulic cutterhead	15	N/A	Maximum and average depths prior to dredging were 1.22 and 0.46 m, respectively	Carline and Brynildson (1977)
Long Lake (Michigan)	59	841,000	Hydraulic cutterhead	30	1590[b] (210)[d]	Maximum and average depths prior to dredging were 2.1 and 0.7 m, respectively	Spitler (1973)
Petite Lake (Minnesota)	16	382,280	Hydraulic cutterhead	25	N/A	Depth prior to dredging was 0.2-3.6 m; 75% of lake area was dredged	Pierce (1970)
Field Memorial Lake (Wisconsin)	17	71,940	Scrapers, bulldozers	-	-	N/A	Pierce (1970)
North Twin Lake (Iowa)	206	1,523,287	Hydraulic cutterhead	30,35	N/A	Minimum dredging depth of the 30-cm dredge was approximately 1 m	Pierce (1970)
Lake George Lake Sisseton (Minnesota)	N/A	382,280[e]	Hydraulic cutterhead	30	2290[f]	Original depth reported as 1.8-2.0 m	Pierce (1970)

[a] Average volume of sediment discharged for six months in each of 3 years.

[b] Maximum rating of suction pump.

[c] Total for 4 years.

[d] Actual production rate. Value based on estimated volume of material dredged (841,005 m^3) ÷ operating time (4,000 h).

[e] Estimate per year.

[f] Average daily production rate based on the hours operated yearly; estimate is referred to by Pierce (1970) as reasonable.

hydraulically dredged (Pierce 1970). Another alternative that can be employed if the project requires extensive dredging in shallow water is the use of small portable cutterhead dredges that have recently been manufactured by many dredge-building companies (Roberts 1976). Designed for one-man operation, these dredges are similar to those described by Carline and Brynildson (1977) and can be readily installed on very small water bodies (Table 2). A typical one has a 20-cm (8-in.) rotating cutterhead surrounding the intake end of the suction pipe, an average production of 75-150 m^3/h (100-200 yd^3/h), and can discharge sediment up to a distance of 925 m (3000 ft) (Roberts 1976).

Several unconventional dredging systems have also been developed within the past 5 to 10 years. These systems are designed to pump dredged material with a high solids content and/or minimize turbidity (Barnard 1978). Several of these devices are modifications of the hydraulic cutterhead dredge (e.g., Mud Cat, Waterless, Delta, Bucket Wheel), while others use compressed air instead of centrifugal motion to pump slurry through a pipeline (e.g., Pneuma, Oozer). One of the most popular of these unconventional dredges is the Mud Cat (Table 2), a small portable dredge available from the Mud Cat Division of National Car Rental System, Inc. Turbidity is minimized by covering the cutterhead with a retractable mud shield, a device that has been able to confine the turbidity plume to within 6 m (20 ft) of the dredge (Nawrocki 1974, as cited in Barnard 1978). Low levels of suspended solids in the vicinity of the dredging operation were also observed with many of the other unconventional dredges (Barnard 1978). Of particular significance in this regard were the results obtained in a study of the release of polychlorinated biphenyls (PCBs) during dredging operations. Hafferty et al. (1977) found the Pneuma dredge to be effective in removing contaminated sediments and minimizing the resuspension of sedimentary material.

The many unconventional dredging systems available today are discussed in detail by Barnard (1978). For additional information on the types of dredges that have been used or could be used on small lakes and reservoirs, see Peterson (1979), which includes a good summary of unconventional dredges taken from Barnard (1978), or Pierce (1970).

Mechanical dredging might also be employed at some sites. Mechanical methods are used especially in congested harbor areas for very small dredging projects, dredging of oversized debris, and for secondary tasks such as dike building (Lee et al. 1976). At small hydro sites where only a limited area near the dam or penstock must be dredged, a bucket dredge or dragline might be the most practical method. A floating mechanical dredge, such as the clamshell, would be effective if the lake/impoundment bottom has numerous underwater logs, stumps, or boulders (Pierce 1970). Mechanical dredging might especially be necessary if no large disposal area (for hydraulically dredged material) is available in close proximity to the lake or reservoir. The lack of available disposal sites was identified more than 10 years ago as the most prevalent problem in lake dredging projects (Pierce 1970).

Finally, because water levels in many impoundments can be lowered considerably, it is conceivable that dredging might be supplanted in some cases by mechanical earthmoving equipment (Table 2; also Peterson 1979). Wheeled or tracked vehicles could be used, and the sediments deposited in a landfill. Although this could simplify construction by providing for smaller cofferdams, improving access for construction equipment, and avoiding problems associated with the resuspension of dredged sediments, this type of operation would not be feasible in those cases where the reservoirs were used for water supply and recreation (U.S. Department of Energy 1979a).

3. ENVIRONMENTAL EFFECTS OF DREDGING AND DREDGED MATERIAL DISPOSAL

The intent of this section is not to provide the reader with an exhaustive review of the literature on the biological effects of dredging and dredged material disposal. Rather, the objective is to briefly describe these effects and identify the important references, should additional information be required. Potential developers of small hydro projects are encouraged to consult the Dredged Material Research Program (DMRP) publications (250 detailed technical reports and 21 synthesis documents) and other U.S. Army Corps of Engineers (COE) and EPA reports, including the joint EPA/COE technical committee annual reports (e.g., Wilkes and Engler 1977). An index and retrieval system and a final report on the DMRP will also be available in the near future. Also, a critical review of the various research projects included in the DMRP can be found in Lee (1976b), and the results of the DMRP studies are summarized in U.S. Army Corps of Engineers (1979b). Although many of these studies focus on dredging in estuaries and open water disposal, much of the information is relevant to the types of effects encountered at inland freshwater dredging and disposal sites and has consequently been cited. Also relevant is the literature on stream channelization, lake restoration, and construction activities near streams (see especially Darnell 1976).

3.1 DREDGED MATERIAL RESEARCH PROGRAMS

Over the past decade intensive research efforts have been directed at evaluating the environmental effects of dredging and dredged material disposal. Impetus for much of this work was the result of legislation passed during this period, particularly the National Environmental Policy Act (NEPA) of 1969 (P.L. 91-190), the River and Harbors Act of 1970 (P.L. 91-611), the Marine Protection, Research, and Sanctuaries Act of 1972 (P.L. 92-532) and the Federal Water Pollution Control Act Amendments of 1972 (P.L. 92-500). The largest and most significant research program in this area was established as a result of the River and Harbors Act of 1970 which authorized the U.S. Army Corps of Engineers (COE) to initiate a comprehensive nationwide investigation of the characteristics of dredged material and alternative methods of disposal (Morton 1977; U.S. Army Corps of Engineers 1979b). Initiated in March 1973, the Dredged Material Research Program (DMRP) was a 5-year, 32.8 million dollar effort conducted at the COE Waterways Experiment Station at Vicksburg, Mississippi. Generally, the DMRP focused on the development of guidelines that would be utilized by the COE districts to evaluate the environmental aspects of dredging and dredged material disposal (Lee 1976b).

In late 1975, a joint EPA/COE technical committee was established to develop comprehensive manuals for the implementation of all technical phases of Public Laws 92-500 and 92-532 (Wilkes and Engler 1977). An interim guidance manual pursuant to Section 404 of PL 92-500 was issued in 1976 (U.S. Army Corps of Engineers 1976a). This interim manual was required for immediate implementation of the technical portions of Section 404 of PL 92-500. Issuance of an Implementation Manual, a comprehensive guidance manual, will require several years (2 to 3) of additional research and development prior to publication

(Wilkes and Engler 1977). The interagency committee has recommended future research priorities, and these, as well as current research activities, are described in Wilkes and Engler (1977). Thus, the DMRP and the joint EPA/COE research programs have sponsored much of the research that has and still is being conducted in the areas of dredging and dredged material disposal.

3.2 MAJOR ENVIRONMENTAL EFFECTS OF DREDGING AND DREDGED MATERIAL DISPOSAL

There is a sizable and increasing body of technical literature which addresses the environmental (physical, chemical, and biological) effects of dredging and dredged material disposal (c.f. Morton 1977; COE and DMRP reports). Most of the physical and chemical effects outlined below are reasonably well understood, although specific effects at specific sites cannot always be predicted with any certainty. The biological effects, which are engendered by the physical and chemical effects, are thus even less predictable on a site-specific basis. According to Wilkes and Engler (1977), methodologies are currently available to predict general physical changes caused by discharge of dredged and fill material, but relatively fewer methods exist to adequately describe and predict chemical and biological effects. By confining consideration to only certain types of sites, dredging, and disposal methods, the range of possible effects can be considerably reduced. Consequently, the discussion that follows has been restricted to those physical and chemical effects associated with the methods of dredging and disposal likely to be used at small hydro sites (see Section 2.4). A summary of the physical, chemical, and resultant biological effects, including their causes and the factors that influence the severity of the effects, is presented in Table 3.

Biological impacts of dredging result from physical and chemical changes that occur both during and after dredging and dredged material disposal. The discussion of physical, chemical, and biological effects that follows, therefore, is organized on an issue-by-issue basis. Issues are those changes or perturbations (usually physical or chemical) that occur as a direct result of either dredging or disposal. Following a description of the physical or chemical changes is (1) discussion of the reasons for concern, i.e., the biological effects (or impacts) of the perturbation and (2) identification of those factors that affect the significance of these impacts.

3.2.1 Effects Due to Increased Suspended Solids

During all types of dredging operations, bottom sediments are mechanically disturbed and resuspended, creating the most visually obvious physical effects of water discoloration and reduced light penetration. It is this reduction in light penetration that is commonly referred to as turbidity (Darnell 1976). Decreased light availability restricts photosynthesis by algae and submergent macrophytes. Since visibility is lower in turbid waters, some interference with the normal behavioral patterns of fishes and invertebrates may occur. Predation success, for example, may be affected for those organisms that use visual cues in searching for food (Heimstra et al. 1969).

Table 3. Summary of the potential environmental effects of dredging and dredged material disposal, their causes, and the major factors contributing to the severity of the effects

Physical/chemical effect	Causes	Resultant biological effects	Factors affecting severity
1. High suspended matter and high turbidity	• Resuspension of bottom sediments • Inadequate disposal • Fine-grained sediments	• Reduced primary productivity • Disorientation of visual predators • Death or stress due to clogging of gills (fish) and feeding structures (mussels, zooplankton)	• Particle size and cohesiveness of dredged sediments • Presence of flocculants • Local hydrodynamics • Time or duration of turbidity
2. Low dissolved oxygen	• O_2 consumption by oxidation of resuspended organic matter and reduced chemical species	• Death or stress to fish, plankton and benthic macroinvertebrates	• Nature and content of organic matter • Content of reduced chemical species in sediment and interstitial water
3. High concentrations of inorganic plant nutrients (N,P)	• Release of inorganic nutrients from dredged sediment and interstitial water	• Increased algal and bacterial productivity	• Content and availability of inorganic nutrients
4. High concentrations of toxic contaminants	• Release from interstitial water and dissolution/desorption from dredged sediments	• Short-term: Death or stress to exposed biota • Long-term: Entry into and accumulation in food chains (heavy metals, chlorinated (hydrocarbons)	• Content and availability of heavy metals and chlorinated hydrocarbons • Content of H_2S, CH_4, and NH_3 • Post-dredging release of toxicants due to redistribution of previously buried contaminated sediments
5. Silt deposition below dam	• Resuspension and transport of bottom sediments • Erosion/runoff or overflow from disposal site	• Destruction of fish spawning areas and habitats • Smothering of mussels and *other benthic invertebrates*, benthic algae, submerged macrophytes, fish eggs and larvae • Shifts in species composition, distribution and abundance	• Amount and rate of sediment transport • Presence and proximity of *refugia* (source for repopulation) to dredging site • Velocity, turbulence and flow rates of stream • Frequency and magnitude of naturally occurring spates
6. Alteration of substratum	• Removal of sediments	• Removal of benthic invertebrates • Shifts in species composition, distribution and abundance	• Presence and proximity of refugia (source for repopulation) to dredging site • Nature and amount of sediments removed • Nature of exposed substrate

• Dredging/disposal method

• Area/volume dredged

• Nature of pre-existing environmental stresses

• Duration of exposure period

• Time of year

• Species and life stage affected

In addition to the effects from decreased light penetration, there may also be direct effects due to the presence of high levels of suspended solids in the water column. Particles of suspended solids have a mechanical or abrasive action which may irritate, damage, or cause clogging of the gills or feeding structures of fish, bivalve mollusks, and zooplankton (see especially Wallen 1951, Loosanoff 1961, Sullivan and Hancock 1977). Experimental studies conducted by Sherk et al. (1976) on several species of fish and zooplankton also indicated an impairment of normal respiratory and excretory function. Finally, high levels of suspended solids may reduce or inhibit feeding by filter-feeding organisms, such as oysters and mussels, thus causing nutritional stress and eventual mortality (Loosanoff 1961).

The magnitude of the biological response will be influenced by the length of time that biota are exposed, the concentration during exposure, and the extent of the turbidity plume. The levels of suspended solids produced during dredging are, in turn, highly dependent upon the type of dredge employed and the characteristics of the bottom sediment (see Section 2.3). Wide fluctuations in suspended solid loads are a natural occurrence in most aquatic systems, and most organisms have developed mechanisms to protect against temporarily elevated levels of suspended solids. For example, bivalves can close their shells and cease pumping, and most fishes secrete an external layer of mucus that carries away particulate matter. However, substances in water from highly contaminated areas, such as harbors, could affect this mucus flow, resulting in injury to the gills even during brief periods of high suspended solids levels.

The actual levels of suspended sediment required to demonstrate acute effects and the limited exposure times have led some investigators to conclude that increases in the concentration of suspended solids as a result of dredging uncontaminated sediments would be unlikely to create a significant hazard to biota (Auld and Schubal 1978, Peddicord and McFarland 1978, Sullivan and Hancock 1977). Several field studies conducted during and after dredging operations have resulted in the same conclusions (Flemer et al. 1968; Ingle 1952; Lunz 1938a,b; McKinney and Case 1973; Stickney 1972; Wilson 1950). However, Sherk et al. (1976) found that lethal and sublethal effects on some estuarine fish do occur at suspended solids levels that could be expected during natural flooding or dredging operations; the 24-h LC_{10} for Atlantic silversides, for example, was 0.58 g/liter (i.e., a concentration of 0.58 g/liter resulted in 10% mortality after 24 h). Auld and Schubal (1978) found that survival of striped bass and yellow perch larvae was significantly reduced, within 48-96 h, by concentrations of 0.5 g/liter. Although we recognize that these concentrations are at the upper range of those measured during dredging operations (Barnard 1978) and that continuous laboratory tests do not replicate the intermittent nature of dredging, nevertheless, some stress or even death could occur to the larvae of sensitive species as a result of dredging. Thus, the use of equipment and procedures that minimize turbidity is strongly recommended.

In any ecosystem, some species and life stages will be less tolerant of elevated levels of suspended solids than others. In general, filter-feeding organisms are likely to be the most sensitive, whereas bottom-dwelling species that normally inhabit fine particulate sediments are likely to be the most tolerant. Moreover, early life stages are generally more sensitive than adults (Sherk et al. 1976), and young larvae may be more sensitive than eggs (Auld and Schubal 1978).

Several extensive literature reviews exist that discuss in greater detail many of the biological concerns relating to turbidity and suspended solids (e.g., see Cairns 1968, Darnell 1976, Hollis et al. 1964, Sorenson et al. 1977, Stern and Stickle 1978). An excellent review recently published by the EPA (Muncy et al. 1979) discusses the effects of suspended solids and sediment on various life stages of warmwater fishes and incudes lists of both tolerant and intolerant species.

3.2.2 Effects Due to Increased Downstream Siltation

Dredging to increase storage capacity or to clear existing intakes will likely be conducted near the dam. Thus, the potentially high levels of suspended solids resulting from such dredging could cause increased siltation downstream of the impoundment. Depending upon the location and containment of the dredged material disposal area, overflow or erosion from this area could also contribute to downstream siltation. In some instances, this siltation could have potentially serious biological consequences.

Siltation affects the biota of a stream either directly by smothering the organisms or indirectly by altering the substratum. Benthic primary production could be reduced if a blanket of silt is deposited and the benthic algae are smothered. Siltation is also believed to cause a reduction in the production of macrophytes (Edwards 1968). Benthic invertebrates that are sessile or have limited mobility (e.g., bivalve mollusks) could be smothered if siltation were severe. The more motile organisms (e.g., insects, crayfish) tend to increase their activity and either migrate or drift out of areas disturbed by an increase in turbidity and sedimentation (Rosenberg and Wiens 1978). Siltation can affect fish populations either directly by smothering and killing the eggs and larvae or indirectly by (1) reducing food availability, vis-a-vis a reduction in the plankton and benthic macroinvertebrate populations, or (2) filling of interstitial spaces in a gravel and rubble substratum, thus potentially eliminating both spawning beds and habitat critical to the survival of young fishes.

The amount of downstream siltation produced as a result of dredging and, consequently, the magnitude of the biological effects will be a function of several factors. These include (1) composition of the bottom material, (2) type of dredge used, (3) quantity of material dredged, (4) proximity of the dredging operation (and, therefore, the turbidity plume) to the dam, (5) level of water withdrawal at the dam (surface vs subsurface), and (6) hydraulic properties, such as velocity, turbulence and flow rate, of the stream below the dam. In those cases where impoundments must be dredged to reclaim lost storage capacity, it is assumed that the bottom material will consist primarily of silt and clay and, as a result, that turbidity plumes will occur. The extent of the plume and the proximity of the dam to the dredging site will determine whether or not the suspended solids settle out before reaching the dam. Because the highest levels of suspended solids occur near the bottom during dredging, subsurface (or hypolimnetic) releases from the dam could result in the transport of large quantities of silt downstream. Although the magnitude of downstream siltation might be reduced if surface water were withdrawn, in most cases, water released at the dam during dredging operations will carry increased silt loads to the tailwaters. In low-flow periods, this silt load would rapidly settle out in the immediate downstream area, but during periods of high flow, the silt would be distributed over a greater area.

Long-term biological effects of siltation would be influenced by the probability of occurrence of freshets or floods of sufficient magnitude and duration to scour out the silted areas. If stream flow is regulated at a more or less constant level by an upstream dam, then silt may not be flushed out, and relatively permanent shifts in substrate composition could occur (Eustis and Hillen 1954). Because stream habitats comprised mostly of sand or silt generally have lower species diversity and biomass than do habitats consisting of rubble and gravel (Brusuen and Prather 1971), siltation below a dam where flows are controlled could adversely affect the benthic biota. On the other hand, when freshets or floods scoured the area and an upstream source of drifting insects was available, benthic populations have shown a relatively rapid recovery (6-12 months) from the effects of logging, road construction, or channelization (Barton 1977, Burns 1972).

Short-term effects will be influenced by the amount of siltation that occurs. The levels of siltation that would be harmful are site-specific and would be dependent upon the annual variability in maximum sediment loads and the seasonality associated with high sediment transport rates. Short-term effects from siltation would be greatest during the peak spawning and reproductive periods of fishes and benthic invertebrates. For many species, this period occurs in the spring and early summer. Fishes most likely to be affected are those which lay their eggs in clean gravel areas, or gravel nests, including many of the sunfishes, catfishes, darters, and minnows (Muncy et al. 1979).

In streams with endemic mussel populations, however, high siltation at any time of year could be harmful. Mussels have been reported to be unable to survive in a layer of silt greater than 0.6 cm (Ellis 1936) and is one reason why few species are able to survive in impoundments. However, mussel populations have often been found to occur in spillways below small dams (Fuller 1974). By functioning as silt traps, these dams reduce downstream siltation in rivers and streams that had previously been degraded by heavy sediment loads from erosion and runoff in the watershed. These spillway populations are frequently the only remaining mussels in these rivers and often consist of rare or endangered species. Such a phenomenon has occurred below dams on the Duck River in central Tennessee (Tennessee Valley Authority 1979) and the Olentangy River near Columbus, Ohio (Stein 1972). Channelization and road construction above the dam on the Olentangy River apparently destroyed the diverse mussel populations that existed below the dam (Stein 1972). In cases such as these, recovery is unlikely due to the absence of refuge populations. Thus, the destruction of mussel populations could be the most significant impact from increased siltation downstream of the dredging site.

Further discussions of the general effects of siltation may be found in Brusuen and Prather (1971), Chutter (1968), Cordonne and Kelly (1960), and Ellis (1936).

3.2.3 Effects Due to Substrate Removal

Removal of the substrate may involve changes both in circulation patterns and in the properties of bottom sediments. For small hydro projects, changes in circulation patterns are likely to be of little consequence, except in those cases where the increase in water depth due to dredging results in thermal stratification which did not occur previously. Changes in mechanical properties of bottom sediments, such as particle size distribution and porosity, may follow dredging operations and may influence (1) movement of soluble contaminants across the sediment-water interface, (2) resistance of remaining sediments to resuspension and redistribution by local currents, and (3) distribution of benthic organisms.

Because benthic organisms are an important food resource of fishes, a reduction or change in this resource could interrupt food chain dynamics (i.e., fish-benthos interactions) in the reservoir, particularly if the dredged area is large in relation to the reservoir as a whole. This interaction might also be affected by alterations in the nature of the substratum exposed by dredging. Substrate particle size is one of the primary factors influencing the distribution of benthic invertebrates (Cummins and Lauff 1969). Consequently, alteration of the substrates may result in the reestablishment of a different benthic community than existed prior to dredging. The severity of this disruption in food chain dynamics could well depend upon both the availability of secondary or alternate food resources (e.g., zooplankton, forage fishes) and the ability of species to compete for and utilize these resources.

An important factor affecting the magnitude of the impact caused by removal of the substrate is the length of time required for recolonization/recovery of the benthic populations. The recovery rate will be dependent upon (1) the existence of undisturbed areas either in the reservoir or upstream that could serve as sources for recolonization of the dredged area and (2) reproductive rates and motility of the recolonizing species. To restore or enhance trout fisheries, small north-central Wisconsin ponds are hydraulically dredged (see Section 2.4), and areas are left undisturbed in order to aid in the rapid reestablishment of the benthic fauna (Carline and Brynildson 1977). When the entire pond was dredged, as was done initially, benthic populations required several years before reaching pre-dredging densities. Other studies of disturbances caused by channelization, dredging, or draining together with dredging, indicated that repopulation of the disturbed area was rapid; a return to pre-impact levels generally occurred by the next breeding season due to drift from upstream areas (Andersson et al. 1975, Crisp and Gledhill 1970, Duvel et al. 1976, Pearson and Jones 1975). Studies of Lake Trummen in Sweden, which was highly polluted by sewage effluent but was restored by dredging, indicated that dredging significantly improved the water quality of the lake without resulting in any serious impacts on the benthos (Andersson et al. 1975, Bengtsson et al. 1975, Cronberg et al. 1975). Apparently, dredging did not alter the particle size distribution of the sediments but simply removed the nutrient-rich upper layers.

Finally, removal of the substratum during the dredging operation also results in the removal and displacement of the organisms associated with it. The impact resulting from this removal of bottom-dwelling organisms will be determined, to some extent, by the type of dredge used. Pearson and Jones (1975) have suggested that much of the substratum and associated fauna fall from the buckets during drag line operations. Under these conditions, repopulation is likely to occur rapidly. Although hydraulically operated dredges are likely to be much more efficient at removing bottom materials (and organisms) without extensive spillage, they may disturb and displace the lighter materials resulting in the burial of organisms outside the immediate dredging zone.

3.2.4 Effects Due to Chemical Changes in the Water Masses and Sediments at the Dredging and Disposal Sites

3.2.4.1 Nature of the chemical changes

Dredging and dredged material disposal may result in at least temporary changes in the chemistry of the water masses associated with both the dredging and disposal sites as well as in the chemistry of the sediments at both sites. Many chemical constituents associated with buried sediments and sediment interstitial water are not in chemical equilibrium when these are mixed with surface waters during dredging and disposal. For example, undisturbed lake and reservoir sediments typically exhibit a gradient from oxidized deposits near the surface to increasingly reduced sediments with greater depth in the deposit.

Thus, sediments from deeper layers create an oxygen demand when they are exposed, via resuspension, to the aerobic environment of the overlying water body. Reductions in dissolved oxygen (DO sag) due to the high oxygen demand when deeper sediments are exposed during dredging have been reported by Windom (1975). Studies cited in Darnell (1976) have shown that (1) some dredged materials may require more than 500 times their own volume of oxygen for complete oxidation, and (2) oxygen levels near the dredging site may be 18 to 83% below normal. Levels of dissolved oxygen at the dredging site are influenced by (1) amount of sediment resuspended, (2) redox potential of the sediment, (3) amount of organic matter in the sediments, (4) chemical composition of the sediments, (5) stimulation (or inhibition) of bacterial or algal production, and (6) degree of hydrologic flushing that occurs at the dredging and disposal sites.

The physicochemical environment within undisturbed lake sediment may also allow many chemical species to attain higher concentrations in the interstitial water than in the overlying water. Interstitial water concentrations of nutrients (inorganic forms of nitrogen and phosphorus), metals (especially manganese and iron), and trace gases (ammonia, hydrogen sulfide, and methane) often greatly exceed concentrations in the overlying water. Thus, when interstitial water is mixed with overlying water during dredging operations, the concentrations of these chemical species may increase temporarily in the vicinity of the dredging site and/or disposal area. In addition, exposure of buried sediments to aerobic conditions may result in transformations of some solid phases to either more- or less-soluble forms. For example, where a metal is bound in the solid phase as a metallic sulfide, oxidation of the relatively insoluble sulfide may result in either formation of more-soluble solid phases, such as sulfates or carbonates, or transfer of the metal ion to an adsorption site. Although oxidation of metallic sulfides appears to be slow, certain dredged material disposal practices (e.g., confined upland disposal) could potentially provide sufficient time for some sulfide oxidation to occur (Chen et al. 1978, Brannon 1978, Gambrell et al. 1977). Fortunately, most fine-grained sediments and soils have a high capacity to adsorb most metals under nonacidic conditions. The precipitation of hydrated oxides of iron and manganese, which may accompany the oxidation of soluble iron and manganese in interstitial water injected into overlying water by dredging, also facilitates removal of some metals released to the overlying water. These hydrous oxides have a high capacity to scavenge (by co-precipitation and/or adsorption) metals from the solution phase (Jenne 1968).

Many organic contaminants, including pesticides and polychlorinated biphenyls (PCBs), are also found associated with lake and reservoir sediments. Most of the compounds are very insoluble in water and have a strong affinity for particulate matter. As a consequence, release of organic contaminants into the solution phase during dredging is generally negligible (Brannon 1978, Chen et al. 1978).

Burks and Engler (1978) identified several factors that influence the degree to which pesticides and PCBs are released from the sediments. The amounts released were directly related to the concentration in the sediments, but seemed to be inversely related to the oil and grease content of the sediments. Sediment-water ratios greater than those that occur during hydraulic dredging are apparently necessary to cause the release of any pesticides. However, buried sediments, containing higher levels of these organic contaminants than occur in recently deposited sediments, may be resuspended into the water column and may pose a threat to organisms which ingest particulate matter either from the water column or from superficial sediments. In addition, dissolved organic matter, especially fulvic acids, can significantly increase the solubility of some otherwise nearly insoluble chlorinated hydrocarbons (Goldberg 1976).

The release and persistence of many contaminants from the solid phases of dredged material to the solution phase of overlying water or groundwater is highly dependent on pH (negative logarithm of hydrogen ion concentration) and Eh (oxidation-reduction potential). These parameters also strongly regulate the chemical form, and thereby the toxicity, of many contaminants. In general, metals tend to be released (solubilized and/or desorbed) under acidic (low pH) and reducing (low or negative values of Eh) conditions and can persist in the solution phase under these conditions (cf. Gambrell et al. 1978). Hydrogen sulfide is formed under reducing conditions from the reduction of sulfate and organic matter and may persist dissolved in interstitial and overlying water under these conditions. Ammonia, as the highly toxic un-ionized NH_3, can accumulate under alkaline (high pH) conditions in aquatic systems with restricted exchange with the atmosphere, but becomes increasingly unstable (transforms to the nontoxic NH_4^+) as conditions become more acidic. Neither hydrogen sulfide nor ammonia can persist for very long in well-ventilated waters. The concentration of soluble phosphate released from interstitial water is often regulated to low levels under oxidizing conditions (high Eh) and neutral pH because of the formation of relatively insoluble iron phosphate.

It is not possible to predict which of the many contaminants potentially mobilized during dredging and dredged material disposal will present an environmental problem without some prior knowledge of the chemical nature of the sediments to be dredged. Such knowledge may be obtained from two sources: (1) the recent and historical land use in the tributary watershed and (2) laboratory studies of the sediment to be dredged. Obviously, the former source provides only a crude estimate of the kinds of contaminants which may be found in reservoir sediments and mobilized in a reservoir dredging operation. For example, the presence of a pulp mill upstream of a reservoir may suggest a high probability of mobilizing excessive hydrogen sulfide and mercury into surface waters if reservoir sediments are disturbed by dredging. Similarly, where watershed land use has been heavily agricultural, one might expect excessive mobilization of plant nutrients and persistent pesticides.

Laboratory studies of lake and reservoir sediments may also be used to assess the kinds and levels of contaminants likely to be mobilized during dredging and dredged material disposal. One common laboratory procedure is the Standard Elutriate Test, jointly developed by the EPA and the U.S. Army Corps of Engineers. The purpose of this test is to classify sediments as "polluted" or "not polluted" to allow prediction of water quality impacts due to dredging and dredged material disposal prior to the commencement of dredging (U.S. Environmental Protection Agency 1975). The procedure is basically aimed at extracting, identifying, and measuring (1) the chemical constituents already dissolved in the interstitial water of sediments and (2) those constituents which are rather loosely bound or sorbed to the sediment. Results of this elutriate test provide an estimate of the short-term releases of contaminants from sediments to be dredged. Although less intensively evaluated, this test has also had some success in estimating long-term releases of contaminants (cf. Lee and Plumb 1974). Column leaching tests have been used to evaluate short- and long-term releases of contaminants from dredged material disposal at upland (subaerial) sites (Mang et al. 1978, Yu et al. 1978).

3.2.4.2 Types of biological effects

Exposure of biota to low concentrations of dissolved oxygen may be stressful or may result in mortality if the levels are very low for extended periods of time. Dissolved oxygen levels below 4 mg/liter for more than 24 h can be considered unfavorable for most fish species and many stream-inhabiting invertebrates. Early developmental stages (eggs and larvae) tend to be most sensitive, but sensitivity will vary among species. Some invertebrates, such as burrowing mayflies and bloodworms, may be able to tolerate dissolved oxygen levels of 2 mg/liter or less almost indefinitely. Organisms stressed by low levels of dissolved oxygen are less able to cope with any additional stresses, such as high temperatures, low pH, chemical pollution, or turbidity (Darnell 1976).

Elutriate tests have shown that ammonia (NH_4^+) is frequently released from sediments during dredging (Brannon 1978). Under conditions of high pH, NH_4^+ can be converted to the highly toxic un-ionized NH_3. Short-term experimental tests have shown that lethal concentrations of NH_3 vary from 0.2 to 2.0 mg/liter; trout were the most sensitive species while carp were the most resistant (U.S. Environmental Protection Agency 1976). However, adverse physiological effects may occur at concentrations below 0.2 mg/liter. Studies conducted during open-water disposal operations have shown that lethal concentrations of ammonia (NH_3) are present for short periods of time (hours or less)(Burks and Engler 1978). In one typical field test, the safe chronic exposure level of 0.02 mg/liter was exceeded for only 12 min (Brannon 1978).

Hydrogen sulfide (H_2S), a by-product of the anaerobic decomposition of organic matter and the reduction of inorganic sulfur sources such as sulfate, is very toxic at low (μg/liter) concentrations and may be released during dredging. Reducing conditions and low pH levels favor the persistence of hydrogen sulfide released from anaerobic sediments. The toxicity of hydrogen sulfide is demonstrated by the fact that 96-h-LC_{50} values ranged from 17 to 32 μg/liter for northern pike fry, while long-term exposure to very low levels (1 μg/liter) reduced egg deposition in bluegill and egg development in white suckers (U.S. Environmental Protection Agency 1976). Although the release of hydrogen sulfide has not previously been identified as a problem associated with dredging, reducing conditions and low pH levels which favor the persistence of H_2S in the water column may exist in small impoundments affected by pulp mill wastes and acid mine drainage. A careful evaluation of potential impacts should be made if the potential small hydro site is located in a watershed where strip-mining operations currently exist or have existed in the past.

The biological effects due to the release of phosphorus from dredged sediments are primarily related to the enrichment effect of phosphate phosphorus (PO_4-P). Because PO_4-P is an important plant nutrient and may be the limiting nutrient in many lentic ecosystems, its release could stimulate the growth of both algae and macrophytes. On the other hand, high turbidity at the dredging and disposal sites could depress productivity by reducing light penetration, thus counteracting any potential stimulatory effects due to a release of phosphorus. Very few studies have demonstrated an increase in phytoplankton standing crop or a "bloom" as a result of dredging. Rather, most studies have indicated that no net effects were detected (Stern and Stickle 1978). However, elutriate tests (Flint and Lorefice 1978) and studies of completely dredged lakes (Cronberg et al. 1975) did find increases in heterotrophic bacteria. Whether or not this increase was due to elevated levels of phosphorus or other nutrients was not determined. The increased heterotroph production could provide additional food for filter-feeders and/or lower the dissolved oxygen content of the water.

Dredging and dredged material disposal can potentially result in releases of soluble toxic substances and the resuspension of particulate matter containing high levels of heavy metals and toxic organic compounds such as pesticides and PCBs. The biological effects from these elevated contaminant levels are of two general types. Acute toxicity can occur if the contaminants are present in the dissolved form in high concentrations for brief periods of time (several days). Chronic toxicity, on the other hand, results when biota are exposed to relatively low concentrations for several weeks or longer. Whereas short-term exposures to very high concentrations can be lethal, chronic effects are often sublethal (e.g., reduced growth and reproduction) and associated with the bioaccumulation of these contaminants in body tissues. Both acute and chronic effects of PCBs and pesticides, especially the organochlorine insecticides such as DDT, are discussed in detail in Appendix A. A summary of the biological effects of these contaminants follows.

Chlorinated hydrocarbons

Chlorinated hydrocarbons such as PCBs and most of the organochlorine insecticides (e.g., DDT, dieldrin, toxaphene) not only are toxic at very low concentrations but also can be bioaccumulated in the tissues of freshwater biota. Unlike heavy metals which are much more soluble in water, PCBs and DDT are relatively insoluble in water but are highly soluble in lipids. Consequently, they are strongly partitioned from water into lipids (fats) of aquatic biota, resulting in greater bioaccumulation than that of most metals. Chlorinated hydrocarbons can bioaccumulate in tissues of both invertebrates and fishes to levels that greatly exceed those in water. In this regard, the concentration factor (CF) or bioconcentration factor (BCF) is often used to relate the concentrations in biota and water. The BCF is expressed as a ratio of the concentration of a substance in the organism to the concentration in water and can be derived from either laboratory or natural exposures to a particular toxicant (Phillips and Russo 1979). Laboratory studies have shown that BCFs for many freshwater invertebrates and fishes typically fall in the range of 10^3 to 10^5 for many chlorinated hydrocarbons. While BCFs based on natural exposures have been hypothesized to be greater than laboratory-derived values, some evidence suggests that the differences between the two estimates may be minimal (see Appendix A).

The widespread distribution and persistence of these compounds is a result of their resistance to metabolic degradation. Even though muscle tissue has been shown to have the lowest concentrations of PCBs and DDT of most tissues, the rapid uptake and retention of these compounds can lead to significant muscle tissue concentrations that exceed the FDA limits for both PCBs and DDT (5 μg/g wet wt in the edible portions of fish). Their high toxicity and bioaccumulation potential place these compounds near the top of the list of contaminants with the greatest potential for environmental impact.

Heavy metals

Because of the extensive literature on the acute and chronic toxicity of metals to aquatic biota, no attempt has been made to review all of it in this report. Instead, the reader is encouraged to consult the document 'Quality Criteria for Water' (U.S. Environmental Protection Agency 1976) and Leland et al. (1979) for a review of the toxicity of various dissolved metals that occur in natural waters. Very extensive and recent reviews also exist for the more toxic heavy metals (e.g., see Hammons et al. 1978 and U.S. Environmental Protection

Agency 1977, both as cited in Spehar et al. 1979, for reviews of cadmium and mercury, respectively). The literature on the bioaccumulation of metals in aquatic biota has been reviewed by Phillips and Russo (1979), and a summary of this topic taken from that review follows.

The degree to which various metals are bioaccumulated in the tissues of freshwater biota varies widely (Table 4). Some metals such as arsenic, cadmium, and lead accumulate in the tissues of invertebrates, but do not tend to accumulate in the muscles of vertebrates such as fish. Consequently, these elements are less of a hazard to humans (who are major consumers of fish) than is mercury, which not only is toxic at low concentrations but also accumulates in muscle tissues. Currently, the FDA action level for mercury in the edible portions of fish is 1.0 μg/g wet wt.

Bioconcentration factors for most metals are generally much lower than those reported for the chlorinated hydrocarbon compounds. Some of the highest BCF values were reported for the uptake of copper and zinc by oysters (28,000 and 26,000, respectively) and the uptake of mercury by fathead minnows (83,000). Evidence demonstrating that copper and zinc are homeostatically controlled in fish has been presented by Wiener and Giesy (1979). However, other elements, especially mercury, can accumulate to significant levels in fish. Of the elements listed in Table 4, mercury has the greatest potential for creating adverse effects on aquatic biota as a result of dredging. Because cadmium is toxic at very low levels (1-10 μg/liter) and can accumulate in the tissues of aquatic invertebrates (BCF = 10^2-10^3), it should also

Table 4. Major sources to natural waters, bioaccumulation potential, biological half-time and significance, and toxicity to humans of selected metals. Source: Phillips and Russo (1979).

Metal	Major sources to natural waters	Bioaccumulation potential: Invertebrates	Bioaccumulation potential: Fish muscle	Biological half-time (d)[a]	Biological requirement[b]	Toxicity to humans[c]
Al	. Industrial wastes . Water treatment facilities . Strip mining . Oil shale mining	N/A[d]	High	N/A	E	Low
As	. Atmospheric fallout from ore smelting and fossil fuel combustion . Industrial outfalls . Improper application of arsenical herbicides or pesticides	N/A	Low	7 (green sunfish)	NE	High
Cd	. Effluents from electroplating and smelting industries . Runoff from agricultural areas where phosphate fertilizers are used	High	Low	378 (shrimp) 1254 (mussels)	NE	High
Cr	. Industrial (e.g., electroplating, steelmaking, photographic) wastes . Nuclear effluents	Low	Low	123 (polychaete worms)	NE	Low
Cu	. Acid mine drainage Algicides	High	Low	N/A	E	Low
Fe	. Corrosion . Steel pickling . Mineral processing . Acid mine drainage	High	High	N/A	E	Low
Pb	. Runoff from highways, lead mines . Atmospheric fallout . Exhaust from outboard motors, snowmobiles	High	Low	Long?	NE	High
Mn	. Acid mine drainage . Industrial outfalls	High	Low	333 (plaice)	E	Low
Hg	. Effluents from chlor-alkali and pulp and paper industries . Combustion of fossil fuels . Natural weathering	High	High	100-400 (mussels) 365-1100 (fish)	NE	High
Ni	. Coal combustion emissions . Effluents from metal plating industries	Low?	Low	N/A	NE	Low
Se	. Combustion of fossil fuels . Agricultural and industrial wastes . Natural sediments	High	Low	37	NE	High
Ag	. Effluents from photoprocessing and electroplating industries . Natural weathering . Cloud-seeding	Low?	Low	Very short	NE	Low
Zn	. Acid mine drainage . Numerous industrial effluents	High	Low	255 (Pacific oysters) 235 (Mosquitofish) 313 (plaice)	E	Low

[a]The amount of time required for an organism to eliminate half of the total body burden of an accumulated substance.

[b]E = Essential to physiological function; NE = Nonessential.

[c]From oral ingestion.

[d]N/A = Information not available from source reference.

warrant concern. All of the remaining metals could have locally high concentrations at the dredging site and therefore could, depending on their availability, represent potential sources of biological impact during dredging and disposal operations. Their significance as a source of impact can only be determined on a site-specific basis. More information on the toxicity and bioaccumulation of heavy metals and chlorinated hydrocarbons as a result of dredging and disposal operations can be found in Hersh et al. (1978).

3.2.4.3 Factors influencing the magnitude of the biological effects due to chemical changes

Several factors play a role in determining the significance of the biological impacts attributable to alterations in water chemistry. For example, responses of biota to elevated concentrations of various contaminants will vary greatly between species and among different life stages of the same species (see Section 3.2.1 and Tables A-1 and A-2). The nature and magnitude of the response (e.g., death, avoidance of the dredging site, reduced growth and/or fecundity) will also be dependent upon the specific contaminant and its availability to biota, i.e., its presence in the particulate and/or dissolved fraction (Section 3.2.4.2).

Laboratory tests of resuspended sediments have shown the release of pesticides, PCBs, and other organic contaminants into the solution phase to be negligible (Fulk et al. 1975, Lee et al. 1975). Most of the contaminants existed in association with suspended particulates. The resuspension of contaminated substrates will increase the opportunity for ingestion of sediment-adsorbed organics by filter-feeding and deposit-feeding biota. The availability to aquatic biota of particulate-associated contaminants, however, is not well understood (Brannon 1978). Very little is known about either the rate of uptake and bioaccumulation of pesticides from sediments (Nathans and Bechtel 1977) or about the toxicity of various organic compounds adsorbed to suspended particulates (Lee et al. 1975). In laboratory studies that must be considered preliminary, Nathans and Bechtel (1977) concluded that annelids (worms) accumulated DDT from sediments but that the uptake was slower and resulted in lower whole-body concentrations than would have occurred from the same concentration of DDT in water. These results are similar to those described in studies of the uptake of PCBs and pesticides from dietary sources (see Appendix A). Munson et al. (1976) found that PCBs, DDT, and chlordane were bioconcentrated by a factor of 5 to 8 in passing from the suspended sediment into zooplankton. They concluded that the movement of chlorinated hydrocarbon compounds into the biological from the nonbiological system (i.e., from suspended sediments) was not influenced by changes in the concentration of the latter but was regulated by those factors that control the zooplankton. Thus, conditions that increase zooplankton populations would probably increase the movement of organic contaminants into the biological system.

Of the many factors affecting the magnitude of the effects on aquatic biota from dredging-induced changes in water chemistry, two of the most important are the concentration (particulate or dissolved) to which the organisms are exposed and the length of the exposure period. The concentration of various contaminants in the sediments and the impacts associated with the resuspension of these sediments during dredging will be highly site-specific. The initial concentrations that will occur at the dredging site will be determined by the complex interaction of many parameters (see Section 3.2.4.1). The concentration to which the biota are actually exposed will depend upon not only the physical and chemical nature of the substratum but also the degree of mixing and dilution that occurs. If the substrate consists primarily of silt and clay which settle out very slowly, then the turbidity plume could be very large and persistent (Barnard 1978). As a result, organisms would be exposed to higher concentrations for longer periods of time than if the sediments contained mostly coarse particulates.

The exposure period will also be affected by the type of dredging operation. Relatively short-term operations to clear existing intakes, repair dams, or construct/repair powerhouses, headraces, and tailraces might require only several days or weeks to complete. During these times, aquatic biota would be exposed to brief periods of elevated contaminant levels that would not result in a significant level of bioaccumulation. If however, the purpose of dredging were to increase storage capacity in the reservoir, then the biota could be exposed to elevated concentrations for a period of several months. With high toxicant concentrations and continuous 24-h dredging in the impoundment, bioaccumulation could be significant.

In summary, many factors will influence the extent to which chlorinated hydrocarbons and heavy metals are toxic to and bioaccumulate in aquatic biota. Because a large fraction of these contaminants may exist in the particulate form, any analysis of the magnitude of these biological effects is difficult due to the paucity of information on the toxicity, uptake, and accumulation of suspended particulate contaminants. However, based on the results of studies by Brannon (1978) with unfiltered elutriates (suspended particulate phase), short-term acute toxicity resulting from dredging and disposal operations may be of low concern at most sites. Long-term chronic effects, on the other hand, have not been studied as extensively but would be expected to occur in those cases where (1) dredging to increase storage capacity was a continuous operation that occurred over several months, or (2) effluent runoff entering the reservoir from the disposal area usually contained large quantities of fine particulates. In both cases, the dredged material is assumed to be highly contaminated.

3.2.5 Effects Due to Upland Dredged Material Disposal

The method of disposal most likely to be employed if dredging is required at small hydro sites would be upland disposal (see Section 2.2). Several potential effects could be associated with this type of disposal. Contaminants in the dredged material (e.g., heavy metals, chlorinated hydrocarbons) could be transported from the disposal area by several mechanisms. These include (1) leaching into groundwater, (2) surface runoff of constituents in either dissolved or suspended particulate form, (3) plant uptake and subsequent cycling through food webs, and (4) direct uptake by animals living in close association with the soil (Gambrell et al. 1978). The potential adverse impact caused by leaching into groundwater is the contamination of drinking water supplies due to the presence of various toxicants.

Studies conducted by Chen et al. (1978) of leachates from a freshwater disposal site showed that most of these leachates contained ammonia, nitrate nitrogen, iron, and manganese in concentrations that exceeded drinking water standards.

Surface runoff from disposal sites can result in erosion and the ultimate transport of contaminants to nearby water bodies. Initially, the runoff may carry substantial quantities of toxicants, nutrients, and high suspended sediment loads. The biological effects of these constituents upon entering a water body would be similar to those described in Sections 3.2.1 through 3.2.4. However, the exposure of biota to contaminants adsorbed to fine particulates in the effluent from the disposal area can occur over long periods of time, thus constituting a potentially greater threat than exposure at the dredging site. Results from column settling tests suggest that the release of PCBs and other chlorinated hydrocarbons into the solution phase should be negligible during the usual detention period for confined disposal areas (Chen et al. 1978). However, suspended solids levels in the effluent were almost two orders of magnitude higher than ambient water concentrations. Because most trace contamiants are associated with suspended particulates, very long detention times or the use of flocculants may be required if large quantities of low-density solids comprise the dredged material (Chen et al. 1978).

The uptake mechanisms by plants and animals inhabiting the disposal area are not well understood. Similarly, the rates of uptake and the consequences of the recycling of many of these constituents through the terrestrial ecosystem have not been thoroughly studied.

Toxic heavy metal uptake by crop plants from contaminated or sludge-amended soil, however, has been studied extensively (Gambrell et al. 1976). These studies have shown that (1) increased heavy metal concentration in soil can lead to increased levels in crops without causing plant toxicity; (2) cadmium is the element of greatest concern with regard to both uptake by crops and consumption by animals, and (3) variation exists in the uptake and toxicity between plant species (review by Martin et al. 1976). Whether or not a given metal is a potental contaminant will depend greatly on its form and availability rather than on its total concentration (Lee et al. 1976).

A major effect of upland disposal is the loss of habitat, although the area may be reclaimed after disposal is terminated. Disposal of dredged material on land has resulted in killing and stunting many areas of bottomland forest (Brady 1976). The disposal of dredged material on areas in the early stages of succession, such as on a weedy herbaceous area, is less damaging than disposal on areas of mature forests or later seral stages because revegetation may occur more quickly in the former (Brady 1976). In addition, mature forests may be a more valuable habitat for wildlife, although its value would actually be dependent upon specific wildlife management priorities.

Many factors influence the magnitude of these biological effects, but probably two of the most important are (1) degree to which the upland disposal area is confined and (2) how the disposal area is managed (reclaimed). Diked disposal areas with impermeable liners provide major controls over surface runoff and groundwater contamination. The effectiveness of the disposal area in removing particulates from the water column is controlled mainly by the detention time of the containment area (which, in turn, is determined by inflow rates and the size of the disposal area) and the particle size distribution of resuspended sediments (Chen et al. 1978). Fine-grained sediments, such as silts or clay, are generally higher in organic content, cation exchange capacity (CEC), available nutrients, and, in some cases, heavy metal loading. The CEC of a dredged material governs the sorption of ammonium nitrogen, potassium and other cations, heavy metals, and some pesticides (Lee et al. 1976). The particle size distribution of the sediments can also affect reclamation efforts. Fine-grained sediments are much easier to vegetate than coarse or sandy soils, but may require months to dry before they can be worked (Hunt 1976). If the disposal area is unconfined, then rapid revegetation would be important in controlling erosion and runoff.

Dredged material disposal areas can be reclaimed for agricultural production, wildlife or recreational development, and landfill material. These alternatives, which are discussed in detail elsewhere (see Gambrell et al. 1978, Morton 1977), may be a key factor in regulating long-term releases of contaminants. Since very little information exists on long-term releases and the biological effects of those releases, predictions of impact are difficult at best. A joint COE/EPA committee has recently recognized a need for "a chemical characterization procedure for dredged and fill material that can be used to predict or estimate long- or intermediate-term releases of constituents" (Wilkes and Engler 1977).

In summary, the release (or immobilization) of most sediment-associated contaminants is regulated to a large extent by the physicochemical environment (pH, oxidation-reduction conditions, and salinity) and microbial activity associated with the dredged material at the disposal site (Gambrell et al. 1978). Although many properties are important, much can be inferred about the potential for contaminant release from the clay and organic matter content, initial and final pH, and oxidation-reduction conditions. Coarse-grained sediments low in organic matter are less effective in immobilizing metal and organic contaminants than sediments that are biologically and chemically active with high organic content but little or no oxygen (Gambrell et al. 1978). Under some disposal conditions, a well-drained upland disposal site can lead to an oxidizing acidic environment conducive to the leaching of contaminants, particularly heavy metals. Whether or not the leachate will contaminate groundwater is dependent upon the hydraulic transmissivity and absorptive capacity of the natural soils (U.S. Army Corps of Engineers 1979b). Gambrell et al. (1978) is an excellent reference that not only discusses the factors influencing contaminant mobilization and release at disposal sites in considerably more detail than is presented here but also offers guidelines for selecting disposal alternatives for contaminated dredge material to minimize adverse environmental impacts.

4. ENVIRONMENTAL FEASIBILITY OF DREDGING AT SMALL HYDRO SITES

4.1 ENVIRONMENTAL CONSTRAINTS AND MITIGATION

The environmental constraints associated with dredging at small hydro sites can be significant if a large fraction of the total bottom area of small reservoirs is dredged. Under these conditions, secondary production (benthic invertebrates and fish) in the reservoir could be reduced dramatically. Obviously, the greatest direct impact would be on the benthic invertebrates, with recovery to pre-dredging population levels requiring as long as several years. Results of this nature were reported when small ponds in Wisconsin were extensively dredged to enhance the trout fishery (Carline and Brynildson 1977). Fish production could also be reduced if a significant decrease in food resources (e.g., benthos) occurred. For example, declines in the growth rates of the trout inhabiting these Wisconsin ponds occurred immediately after dredging and were associated with the decrease in benthos, the primary food of the trout. Mortality to benthos as a result of their removal from the reservoir is a direct consequence of dredging for which no reasonable mitigation exists. However, recovery/re-establishment of reservoir benthic populations will be most rapid if (1) dredging operations are completed shortly before the seasonal increase in biological activity or larval abundance (Hirsh et al. 1978) and (2) some areas of the reservoir are not dredged and can serve as sources for the colonization of dredged areas. It should be noted that most small hydropower redevelopment projects will not involve the removal of huge quantities of sediment from a large portion of the reservoir due to the tremendous economic costs (see Section 4.3).

In reference to dredging activities in the Chesapeake Bay, it has been stated that how the bay is dredged and what is done with the dredged material are more significant than the fact that it is dredged (p. 47 in Massoglia 1977). This statement is also applicable to dredging operations at most small hydro sites. If, however, an endangered mussel species maintained a remnant population immediately below an impoundment, then dredging in the reservoir might adversely affect its habitat and survival.

How the sediments are dredged will affect the degree of environmental impact at both the dredging and disposal sites. Consequently, impacts can be mitigated by the type of dredging operation employed in the project. Mechanically dredged sediments will be compact and will contain only a minimal amount of water when transported to the upland disposal site. Thus, this method of dredging should greatly reduce, if not eliminate, the contamination of nearby waters by the overflow water normally associated with hydraulically dredged sediments (Lee 1976a). If an unconfined upland disposal technique is used (i.e., the sediments are uncontaminated), mechanical dredging could also minimize the effects of erosion and runoff from the disposal site to a greater extent than would hydraulic dredging. A major disadvantage of mechanical dredging is the greater cost associated with rehandling and transporting the dredged material (Section 4.3).

The higher turbidity levels associated with conventional mechanican dredging operations can be minimized by using silt curtains or "diapers," an impervious material suspended vertically in the water column (Mohr 1976). Studies have shown that, if properly deployed, these devices can be effective in currents up to at least 0.26 m/s (0.85 ft/s) (Johanson 1976). Guidance on the selection and use of these devices is provided in Barnard (1978). Other potential methods of controlling turbidity include (1) proper equipment maintenance and operation, (2) use of chemical flocculants, (3) use of one of several specialized dredges that have recently been developed for this purpose and are discussed in Section 2.4 (Barnard 1976, 1978).

How the dredged material is disposed of will greatly affect both the nature and magnitude of potential environmental impacts. Simply stated, placing contaminated dredged material in an unconfined disposal area carries a considerably higher risk of impact than placing the spoils in a confined (or diked) disposal area. Impacts of contaminated spoils in confined disposal areas can be minimized in several ways. Proper dike construction will prevent seepage and reduce the risk of structural failure. Effects of overflow water on nearby waters can be minimized by ensuring that the disposal areas are of sufficient size and depth so that most of the fine particulates (and the heavy metals and chlorinated hydrocarbons sorbed to them) have settled out prior to discharge from the disposal area. Also, flocculants and filtration can be used to reduce the concentration of suspended solids in the overflow [see Barnard and Hand (1978) for guidance in this area].

Potential groundwater contamination can be reduced by using synthetic or clay liners in the basin. Leaching can also be minimized by employing various dewatering/densifying techniques when hydraulically dredged materials are placed in confined disposal areas as a slurry. These techniques are described in detail by Haliburton (1978). By promoting shrinkage and consolidation, surface runoff would be reduced and the dredged material could be more readily used for agricultural soils amendment, upland habitat development, surface mine reclamation, landfill and construction material, and sanitary landfill (Spaine et al. 1978). Uptake of contaminants by plants can be minimized by (1) planting fiber rather than food crops, or (2) selecting crops in which heavy metals tend to accumulate in plant tissue that is not harvested (Gambrell et al. 1978). Liming can also be an effective method of reducing the availability of many heavy metals to plants (Gambrell et al. 1978). Thus, some mitigation measures can be applied to all three potential problem areas associated with confined upland disposal (plant toxicity and surface-water and groundwater contamination).

Another impact of dredged material disposal is the loss of habitat. In the case of upland disposal areas, the initial loss can be mitigated by reclamation and/or the use of agronomic/wildlife management techniques. Upland habitat development is a low cost method that is based on the application of well-established agricultural and wildlife management techniques, the principles and applications of which are adaptable to virtually any upland disposal site (Smith 1978). In the process of developing an upland habitat, a vegetation cover is established that would reduce erosion potential, thus stabilizing the dredged material and preventing its return to the waterway (Smith 1978). However, the potential for uptake and recycling of contaminants through the terrestrial ecosystem still remains. Upland habitat development techniques were addressed in the U.S. Corps of Engineer's Dredged Material Research Program and several good references are available (Hunt et al. 1978, Lunz et al. 1978, Smith 1978).

One of the best methods of minimizing environmental impact is to avoid dredging during the period of high biological productivity (spring through late summer). For example, dredging during the peak spawning period of those fishes that spawn at or near the dredging site or in the downstream regions of the river below the dam should be avoided. Rosenberg and Wiens (1978) have recommended that disturbances resulting in sediment addition to streams should be carried out during a period when active stages of the benthic insect fauna are low in number, such as late summer, provided, of course, that river discharge could adequately transport the sediments that are inadvertently released during the dredging operation in the reservoir. Studies conducted during dredging operations in the Chesapeake Bay suggest that October and November would be the times when dredging and dredged material disposal would have the least damaging effect (Flemer et al. 1968). Other investigators have also recommended that dredging should coincide with minimal biological productivity (Peddicord and McFarland 1978) and should avoid periods of fish spawning (O'Connor et al. 1976).

Such mitigation, however, could conflict with construction schedules. For example, the construction period in Maine is the summer (U.S. Department of Energy 1979c), and difficulties may be encountered in dredging at other periods due to cold weather and ice cover. The time required for dredging is a critical factor. Attempts to avoid dredging during the spring and summer conflict with the best time for dewatering dredged material (i.e., during the warmest periods of the year when evaporation rates are highest). In the Southeast, the peak drying period extends from March to October, thus coinciding with the period of high biological productivity. In most cases, a greater priority should be placed on dredging during periods of low productivity, since the size of the disposal site can be enlarged to compensate for the minimal dewatering of dredged material that would result from dredging during the colder, less productive periods of the year.

Mitigation of impacts of a more sociological nature that are associated with dredged material disposal, including odor, mosquitoes, fear of pathogens, land use, and aesthetics, are addressed in Ezell (1978), Harrison and Chisholm (1974), and Harrison et al. (1976).

The type and degree of mitigation that might be required is very site-specific and dependent primarily on the characteristics of the sediments at the site. However, one of the most effective and least expensive ways to minimize the environmental impact of dredging at nearly all hydro sites is by scheduling operations for the period from late summer to early spring. Another method having applicability to most sites would be the use of silt curtains. Major mitigation efforts directed at minimizing effects from highly contaminated sediments could involve both the type of dredge used and the type and ultimate fate of the disposal site. Such mitigation could be very costly and must be determined on a case-by-case basis.

If fish or mussel species listed as threatened or endangered (on either state or federal lists) are present in the reservoir or in the river below the dam, then dredging operations should be undertaken only after thorough reconsideration of the environmental costs. The presence of freshwater mussels is indicative of an ecosystem that may be particularly sensitive to high levels of suspended solids because both the mussels themselves and the fish species that serve as hosts to certain life history stages of mussels are not tolerant of silt (Yokley 1976).

4.2. ENVIRONMENTAL REGULATIONS RELATED TO DREDGING AND DREDGED MATERIAL DISPOSAL

4.2.1 Regulation of Dredged Material Disposal

Section 404 of the Federal Water Pollution Control Act Amendments (FWPCAA) of 1972 established a permit program to regulate the discharge of dredged or fill material into U.S. waters or wetlands. Although the U.S. Army Corps of Engineers (COE) has primary responsibility for the permit program, it is administered by both the COE and the U.S. Environmental Protection Agency (EPA). Developers of small hydro projects that will require the disposal of dredged material or the placement of fill or any material in a stream, such as would be necessary, for example, in the construction of a powerhouse at any existing dam (Corso 1979), must obtain a Section 404 permit from the District Engineer having jurisdiction over the waters in which the activity is proposed (Wood and Hill 1978). Permit approval by the District Engineer must comply with the guidelines that were established by the EPA (in conjunction with the COE) to implement Section 404(b) of the FWPCA (U.S. Environmental Protection Agency 1975). As provided for in the guidelines, these two agencies will publish procedures manuals to be used for the evaluation of proposed discharges of dredged or fill material to navigable waters. In the meantime, interim guidance to permit applicants concerning the applicability of specific approaches or procedures to be used in conducting an ecological evaluation of proposed dredged material discharges is available from the District Engineer (U.S. Army Corps of Engineers 1976a).

The COE has stated that the interim guidance is not intended to establish standards or rigid criteria, but rather it attempts to provide a balance between the technical state-of-the-art and routinely implemental guidance for using the procedures described in U.S. Environmental Protection Agency (1975). Procedures presented in the Interim Guidance Manual are to be used to evaluate (1) the discharge and overflow from hopper dredges and bottom- or end-dump barges and scows, (2) the discharges of hydraulic dredges, and (3) the runoff, effluent or overflow from a contained land or water disposal area (U.S. Army Corps of Engineers 1976a).

Dredging operations at small hydro sites may require other permits in addition to certification under Section 404 of the FWPCA. Because each discharge of dredged or fill material into navigable water is, in effect, the discharge of a pollutant to the water, a state water quality certification under Section 401 of the FWPCA would be required (U.S. Environmental Protection Agency 1975). Because this provision has been incorporated in the Corps of Engineers regulations (U.S. Department of Defense, Corps of Engineers 1975), any state may cause the denial of a Section 404 permit if the Section 401 permit is denied. Also, some states have existing regulations governing the same types of activities that are regulated by Section 404 of the FWPCA. Thus, if a state denies a permit, the COE will not issue a Section 404 permit (U.S. Environmental Protection Agency 1975). Finally, discharges of pollutants into navigable waters, resulting from the subsequent onshore processing of dredged material that is extracted for any commercial use (other than fill), are not included in the definition of the term 'discharge of dredged material' in the regulations of the U.S. Environmental Protection Agency (1975). Such discharges are subject to Section 402 of the FWPCA, even though extraction of such material may require a permit from the COE under Section 10 of the River and Harbor Act of 1899 (U.S Environmental Protection Agency 1975).

The disposal of dredged material that is highly contaminated by PCBs or other hazardous wastes may be controlled by the EPA under regulations promulgated pursuant to Section 6(e)(1) of the Toxic Substances Control act (TSCA) and the Solid Waste Disposal Act as amended by the Resource Conservation and Recovery Act (RCRA) of 1976. Final regulations relating to the disposal of PCBs were issued by the EPA on February 17, 1978 (U.S. Environmental Protection Agency 1978a). If the dredged material contains 0.05% or greater of PCB chemical substances, on a dry weight basis, it can be defined as a PCB mixture and must be disposed of in an incinerator, a chemical waste landfill, or in a manner determined by the Regional Administrator in the EPA region in which the PCB mixture is located [40 CFR 761.10(b)(4)]. Proposed guidelines and regulations for hazardous wastes were issued by the EPA on December 18, 1978 (U.S. Environmental Protection Agency 1978b). Certain dredged material may prove to be hazardous and therefore subject to these regulations (U.S. Environmental Protection Agency 1978b). The EPA has noted, however, that little information is available on not only hazard levels and potential threats to human health and the environment associated with onland disposal of these wastes but also acceptable waste management techniques and economics. Consequently, no decision has been reached on how these wastes should be managed, and comments are invited. The two alternatives considered by the EPA are (1) designation of dredged material as a special waste under Section 250.46 of the RCRA, or (2) exemption of these wastes from RCRA requirements and regulating them under Section 404 of the Clean Water Act.

Small hydro projects will likely require a Section 404 permit regardless of whether or not extensive dredging to increase reservoir storage is required. Developers should, therefore, become familiar with the Section 404 permit program during the initial stages of project development. The Interim Guidance Manual (U.S. Army Corps of Engineers 1976a) and the guidelines/regulations promulgated by the EPA in conjunction with the COE for evaluating proposed discharges of dredged and fill materials to navigable waters and wetlands (U.S. Environmental Protection Agency 1975) should be reviewed. The best introduction to the permit program, however, is a pamphlet published recently by the EPA and entitled "A Guide to the Dredge or Fill Permit Program" (U.S. Environmental Protection Agency 1979). Both the COE and the State permit review processes are outlined and discussed. Specific information on procedures to follow when applying for a Section 404 permit is provided in the COE regulations (U.S. Department of Defense, Corps of Engineers 1975) related to the issuance of permits for activities in navigable waters or ocean waters [see 33 CFR 209.120(f)].

4.2.2 Protection of Wetlands

Wetlands are important national resources that are declining at an alarming rate. The United States has already lost 40% of the 49×10^6 ha (120×10^6 acres) of wetlands inventoried in the 1950's (Executive Order 11990, May 24, 1977). In his Environmental Message to Congress, President Carter reported that wetlands are currently being lost at the rate of 121,000 ha (300,000 acres) per year (Miller 1977). Legislation passed during the past 50 years provided for the preservation of these areas through direct acquisition (Migratory Bird Conservation Act of 1929, the Migratory Bird Hunting Stamp Act of 1934 and amended in 1958, and the Water Bank Act of 1973). Another more recent strategy employed to preserve wetlands is the enactment of legislation making it unlawful, except as provided, to destroy specific wetlands (Miller 1977). Important wetlands legislation includes Executive Order 11990 (Protection of Wetlands), the River and Harbor Act of 1899 (Section 10), and the Federal Water Pollution Control Act Amendments (FWPCAA) of 1972 (Section 404).

Dredging operations at small hydro sites would not be affected by Executive Order 11990, since the Order "does not apply to the issuance by federal agencies of permits, licenses, or allocations to private parties for activities involving wetlands on non-Federal property" (Executive Order 11990, Section Ib). The U.S. Army Corps of Engineers administers the only nationwide regulatory program that controls development activities in U.S. waters and wetlands, and the authorities for this regulation are Sections 10 and 404 of the River and Harbor Act of 1899 and the FWPCAA of 1972, respectively (Wood and Hill 1978).

Considerable emphasis has been placed on the potential impacts of construction activities in wetlands. Darnell (1976) concluded that the most important impact of construction activity upon aquatic environments is wetland habitat loss. Wetlands guidelines were published by the U.S. Fish and Wildlife Service in 1971 to discharge its responsibility under the Fish and Wildlife Coordination Act. These guidelines required that an applicant seeking a federal permit involving the alteration or destruction of valuable wetland areas had to show, inter alia, that there were no alternate upland sites available. Moreover, spoil and dump sites were included in a list of structures, facilities, or activities that were considered unacceptable in wetlands (Wood and Hill 1978). The EPA guidelines for evaluating proposed discharge of dredged or fill material in navigable water (U.S. Environmental Protection Agency 1975) point out that "from a national perspective, the degradation or destruction of aquatic resources by filling operations in wetlands is considered the most severe environmental impact covered by these guidelines" [40 CFR 230.4-1(a)(1)]. Finally, Klock (1979) stated that small hydropower development in New England would conflict with at least one of several water resource management policies of the New England Water Basin Commission (NEWBC). Dredging at small hydro sites could potentially conflict with the NEWBC policy on maintenance and enhancement of wildlife habitat, one of the most important of which is wetlands.

An excellent general reference on wetland ecosystems is Good et al. (1978). The reader should consult Darnell (1976) for additional information on the nature of wetland impacts or Wood and Hill (1978) for a more detailed discussion of the regulatory role of the U.S. Army Corps of Engineers with regard to wetland protection policies.

4.3 ECONOMIC COSTS ASSOCIATED WITH DREDGING AND DREDGED MATERIAL DISPOSAL

Dredging to increase reservoir storage capacity may place a substantial economic constraint on the development of small hydro projects. Although the evaluation presented in this section is preliminary and is not intended to be the final word with regard to the economic feasibility of dredging, developers should carefully consider the factors that will ultimately determine the feasibility of reclaiming reservoir storage in this manner. Much of this information may also be applicable to small-scale dredging operations that will occur, for example, during construction and/or repair of structures such as powerhouses, spillways, and penstocks. Because these latter activities will generally require the removal of substantially less bottom material, the economic costs are likely to be minor. Only if the sediments to be dredged contain high concentrations of various contaminants will the costs of dredging and disposal of small volumes of material be a serious economic constraint on project development. The direct costs that will determine the economic feasibility of dredging and disposal are those associated with (1) acquisition of land for the disposal area; (2) construction of dikes and installation of weirs at the disposal site; (3) mobilization and demobilization of dredging equipment; (4) operation of the dredge, including materials, fuel, and labor; (5) transport of the sediments from the dredging site to the disposal area; and (6) acquisition of all necessary state and/or federal permits.

The most expensive aspect of confined dredged material disposal can be land acquisition (U.S. Army Corps of Engineers 1979b). In most cases, the best disposal site from an economic standpoint is one located adjacent to the reservoir, since no transportation costs would be incurred if the dredged material could be pumped directly to the disposal site. Land adjacent to the reservoir may be more valuable than land located some distance away. Waterfront property may be subject to economic returns as industrial or residential sites or may be a valuable wetland area of high biological productivity (Pope 1976). Costs will depend not only on the present use or potential future uses of this land but also on the amount of land needed. The actual acreage required for disposal will be determined by both the physical and chemical characteristics of the sediment and the volume of material to be dredged. Various methods are available to dewater and densify dredged material to increase the storage capacity of the disposal site and thus reduce or minimize the area required for disposal (see Haliburton 1978, Palermo et al. 1978).

Confining dredged material behind dikes or levees is approximately 2.5 times more expensive than open-water disposal (Morris 1974 as cited in Brady 1976). The COE had estimated that replacing open-water disposal with confined disposal of dredged material from the Great Lakes would increase the cost of dredging by a factor of 3.5 (U.S. Comptroller General 1972). Obviously, costs are site-specific, and in some cases, confined disposal is less expensive than open-water disposal if the latter involves long haul distances (C. Calhoun, personal communication). The size of the dikes and weirs will be affected by the amount of settling time required which, in turn, is dependent upon the nature and volume of the dredged material. Additional costs will be incurred if it is necessary to install a liner in the basin to prevent leaching and possible contamination of groundwater.

High initial costs are associated with transporting the dredging plant to and from a project (Pearce 1976). For small hydro projects where considerably smaller equipment (e.g., portable dredges) might be used, the high costs of mobilization and demobilization could be reduced. Obviously, a major determinant of these costs will be the distance the equipment must be moved. The regional distribution of the private dredging fleet in 1972 showed 65% of the 264 hydraulic cutterhead dredges to be located on the East (37%) and Gulf (28%) Coasts, but less than 30% of the 161 clamshell dredges were located in these areas (15 and 14%, respectively). For additional information on the regional distribution of contractor dredges, see Murden and Goodier (1976). The costs of such pre-dredging preparatory activities as dike construction, installation of weirs, and placement of the pipeline can also be included as part of the mobilization and demobilization costs (Pearce 1976).

Dredging costs vary widely depending upon the type of dredge used (and the disposal method) and the volume of material that is dredged.

Mechanical dredging involves a greater cost than hydraulic dredging due to lower production rates and the necessity of rehandling and transporting the dredged material. For example, in the St. Paul District, the COE can move material by a hydraulic cutterhead dredge at a cost of $\$0.43/m^3$ ($\$0.33/yd^3$) compared with $\$2.05/m^3$ ($\$1.57/yd^3$) if a mechanical clamshell dredge is used (U.S. Army Corps of Engineers 1976b). The size (capacity or pumping rate) of the dredge will also affect the costs per unit volume. Carline and Brynildson (1977) reported that (1) projects utilizing a hydraulic cutterhead dredge with a 15-cm diameter intake were considerably more costly than those in which a 20-cm dredge was used, and (2) there was, in general, an inverse relationship between increases in pond volume (or amount of material dredged) and unit costs.

The cost per unit volume of several dredging projects are presented in Table 5. In extrapolating from these values, two important facts should be kept in mind. First, the costs shown in the table do not include the purchase of land for the dredged material basin(s). Second, with the exception of the Long Lake restoration project in Michigan, the dredging equipment was privately owned and operated; i.e., the work was not performed under a contract to a private dredging firm. If these data are used directly (after conversion to present-day dollars) to estimate the cost of a small hydro dredging operation (that would likely require outside contracts for the work), the estimates may be low. Furthermore, if the dredged material cannot be pumped directly to the disposal area, as was probably done in the majority of the projects listed in Table 5, then

Table 5. Comparison of costs (dollars/m^3) of various dredging projects. N/A = No information available.

Type of dredge used	Discharge diameter (cm)	Dredge owner and operator	Purpose of project	Location	Surface area of lake/reservoir (ha)	Year(s) dredged	Volume dredged (m^3)	Type of disposal	Cost per m^3	Reference	Comments
Cutterhead pipeline	15	DNR[a]	Pond restoration	North-central Wisconsin	0.4	1970	5,275[b]	Onshore adjacent to pond	$0.73[b]	Carline and Brynildson (1977)	(a) Costs include materials, fuel, and all labor for area construction, dredge operation, and supervision. (b) Depreciation of dredge was based on $10/h of operating time. (c) Costs represent the money required to increase pond volume by 1 m^3 and do not include easements or outright purchase of land.
	15	DNR[a]	Pond restoration	North-central Wisconsin	0.4	1971	4,610[b]	Onshore adjacent to pond	$2.50[b]		
	15	DNR[a]	Pond restoration	Wisconsin (7 ponds)	N/A	N/A	2,240-24,390[b]	N/A	$2.68-$0.53[b]		
Cutterhead	30	Private contractor	Lake restoration	Eastern Michigan	59.1	1961-65	841,000	Diked onshore area (spoils covered 22.7 ha of land)	$0.22	Spitler (1973)	(a) No detailed information available on location of spoils area except that one "was selected away from lake." (b) Volume dredged computed as difference between pre- and post-dredging water volumes.
Mud Cat	15	City of Oakland	Reclamation reservoir storage for water supply	East-central Illinois	10.5	1972-75	72,784	Confined in 6-ha area adjacent to lake	$0.99	Roberts (1976)	(a) Volume dredged is dried sediment; volume of the slurry was 255,701 m^3. (b) Cost includes amortization of dredging equipment over 10-year period.
Sidecasting dredge (Merritt) with two hopper barges (239-m^3 capacity)	30	COE	Maintenance of navigation channels through coastal inlets	South Atlantic coast	-	1975	285,347	Open water (beach zone)	$2.33	Sanderson (1976)	(a) Cost for hauling only (with two barges) was $1.16/m^3. (b) Cost in 1975 dollars.

[a]Wisconsin Department of Natural Resources.

[b]Values derived from Figure 17 of Carline and Brynildson (1977).

substantial additional costs to transport the material could be incurred. In all likelihood, transportation costs alone would exceed the costs of actual dredge operation (Table 6). Because disposal costs will be very site-specific, the maximum distance that could occur between the dredging and disposal sites without affecting the economic feasibility of dredging to reclaim lost storage capacity, can not be estimated. For additional information on long distance transport of dredged material, see Souder et al. (1978).

Estimated costs to dredge many of the lakes included in the EPA's Lake Restoration Program are presented in Peterson (1979). These costs, which do not include activities such as dike construction, treatment of return water, mobilization, etc., are based on proposal estimates and bids, since most of the projects have not been completed. The estimates ranged from \$0.89/m^3 (\$0.68/yd^3) for the removal of 78,794 m^3 (103,059 yd^3) of sediment from a 21-ha (52-acre) lake with a Mud Cat dredge to \$13.73/m^3 (\$10.50/yd^3) to remove 12,682 m^3 (16,588 yd^3) of sediment from a 1.9-ha (4.7-acre) lake with a bulldozer. The latter project includes the removal of dredged material to a remote location by watertight tank trucks. Both sediment removal from freshwater lakes and dredging activities on navigation projects were, on the average, 2 to 4 times more costly in the Northeast compared to other regions of the country (Peterson 1979).

Long transport distances can also affect the economics of dredging by affecting the production rates of hydraulic dredges. With these dredges, transport distance is limited by dredge pump and auxiliary pump power and production, the latter of which will decrease with

Table 6. Comparison of costs (dollars/m^3) to transport 382,275 m^3 (500,000 yd^3) of dredged material for varying distances and with various transport systems. Costs adjusted to March 1978 dollars. NF = not economically feasible. Source: Spaine 1978.

Transport distance (km)	Transport system				
	Pipeline	Rail	Barge	Belt conveyor	Truck
16	3.23	NF	3.23	11.74	5.98
32	4.11	NF	4.11	19.82	8.65
161	12.48	9.39	6.16	NF	17.91
400	NF	12.19	9.69	NF	NF

pipeline length (Mohr 1976). Mechanical dredge production, on the other hand, is independent of transport distance because variations in distance can be adjusted by changing the number and sizes of the transport plant.

The relationship between production rate and economic cost is obvious. At high rates of production, the time and effort expended on a project are minimized. However, transport distance is not the only factor affecting production rates. In addition to the type and size of dredge used, the nature of the bottom material will also affect production rates (Gren 1976, Pearce 1976). For example, a 69-cm cutterhead dredge can normally produce 115 m^3/h (150 yd^3/h) in blasted rock to nearly 1529 m^3/h (2,000 yd^3/h) in mud and soft clays (Gren 1976).

Finally, dredging will require at least one or more permits, and there will be some costs associated with their acquisition. The cost will vary depending upon the nature of the project and the state in which the site is located. Assuming that no enviromental impact statement is required but that an initial survey and some monitoring during the dredging operation would be required, the costs to meet all applicable regulations for a minor (7646 m^3 or 10,000 yd^3) and major (382,300 m^3 or 500,000 yd^3) dredging project in the state of California was estimated to be approximately $12,000 and $125,000, respectively (Boerger and Cheney 1976). The authors present a detailed breakdown of these costs, including estimates of the time required for various steps in the permit application process, if additional information on the economic impact of dredging regulations is desired.

Increasing the storage capacity of reservoirs could provide developers of small hydro projects with the option of operating the facility in a daily peaking mode, thus providing more firm energy with a higher economic value than the less dependable energy produced if the project were operated as a run-of-river facility. Increased pondage would also enable a greater range of river flows to be utilized. Consequently, the number of operating hours per day could be increased. Several questions must be addressed in evaluating the benefits of increasing reservoir storage capacity. How much additional storage capacity can be obtained? How often will this capacity (or volume of water) be available? What is the value (mils/kWh) of this additional volume of water? Although the answers to these questions can only be determined on a site-specific basis, several aspects of the economic feasibility of dredging at small hydro sites are worth noting.

Forty-eight projects involving 82 sites were evaluated to determine their feasibility for electrical power generation (U.S. Department of Energy 1979c). Twenty-two of these sites (27%) were not found to be economically feasible. Of the 41 sites that had a generating potential estimated to be $\lesssim$ 1 MW, 18 (44%) were not feasible compared to only 14% of the sites with capacities ranging from 1.1 to 5.0 MW. All of the sites with capacities greater than 5.0 MW (15% of the 82 sites) were economically feasible. Thus, the majority of the sites that were not feasible for development of small hydroelectric power generation were $\lesssim$ 1 MW, and all of these sites were found to be infeasible due to economic reasons.

The cost of developing an economically feasible small hydro site with a generating potential of less than 1 MW ranged from $175,000 (for a 0.25-MW site) to $1,337,000 (0.44 MW) (U.S. Department of Energy 1979c). The major cost associated with small-scale hydroelectric development is the initial capital investment, including structure costs and the cost of generating machinery (U.S. Department of Energy 1979b). If additional civil works are required, the impact on cost can be major (O'Brien et al. 1979). For sites with a potential of less than 1 MW, it is doubtful that the benefits derived from dredging (i.e., an increase in the amount of firm energy produced as a result of the increased storage capacity in the reservoir) would outweigh the costs. Since the amount of energy that could be produced depends on several site-specific factors, no economic evaluation of the benefits of dredging was attempted.

It should be noted, however, that added benefits could be derived if a market for the dredged material could be located. A majority of the sites will provide good soil base material with excellent agronomic characteristics (R. M. Engler, personal communication). By selling the dredged material or leasing the disposal area for agricultural uses, the social costs associated with disposal can actually present an opportunity for economic gain (Pope 1976). Although benefits were realized from the dredging of Illinois water supply reservoirs to reclaim lost storage capacity (Roberts 1976), it is doubtful that they would add significantly to the overall benefit/cost ratio for dredging at small hydro sites. The difficult question of sediment contamination would have to be carefully evaluated before the dredged material could be sold commercially. Another equally important potential constraint is the availability of a transportation system at reasonable cost (Pope 1976). Because transport costs are high (see Table 6), the commercial sale of dredged material may not be economically feasible. For additional information and guidance on productive uses and land improvement techniques associated with dredged material disposal areas, see Spaine et al. (1978) and Walsh and Malkasian (1978).

In summary, the decision to dredge an impoundment to increase storage capacity must be made carefully. Because of the high costs of dredging and dredged material disposal, it is unlikely that such a method would be employed to reclaim lost capacity, especially at very small ($<$1-MW) projects. Even for larger projects, economic feasibility of large-scale dredging operations (e.g., $> 100{,}000/m^3$) might be difficult to achieve if a local disposal area cannot be found. Other less expensive methods exist to reclaim reservoir storage capacity, including the use of flashboards to raise the height of the dam. Since the greatest storage is in the upper level of the impoundment, the addition of one foot of storage above the spillway often compensates for the lifetime loss of volume caused by sediment accumulation (Roberts 1976). Although adequate mitigation currently exists to minimize the environmental effects of dredging and dredged material disposal, the cost of this mitigation may place a significant economic constraint on the use of dredging as a means of increasing storage capacity.

4.4 GUIDELINES FOR EARLY EVALUATION OF THE ENVIRONMENTAL FEASIBILITY OF DREDGING

If the development of a small hydro site includes the need for dredging, regardless of the scale of the operation (i.e., extensive dredging in order to increase storage capacity or minor, short-term dredging for clearing intakes or construction and/or repair of the powerhouse, dam, or spillway), then an evaluation of the potential environmental impacts of dredging and dredged material disposal should be conducted during the initial stages of project planning and development. Such an evaluation, for example, should be included in the feasibility study. Moreover, the assessment should be conducted at a level that is commensurate with establishing the magnitude of the problem. After assessing the magnitude of the potential impacts, a more detailed evaluation can be presented in the application for a license from the Federal Energy Regulatory Commission.

Possibly the most critical questions that must be addressed during any evaluation of the environmental effects of dredging concern (1) the quantity of material that will be removed and the extent of the area to be dredged and (2) the degree of contamination of the dredged material, especially the mobility and bioavailability of such potentially toxic constituents as heavy metals and various chlorinated organic compounds, especially PCBs and pesticides. Factors such as the quantity and composition of the dredged material will be important determinants of the magnitude of the environmental impacts associated with dredging and dredged material disposal and, in turn, will determine the measures required to mitigate these impacts.

Estimation of the quantity of sediment to be dredged will be based on a number of site-specific factors which the individual developer considers important. The compositional characteristics of the sediments will reflect both historical and present land-use practices in the watershed. Land use at the 56 sites where feasibility studies were conducted is typically a mixture of low-density residential and low-intensity agriculture (U.S. Department of Energy 1979a). If the river basin is heavily industrialized, then the potential for significant sediment contamination exists. All available local, state, and federal sources of both water quality and sediment data should be searched in an attempt to establish an historical inventory of potential contaminants. Available sources include the water quality monitoring programs conducted by the U.S. Geological Survey, the EPA, and the State, as well as NPDES compliance monitoring programs conducted by various public and private industries as required under Section 402 of the FWPCA. Depending upon the availability of these types of information, it may be necessary to determine the chemical composition of the sediments at the site and the potential for impact by the Elutriate Test (U.S. Environmental Protection Agency 1975). Bulk sediment analysis, however, should not be used to predict the impacts of dredging and disposal operations (Brannon 1978, Lee and Plumb 1974, Lee et al. 1975).

The permitting authority may also require that bioassays be performed to evaluate potential impacts due to both the physical presence of suspended particulates and any biologically active contaminants associated with the particulate and/or dissolved fractions. Furthermore, such biological evaluations must also include an assessment of the bioaccumulation potential of the contaminants in the dredged material (Engler, in press). For additional information on how and when these tests should be conducted and a discussion of the factors influencing test results, see Engler, in press; Fulk et al. 1975; Lee and Plumb 1974; Lee et al. 1975; Plumb 1976; Shuba et al. 1977; and U.S. Environmental Protection Agency 1975.

When the quantity of sediment to be removed has been estimated and data on the physical and chemical characteristics of the sediments have been obtained, the difficult task of how the material should be dredged and disposed of must be undertaken. The information presented in Sections 2 and 3 should be helpful in this regard. An excellent guidance manual (Gambrell et al. 1978) exists to assist in selecting disposal alternatives for highly contaminated dredged material to minimize adverse environmental effects (see also Palermo et al. 1978). If the dredged material is to be used for other purposes, several references exist which present a list of the procedures to follow in evaluating various alternative uses. For example, see Lunz et al. (1978) and Smith (1978) if the dredged material will be used for habitat development, or Spaine et al. (1978) and Walsh and Malkasian (1978) if various land improvement alternatives (e.g., landfill and construction material, surface mine reclamation, sanitary landfill, and agricultural land enhancement) are to be evaluated. A site-specific evaluation of potential upland disposal sites should consider many factors, and these are outlined in Chen et al. (1978).

Guidance in assessing the environmental impacts of dredging and dredged material disposal is provided in the interim guidance manual

(U.S. Army Corps of Engineers 1976a). The regulations discussed in Section 4.2 of this document also contain valuable information related to the procedures involved in evaluating the environmental effects of dredging and dredged material disposal (e.g., see U.S. Department of Defense, Corps of Engineers 1975; U.S. Environmental Protection Agency 1975). An excellent source of information on the dredged material permit program is the pamphlet recently published by the EPA (U.S. Environmental Protection Agency 1979).

During the period when initial decisions are made regarding (1) method of dredging and type of disposal; (2) location, size, and structural characteristics of the disposal basin; (3) ultimate fate of the dredged material; (4) mitigation or environmental control measures that will be employed; (5) quantity of materials to be dredged and area(s) of the impoundment where dredging will occur; and (6) time of year and duration of the dredging operations, a dialogue should be established with appropriate personnel in local, state, and federal agencies. Initiating contacts with those agencies responsible for assessing environmental impacts and with the general public is an important aspect of project development (Corso 1979, U.S. Environmental Protection Agency 1973, Marker 1978). Local, state, and federal agencies with jurisdiction over placement of waste, water quality, zoning, and other environmental issues should be contacted for laws and policies on land activities related to a specific dredged material containment plan (Spaine et al. 1978). Furthermore, since legal constraints may be imposed on land application of solid wastes, all aspects of the environmental impact of land application of dredged material should be addressed, including sociological impacts (e.g., fear of odors or high levels of toxic substances), land-use and aesthetic impacts, economic impacts (i.e., shift in land values), and public health impacts (e.g., impacts on groundwater quality, chemical contamination of crops) (Harrison and Chisholm 1974). Because these impacts are most often the issues of greatest concern to the public, all attempts should be made to open effective channels of communication with the public in the early stages of project development.

5. CONCLUSIONS AND RECOMMENDATIONS

Retrofitting existing small dams for hydroelectric power generation ($\lesssim$15-MW capacity) may require dredging in order to (1) reclaim lost storage capacity in the reservoir, (2) clear intake structures or penstocks, and (3) construct or repair powerhouses, headraces, and/or tailraces. Using the extensive literature available on the impacts associated with dredging and dredged material disposal operations, especially the results from the U.S. Army Corps of Engineers Dredged Material Research Program, an analysis of the environmental issues associated with dredging at small hydro sites was performed. The conclusions listed below are based on the results of this analysis.

1. Hydraulic cutterhead dredging with confined upland disposal are likely to be the methods used at the majority of the sites. Very few projects will involve extensive dredging to reclaim lost storage capacity in the reservoir. The high costs (relative to the cost of the entire project) associated with the removal of large quantities of sediment and the need for large disposal areas near the reservoir make this type of operation a less attractive alternative to the installation of flashboards to increase the height of the dam.
2. The major environmental issues (or impacts) associated with dredging and disposal operations at small-scale hydropower re-development projects will be (1) increased levels of suspended solids and downstream siltation, (2) substrate removal, and (3) deleterious chemical changes in the water masses and sediments.

3. At most sites, the biological effects resulting from elevated suspended solids concentrations and substrate removal either will not be significant or can be easily mitigated (e.g., use of silt curtains or specialized dredging equipment; dredging during periods of low biological productivity).
4. Downstream siltation will have the greatest effect on filter-feeding species, especially mussels, that inhabit areas immediately below the dam. Although siltation can be reduced by various mitigative measures, the existence of threatened or endangered species may preclude any extensive dredging operations in the reservoir.
5. Toxicity and bioaccumulation of various contaminants (heavy metals, chlorinated hydrocarbons) constitute the most important potential biological effects caused by chemical changes in the water masses and sediments. Although lethal effects due to short-term exposures to these contaminants may be minimal, long-term chronic exposures could result in their bioaccumulation in the tissues of aquatic biota at those sites with highly contaminated sediments. The magnitude of these effects will be site-specific and dependent upon the type of dredging and disposal operation, the physical and chemical nature of the sediments, and the mobility and availability of these contaminants to biota. The levels of dissolved oxygen, un-ionized ammonia, and hydrogen sulfide produced during dredging will also be site-specific but, in all likelihood, would only be acutely toxic to the early life history stages of some of the more sensitive species inhabiting the site.

It should be emphasized that the changes in water quality resulting from dredging and dredged material disposal will be influenced by many site-specific factors, as discussed in Section 3. Consequently, to generalize about the magnitude of the resultant biological effects is difficult, and the conclusions stated above must be considered in this light. Obviously, however, these effects will be of greatest concern at those sites where high levels of contamination exist and extensive dredging to increase reservoir storage capacity is proposed.

Because of the site-specific nature of the conclusions reached from our analysis of the environmental issues related to small-scale hydropower development, the task of making specific recommendations is equally difficult. Some general recommendations applicable to most sites are listed below.

1. Dredging during the period of high biological productivity (usually spring through mid-summer) should be avoided.
2. Turbidity and downstream siltation should be minimized, especially if:
   a. the impoundment is small and a large portion of it will be dredged;
   b. highly contaminated sediments are present;
   c. dredging is not conducted during periods of low productivity and/or if sensitive life stages/species are present; and
   d. dredging does not coincide with the period of high sediment loading to the river.

3. Downstream siltation should be prevented if threatened or endangered mussel species inhabit areas below the dam.
4. Confined upland disposal areas should be designed and constructed to minimize the levels of suspended solids in the effluent returning to the water body. The presence of very fine particulates may require very long detention times or the use of flocculants.
5. The disposal of dredged material in wetland areas should be avoided.
6. Information on historical land use in the watershed, including present industrial and municipal effluent sources, should be obtained during the feasibility study. The availability of water quality data for the watershed should be investigated. If no prior data on the chemical composition of the sediments exist, then an inventory of the sediments to be dredged should be considered. Bulk sediment analysis, however, should not be used to predict the impacts of the proposed dredging and disposal operations.
7. Prior to dredging, chemical changes in the water column should be assessed using the Elutriate Test and the results compared to appropriate water quality criteria. If the results indicate tht contaminants will be released to the water column during dredging, then:
   a. bioassays should be performed using sensitive species that inhabit the site, and
   b. an evaluation of bioaccumulation potential should be conducted.
8. All components of the operation, including excavation, transportation, and disposal must be considered as a total integrated system; the best dredging operation may not be

compatible with the best disposal operation (Barnard 1978). Because many of the biological effects from dredging and dredged material disposal are the result of site-specific changes in sediment and water chemistry, appropriate mitigation must also be considered on a site-specific basis.

At most sites and for most of the issues discussed in this report, adequate mitigation exists. The cost of this mitigation may, in some cases, be very high. For example, if the sediments are highly contaminated, specialized dredging equipment as described in Barnard (1978) may need to be used. The location of disposal areas at some distance from the dredging site will result in high transportation costs. Thus, a careful evaluation of the dredging operation from both an economic and an environmental standpoint is critical and should be done during the early stages of project development, preferably during the feasibility study.

6. LITERATURE CITED

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APPENDIX A

PHYSICAL, CHEMICAL, AND BIOLOGICAL PROPERTIES OF CHLORINATED HYDROCARBONS

PHYSICAL, CHEMICAL, AND BIOLOGICAL PROPERTIES OF CHLORINATED HYDROCARBONS

Chlorinated hydrocarbons, such as polychlorinated biphenyls (PCBs) and various pesticides, particularly the organochlorine insecticides (e.g., DDT, dieldrin), are major contaminants of natural waters. Their widespread distribution in water, sediments, and biota throughout the United States, especially in the large river basins east of the Mississippi River, has been well documented (e.g., Dennis 1976, Walker 1976a). In field investigations of 11 confined disposal sites, Chen et al. (1978) reported that 99% of the chlorinated hydrocarbons in all samples were DDT (and its derivatives) and PCBs. Sampling at five open-water disposal areas indicated that the most abundant and widely distributed chlorinated hydrocarbon was Aroclor 1254, a PCB mixture (Fulk et al. 1975). Because of their widespread distribution, environmental persistence, and toxicity, PCBs and the organochlorine insecticides, especially DDT, have been intensely investigated. Consequently, the discussion that follows will focus on these compounds, although much of the information on DDT may also be applicable to many of the other organochlorine pesticides.

Polychlorinated biphenyls (PCBs)

Polychlorinated biphenyls (PCBs) are actually mixtures of chlorinated biphenyl isomers and are used primarily as insulating fluids for transformers and capacitors. Most PCBs are produced commercially by the Monsanto Company under the trade name Aroclor®. A given mixture (or Aroclor species) is identified by a four-digit number, the last two digits of which refer to the percentage of chlorine, by weight, in the mixture (except Aroclor 1016 which was only introduced in the early 1970's and contains 40%, by weight, of chlorine). All PCBs have a very low solubility in water, but the solubility decreases with an increase in the percentage of chlorine in the mixture. Also, the low chlorine compounds easily undergo microbial degradation (Baxter et al. 1975), whereas the compounds with the highest number of chlorine atoms are more chemically stable. Differences in the solubility and persistence of the various PCB mixtures have important biological implications as discussed below.

With few exceptions, PCB concentrations in the range of approximately 0.1 to 15 μg/liter were found to be toxic to many freshwater fishes and invertebrates (Table A-1). Exposures for longer periods of time ($>$ 96 h) reduced the LC_{50} values, thus pointing out the greater threat to biota from longer exposure periods and the inadequacy of using acute toxicity tests alone to evaluate potential adverse effects. A review by Nebeker (1976) indicated that invertebrates, especially newly hatched fish larvae, small insects, and crustaceans with short life cycles, are the most sensitive and susceptible to acute toxic effects. In larger animals or those with longer life cycles, toxic effects are delayed, and only the long-term orchronic bioassay adequately reflects the effects that could occur (Nebeker 1976). In addition to those sublethal effects described in Table A-1 (reproductive impairment and reduced growth), Cutkamp et al. (1972) found that chronic exposure of fathead minnows to 0.93 μg/liter of Aroclor 1242 for four months resulted in a 56% inhibition of mitochondrial Mg^{2+} ATPase activity in the kidney.

Table A-1. Acute and chronic toxicity of various PCBs to freshwater biota as determined from continuous-flow bioassays

Species	Aroclor® mixture	Concentration (μg/liter)	Effect
Daphnia magna[a] (water flea)	1248	2.6	50% mortality after 14 d
	1254	1.8	50% mortality after 14 d
	1254	1.3	50% mortality after 21 d
	1248	7.5	97% reproductive impairment after 14 d
	1248	2.1	50% reproductive impairment after 14 d
	1248	1.0	16% reproductive impairment after 14 d
	1254	3.8	100% reproductive impairment after 14 d
	1254	1.1	50% reproductive impairment after 14 d
	1254	0.48	16% reproductive impairment after 14 d
Gammarus pseudolimnaeus (scud)	1242[a]	29	50% mortality after 4 d
	1242[b]	5.0	50% mortality after 10 d
	1242[b]	10.0	50% mortality after 4 d
	1248[a]	73	50% mortality after 4 d
	1242[a]	8.7	52% mortality after 60 d
	1248[a]	5.1	53% mortality after 60 d
Ischnura verticalis[b] (damselfly)	1242	400	50% mortality after 96 h
	1254	200	50% mortality after 96 h
Oroconectes nais[b] (crayfish)	1254	80	50% mortality after 7 d
Tanytarsus dissimilis[a] (midge)	1254	0.65	50% larval mortality after 21 d
	1254	0.45	50% pupal mortality after 21 d
	1254	3.5	Number of larval and pupal cases were 7.4 and 6.8% of the controls, respectively
	1254	1.2	Number of larval and pupal cases were 35 and 18% of the controls, respectively
	1254	0.45	Number of larval and pupal cases were 52 and 55% of the controls, respectively
	1254	3.5	No emergence
Jordanella floridae[c] (flagfish)	1248	2.2	15% mortality to newly hatched larvae after 40 d; mean weight was 70% of controls
	1248	5.1	65% mortality to newly hatched larvae after 40 d; mean weight was 14% of controls
Pimephales promelas[c] (fathead minnow)	1242	15.0	50% mortality to yolk-sac larvae (<24 h old) after 4 d
	1254	7.7	50% mortality to yolk-sac larvae after 96 h
	1242	300	50% mortality to juveniles (3 months old) after 96 h
	1248	4.7	50% mortality after 30 d
	1254	>33	50% mortality to juveniles (2 months old) after 96 h
	1254	1.8	Spawning occurred but was significantly less than that at lower concentrations; hatching and fry survival good
	1260	3.3	50% mortality after 30 d
Salvelinus fontinalis[d] (brook trout)	1254	6.2	50% mortality after 128 d

[a]Experimental temperatures were 18 ± 1°C. Tabular data obtained from Nebeker and Puglisi (1974).

[b]Experimental temperatures were 15.6°C. Tabular data obtained from Stalling and Mayer (1972). *Gammarus fasciatus*, not *G. pseudolimnaeus*, used.

[c]Experimental temperatures were 24 ± 1°C. Tabular data obtained from Nebeker et al. (1974).

[d]Experimental temperatures were 12°C and 25°C for brook trout and fathead minnows, respectively. Tabular data obtained from Spehar et al. (1979).

Another important property of PCBs, in addition to their acute and chronic toxicity, is their tendency to accumulate in the tissues of aquatic biota to levels that greatly exceed the concentration of PCBs in the surrounding water. Because of their high solubility in lipids and their low solubility in water, PCBs are strongly partitioned from water into lipids (fats) of aquatic biota, resulting in large degrees of bioaccumulation or bioconcentration (U.S. Environmental Protection Agency 1977). Many freshwater invertebrates, can accumulate PCBs to appreciable concentrations over short periods of time, as shown by Sanders and Chandler (1972). For example, continuous exposure of late-instar mosquito larvae (Culex tarsalis) to water containing 1.5 ± 0.3 μg/liter of Aroclor 1254 resulted in total body residues of 19 μg/g after only 24 h, a 12,600-fold increase. The scud, Gammarus pseudolimnaeus, and the filter-feeding zooplankter, Daphnia magna, had concentrations of the same compound that were, respectively, 24,000 and 47,000 times greater than the ambient water concentration (1-2 μg/liter) after four days. Uptake from the water by D. magna approached a steady state at this time, whereas uptake by G. pseudolimnaeus reached a steady state after 14 days exposure (body residues were 44 μg/g, resulting in a concentration factor of 27,500), and no further accumulation was found with an additional 21-d exposure. Using the same species of scud, Nebeker and Puglisi (1974) found similar concentration factors (16,000-36,000) after 60-d exposure to Aroclor 1242 concentrations ranging from 2.8-26 μg/liter. Uptake of Aroclor 1248, on the other hand, resulted in tissue residues that were 28,000 to 108,000 times the ambient water concentration.

Studies of PCB uptake in fishes have shown that the steady state condition is not reached for several months or longer, depending upon the size (or age) of the species (DeFoe et al. 1978, U.S. Environmental Protection Agency 1977). Fathead minnows exposed to Aroclor 1242 for 8.5 months had whole body residues that were 32,000 to 274,000 times the concentration of the surrounding water, while similar tests with Aroclor 1254 resulted in even higher concentration factors (156,000-238,000). Whole body residues at the end of these tests were approximately 2 to 4 times higher than the residue concentrations present after one month (U.S. Environmental Protection Agency 1977). The significance of bioaccumulation in fishes was shown by DeFoe et al. (1978) who estimated that a 100-d exposure of fish to 0.004 μg/liter of Aroclor 1248 or 0.002 μg/liter of Aroclor 1260 in water could result in tissue concentrations of 0.5 μg/g.

Both the accumulation and retention of PCBs in the tissues of aquatic biota is dependent upon the nature of the isomers that exist in a particular mixture. The more highly chlorinated PCB mixtures such as Aroclor 1254, which consists primarily of tetra-, penta-, and hexachlorobiphenyls (21, 48, and 23%, respectively), not only are accumulated to a greater extent in the lipids of fish than are the less chlorinated isomers (e.g., DeFoe et al. 1978, Walker 1976b) but also are less easily degraded. Sanborn et al. (1976), using pure isomers to examine uptake and retention in green sunfish, found that trichlorobiphenyls could be degraded, but the tetra- and pentachlorobiphenyls, especially the latter, were much less susceptible to metabolism.

Many investigations of toxicity, uptake, and accumulation of PCBs by aquatic biota, especially those conducted in the early and mid-1970's, have led to a more complete understanding of the ecological effects of these toxicants. Although investigators have known for some time that uptake of PCBs by biota could occur from water, food, or sediments, recent studies have suggested that direct water exposures represent a greater hazard to fish than dietary exposures because direct uptake from water is more rapid and leads to a much higher accumulation in the tissues (Nebeker 1976, Stalling and Mayer 1972). Levels of PCBs in the tissues of aquatic organisms were found to be directly proportional to the concentration in the water for both invertebrates (Hansen et al. 1974, Nebeker and Puglisi 1974) and fishes (Kimura et al. 1973, Nebeker et al. 1974, Snarski and Puglisi 1976). The concentration factor, however, was independent of the concentration of PCBs in the surrounding water (e.g., Nebeker and Puglisi 1974, Nebeker et al. 1974, Hansen et al. 1974, DeFoe et al. 1978).

Although increasing the concentration of PCBs in the diet also led to higher whole-body residues (e.g., Fig. 2 and 3 in Walker 1976b), uptake from dietary sources led to concentrations in fish tissues that were usually less than an order of magnitude higher than the levels in the food. Long-term exposure (240 d) of coho salmon to Aroclor 1254 concentrations in food of 0.4 to 580 μg/g resulted in whole-body residues that were 0.9 to 5 times the exposure levels (unpublished data from Mehrle and Grant as cited in Stalling and Mayer 1972). Similarly, juvenile lake trout fed 0.1 and 6 mg/kg of Aroclor 1248 had whole body residues of 0.13 and 9.7 mg/kg, respectively, after 320 d (Meyer et al.

as cited in Walker 1976b). Finally, in a study by Schottger et al. (as referenced in Nebeker 1976), concentrations of 1.2, 3.8, and 12 μg/g of Aroclor 1248 were provided in the diet of lake trout for three months, resulting in weight gains that were 6, 10, and 28%, respectively, lower than that of controls.

Although the uptake of PCBs from dietary sources results in much lower concentration factors than were reported for the uptake of water, these sources may nevertheless be significant. Polychlorinated biphenyls have a high affinity for sediments and can readily enter the food chains (Nebeker 1976, Stalling and Mayer 1972). Although PCBs are usually present in natural waters at low concentrations, aquatic invertebrates are capable of accumulating PCBs to appreciable concentrations, thus exposing organisms at higher trophic levels (e.g., fishes) to significant amounts of PCBs via the food chain (Sanders and Chandler 1972). The uptake of PCBs from dietary sources alone, however, would be unlikely to result in whole body concentrations that have been found to be associated with mortalities in chronic continuous flow exposures (500-600 μg/g) (Stalling and Mayer 1972).

Finally, it should be noted that the bioaccumulation or concentration factors measured in laboratory studies may represent an underestimation of the degree of PCB accumulation that can occur in natural waters because (1) wild fish are exposed to PCBs in food and sediments in addition to water, (2) fish can accumulate PCBs over much longer exposure periods than those used in laboratory tests, and (3) the levels found in wild fish may reflect an integrated history of exposure due to the patchy distribution of PCBs in the environment

(Hansen 1976, U.S. Environmental Protection Agency 1977). Evidence presented at the EPA hearings on the proposed effluent standards for PCBs indicated bioaccumulation factors in the range of 1 to 10 million for fishes in Lake Ontario (U.S. Environmental Protection Agency 1977). In contrast to these findings, the review by Nebeker (1976) discussed the results of two field studies and suggested that the bioconcentration factors measured in the laboratory are essentially the same as those found in the river environment.

DDT and other organochlorine insecticides

Many of the organochlorine insecticides have physical and chemical properties similar to those described previously for PCBs. Both groups have relatively low solubilities in water but are highly soluble in organic solvents (e.g., lipids), are resistent to degradation (to nontoxic compounds), and are readily adsorbed onto particulate matter (Hamelink and Waybrant 1976). As a result, many of the biological properties of these insecticides (e.g., acute and chronic toxicity, bioaccumulation) are similar to those of PCBs. A brief summary of these biological properties follows.

Most of the organochlorine insecticides are toxic to fish and freshwater invertebrates at concentrations in the same range as that for PCBs (Table A-2). Endrin, toxaphene and DDT, however, are generally more toxic to both invertebrates and fishes than PCBs, including the more highly chlorinated mixtures (Table A-1). Acute toxicities of DDT and Aroclor 1254 differed by more than order of magnitude in studies with Daphnia (Maki and Johnson 1975) and coho salmon fry (Halter and Johnson 1974). Moreover, the organochlorine insecticides, as a group, are generally more acutely toxic than many other insecticides and herbicides. Macek and McAllister (1970) determined the acute toxicity of three groups of insecticides to 12 fish species. The organochlorine compounds were generally 10^2 to 10^3 times more toxic than the organophosphate compounds and 10^3 to 10^4 times more toxic than the carbonate insecticides.

The acute toxicity of the various organochlorine insecticides may vary by more than an order of magnitude (Table A-2, Macek and McAllister 1970, Naqvi and Ferguson 1970), but the variability in chronic toxicity levels is minimal. Except for toxaphene, all maximum acceptable tolerance concentrations (MATC), which are based on long-term exposures, were in the range of 0.1 to 1.0 μg/liter for fathead minnows and brook trout. In addition to such sublethal effects as reduced growth and reproduction, long-term exposure to DDT has been shown to impair the establishment of locomotor patterns in goldfish (Davy et al. 1972) and to delay the appearance of behavior patterns and impair balance in Atlantic salmon alevins exposed to DDT during the egg stage (Dill and Saunders 1974).

Many of the organochloride insecticides can persist for long periods of time in aquatic ecosystems. Because of their high adsorptive affinity for suspended particulates, chlorinated hydrocarbon compounds tend to accumulate in the sediments at the bottom of rivers and reservoirs. Furthermore, both PCBs and DDT can persist in the sediments and fish for years after marked declines in inputs to aquatic systems have occurred. For example, since 1962, the inputs of PCBs to the Southern California Bight have decreased by a factor of 14, while

Table A-2. Acute and chronic toxicity of various insecticides and herbicides to freshwater biota as determined from continuous-flow bioassays. N/A = Information not available in reference. LC_{50} = the concentration that will result in 50% mortality to the test organisms after a specified period of exposure.

Chemical	Test temperature (°C)	Species	Initial size/age	LC_{50} Concentration (μg/liter)	LC_{50} Exposure period (d)	MATC[a] Concentration (μg/liter)	MATC[a] Exposure period (d)	Reference
ORGANOCHLORINE INSECTICIDES								
Chlordane	20-21	Daphnia magna (water flea)	N/A	28.4	4	<21.6[b]	28	Cardwell et al. (1977)
	16	Hyallela azteca (amphipod)	Juvenile	97.1	7	<11.5[b]	65	Cardwell et al. (1977)
	25	Chironomus (midge)	Newly hatched larvae	N/A	N/A	< 1.7[b]	21	Cardwell et al. (1977)
	25	Lepomis macrochirus (bluegill)	3 months	59 40	4 6	<1.22[b]	290	Cardwell et al. (1977)
	15	Salvelinus fontinalis (brook trout)	24 months	47 25	4 7	<0.32[b]	395	Cardwell et al. (1977)
	25	Pimephales promelas (fathead minnow)	3 months	36.9 32.1	4 8	N/A		Cardwell et al. (1977)
DDT	15.6	Gammarus fasciatus (scud)	N/A	0.6	5	N/A		Stalling and Mayer (1972)
	15.6	Palaemonetes kadiakensis (glass shrimp)	N/A	1.3	5	N/A		Stalling and Mayer (1972)
	18	Fathead minnow	~30 mm (SL)	>40[c]	2	N/A	N/A	Lincer et al. (1970)
	25	Fathead minnow	45 ± 3 d	48	4	0.9[d], 0.4[e]	266	Jarvinen et al. (1976)
Endrin	18	Fathead minnow	~30 mm (SL)	0.57 0.39	2 4	N/A		Lincer et al. (1970)
	25	Jordanella floridae (flagfish)	N/A	0.85	4	0.22-0.30	110	Spehar et al. (1979)
Endosulfan	19 ± 1	Daphnia magna	<24 h	N/A		2.7-7.0	64	Macek et al. (1976)
	25 ± 1	Fathead minnow	53 d	0.86	7	0.20-0.40	280	Macek et al. (1976)
Heptachlor	19 ± 1	Daphnia magna	<24 h	N/A		12.5-25.0	64	Macek et al. (1976)
		Fathead minnow	60 d	7.0	10	0.86-1.84	280	Macek et al. (1976)
Methoxychlor	20 ± 2	Fathead minnow	0.4-0.8 g	7.5	4	N/A		Merna et al. (1972)
	20 ± 2	Perca flavescens (yellow perch)	4.0-6.0 g	20	4	N/A		Merna et al. (1972)
Toxaphene	16	Brook trout	16 months	10.8	4	<0.039[f]	365	Mayer et al. (1975)
	25	Fathead minnow	N/A	4.8	10	0.025-0.054	259	Spehar et al. (1979)
	25	Ictalurus punctatus (channel catfish)	N/A	15	9	0.129-0.299	240	Spehar et al. (1979)
ORGANOPHOSPHATE INSECTICIDES								
Diazinon	12 ± 0.5	Brook trout	13-20 weeks	770	4	<<0.55[f]	173	Allison and Hermanutz (1977)
	25 ± 0.5	Bluegill	12 months	460	4	N/A		Allison and Hermanutz (1977)
	12 ± 0.5	Fathead minnow	12 months	7800	4	<3.2[f]	274	Allison and Hermanutz (1977)
Malathion	25	Flagfish	N/A	349	4	8.6-11	110	Spehar et al. (1979)
HERBICIDES								
Acrolein	19 ± 1	Daphnia magna	<24 h	N/A		16.9-33.6	64	Macek et al. (1976)
	25 ± 1	Fathead minnow	51 d	84	6	11.4-41.7	245	Macek et al. (1976)
Trifluralin	19 ± 1	Daphnia magna	<24 h	N/A		2.4-7.2	64	Macek et al. (1976)
	25 ± 1	Fathead minnow	44 d	115	12	1.9-5.1	427	Macek et al. (1976)

[a]MATC = Maximum Acceptable Toxicant Concentration.

[b]Values represent the lowest concentration of technical chlordane found to cause major chronic effects.

[c]Statistical analysis of the results indicated that none of the concentrations employed had any significant deleterious effects on any of the life-cycle stages of this species.

[d]For fish exposed to DDT in water only.

[e]For fish exposed to DDT in both water and diet.

[f]Lowest concentration tested; since these levels caused deleterious effects, no MATC could be established.

the concentrations in the sediments and fish have only been reduced by a factor of 1.2 and 1.9, respectively; similar findings were obtained for DDT and metabolites (U.S. Environmental Protection Agency 1977). Knowing the precise history of DDT treatment to several small watersheds, Dimond et al. (1971) found that DDT persisted in the streams for at least 10 years following light applications to the forest. Residues declined sharply within two or three years after application. Tissue concentrations of DDT and metabolites in biota were approximately an order or magnitude lower than initial levels after five years but, after ten years, were still approximately an order of magnitude higher than the concentrations in biota from the control streams.

The resistance of many chlorinated hydrocarbons to degradation by enzymatic processes is a major factor controlling their accumulation in the tissues of biota (Metcalf et al. 1976). DDT is readily converted to several persistent by-products. Studies have shown that DDE is the most common and persistent of these degradation products (e.g., Dimond et al. 1971, Grzenda et al. 1970, Johnson et al. 1971). Metcalf et al. (1976) found DDE to be extremely stable in the tissues of aquatic biota and approximately as persistent as pentachlorobiphenyl. By comparison, the degradation of DDT in sediments (mud) apparently occurs at an even slower rate than that observed for animal tissues (Dimond et al. 1971).

The metabolism of DDT to DDE can be very rapid. Johnson et al. (1971) studied the degradation of DDT in seven species of aquatic invertebrates and found that the conversion of DDT to DDE was 85% complete after three days of continuous exposure to DDT. Another degradation product, DDD, was found in only two of the seven species and never accounted for more than 7% of the total residues (DDT + DDE + DDD). Some degree of conversion of aldrin to dieldrin has also been observed (Johnson et al. 1971). The rate of degradation of DDT in goldfish was reported to vary among the twelve tissues examined, with the highest percentage of metabolites (70% DDE + DDD) occurring in the liver and the lowest in the immature ovary (30%) after 8-d exposure to DDT in the diet (Grzenda et al. 1970). Also, the percent composition of DDT, DDE, and DDD in the goldfish tissues was found to vary with time; i.e., the percentage of metabolites increased with continued exposure.

Another important biological property of DDT and some of the other organochlorine insecticides is the fact that these compounds, like PCBs, can accumulate in the tissues of freshwater biota. Accumulation of compounds such as DDT, DDE, and aldrin from the water by freshwater invertebrates has been shown to increase with an increase in both exposure time and concentration in the water (Derr and Zabik 1972, Johnson et al. 1971, Wilkes and Weiss 1971). In short-term continuous exposures to DDT concentrations of 0.1 μg/liter, the bioconcentration factors (BCF) for seven invertebrates ranged from 2900 (crayfish) to 114,100 (*Daphnia*) after three days. Bioconcentration factors for aldrin were similar, and the BCFs for both DDT and aldrin increased over the 3-d exposure period (Johnson et al. 1971).

Bioconcentration factors for fish are similar to those found for many invertebrate species. Continuous 32-d exposures of fathead minnows to various organochlorine insecticides at levels of 3-7 μg/liter in the water resulted in low BCFs for lindane, methoxychlor, and heptachlor (180, 8300, 9500, respectively), but higher BCFs for DDT (p, p' DDT = 29,400; o, p' DDT = 37,000), chlordane (37,800), and p, p' DDE (51,000) (Veith et al. 1979). Bioconcentration factors for DDT as high as 100,000 have been reported for fathead minnows (Jarvinen et al. 1976) and golden shiners (Courtney and Reed 1972). Veith et al. (1979) found that the BCF was independent of the age of the fish but was dependent on both temperature and species.

In addition to the direct uptake from water, pesticide residues can bioaccumulate in fishes as a result of uptake from food. Laboratory studies of the uptake of DDT from dietary sources have shown an initial period of rapid uptake (approximately the first 2-4 weeks), followed by a period of slower uptake until an equilibrium state was reached and no additional accumulation with continued exposure was observed (Grzenda et al. 1970, Macek et al. 1970). The time required to reach equilibrium varied among tissues. In skeletal muscle, no significant additional accumulation was observed after 28 d in goldfish (Grzenda et al. 1970) and after 32 d in rainbow trout (Macek et al. 1970). Also, the residue concentration in the tissues increased with increased concentrations in the diet for both DDT (Macek et al. 1970, Warlen et al. 1977) and dieldrin (Grzenda et al. 1971). Finally, when fish were fed uncontaminated food after the equilibrium state had been reached, the average half-life of DDT was found to be 29 d in goldfish (Grzenda et al. 1970), 56 d in fathead minnows (Jarvinen et al. 1976), and was predicted to be 160 and 40 d for DDT and dieldrin, respectively, in rainbow trout (Macek et al. 1970).

A comparison of the DDT residues in the tissues of biota with residues in the diet suggests that concentration factors for fish are low, ranging from 0.6 to 1.2 in several laboratory studies (Grzenda et al. 1970, Jarvinen et al. 1976, Macek and Korn 1970). Similar results were obtained in studies of both invertebrates and fish that were fed dieldrin-contaminated diets (see review by Jarvinen et al. 1976). Of particular significance is the 10-year study by Dimond et al. (1971) of DDT residues in the biota of several small watersheds where the history of DDT treatment was known. A comparison of residue levels in fish and fish-eating birds indicated a concentration factor of 13. Residues in trout were approximately 2 to 10 times greater than the residues found in aquatic insects in the first six years but were only 1.0 to 1.8 times greater during the last four years of the study when lower levels of contamination existed. Evidence for the existence of higher concentrations of DDT in fish of higher trophic levels has been contradictory (e.g., see review in Grzenda et al. 1970, Klaassen and Kadoum 1975, Reinert and Bergman 1974).

In general, there are two schools of thought regarding the relative importance of uptake from water vs uptake from food to the accumulation of pesticides in biota at higher trophic levels. Much of the earlier work on the distribution and persistence of pesticides in aquatic environments has emphasized that the food chain is the major source of transfer to higher trophic levels (e.g., Macek and Korn 1970, Johnson et al. 1971). Thus, the ingestion of pesticides from the food supply accounts for the high pesticide residues often found in many fishes. A second hypothesis suggests that the levels found in various trophic levels are dependent upon those factors that control the concentration of the pesticide in the surrounding water (e.g., Grzenda et al. 1970, Hamelink et al. 1971, Hamelink and Waybrant 1976). The persistence of chlorinated hydrocarbon compounds such as DDT is regulated by physical properties of the chemical, such as water solubility and adsorptive affinity. Thus, according to this hypothesis, exchange equilibria (e.g., between lipids in tissues and water) control the accumulation of these compounds in invertebrates and fish.

Obviously, the uptake from both water and food contribute to the accumulation of residues in the tissues of biota and are therefore important. At very low concentrations in the water, uptake of DDT from dietary sources may be important (Macek and Korn 1970). Even though DDT uptake from water may be greater, sources of DDT in the food can also lead to high residue concentrations in the tissues (Jarvinen et al. 1976). In a contaminated environment, both uptake pathways are operating, but the significance of one over the other will be dependent upon the degree of contamination (i.e., the concentration of the contaminant in the water and the length of the exposure period). At low concentrations in the water, the importance of DDT-contaminated food was greater, but as the concentration of DDT in the water increased, the importance of food sources decreased (Jarvinen et al. 1976).

Muscle tissue has been found to be among the tissues with the lowest levels of DDT (Grzenda et al. 1970) and dieldrin (Grzenda et al. 1971). Macek et al. (1970), on the other hand, reported that 70% of the total residues of both DDT and dieldrin were found in the edible tissues of rainbow trout after a 140-d exposure. The FDA limits on the concentration of DDT in fish is 5 μg/g (wet wt), the same as that set for PCBs, while the limit for dieldrin is 0.3 μg/g (wet wt).

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