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# Analysis of Environmental Issues Related to Small-Scale Hydroelectric Development

## III: Water Level Fluctuation

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ENVIRONMENTAL SCIENCES DIVISION
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# ANALYSIS OF ENVIRONMENTAL ISSUES RELATED TO SMALL-SCALE HYDROELECTRIC DEVELOPMENT

## III: Water Level Fluctuation[1]

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ABSTRACT

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This report identifies potential environmental impacts in reservoirs and downstream river reaches below dams that may be caused by the water level fluctuation resulting from development and operation of small scale hydroelectric projects. (Small scale hydroelectric projects are defined as existing dams that can be retrofitted for hydroelectric power generation to a total site capacity of $\leq$25 MW.)

The impacts discussed will be of potential concern at only those small-scale hydroelectric projects that are operated in a store and release (peaking) mode. Potential impacts on physical and chemical characteristics in reservoirs resulting from water level fluctuation include resuspension and redistribution of bank and bed sediment; leaching of soluble organic matter from sediment in the littoral zone; and changes in water quality resulting from changes in sediment and nutrient trap efficiency. Potential impacts on reservoir biota as a result of water level fluctuation include habitat destruction and the resulting partial or total loss of aquatic species; changes in habitat quality, which result in reduced standing crop and production of aquatic biota; and possible shifts in species diversity.

The potential physical effects of water level fluctuation on downstream systems below dams are (1) streambed and bank erosion and (2) water quality problems related to resuspension and redistribution of these materials. Potential biological impacts of water level fluctuation on downstream systems below dams result from changes in current velocity, habitat reduction, and alteration in food supply. These alterations, either singly or in combination, can adversely affect aquatic populations below dams.

The nature and potential significance of adverse impacts resulting from water level fluctuation are discussed. Recommendations for site-specific evaluation of water level fluctuation at small-scale hydroelectric projects are presented.

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# 1. INTRODUCTION

In 1977 the U.S. Department of Energy (DOE) initiated a program to stimulate the redevelopment of existing dams for hydroelectric generation. The DOE Small-Scale Hydroelectric Development Program is directed toward dam and impoundment systems that have a potential capacity of 25 MW or less. Through both technical support and financial assistance, the DOE goal is to stimulate development by the public and private sector of 1000 MW of capacity by 1985. The DOE Small-Scale Hydroelectric Program includes an Environmental Subprogram for identifying and analyzing potential environmental issues related to small-scale hydroelectric (SSH) development. This report addresses the potential ecological concerns related to water level fluctuation that may occur as a result of the operation of hydroelectric facilities in support of the Environmental Subprogram Plan.

The mode of operation of a hydroelectric facility is the prime determinant of the potential significance of ecological impacts related to water level fluctuation. Three general classes of operation (Linsley and Franzini 1972) can be identified for hydroelectric facilities: (1) run-of-river, (2) storage, and (3) pumped-storage. A strict run-of-river facility generally has extremely limited water storage capacity and only uses normal river flow for hydroelectric generation. Strict run-of-river hydroelectric operation should not increase water level fluctuation beyond fluctuations already present in the drainage basin. However, some hydroelectric facilities classed as run-of-river have enough water storage capacity (pondage) to permit storing water during off-peak hours for use during peak hours of the same day. A limited peaking operation of this type will result in some water level fluctuation that may cause environmental impacts.

A true storage hydroelectric facility includes a reservoir of sufficient size to permit water storage from the wet season to the dry season. This storage capability can provide consistent flows for hydroelectric generation, which are considerably greater than the minimum natural flows in the basin. Storage hydroelectric systems can also be operated in a peaking mode. Water level fluctuation will be associated with the operation of storage hydroelectric facilities.

A pumped-storage facility includes an upper and a lower reservoir. Power is generated for peak demand, but during off-peak hours, secondary power is used to pump water from the lower reservoir to the upper reservoir (Linsley and Franzini 1972). Pumped-storage operation can cause considerable water level fluctuation.

About 5000 existing dams within the United States have been identified as having the potential for additional hydroelectric generation (U.S. Army Corps of Engineers 1979). The percentage of these sites that would be developed to operate in run-of-river or storage mode is not known. At those sites developed in a strict run-of-river mode, water level fluctuation should not occur. Development of sites as storage or peaking facilities would cause water level fluctuation and could cause environmental impacts.

This report identifies and discusses generic ecological issues related to water level fluctuation in both reservoir ecosystems (Sections 2 and 3) and riverine systems below a dam (Sections 4 and 5). Physical, chemical, and biological concerns are discussed. Section 6 offers general guidance for analyzing water level fluctuation issues on a site-specific basis. Appendix A presents a quantitative methodology for predicting loss of shore zone habitat in a reservoir resulting from water level fluctuation.

This report is the third in a series analyzing environmental issues related to small-scale hydroelectric development. The first report in the series (Loar et al. 1980) examines the topic of dredging. The second report (Hildebrand 1980) addresses design considerations for facilities to pass fish upstream around dams. All three reports are available for purchase from the National Technical Information Service, U.S. Department of Commerce, 5285 Port Royal Road, Springfield, Virginia.

## 2. IMPACTS OF WATER LEVEL FLUCTUATION ON PHYSICAL AND CHEMICAL CHARACTERISTICS OF RESERVOIRS

R. R. Turner

Section 2 presents and generically evaluates the potential abiotic (i.e., physical and chemical) impacts of water level fluctuations on reservoirs (impoundments). Potential environmental issues that may arise at any small hydroelectric site are identified and analyzed in the context of the controlling characteristics and variables that determine the nature and severity of any environmental impact. The most important controlling characteristics determining the nature and severity of impacts are (1) the range of water level fluctuations and (2) impoundment morphology. Appendix A contains a comprehensive and detailed treatment of the geometric relationships between these variables and should be consulted as a prelude and companion to this section. The goal of Section 2 is to provide the potential developer of a small hydroelectric site with a convenient and rapid methodology for identifying potential abiotic environmental issues that may arise at the site because of an unfavorable combination of physical and chemical characteristics.

The primary abiotic effects of water level fluctuations within impoundments not formerly exposed to such fluctuations, or exposed to different temporal patterns and ranges of fluctuations, are expected to involve water quality and the stability of shoreline (bank) and substrate (bed).

Specifically, these effects may be summarized as:

1. Alteration of the development and persistence of thermal stratification and alteration of the water quality parameters that are coupled to stratification.

2. Resuspension and redistribution of bed and bank substrate materials (soils and sediments) within the new littoral (or shore) zone defined by the water level fluctuations imposed by hydroelectric operations.

3. Leaching of soluble matter from substrate material in the littoral zone as water moves into and out of the interstices (bank storage) of this substrate in response to water level fluctuations.

4. Changes in sediment and nutrient retention (trap efficiency) by the impoundment, and the resulting changes in impoundment water quality, as a consequence of changes in circulation pattern and hydraulic flushing rate imposed by the hydroelectric generation.

Water level fluctuations effectively alter the size of the impoundment littoral zone and temporarily alter the volume of impounded water. A widened littoral zone provides a larger surface area over which abrasional processes (waves, currents, and ice) may operate and, in some circumstances, may increase the relative importance of bank storage in the reservoir water balance and water quality. Volume changes may directly affect the circulation pattern, hydraulic efficiency, and thermal regime of an impoundment. The nature and severity of physical and chemical effects resulting from water level fluctuation are determined by one or more of the following features (determinant characteristics) of an impoundment: morphology, geographic location (latitude and climate zone), tributary hydrology, timing and range of water level fluctuation, character of bed and bank substrate, and wind conditions (speed, direction, and duration).

In the following subsections, determinant characteristics (e.g., morphology) that affect each impoundment response characteristic (e.g., thermal regime) are discussed in the context of the four potential abiotic effects of water level fluctuations summarized above. If possible, the direction and magnitude of the changes in impoundment response characteristics engendered by changes in the determinant characteristics are indicated. With this approach combinations of physical features of a candidate small hydroelectric site that may either preclude, restrict, or encourage development on environmental grounds can be identified.

## 2.1 Impacts on Thermal Regime and Circulation Pattern

The thermal regime (content and distribution of heat or temperature structure) of an impoundment varies in response to circulatory processes and seasonal and other episodic inputs and outputs of heat. In some cases, the thermal regime of impoundments may be significantly influenced by discharge of heated effluents from power plants (Benedict et al. 1973) or other industrial sources. The potential effects of water level fluctuations on thermal regime must be evaluated against this background.

The chief environmental issue related to the thermal regime of an impoundment is the water-quality problems caused by thermal stratification (i.e., low or absent concentrations of dissolved oxygen and high concentrations of minerals--particularly, Fe, Mn, $NH_3$, and $H_2S$--in hypolimnetic water). As in natural lakes in temperate regions, the morphometric properties of depth and surface area largely determine whether an impoundment with a surface outlet will thermally stratify. If other factors are equal, a deeper, more sheltered (from wind fetch) impoundment with a smaller surface area is more likely to stratify than is a shallow, exposed impoundment with a large surface area.

In contrast to natural lakes with low hydraulic flushing rates and heat budgets dominated by direct solar inputs, the thermal regime of impoundments with high hydraulic flushing rates can be dominated by the temperature characteristics of tributary streams (advective heat inputs) and by the circulation patterns induced by these streams (Carmack et al. 1979). Seasonally cold influents (e.g., snow-melt waters) can act to prolong overall spring warming of an impoundment, delay development of stratification in the spring, and hasten cooling and destratification in the fall (Hutchinson 1957). If snow-melt waters are not a component of tributary inflow to an impoundment, spring inflows are likely to be warmer (and less dense if temperature is greater than 4°C) than most of the currently pooled water and may result in faster warming of surface water and development of stratification. Depending on the depth of withdrawal from the impoundment, warm spring inflow may flow completely over underlying water layers, which are colder and denser, without significant mixing and create thermal stratification, which is induced almost entirely by tributary inflow (Churchill 1958, Huber et al. 1972).

Wind velocity, wind direction, and the fetch (length of water surface over which the wind is blowing) directly affect thermal structure by the (1) physical mixing of surface water by wind-driven waves and currents and (2) convective mixing associated with evaporative and radiative heat losses. Given the same fetch over open water, higher wind velocities may be expected to result in greater thickness of the epilimnion (mixed layer). Also, winds of extended duration from the same direction may induce increased mixing of deeper, cooler water into the epilimnion, particularly on the windward side of an impoundment. Surface water cooling associated with evaporative heat loss is a function of air temperature, water temperature, solar radiation, relative humidity, and wind speed (Wunderlich 1971). Depending on prevailing air and water temperatures, solar radiation, and humidity, increasing wind speed is expected to increase the rate of surface water cooling and also increase the rate of convective mixing.

The range of water level fluctuations may influence thermal regime, mainly by raising and lowering an imaginary horizontal plane within an impoundment above which wind waves can significantly mix the water column. If the wave height, length, and period are known, the theoretical maximum depth of significant influence by wind waves and the position of the imaginary plane can be calculated. Generally, below depths approximating one-half the wave length, wave-induced mixing is minimal (Chow 1968). This imaginary plane, or "wave base," intersects the bottom, dividing the impoundment into two zones, the relative volumes of which depend highly on impoundment morphometry. In some shallow impoundments, the elevation of the theoretical wave base may be below the greatest depth of the impoundment, even at maximum pool elevation, and all parts of the water column would be theoretically within range of mixing by wind waves of sufficient dimensions and periods. This does not imply that wind waves will occur continuously enough to maintain complete mixing of the entire water column, or that other factors, such as sharp changes in the water density versus water depth profile, can be ignored. Because wave base at any given time is determined by the properties of the extant wind waves, wave base may be expected to vary hourly, daily, and seasonally as wave properties vary. Also, sharp density gradients (e.g., at a thermocline) can restrict effective wave mixing to higher elevations than would be predicted from the position of the theoretical wave base.

The impoundments of most interest in evaluating the effects of water level fluctuations associated with small hydroelectric operations are those impoundments where some substantial volume is either rarely or never within the depth range of significant wave mixing at either maximum, or normal, pool elevation, but is brought into range episodically by water level drawdown for hydroelectric generation. Such a situation is most likely to occur with an impounded river with a reasonably wide, but drowned, floodplain and an incised former river channel. Water level fluctuation might be limited to elevations on the steeper valley sides to avoid exposing the flat, former floodplain (see Figure 1), but could conceivably involve all of the water column above the former floodplain in periodic turbulent mixing by wind waves. Thermal stratification would probably be confined to water occupying the former river channel.

In the presence of other favorable factors, a shift from stable to fluctuating water levels could reduce the tendency for much of the reservoir volume to become thermally stratified and, thus, less likely to experience the water quality problems that often accompany thermal stratification. If the period of maximum windiness does not coincide in time with the period of likely development of thermal stratification, any benefit from more complete wind mixing will be reduced or eliminated. Similarly, if water level drawdown is infrequent, of short duration, or of limited range, opportunities for realizing the benefits of improved wind mixing may be insufficient.

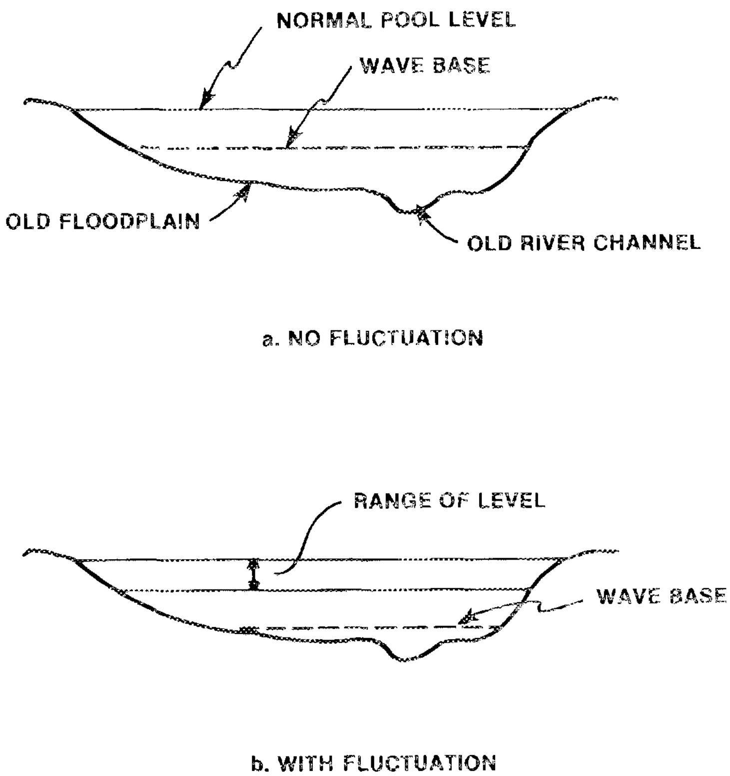


Figure 1. Theoretical cross section of a reservoir showing how water fluctuation could expand the influence of wave mixing.

In summary, thermal regime and circulation pattern in a small impoundment retrofitted to produce hydroelectric power are not expected to be affected in a manner likely to lead to water quality problems that were not previously present at the site. In some cases, the probable improved mixing associated with water level fluctuation may reduce the tendency towards, and persistence of, thermal stratification.

## 2.2 Impacts on Bank and Bed Stability

Soils and sediments within the littoral zone of an impoundment are periodically exposed to the erosive forces of wind waves, currents, and ice. After initial exposure to these forces, the texture (particle size distribution) and angle of repose (topographic profile) of these materials will change until some equilibrium state is established between the acting forces and the available substrate. After this equilibrium is established, the topographic profile and physical character of the littoral zone will remain essentially unchanged, except in response to a change in water level or the occurrence of large storms which may temporarily increase the erosive forces applied to the littoral zone by waves.

A common feature of littoral zones that are in equilibrium is the parallel development of wave-cut terraces and wave-built terraces. Generally, some of the bank material removed by waves in the process of terrace cutting comprises the material of the adjacent wave-built terrace. If wave-driven currents (e.g., longshore drift) or tributary inflow currents are present, considerable material from cut terraces may comprise submerged offshore bars and spits.

Impoundments, particularly the smaller and more sheltered ones, do not usually possess very high wave energy, except perhaps where boat traffic is heavy or the wind fetch over open water favors good wave development. Gradually shoaling bottom contours may also dissipate wave energy before waves reach shore. Furthermore, the growth of rooted aquatic plants and riparian vegetation where stable water levels exist may protect an otherwise vulnerable shoreline from the occasionally energetic assaults of waves and currents. Thus, the classic development of wave-cut and wave-built terraces may not always exist over the entire wetted perimeter of an impoundment. This lack of terrace development is apparent in areas where water level does not fluctuate significantly and where the bank is naturally rocky, only gradually shoaling, or otherwise protected against the erosive forces of waves.

In areas where water level does fluctuate, especially in reservoirs where bank materials are not very well consolidated (e.g., alluvial floodplain deposits), these materials are continually resuspended and redistributed in the littoral zone. The net result of this resuspension is the preferential removal (winnowing) of the finer, less dense particles (silt, clay, organic matter) from the littoral zone and redeposition in deeper and more quiescent water (Hynes 1961, Hunt and Jones 1972, Grimas 1962). Complete loss of these particles from the impoundment in the outflow can also occur. The material remaining in the littoral zone after extended exposure to a fluctuating water level is coarser in particle size than would be the case with a stable water level.

Alluvial sediments, containing very little coarse material that can withstand the erosive forces of waves and currents, may be periodically exposed by a fluctuating water level. In this case, banks exposed at low water levels may be submitted to continuous undercutting and collapse, with the fine-grained eroded material contributing to increased turbidity of the water column and accelerated sedimentation rate in deeper water (Rao and Palta 1973, Grimas 1962, Grimas 1965).

As reservoirs age, they fill with sediment in a complex manner, responding to the site-specific spatial distribution of major sediment sources (tributary inflows) and to numerous other factors, including morphometry, water level, and hydraulic flushing rate. Because inflowing streams deposit the bulk of their sediment load near the inflow points (Neel 1963), reservoir volume is often reduced initially by siltation near the inflow (deltaic sedimentation). With stable water levels, the resulting delta advances gradually down the longitudinal axis of the reservoir, often leaving behind greatly reduced water depth, except in areas where the higher velocity of inflowing tributaries maintains a deeper channel or network of channels. Imposing water level fluctuations on an impoundment in which sediments were previously distributed under a stable water level may lead to major redistribution of delta sediments near tributary inflows. Lowering the water level effectively lowers the local "structural base level," defined as the depth at which erosion by flowing water is balanced by deposition, with the consequence that previously deposited sediments can be resuspended and transported.

Thus, as demonstrated by Lara (1973), regular drawdowns of a reservoir can profoundly affect the longitudinal and lateral distribution of sediment. These effects may be beneficial to prolonged reservoir life time because previously deposited sediments can be sluiced out of the reservoir, a process that ordinarily requires a deep outlet. However, the effects of sediment redistribution could be deleterious because of the excessive resuspension of sediments into the water column and the potential clogging of outlet structures (e.g., Nolichucky Project [TVA 1978]). For a surface discharge reservoir subjected to water level fluctuations for the first time, the sediment redistribution may substantially change the usable water storage (e.g., conservation storage) below a certain elevation (i.e., below minimum operating pool, but above dead storage). Although these changes may not directly affect routine reservoir operations, they could affect occasional operations such as flood control preparation or drought-year flow augmentation.

In summary, imposing water level fluctuations on an impoundment with no previous history of such fluctuations may increase erodibility of exposed banks (beaches) over the entire wetted perimeter of an impoundment and resuspend and redistribute delta deposits. Both effects can lead to increased turbidity of the water column and may lead in some cases to undesirable redistribution of storage capacity in an impoundment.

## 2.3 Impacts on Leaching of Bank and Bed

The alternate inundation and drainage to which soils and sediments in the littoral zone in impoundments are exposed when water level fluctuates may lead to increased leaching of soluble matter from the exposed substrate. Sediments or soils that are continuously inundated (saturated with water) and that have organic matter present have a strong tendency to develop anaerobic (chemically reducing) conditions. These conditions are highly conducive to the solubilization of some plant nutrients and metals; concentrations of these nutrients and metals may increase in the interstitial water of flooded soils and sediments (Rittenberg et al. 1955, Reddy and Patrick 1975). In shallow water, where aerobic (chemically oxidized) conditions are likely to prevail in the water overlying flooded soils and sediments, a sharp gradient in the concentrations of nutrients and metals is likely to exist across the substrate-water interface (e.g., Lee 1970). When the water level is lowered, some of the interstitial water, containing higher concentrations of nutrients and metals, may drain onto the surface of the substrate and enter the adjacent surface water body. If this drainage water is highly enriched in soluble plant nutrients or other substances that may be deleterious (such as heavy metals), the nearshore water quality may be adversely affected.

Prolonged or repeated exposure of formerly inundated substrate by water level drawdown facilitates more complete drainage, desiccation, shrinkage, and cracking of the exposed bottom, especially if the substrate is composed of cohesive mud. Reduced chemical species in the substrate are subjected to oxidation and may become either more or less soluble when rising water levels reinundate the substrate.

Water level drawdown is often used in natural lakes to help improve water quality and substrate stability (Fox et al. 1977) or to reduce nuisance aquatic vegetation (Beard 1969, Beard 1973, Cooke 1980). Generally, these drawdowns span several months when used to substantially dry and consolidate the exposed substrate or to completely destroy aquatic vegetation. These drawdowns appear to be most successful in improving water quality and benthic habitat if large areas of mucky lake bottom are exposed by the drawdown. Water level drawdowns that are of limited duration and that expose only sandy or rocky substrate or only a small fraction of the total bottom area, such as occur in hydroelectric impoundments operated in a peaking mode, would not be expected to substantially improve water quality or habitat. Drawdowns may serve in some cases to control growth of certain aquatic vegetation within the littoral zone (Sect. 3) and may lead to a physically firmer substrate within the littoral zone (Sect. 2.2). However, the periodic leaching of exposed substrate and the drainage of interstitial water into nearshore surface water are likely to be the most important factors for consideration when impoundments are subjected to water level fluctuations.

As with the physical stability of bed and bank substrate, the short-term effects on leaching after a change from historically stable water levels to fluctuating levels should be the main environmental concern. Flooded soils and sediments that are periodically exposed to drainage and leaching should, in most cases, reequilibrate both physically and chemically with any imposed pattern of regular water level fluctuation. Exceptions are likely to be impoundments that are used only seasonally for peaking power (e.g., an annual operational cycle characterized by a prolonged period of stable water level followed by a period of daily [or shorter interval] water level fluctuation).

## 2.4 Impacts on Trap Efficiency for Sediments and Nutrients

The ability of a reservoir to trap and retain sediment, known as the "trap efficiency," is expressed as the percentage of total inflowing sediment that is retained (deposited) in the basin. The degree to which a reservoir traps inflowing sediment is the prime determinant of the useful life span of the reservoir. It is an important factor affecting water quality, especially turbidity, both within and downstream from the reservoir. Ironically, the effective trapping of sediment by a reservoir often increases erosion of bank and bed sediments downstream from the reservoir (Neel 1963, Baxter 1977).

Trap efficiency for sediment varies as a function of (1) inflowing sediment particle size, (2) reservoir capacity/annual inflow ratio (often called the C/I ratio, but also called the theoretical water renewal time or the hydraulic flushing rate), (3) location and operation of the reservoir outlet, (4) reservoir shape, and (5) chemical properties of the water (American Society of Civil Engineers 1973, Chow 1968). As stream flow enters a reservoir, the cross-sectional area of flow is normally increased, thus reducing velocity and decreasing sediment transport capacity. Coarse-grained sediment particles are deposited immediately near the head of the backwater, whereas finer-grained particles with lower settling velocities remain in suspension until they are deposited or carried out of the reservoir in the outflow.

Sediment trap efficiency depends primarily on the fall velocity of the sediment particles and the mean flow velocity through the reservoir. The fall velocity of sediment particles in water depends on several factors, including the size and shape of the particles and the chemical composition and viscosity of the water. Chemical composition of water affects the fall velocity of fine-grained sediments, such as clays and colloids, which tend to aggregate (flocculate) or disperse in response to the character and quantity of dissolved solids in the suspending water (e.g., the calcium/sodium ratio). Divalent cations (such as calcium) are more effective in flocculating fine-grained sediments (clays, silts) than are monovalent cations (such as sodium).

The mean flow velocity of water through a reservoir depends on the water inflow rate, available storage, rate of outflow, and sometimes the reservoir morphometry. Flow velocity is probably the single most important factor affecting sediment trap efficiency. Data that can be used to reliably estimate flow velocities through reservoirs are not usually available, and actual velocity measurements are comparatively rare (Wunderlich and Elder 1973). Thus, some related parameters have been used to estimate flow velocities, including the reservoir C/I ratio (Brune 1953) and the sedimentation index (the ratio of the period of retention to the theoretical mean flow velocity through the reservoir [Churchill 1948, cited in Brune 1953]). To obtain the sedimentation index, the period of retention is determined by dividing reservoir capacity at mean operating pool by mean daily inflow rate. Mean velocity is estimated by dividing mean daily inflow by the average cross-sectional area of the reservoir (which can be calculated as reservoir capacity divided by reservoir length at mean operating pool level). Dendy (1974) has treated these relationships more thoroughly and has compared their effectiveness in predicting sediment trap efficiencies for small reservoirs.

If storage volume is not regulated by varying water level, both the C/I ratio and the sedimentation index for a given reservoir are determined entirely by the water inflow rate. If the water level is varied, for example, as a consequence of hydroelectric seasonal or peaking operations, the C/I ratio and sedimentation index will also vary proportionally. However, trap efficiency for sediment does not appear to vary as a simple linear function of C/I ratio or sedimentation index (Brune 1953) and may exhibit considerable variability at low values of the C/I ratio. At low values of the C/I ratio (i.e., reservoirs with extremely high hydraulic flushing rates), additional factors such as reservoir shape and sediment characteristics may become more important in determining trap efficiency.

Most of the relationships between trap efficiency and reservoir characteristics have been derived from long-term average annual conditions in a large variety of reservoirs. In considering the potential effects of water level fluctuation on trap efficiency of a candidate small hydroelectric site, information on the annual, and shorter period, variability in trap efficiency for an individual reservoir may be more important than the average data from many reservoirs. In particular, the ability to predict sediment trap efficiency, given water inflow rates, sediment characteristics, water level fluctuation regime, and location and operation of the reservoir outlet, would be helpful. Unfortunately, only a few attempts have been made to achieve such predictive capability for individual reservoirs (Borland 1951, cited in Brune 1953; Rausch and Heinemann 1975), and these few have been successful in predicting only the direction, not the absolute magnitude, of change in sediment trap efficiency caused by a change in one or more of the determinant variables.

Generally, in individual reservoirs sediment trap efficiency decreases with a (1) decrease in water retention time (estimated by the C/I ratio), (2) decrease in sediment particle size, and (3) change from surface-water to deep-water withdrawal. Imposing water level fluctuation on a formerly unfluctuated reservoir would directly affect the water retention time as a consequence of the changes in average storage volume; the overall effect would be a probable reduction in sediment trap efficiency. Also, periodically lowering the water level may in some circumstances permit coarser-grained sediments in the tributary stream channels to reach, and be transported further into, the reservoir basin (Sect. 2.3), thereby initially (during the reequilibration period) increasing trap efficiency. However, this increase in trap efficiency would also lead to a faster loss of reservoir storage capacity. Thus, any apparent increase in trap efficiency resulting from the coarser sediment input may ultimately be offset by the decrease in retention time caused by the decrease in storage volume. Viewed in another way, the reduction in average cross-sectional area of an impoundment caused by water level fluctuations must necessarily lead to higher mean water velocity through the reservoir and, thus, less efficient trapping of certain sediment particle sizes.

The term "trap efficiency" has also been used in the context of nutrient retention by reservoirs (Glymph 1973, Rausch and Shreiber 1977). The efficiency with which impoundments trap inflowing nutrients is an important environmental issue because water quality and productivity, both within and downstream from impoundments, is partly determined by trap efficiency. Impoundments trap or retain nutrients by two processes: (1) sedimentation of nutrient-bearing particulate matter and (2) transformation of soluble (dissolved) nutrient forms to particulate forms, which are then subject to sedimentation. Soluble nutrient forms may be converted to particulate forms by adsorption, precipitation, or uptake and transformation by organisms (plants, bacteria). In particular, photosynthesis by aquatic plants, which is often favored in the lakelike environment of impoundments, results in conversion of soluble nutrients to particulate matter (Neel 1963, Bachman 1978). In impoundments with high concentrations of suspended sediment input from tributary streams, adsorption of dissolved plant nutrients on suspended sediment, which subsequently settles, can also be an effective mechanism of nutrient retention by impoundments (Wang 1974, Gill et al. 1976). Finally, the chemical transformations that occur in hypolimnetic waters of stratified impoundments during summer and early fall, particularly the solubilization of iron, can result, under some circumstances, in significant removal of soluble phosphate by precipitation of ferric phosphate.

The factors affecting the nutrient trap efficiency of reservoirs are less well studied than those affecting sediment trap efficiency. However, nutrient trap efficiency appears to vary in response to the same factors that affect sediment trap efficiency and in response to sediment trap efficiency itself (Prochazkova 1975, Gill et al. 1976, Rausch and Schreiber 1977). Thus, any changes in water retention time, which is the primary determinant of sediment trap efficiency, induced by water level fluctuations will also affect nutrient trap efficiency.

Significantly, although no intensive efforts have been directed at developing a capability for predicting nutrient retention by impoundments, retention of phosphorus by natural lakes has been extensively studied (e.g., Kirchner and Dillon 1975, Dillon 1975, Chapra 1975, Larsen and Mercier 1976). Phosphorus has been viewed as the key nutrient responsible for accelerated eutrophication of natural lakes (Vollenweider 1968, National Academy of Science 1969, American Society of Limnology and Oceanography 1972). The quantity of phosphorus trapped annually in the sediments of natural lakes in relation to the quantity of input phosphorus determines the steady-state concentration of phosphorus in lake water. Thus, this parameter is an important component of several numerical models of lake eutrophication (Vollenweider 1975, Dillon 1975). Water renewal time, in various forms, is a principal variable in these models. Unfortunately, these models have not been very successful in accurately predicting the trophic state of impoundments with high flushing rates, but appear to be reasonably successful in predicting phosphorus concentration (Goodwyn 1975, Lind 1979). Limitation of plant growth by low light penetration may be a major reason for the failure of these models to predict the trophic state of these reservoirs.

Other factors, in addition to water renewal time, are important in determining nutrient trap efficiency of impoundments. As with sediment trap efficiency, morphometry and outlet characteristics may be especially important, although little empirical study has been conducted on the effects of these factors on nutrient retention. Wright (1967) concluded that impoundments with surface water outflow tend to trap nutrients, whereas impoundments with subsurface outflow tend to "dissipate" nutrients. Martin and Arneson (1978) presented data for two impoundments in Montana that strongly supported Wright's earlier conclusion.

In summary, the main factors affecting nutrient trap efficiency by impoundments appear to be water renewal time (flushing rate) and depth of the outlet. Thus, nutrient retention by an impoundment would be affected by imposing water level fluctuations only to the extent that water renewal time is affected. As with sediment trap efficiency, imposing water level fluctuations on a formerly unfluctuated reservoir would directly affect water renewal time as a consequence of changes in the average storage volume. The overall long-term effect would probably be a reduction in nutrient trap efficiency.

# 3. IMPACTS OF WATER LEVEL FLUCTUATIONS ON BIOLOGICAL CHARACTERISTICS OF RESERVOIRS

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Reservoir ecosystems can be structurally similar to lake ecosystems, regardless of basic differences in morphometry and water renewal times, if water level fluctuations do not exceed those induced naturally. Both lakes and reservoirs have a tendency to stratify if sufficiently deep, and both have similar ecological zones that can be affected by water level fluctuations. Accordingly, the terms and definitions of ecological zones developed from classical studies of lake communities have also been applied to reservoirs. The major ecological zones defined in most limnological texts (e.g., Hutchinson 1967, Reid 1961, Ruttner 1953, Welch 1952) consist of (1) the open water area (limnetic or pelagic zone), (2) the bottom area below the level of light penetration (profundal or bathyl zone), and (3) the bottom area within the level of light penetration (littoral zone). While definitions of the littoral zone and subzones vary, most definitions are highly influenced by classical studies of lakes with shorelines protected from wind and wave action and nonfluctuating water-level conditions. These conditions promote high biological productivity in the littoral zone. Thus, the most obvious effect of water level fluctuation is modification of the littoral zone from an area supporting luxuriant (sometimes nuisance) vegetation to an area of barren substrate exposed during low water levels.

Section 3.1 emphasizes the effects of fluctuating water levels on the biology of the littoral zone, not only because it is the obvious zone affected, but also because the effects are reasonably well documented. Effects of water level fluctuations on other zones in a typical reservoir system will be addressed in Sect. 3.2. Discussions on impacts other than those resulting from exposure of the littoral area are more speculative because of the small number of available studies directly addressing those topics. However, our speculations are inferred from basic studies of reservoir ecology and fisheries biology and from the physical and chemical effects discussed in Sect. 2.

Additional factors affecting biological response to water level fluctuation are the timing and frequency of fluctuations. Operation of small hydroelectric facilities may result in a regime of daily fluctuations, annual fluctuations, or both. The extent of biological change that occurs will depend, in part, on previous fluctuation levels and timing. Most studies of biological responses to water level fluctuation have been conducted at sites where seasonal fluctuations predominate, although the timing of high and low water periods has varied considerably. Relatively little information has been gathered on the effects of more frequent (e.g., daily) fluctuations. Discussions in this area will be speculative, but the information gained from data on seasonal fluctuations will be applicable.

## 3.1 Impacts on the Littoral Zone

### 3.1.1 Characteristics and Significance of Littoral Zones

Welch (1952) defines the littoral zone simply as the area extending from the water's edge to the lakeward limit of aquatic vegetation. By inference, it is the zone where light penetration is sufficient for photosynthesis of benthic plants. Within that zone are several characteristic assemblages of aquatic macrophytes. Closest to the shore are the stands of emergent rooted vegetation. In slightly deeper waters, ranging from depths of about 10 cm to 2.5 m, plants with floating leaves predominate. From about 2.5- to 6-m depths is the zone of totally submerged hydrophytes. The submerged hydrophytes often form large dense mats in late summer. In areas where such vegetation zones exist, the substrate will often contain high levels of organic detritus resulting from dead and decaying plant material. The macrophytes also serve as traps for detritus entering the lake or reservoir from inflowing streams and shoreline areas.

Associated with the vegetation or any submersed stones and logs is a community of attached, but nonpenetrating organisms, which is called aufwuchs (Reid 1961). This community is composed typically of an assortment of unicellular and filamentous algae as well as various protozoans, bryozoans, and rotifers. Reid's (1961) definition of the aufwuchs community also includes organisms that move on the substrate, including roundworms, rotifers, annelid worms, crustaceans, and insects. Predatory stalking insects, such as dragonflies, damselflies, and water bugs are often abundant on and among the vegetation. As described by Reid (1961), the plankton of the littoral zone is typically rich in numbers of species. Part of this plankton may consist of organisms that have been displaced from the aufwuchs community. Some algal forms may often be more abundant in the littoral zone than in the limnetic area. The zooplankton of the littoral zone includes many forms that do not commonly occur in the limnetic region. These include mites, ostracods, cyclopoid and harpacticoid copepods, and certain cladocerans.

The benthic fauna associated with a littoral zone supporting aquatic vegetation is generally diverse and abundant. In tropical Lake Kariba, benthic biomass of bare mud zones was estimated to average 380 mg/m$^2$ (dry weight). With the invasion of macrophytes, the population density of benthic species increased by 500%. The number of species also increased from 30 to 49 species. The total biomass in mud flats resulted almost entirely from the presence of chironomid larvae, whereas the major faunal biomass in zones with aquatic plants was due to groups other than Chironomidae (McLachlan 1969).

The density of benthos in a nonfluctuating lake or reservoir is characteristically highest in the littoral zone, decreasing with depth (Grimas 1962). This decrease results partly from substrate changes and the decreased oxygen levels that occur with increasing depth as a result of stratification.

The actual extent of the littoral zone of reservoirs and lakes depends on the morphology of the reservoir, the nature of the bottom sediments, and light conditions. The contribution of the littoral zone to total productivity is greater in clear reservoirs, which have shallow, gently sloping shoreline morphometry compared with turbid reservoirs, which have little or no littoral zone. Whether extensive or limited, the littoral zone plays an important role in reservoir dynamics. Aquatic plants are used as a food source by many birds and mammals (Sculthorpe 1967).

Standing crops of rooted aquatic plants (macrophytes) vary greatly, depending on species and environmental conditions, but commonly range from 500 to 1500 $g/m^2$ (Boyd 1971). The digestibility of macrophytes as a food source varies greatly, with submerged and floating-leaf species having higher nutritive value than more highly productive, emergent plants (Boyd 1971). Aquatic organisms that feed directly on aquatic plants include crayfish, mollusks, some insects, and a few fish such as some carp species (Berg 1949, Smirnov 1961). Plants that are eaten include the fragile and succulent free-floating plants and some of the linear-leaved submerged species (Sculthorpe 1967).

An important secondary function of macrophytes is the provision of support and shelter. The growth of attached algae (periphyton), fungi, bryozoans, and chironomids on the stems and leaves of plants may amount to 100 to 500 g dry $wt/m^2$ during some periods of the year (Westlake 1966). While periphyton also grows on stones, logs, or any other submerged material, the large surface area provided by vegetation greatly increases production.

Shelter afforded by the littoral zone is used by fish as well as invertebrates. The combination of high invertebrate production, shelter, and warm temperatures probably explains why most fish species use the littoral zone as a nursery area (Table 1). In the absence of aquatic plants, juvenile fish still reside in the littoral zone, using whatever shelter (tree stumps or artificial reef) is available.

Table 1. Trophic position, feeding, spawning, and nursery habitats of fishes most commonly found in reservoir ecosystems

| Common name | Scientific name | Optimum water temp. range | Trophic position | Feeding habitat | Nursery habitat | Spawning | |
|---|---|---|---|---|---|---|---|
| | | | | | | Habitat | Substrate |
| Paddle Fish | Polyodon spathula | Warm | Planktivore, filter feeder | Limnetic | ? | Tributaries | Gravel shoals |
| Bowfin | Amia calva | Cool-warm | Predator, omnivore | Littoral/benthic | Littoral | Littoral | Vegetation |
| Gars | Lepisosteidae | Cool-warm | Predator, piscivore | Littoral[a] | Littoral | Littoral/tributaries | Vegetation/gravel |
| Gizzard shad | Dorosoma cepedianum | Warm | Planktivore, filter feeder | Limnetic[a] | Littoral | Littoral | Indiscriminate |
| Threadfin shad | D. petenense | Warm | Planktivore, filter feeder | Limnetic[a] | Limnetic | Littoral | Submerged objects |
| Trout | Salmonidae | Cold | Predator, piscivore | Limnetic[a] | Littoral | Tributaries | Clean gravel |
| Pike | Esocidae | Cool-warm | Predator, piscivore | Limnetic[a] | Littoral | Littoral | Vegetation |
| Suckers | Catostomidae | Cool-warm | Omnivore | Littoral/benthic | Littoral | Tributaries/littoral | Gravel/vegetation |
| Carp | Cyprinus carpio | Cool-warm | Omnimore | Littoral | Littoral | Littoral | Indiscriminate |
| Minnows | Cyprinidae | Warm | Planktivore, omnivore | Littoral | Littoral | Tributaries/littoral | Gravel varied |

Table 1 (continued)

| Common name | Scientific name | Optimum water temp. range | Trophic position | Feeding habitat | Nursery habitat | Spawning Habitat | Spawning Substrate |
|---|---|---|---|---|---|---|---|
| Catfish | Ictaluridae | Warm | Predator, omnivore | Littoral[a] | Littoral | Littoral | Cavities under logs |
| Bass | Serranidae | Warm | Predator, piscivore | Littoral[a] | Littoral | Littoral | Gravel (white bass only) |
| Rock bass | Centrarchidae | Warm | Predator, omnivore | Littoral[a] | Littoral | Littoral | Gravel near vegetation |
| Sunfishes | Centrarchidae | Warm | Predator, omnivore | Littoral | Littoral | Littoral | Sand or gravel near vegetation |
| Sauger | Percidae | Cool-warm | Predator, piscivore | Littoral/ benthic | Littoral | Tributaries/ littoral | Gravel |
| Walleye | Percidae | Cool-warm | Predator, piscivore | Littoral/ benthic | Littoral | Tributaries/ littoral | Gravel |
| Drum | *Aplodinotus grunniens* | Cool-warm | Predator, omnivore | Littoral/ benthic | Littoral/ benthic | Limnetic | None |

[a]These species may temporarily change their feeding habitats when the opportunity provides easy prey.

*Source:* Carlander (1969 and 1977), Pflieger (1975).

The successful reproduction of many fish species in lakes and reservoirs depends on their finding suitable spawning areas within the littoral zone (Table 1). Some fish, however, are much more specific in their requirements than others. For instance, carp and gizzard shad spawn in shallow water, but are indiscriminant as to substrate. The short nose gar, pikes, golden shiner, and bigmouth buffalo appear to require submerged vegetation (either aquatic or terrestrial) for spawning (Carlander 1969). Most centrarchids build nests in shallow water in sand or fine gravel near stumps or vegetation clumps. Vegetation may be critical to black crappie spawning (Carlander 1977). Studies on European and Russian reservoirs also indicate that the littoral zone is the spawning and feeding area of almost all commercial fish (Makhotin 1977).

### 3.1.2 Impacts of Water Level Fluctuation on Vegetation (and Associated Biota) of the Littoral Zone

The amplitude, frequency, duration, and timing of water level fluctuations interact to determine the magnitude of vegetation reduction in the littoral zone. Lantz et al. (1967) found that 90% or more of the vegetation was eliminated by lake drawdown if the drawdown lasted for 3 or 4 months during a winter or summer period. Neither a single short drawdown nor a spring drawdown was effective in eliminating vegetation. Quennerstedt (1958) determined that inundation of emergent species (for more than 3 to 4 weeks) as well as desiccation of submergents reduced vegetation zones. He further observed that, when water level fluctuations decreased, some plant species expanded their distribution.

Many rooted macrophytes, particularly emergent or floating-leaved species, can reproduce vegetatively by such structures as rhizomes, rootstocks, or tubers (Schulthorpe 1967). In some plants, these vegetative parts also serve to allow the plants to survive periods of drying, although the extent and season of desiccation are important factors. Tropical plants are adapted to surviving dry seasons while temperate plants rely on these structures to survive cold winters. Thus, a drop in water level after the summer growing season in temperate climates may not prevent redevelopment of macrophyte beds in the following spring. Large seasonal water level fluctuations can be expected to be more effective in reducing vegetative growth than frequent small fluctuations. However, most fluctuating regimes can reduce littoral vegetation.

The attached algae and zooplankton associated with the aquatic vegetation (and with the aufwuchs community living on the vegetation), would be reduced by water level fluctuation in a similar manner as the aquatic vegetation. These groups may be able to rebuild their populations quickly after favorable conditions reappear. Many of the zooplankton species can reproduce parthenogenetically or have resting stages or both (Pennak 1953). Inundation of any submerged structures would likely be followed within a few weeks by a rapid increase in planktonic forms and parts of the aufwuchs community.

### 3.1.3 Impacts of Water Level Fluctuation on Benthic Macroinvertebrates of the Littoral Zone

The macroinvertebrates commonly associated with aquatic vegetation and detritus (including Diptera, Trichoptera, Odonata, and Amphipoda) are the most severely affected by water level drawdowns. These invertebrates cannot survive desiccation by forming a resting stage or burrowing. Attempts to migrate to deeper waters during drawdowns probably result in significant predation mortality. Aggus (1979) states that water level drawdowns result in rapidly increasing prey availability for fish. The low macroinvertebrate diversity in fluctuating reservoirs, noted by several investigators (Fillion 1967, Grimas 1962, Kaster and Jacobi 1978, Hruska 1973), is largely because of the absence of those forms that rely heavily on the substrate and shelter offered by aquatic vegetation.

Redevelopment of hydroelectric power generation on existing reservoirs (if the height of the dam is increased) could potentially result in an initial increase in mean water level and subsequent water level fluctuations about a new mean water level. Changes in the littoral benthic fauna of such a situation have been described by Hynes (1961) and by Hunt and Jones (1972). The water level in Llyn Tegid increased by 2 m, resulting in a change in amplitude from 2 to 4.5 m. Hynes (1961) observed that, as a result of this new regime, the rocky-sandy littoral zone became covered with silt, which resulted in the disappearance of rooted macrophytes and a drastic decrease in the diversity, but not the total density, of benthos. He observed that sponges, flatworms, oligochaetes, leeches, gastropods, Gammarus (scuds) mites, stoneflies, mayflies, and caddisflies were almost or completely eliminated. The loss of these organisms was balanced by a gain in two oligochaete species and several species of chironomids. Hunt and Jones (1972) also found that the density of benthos in Llyn Tegid increased from 1504 to 6488 individuals/$m^2$ in the preregulated periods to 3654 to 9224 individuals/$m^2$ after the increase in both water level and water level fluctuations. The authors state that this increase was accounted for by chironomids and oligochaetes. The increase in the number of chironomids and oligochaetes was attributed to the changes in substrate, specifically to the accumulation of silt and subsequent disappearance of vascular macrophytes.

The organisms that universally appear to be able to survive or take advantage of water level fluctuations are chironomids and oligochaetes. The ability of chironomids and oligochaetes to withstand desiccation of the substrate has been reported by several investigators. Kaster and Jacobi (1978) reported that larvae of Chironomus plumosus were found active at a depth of 8 cm in substrates exposed to air for 21 days. No benthos were observed below 9 cm in substrates exposed to air. However, C. plumosus larvae were found alive at a depth of 15 cm in substrate covered with ice remaining from the receding water during the winter. Chironomid tunnels in the substrate with ice cover were found as deep as 20 cm. Similarly, Limnodrilus specimens were also found burrowing deeper in air-exposed sediments, with smaller individuals being more successful. Grimas (1961) reported that 80% of the chironomid and oligochaete fauna were alive after three months of exposure to ice cover during a winter drawdown in Lake Blasjön. Kaster and Jacobi (1978) further observed that the rate of survival of these organisms, when exposed to air, was higher in organically rich sediments than in sand and silt. However, survival was higher in sand and silt than in organic detritus, when exposed to ice cover, because organic detritus freezes faster and deeper.

Fillion (1967) reported that several organisms were able to tolerate conditions of exposure during winter drawdowns of 50 to 85 days. Objects such as logs, roots, and mats of vegetation sheltered many chironomids and, occasionally, Megaloptera, Plecoptera, and oligochaetes. Tipulidae larvae were found under rocks. Chironomids were also found, alive in the drier, hard-packed mud, until inundation occurred (up to 85 days).

Paterson and Fernando (1969) studied the effects of desiccation on benthic organisms in a small (65-ha), shallow (maximum depth of 3 m) reservoir in southern Ontario, which was entirely drained beginning in September and filled again the next March. This schedule exposed the benthos to desiccation above freezing for about 50 days and to desiccation at temperatures below freezing without any ice cover for an additional 100 days. During this 100-day period, the substrate froze to a depth of 20 cm. In this experiment, Paterson and Fernando (1969) found that the major portion of the benthic fauna, mostly oligochaetes and chironomid larvae, was destroyed. In comparing their results with that of Grimas (1961), who found survival of the benthos to be 80%, Paterson and Fernando concluded that chironomid and oligochaete populations of the littoral and shore area could survive winter drawdown in significantly large numbers by burrowing into the sediments if the exposed area is covered by ice or snow. If an ice or snow cover is lacking, organisms that burrow into the sediments would still die due to desiccation or freezing.

Clams, mussels, and snails, which occupy the littoral zones of lakes and reservoirs, should also be able to withstand some exposure. Several examples of survival of clams and mussels, for periods ranging from days up to one year, were reported by McMahon (1979). Variation was related to species differences, relative humidity, and temperature. Most clams, mussels, and snails are also capable of burrowing into the substrate to avoid desiccation (Pennak 1953). Burrowing mayflies can also withstand desiccation for a few days by burrowing. However, Hexagenia species have been observed to desert their burrows and attempt to migrate in response to exposure of the substrate (Fremling 1960).

Although several organisms can survive exposure of substrate under some conditions, the abundance of benthic organisms appears to be lower in the zone subject to fluctuation. Investigations conducted by Grimas (1962) on two reservoirs in northern Sweden within the same drainage system demonstrated that, in the unregulated lake, the greatest density of invertebrates was found in the 0- to 4-m depth zone. In the regulated lake with fluctuations of 0 to 6 m, density of invertebrates was greatest between 6 and 10 m. This pattern of highest densities of benthos immediately below the low-water mark has been noted by several investigators (Fillion 1967, Nursall 1952, Hruska 1973). Desiccation of benthos exposed in the zone of fluctuation is only one factor contributing to this distribution pattern. As discussed in Sect. 2.1, factors such as wave action resuspend and redistribute substrate materials, and erosive forces may cause undercutting and collapse of banks. Examples of such effects on benthos are given in Cowell and Hudson (1967) and Grimas (1962). Wave action transfers the silt, clay, and organic matter from the zone of fluctuation to the area below the lower limits of drawdown. Thus, organisms that are best suited to silt and clay and organically rich substrates are able to survive or even increase in numbers below the lower drawdown limits.

Changes in the distributional pattern of macroinvertebrates may have far-reaching ramifications. Studies in Lake Francis Case (South

Dakota), which operates with a 6- to 14-m drawdown annually, indicate that several fish species (bluegill, orange spotted sunfish, green sunfish, largemouth bass, black bullhead, stonecat, and flathead chub) became rare (Gassaway 1970). Although the authors did not specifically state the reasons for this decline, the water level fluctuations and possible effects on the benthic and aufwuchs community are a possible cause. Most of these fish are known to depend heavily on benthic invertebrates as a food source. Isom (1971), in describing the effects of water storage on benthos in the mainstream reservoirs of the Tennessee Valley, indicated that benthic invertebrates were of little importance since a truncated plankton-to-fish chain is common in storage impoundments. The secondary effect on fish populations resulting from reductions in benthic invertebrates caused by water level fluctuation merits further research.

### 3.1.4 Impacts of Water Level Fluctuation on Fish Using the Littoral Zone

The best-documented effect of water level fluctuations on fish populations relates to fish spawning success. Il'ina (1962) determined that the extremely low water levels in Rybinsk Reservoir in the spring of 1960 exposed all the vegetation and preferred spawning grounds of most species. Spawning time was greatly prolonged for all species, and a great part of the brood stock did not spawn. Phytophilic species (those that spawn over vegetation), including white bream, blue bream, and pike, were severely affected. Poddubny (1976) reported that the loss of eggs laid by phytophilic species (pike) on certain substrates was as high as 70 to 100% in Khakhoyskii Reservoir. Egg loss was attributed to predation, desiccation, and silt deposits. Nelson (1978) determined that the decline of northern pike and yellow perch in Lake Oahe can be linked to the reduction of the vegetational spawning habitat. He further indicated that maintenance of instream flows in the tributaries was important to sauger and walleye spawning success. In Lewis and Clark Lake, more than 80% of the variability in year-class strength could be predicted from the change in water level over the spawning ground, the reservoir water temperature in June, and the reservoir water exchange rate (Walburg 1972).

The effect of water level fluctuations on spawning is not limited solely to lack of spawning habitat. Water level fluctuations can also influence temperature conditions in the littoral zone. Makhotin (1977), discussing factors that determine spawning efficiency in Kuybishev Reservoir, Russia, noted that the water level regimes directly influence the temperature condition of the water body. Inundation of shallow areas during spring floods results in rapid warming, stimulating the spawning of phytophilous fish and spawning migrations. Conversely, disruption of water warming patterns in the spawning grounds and the absence of requisite substratum lead to large-scale resorption of gonads by spawners or mortality of fertilized eggs due to detachment. Poddubny (1976) reports that studies of gonadal resorption in phytophilic fishes in Rybinsk Reservoir have established that the number of resorbed gonads depends directly on three factors: (1) the actual area of spawning ground, (2) the temperature conditions in spring, and (3) the number of spawners that resorbed gonads in the previous year and released spawn in the next. In particularly unfavorable years, the number of fish resorbing gonads reaches 44% in bream and 60% in blue bream. The timing and amplitude of the fluctuations are particularly critical in determining effects on spawning.

### 3.1.5 Impact of Fluctuations on Substrate Stability

As discussed in Sect. 2, water level fluctuations may result in increased turbidity conditions, especially along the shore. Although such increases may be slight compared with the effect of storm events, it is nevertheless an added stress that could contribute to the poor spawning success of some species. A literature review of the effects of suspended solids on fish (Muncy et al. 1979) concluded that substantial evidence indicates that reproductive behavior varies among warmwater fish in response to suspended solids and sediments. Fish with complex patterns of reproductive behavior are vulnerable to interference by suspended solids at several critical behavioral phases of the spawning process. Short-term exposure to high levels of suspended solids probably does not seriously impede reproductive movements of most warmwater fish (Muncy et al. 1979). Thus, turbidity induced by water level fluctuations of short duration would probably not interfere with reproductive behavior. However, Muncy et al. (1979) report that walleye, northern pike, and yellow perch do sometimes suffer mortality at naturally occurring concentrations of suspended solids. Loss of spawning habitat is the primary impact of water level fluctuations on these phytophilic fish. Increased turbidity may be just one of many factors that cause low spawning success when the eggs are deposited on inappropriate substrates. If water level fluctuations continually alter the stability of littoral substrates, macrophyte colonization on the substrates may be reduced. Indirect effects of increased turbidity levels, including decreased photosynthesis and adverse effects on zooplankton, may occur if the turbidity increase is extensive.

In summary, water level fluctuations can be expected to alter the growth of macrophytes in the littoral zone. This may be beneficial if growth of undesirable species is retarded, or it may be detrimental if growth of a desirable species is retarded. Reduction in macrophytic growth can result in a decreased diversity and abundance of preferred "fish food" organisms. However, the abundance of burrowing benthic organisms may actually increase in reservoirs subject to water level fluctuation if the substrate is suitable. The most serious effect of exposure of the littoral zone is in the reduction of suitable spawning habitat for fish, although the reduction in food and shelter may also limit recruitment of young-of-the-year fish. An initial step in determining the potential significance of water level fluctuation at small hydroelectric sites where reservoir water levels will fluctuate would involve estimating the habitat area affected by such fluctuation. Appendix A details a quantitative methodology to estimate loss of shore zone habitat area as a result of reservoir water level fluctuation.

## 3.2 Impacts on Nonlittoral Zones

In this section, potential biological impacts in areas of a reservoir other than the littoral zone that result from water fluctuation are presented. The potential significance of the impacts discussed are to some degree speculative.

### 3.2.1 Impacts Resulting from Fluctuating Water Volume

Changes in the surface area and volume of a body of water may result in an altered carrying capacity for fish (Aggus 1979). Reductions in water volume may concentrate the fish and subject the smaller and younger individuals to increased predation and competition for food. Consequently, fisheries managers have successfully used water level drawdowns as a means of selectively culling excessive prey species (McGammon and von Geldron, Jr. 1979; Lantz et al. 1967; Bennett 1971). However, the predator fishery resource is enhanced by such drawdowns only if drawdowns are conducted once every few years (Bennett [1974] recommends once every 3 to 4 years, whereas Keith [1975] recommends once every 5 to 10 years). These recommendations imply that more frequent drawdowns would be detrimental to predators. Aggus (1979) states that large fluctuations that occur annually or more frequently create instability in the fish community, resulting in a constantly changing standing crop. No information is available on the effects of small, but frequent fluctuations on predation pressure.

At the least, however, one would expect greater predation pressure on the young-of-year fish compared with nonfluctuating conditions because they would regularly be drawn out of protective cover.

Another mechanism by which small, shore-oriented fish may be affected by water level fluctuation is stranding in temporary water pockets, which later dry up. Green sunfish, young bluegills, and many of the invertebrates that attempt to follow the receding water line have been reported to be susceptible to stranding (Bennett 1971).

Decreasing the volume of water in an impoundment may change its physical characteristics. Section 2 discusses the possibility that, at low volume, the thermocline may be altered or total destratification may occur. Although the probability of this situation occurring is low, biological implications of such an event include increased oxygen levels in the profundal zone, which could result in changes in the distribution and abundance of aquatic biota. Destratification of a lake by aeration results in increases in the benthic populations (Wirth et al. 1970). Destratification in this case was beneficial to the benthic community.

Destratification may also have some adverse effects on the fish community. In general, juvenile fish have higher optimum temperatures than adult fish of the same species (Coutant 1977). This differing thermal preference contributes to possible separation of young-of-the-year fish from adults when stratified conditions allow each group to find its preferred temperature. Destratification may bring different size groups together and thus result in a greater amount of cannibalism. However, no data are available which would directly indicate that this would occur. In a homothermal lake in New York, walleye did cannibalize their young-of-the-year heavily during one year of study (Chevalier 1973), whereas other studies on walleye have indicated that cannibalism occurs infrequently. However, other factors, such as low densities of the normal prey, may have contributed more significantly to cannibalistic behavior than the lack of stratification.

### 3.2.2 Impacts Resulting from Timing of Water Releases

Operation of small hydroelectric operations in a peaking mode may result in daily fluctuations of water. The timing of the fluctuations and the discharge may determine the extent to which fish and invertebrates are lost through turbines. Cowell and Hudson (1967) found that up to 24.1 metric tons of *Hexagenia* (burrowing mayfly) were lost through the turbines of Lewis and Clark Lake between April 12 and July 1 when *Hexagenia* were migrating through the water column. Since migration was strictly nocturnal, the losses occurred at night. Chironomids and ceratopogonids were also lost through turbines at night. The percentage of the total *Hexagenia* population lost through the turbines was estimated to be 7 to 10%. The total annual loss of chronomids was estimated to be nearly equal in weight to the loss of *Hexagenia* mayflies (about 20 metric tons). Benson (1973) reported that, in Lewis and Clark Lake, chironomids were 3 to 8 times more numerous in night samples than in day samples, and burrowing mayflies were collected only at night. Thus, peaking during evening hours could result in a huge loss of invertebrate biomass from a reservoir, whereas midday peaking may have little effect on export of invertebrates from a reservoir. Ichthyoplankton were also lost from Lewis and Clark reservoir through the turbines (Walburg 1971), but this loss occurred at all hours and would not be particularly affected by the timing of peaking operations.

### 3.2.3 Impacts Resulting from Changes in Flushing Time

Imposing water level fluctuations on a formerly unfluctuating reservoir will directly affect the water retention time, as described in Sect 2.1. Decreasing the water retention time may create current velocities that small fish are unable to resist or avoid. In Lewis and Clark Lake, where water retention time varied from 5.5 to 7.2 days depending on water level, losses of young fish from the reservoir were extensive. Estimated peak 24-h losses were 10 million freshwater drum, 800,000 emerald shiner, 700,000 sauger-walleye, and 170,000 channel catfish. Summer survival of age 0 freshwater drum was inversely related to the flushing rate in July and August, when the current velocities approached 3 to 5 cm/s in the reservoir (Walburg 1971).

Water retention time is an important factor determining the amount of phytoplankton and zooplankton produced in a reservoir. In studies on the Vltara cascade of reservoirs in Czechoslavakia, Straskraba and Javornicky (1973) reported that the primary production and zooplankton standing crop in reservoirs with mean retention times above 20 days were comparable with those for lakes. However, in reservoirs with retention times below 3 days, the primary production varied according to whether inflowing water came from the lower strata of an upstream reservoir or from the surface water of a river. In either case, however, the zooplankton production was very low in reservoirs with retention times less than 3 days. Zooplankton production in Lake Sharpe, one of the Missouri River mainstem reservoirs, was also found to be reduced when retention time was reduced from a range of 26 to 50 days to 18 to 22 days (Benson 1973). The effects were most apparent on *Cyclops* and *Daphnia*, both important fish food organisms. Reproduction of daphnids in Lewis and Clark Lake showed a large decrease, with a change of mean retention time from 8.6 to 6.7-6.4 days. The somewhat limited data available indicate that, if hydroelectric power generation decreases retention time in a reservoir, the production of zooplankton, in particular, and probably the overall productivity will be decreased.

In summary, the types of potential impacts discussed above are more speculative and less well documented than the impacts discussed for the littoral zone. However, if the operation of a hydroelectric facility does significantly alter water volume or flushing rate, some of the impacts discussed above could adversely affect reservoir productivity.

# 4. IMPACTS OF WATER LEVEL FLUCTUATION ON PHYSICAL AND CHEMICAL CHARACTERISTICS OF RIVERS DOWNSTREAM FROM DAMS

R. R. Turner

This section deals with the physical, chemical, and biological effects of water level fluctuations in the lotic environments downstream from hydroelectric impoundments. More particularly, it describes the potential effects of those water level fluctuations that occur as a consequence of hydropower generation in a peaking mode of operation, i.e., those fluctuations that are not inherent in the natural (run-of-river) flow from impoundments.

Although the amplitude of hydropower water level fluctuations may be equivalent to the unregulated amplitude of the unimpounded stream, peaking operations usually require a rather drastic alteration in the interval between major fluctuations, the rate of change of water level, and the duration of a given water level (stage height) in the downstream channel. Thus, in contrast to fluctuations within reservoirs (Sect. 2), the effects of fluctuations downstream from impoundments are related more to the interval between and rate of change in water level fluctuations than to the amplitude of fluctuation. Unregulated (i.e., unimpounded) streams, especially the smaller ones, can be subject to enormous natural fluctuations in water level in response to rainfall in their watersheds.

Nonhydroelectric uses (e.g., flood control, irrigation) of impounded streams often reduce the natural amplitude of water level fluctuations in tailwaters by retaining peak flows and augmenting low flows. Fluctuations related to peaking operation ordinarily increase amplitude to near the natural (nonflood) range without any relationship to the natural frequency, timing, and duration of the occurrence of a particular water level in the downstream environment.

The primary physical effects of water level fluctuation downstream of impoundments operated in a peaking mode are expected to include streambed and bank erosion and water quality problems related to resuspension and redistribution of bank and bed materials. Specifically, hydroelectric operation of a reservoir not previously used to meet peak power demands may increase erosion and downstream transport of streambed and bank materials, thus increasing turbidity and altering channel geometry. Also, subsequent redeposition of eroded material further downstream may be deleterious to benthic biota (Sect. 5) and may alter channel geometry by infilling.

In colder regions, daily water level fluctuations can also keep ice broken up for considerable distances below impoundments. Often this physical effect may be augmented in the stream reaches immediately below impoundments by the release of comparatively warmer reservoir bottom water, which retards ice formation (Neel 1963).

### 4.1 Impacts on Streambed and Bank Erosion

Changes in the rates of erosion and sedimentation in streams downstream from dams can have far-reaching and sometimes costly consequences. Accelerated rates of erosion can undermine bridge piers and abutments. Channel degradation of streams that supply water to communities or industry can be serious if, at normal flow, it lowers the water level below the design level of intake structures. The recreational value of some stream reaches may be affected by loss of beaches and boat landings because of accelerated erosion in one area and excessive siltation in another. Degradation of a main stream channel may also have repercussions on the rates of erosion and sedimentation in tributary streams by lowering the local erosional base level for these smaller streams.

The rate of erosion of streambeds and banks depends, among other things, on complex interrelationships of stream velocity, water quality (especially suspended sediment concentration), channel gradient, types of materials composing the streambed and banks, channel configuration, and channel alignment. An exhaustive treatment of these interrelationships is beyond the scope of this document. However, Gottschalk (1968) and Taylor (1978) have provided lucid treatments of erosion and sedimentation processes as they occur downstream from dams, which can be consulted for detailed information.

Where streams have been impounded, the downstream balance between erosion and sedimentation is often disrupted, with the result that formerly stable streambeds and banks may begin to erode and areas further downstream with minimal or no sedimentation may begin to fill with sediment. These changes can be attributed mainly to (1) the discharge of water from dams with greatly reduced sediment load (Sect. 2.4) and (2) altered periodicity and amplitude of water levels and velocities in the downstream area.

Sediment-free water discharged from a dam can be aggressively erosive. Also, if such water is released in sufficient quantity into a channel with a gradient that has become stabilized under prior conditions of the flow of sediment-laden water, active scour of the streambed and bank will begin. This scour will normally begin near the dam, but as the channel gradient decreases, the eroding area will move downstream. Below large dams, scouring may progressively affect many miles of streambed and banks (Borland and Miller 1960).

Alteration of periodicity, duration, and amplitude of water levels and velocities in streams as a consequence of river impoundment, or changes in the operation of an existing impoundment, may also profoundly affect erosion and sedimentation processes in the downstream area. Streambeds and banks tend to evolve to a stable condition in response to the range of hydraulic forces that are applied. Where the nature and strength of these forces are altered, some readjustment of streambeds and banks is likely. Thus, streambeds and channel banks that are stable (neither aggrading nor degrading) under a regimen of unregulated streamflow may become highly unstable under a regimen of regulated flow. Instability may occur as a consequence of changes in stream turbulence or excessive pore water pressure in channel banks. Stream turbulence determines erosive capacity and is partly determined by the rate of rise in water level.

Artificially created surges (sudden rises in water level), such as those caused by peaking operation of hydroelectric facilities, may greatly increase stream turbulence as the power wave passes a point in the stream and may accelerate erosion of the streambed and banks (Brye et al. 1979).

The presence of groundwater in the banks (bank storage) and floodplains adjacent to a channel in alluvium which creates high pore water pressures, can significantly affect the stability of channel banks. Groundwater may enter banks as a result of elevations of the areal water table or as a result of stream water percolating into the bank when the water level is high. Seepage of bank storage back into the stream when water levels are lower creates conditions conducive to collapse and erosion of the channel bank (Burgi and Karaki 1971). Rapid and frequent fluctuations in the surface level of the stream, such as may accompany hydroelectric peaking operations, are especially conducive to channel bank erosion because streamflow may vary from essentially zero (dry channel) to full discharge and back to zero in a matter of hours. Although natural, unimpounded streams and streams below dams operated in run-of-river mode may fluctuate their water levels over similar ranges in response to storm flow, these natural fluctuations in water level are neither as frequent nor as rapid as may occur below dams operated to meet peak power demands.

Stream channels, which cut through bedrock or which are armored naturally or artificially against further erosion by virtue of the large particle size of the bank and bed materials, will likely not be affected by altering the historical periodicity and amplitude of water levels in the channel. Also, even where the types of materials composing the streambed and banks are subject to erosion and redistribution, any recently imposed hydraulic conditions will ultimately lead to a new quasi-equilibrium between erosion and sedimentation processes. The time required for channel readjustment will be highly site-specific and difficult to predict. Where downstream water levels are also regulated by dams or other backwater-creating conditions, the effects of water level fluctuations are more likely to be of the types described in Sect. 2.2.

In summary, water level fluctuations below dams converted from run-of-river to hydroelectric peaking operation may in some circumstances lead to disruption of the downstream balance between erosion and sedimentation. Formerly stable streambeds and banks may begin to erode, whereas formerly silt-free stream reaches may suddenly be subject to excessive siltation. These effects may be chiefly attributed to the altered periodicity, duration, and amplitude of downstream water levels, which force the stream channel to readjust to the imposed hydraulic conditions.

# 5. IMPACTS OF WATER LEVEL FLUCTUATION ON BIOLOGICAL CHARACTERISTICS OF RIVERS BELOW DAMS

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Tailwaters are defined as the portion of streams extending from below dams to their confluence with equal or larger-size tributaries or to the headwaters of other downstream impoundments. The biological community structure in tailwater streams is often altered considerably from the community in an unregulated section as a result of changes in physicochemical characteristics of the water discharging from a reservoir.

A series of papers by Ward (1974, 1976<u>a</u>, 1976<u>b</u>, 1976<u>c</u>) and Ward and Stanford (1979<u>a</u>, 1979<u>b</u>) provides excellent research data and literature reviews on the effects of dams on stream benthic invertebrates. They found that, within temperate regions, temperature and flow regimes are often the major factors affecting benthic communities below dams. Other factors that determine species-specific response to stream regulation include properties of the reservoir (trophic status, depth retention time, temperature profile, and extent of drawdown) and characteristics of the stream (geochemistry, topography, and meteorology of the region under consideration). Channel morphology may also be of critical importance, especially to fish.

The properties of the reservoir and the location of the discharge largely determine the water quality of tailwater streams. Epilimnetic discharge is generally characterized by relatively high densities of planktonic organisms, adequate dissolved oxygen, and a paucity of dissolved plant nutrients. During periods of reservoir stratification, a hypolimnetic discharge may contain low dissolved oxygen, relatively high concentrations of plant nutrients, and other dissolved inorganic solids (iron and manganese), and thus it may degrade water quality in tailwaters (Krenkel et al. 1979). Water quality degradation can affect the biotic communities in the receiving streams, with or without fluctuating water levels.

To establish a basis for discussing tailwaters affected by fluctuating water levels, tailwaters characterized by relatively constant flow conditions are first described. Although tailwater discharges are seldom constant year-round, they may be considerably dampened compared with unregulated streams. Tailwaters below dams with hypolimnetic discharges are usually characterized by relatively constant thermal conditions. Therefore, water temperature may be cooler in summer and warmer in winter compared with that of unregulated streams. Diurnal fluctuations in temperature are also dampened where constant and medium to high discharges are maintained. Separating the effects of flow constancy from thermal constancy is difficult, because they occur together so frequently. Our description of tailwaters with relatively constant flow will address the effects of thermal constancy as well. However, in our discussions of effects of water level fluctuations, only those thermal factors that are influenced by flow changes will be addressed.

## 5.1 Biological Communities of Tailwaters Under Relatively Constant Thermal and Flow Regimes

Reviews of the literature by Ward (1976b) and Ward and Stanford (1979b) have shown that macrobenthic diversity is generally reduced in the immediate tailwaters below storage reservoirs with deep releases compared with that of the stream above the reservoir, of unregulated tributaries, or of tailwaters further downstream. Various investigators have attributed the lower species diversity of macroinvertebrates in tailwaters to the seasonal thermal constancy below deep release dams (Fraley 1979, Spence and Hynes 1971a, Lehmkuhl 1972, Ward 1974). Ward (1976c) further implicated delayed seasonal temperature maximum and diurnal temperature constancy in the effects on diversity. He identifies several effects of the altered thermal regimes: (1) thermal stimuli essential to complete the life cycle are lacking, (2) niche overlap is reduced, (3) competition associated with greater productivity is increased, (4) major invertebrate predators are eliminated, and (5) optimal temperatures for growth are lacking.

Many factors associated with flow constancy are favorable to invertebrates (Ward 1976b; Ward and Stanford 1979b). For instance, increased bank stability favors the establishment of riparian vegetation. Streamside vegetation provides allochthonous organic matter, which is often a significant food source for stream benthos. Both detritivores and filter feeders may directly benefit from such input.

Flow constancy increases bed stability and reduces turbidity, enhancing the production of attached algae and macrophytes and providing additional food and niche diversification (Ward 1976b, Ward and Stanford 1979b). For instance, dense stands of Cladophora often inhabit tailwater areas below TVA dams and provide shelter for large populations of chironomids and amphipods (Pfitzer 1954). Amphipods are poorly adapted for withstanding current, but the constant flows plus shelter favor their occurrence.

The negative effect of seasonal flow constancy is that siltation is occasionally a problem, especially if the current velocity is slower than normal for that stream. Silt gradually fills the gravel interstices, severely reducing benthic habitat and destroying spawning habitat for fish. Stream habitats with little cobble and large amounts of sand and silt were found to have low benthic biomass and species diversity by Brusven and Prather (1971). Hilsenhoff (1971) attributed reduction in species diversity below an impoundment to a combination of nutrient enrichment and increased siltation.

Ward (1976b) summarizes the benthic communities commonly found in streams in which siltation and water quality are not a problem and in which severe daily fluctuations are absent:

> The stream below a deep release dam with a seasonally constant discharge pattern will likely contain dense benthic algae and macrophytes and a rich benthic fauna with low diversity. Chironomids, amphipods, oligochaetes and snails will very likely be present. Certain mayflies may be very abundant, but those utilizing holdfasts will be absent. Stoneflies will probably be absent immediately below the dam. Surface release will modify the fauna somewhat by enhancing filter-feeding benthos.

Streams with a reduced benthic diversity resulting from thermal constancy also tend to have less diverse fish communities. Fish are either eliminated directly as a result of the negative effects of cool summer temperatures on their growth and reproduction or indirectly by elimination of their food resource (Spence and Hynes 1971b, Holden 1979). The introduction of coolwater fish (e.g., rainbow trout) for their sport quality further hastens the elimination of warmwater species. Although tailwater conditions are usually not suitable for trout spawning, the populations are normally maintained by stocking. In streams characterized by flow constancy as well as thermal constancy, increased siltation is often the cause of reduced trout reproduction.

Many factors that affect fish communities in tailwaters result from the dam itself rather than changes in flow or thermal regimes. High densities of fish below dams may be caused by upstream spawning migrations, where the dam acts as an effective barrier for further movement. Conditions are usually not favorable for fish spawning immediately below dams; therefore, the spawning effort of these species may be considerably hindered and wasted. Gas supersaturation, a major mortality factor on young salmon and trout (Holden 1979), is another problem below large dams. Supersaturation is not as important below small dams because it occurs only when water plunges to depths at which increased hydrostatic pressures increase solubility.

Tailwater communities below surface release dams are a combination of stream and reservoir species because of the flushing phytoplankton, zooplankton, and benthos and juvenile fish from the reservoir. The flushing of juvenile fish from surface release reservoirs may be an important source of fish recruitment to the tailwaters (Walburg et al. 1971). It may also be a considerable food resource that attracts large concentrations of piscivorous fish to the tailwaters. Walburg et al. (1971) found that both the concentration and diversity of fish species in the tailwaters of Lewis and Clark Lake varied with the season and was related to the seasonal availability of food discharged from the reservoir as well as to spawning activities and water temperature. The phytoplankton and zooplankton flushed from the reservoirs may also significantly affect benthic invertebrates in tailwaters. Filter-feeding caddisflies (Trichoptera) and blackflies (Simuliidae) often develop high densities in areas where plankton concentrations are high and constant (Ward and Stanford 1979a).

## 5.2 Impacts on Tailwater Communities

Water level fluctuations in tailwater streams vary in amplitude and timing, depending on the mode of hydroelectric generation, the size of the dam, alternative uses of stored water, and the rainfall pattern in the area. If run-of-the-river mode of operation is used, then fluctuations depend entirely on factors such as rainfall and snowmelt. A peaking mode of operation in small impoundments normally requires daily pondage and storage. In extreme cases, little water may be released from the dam during storage, which results in severe reductions in wetted perimeters of the tailwaters, with only pockets of water remaining. In the more common situation, some minimum level of discharge is maintained for the tailwater stream. In either case, daily surges occur at least once or twice a day on week days, but not always on weekends (Krenkel et al. 1979). On larger multipurpose dams, where water is used for irrigation or flood control as well as power generation, the seasonal changes in water flows may be as significant as daily fluctuations in affecting the stream biota. The following discussions indicate under what type of flow regime the information presented has been obtained.

Water level fluctuations are discussed specifically in the context of their effect on tailwater communities generally described in the previous section. The underlying assumption is that retrofitting of hydroelectric power generation on small dams will impose fluctuating conditions on a previously nonfluctuating tailwater environment. Topics to be addressed will be the (1) effects of changes in current velocity, (2) effects of habitat reduction during high or low flows, (3) significance of the unstable nature of food supply to tailwater communities under fluctuating conditions, and (4) recovery potential of fluctuating tailwater streams.

### 5.2.1 Variable Current Velocity

Altering the volume of discharge changes the characteristics of a stream, including water depth, wetted perimeter (the distance measured along the bottom from the water surface on one side to the water surface on the other side), and current velocity. The relative changes in water depth, width, and current velocity will differ with stream morphology, because these parameters vary as a power of the discharge in such a way that the sum of the exponents equals one (Hynes 1970). However, it has been shown that, with a decrease in volume of flow, current velocity is decreased more rapidly than the depth or wetter perimeter. Curtis (1959) observed a 65% decrease in mean current velocity, with an 80% decrease in discharge, while the wetted perimeter and maximum depth decreased only 21% and 17% respectively. Kraft (1972) observed that, in a well-defined channel, a 90% reduction in discharge resulted in a decrease in surface area and mean depth of 42% and a 75% decrease in current velocity. Thus, reduction in current velocity may be the initial or major change associated with nondischarge or low discharge periods at hydroelectric generating stations.

Current velocity is an important factor regulating the occurrence and microdistribution of stream-dwelling invertebrates. Feeding adaptations and respiratory structures of stream invertebrates are specifically adapted for currents, and some species are confined to fairly definite ranges of current speed (Hynes 1970, Ward 1976b). Thus, one obvious effect of radically changing current velocities is that those species limited to narrow ranges will be unable to tolerate periods of unsuitable current velocity. If high flows do not physically remove or damage the organisms, then low flows may result in mortality resulting from insufficient oxygen levels and food availability or downstream displacement via drift.

The effect of high velocities or sudden increases in velocity have often been observed to physically remove or destroy benthos, both in natural streams subject to flooding and below power-generating dams. Briggs (1950) reported that periods of minimum benthic production in a free-flowing stream section coincided with the periods of greatest fluctuation in flows and lowest water temperature. Because the riffles were much more affected than pool areas, he concluded that flushing out the organisms was the primary cause, with low temperatures of secondary importance. Powell (1958) noted that the severe flushing action of daily releases of up to 1850 cfs resulted in mortality of organisms living on the stream substrate below a dam. Mullan et al. (1976) reported that the limiting effect of high water velocities on benthos in some upper Colorado River tailwater areas could be deduced from the fact that slack water areas around a dam diversion tunnel consistently produced a diverse benthic community. The extent of scouring or flushing that results from high water flows will obviously be site-specific, depending on amplitudes of fluctuation and adaptations of the existing community. If fluctuations are introduced to a previously nonfluctuating system, however, some removal of organisms and erosion of the streambed can always be expected. After frequent, periodic fluctuations are established, this effect will become small because only small organisms that can tolerate the velocity variations will remain.

The low flow velocities resulting from regulation of a stream for power production may also adversely affect benthic production. Trotzky and Gregory (1974) compared the macroinvertebrates of a stream section above a dam, where fluctuations only occurred seasonally, with the macroinvertebrates of a tailwater area below the dam, where daily fluctuations in current velocity varied from 0.1 to 0.5 m/s. Although the current velocities in the tailwater during high flow periods were similar to those in the nonfluctuating stream section, many swift-water benthic species found upstream were conspicuously absent from the tailwater during low flow periods. Overall diversity and abundance of macroinvertebrates was further reduced because slow-water species could not invade the stream because of occasional high water flows. Organisms that were successful in the highly fluctuating tailwaters were behaviorally adapted to avoid the full force of the high currents, yet morphologically adapted to feed and respire during low flows.

Fish populations in streams are also affected by fluctuating discharges. Experimental studies by McPhee and Brusven (1976) demonstrated that both decreases and rapid increases in flow displaced fish from test sections. Fish were displaced more rapidly at night than during the day. Such reductions in carrying capacity and resultant displacement of fish are caused by loss of shelter, food, and available space. Frazer (1972) argues that, although shelter is an important determinant of fish carrying capacity, carrying capacity of a unit of streambed can be affected by changes in current velocity alone. In support of his argument, Frazer (1972) mentions studies by Kalleberg (1958) that report a decrease in the size of territories for juvenile salmon and brown trout as a result of decreased current velocity. Thus, even if wetted perimeter and average depth of a stream are not drastically altered by water level fluctuation, fish populations may nevertheless be reduced as a result of competition for space during periods of low current velocity caused by flow reductions.

Downstream displacement via drift in response to low flows appears to be an important mechanism contributing to reductions of benthic fauna in fluctuating systems. Once in the drift, invertebrates are considerably more vulnerable to predation. MacPhee and Brusven (1976) stated that extreme reductions in flow significantly increased the amount of insect drift and the rate of ingestion of drifting organisms by salmon in an experimental diversion channel. Numerous other authors have confirmed that increased drift is associated with reduced velocities (Gore 1977, Minshall and Winger 1968, Radford and Hartland-Rowe 1971, and Armitage 1977). Minshall and Winger (1968), who studied the effects of reduction in stream discharge on benthic drift, found that virtually all bottom-dwelling forms were affected. Entry into the drift appeared to be an active process, occurring even during periods of high light intensity. They noted that periodic reduction of water levels during daylight could increase the drift of invertebrates during periods when fish are actively feeding.

Another issue related to the effects of changes in current velocity is the potential effect on upstream migration and reproductive success of fish. Raymond (1968) found that the rate of migration of chinook salmon was directly related to river flows. Fraser (1972) states that:

> Migrations of fish are affected by the amount of discharge in a number of ways. Discharge can cause migrations to commence, create barriers at high or low flows, cause delays, disrupt normal routing, and change the speed of travel. The role of discharge in inducing migrations of fish is important in many species and may vary between species and between streams for the same species. Most, but not all, salmon migrations occur at times of the year when increasing or seasonally high discharge can be expected.

Several examples provided by Fraser (1972) demonstrate that reduced flow delays both upstream and downstream migration. No indication was given of the effect of daily fluctuations on migration; however, one can reasonably conclude that such fluctuations may at least delay migrations if stream depth remains sufficient for movement. Fraser (1972) also suggests that stream velocity is important to successful spawning and survival of eggs and fry. Salmon and steelhead have rather narrow tolerance to velocity and depth when choosing spawning sites. The velocity of intragravel flow, which is determined by surface flow, is important to the survival of salmon eggs. Several studies cited by Fraser (1972) indicate a strong relationship between discharge and reproductive success in salmon. Although salmon survival is relevant at only a few small hydroelectric generating sites in the United States, other fish that make upstream spawning runs and that require well-aerated gravels may be similarly affected.

### 5.2.2 Unstable Habitat

Large reductions in current velocity are also normally associated with reductions in wetted perimeter, depth, and substrate area. In circumstances of extremely rapid flow reductions, stranding and desiccation of both invertebrates and fish may occur. Kroger (1973) found 55 stranded sculpins in three 0.84 $m^2$ areas of exposed riffle after a flow reduction of 2.8 to 0.3 $m^3/s$ within 5 min. Furthermore, all the macroinvertebrates present in the exposed riffle were left stranded. Powell (1958) reports similar observations for the Blue River, where water depths vary daily (up to 1.2 to 1.5 m within less than 1 min) below Green Mountain Dam. Powell (1958) found stranded trout and dace as well as sculpins. Many insects were destroyed by desiccation, and stoneflies were consumed by birds during attempts to migrate toward the water. These are extreme examples of rapid habitat loss directly below dams. Fluctuations tend to be dampened downstream, (Powell 1958) even in these extreme cases, and are not as severe below all hydroelectric generating stations. The effect of stranding and desiccation is the reduction of benthic and fish communities in the areas affected most heavily. Frequently, extreme fluctuations will prohibit development of an adapted community.

Stream communities that are exposed daily to moderate water level fluctuations are inhabited by species that can tolerate unstable conditions. A distinctive "intertidal" or fluctuation zone has developed below a hydroelectric dam on the Connecticut River that has regular diel variations in discharge (Fisher and LaVoy 1972). Chironomids and oligochaetes predominate in the areas subject to greatest exposure, whereas unionid mussels predominate in the least exposed sites. The persistence of chironomids in fluctuating zones has been noted by other investigators (Ward 1976b, Covich et al. 1978). Chironomids can apparently survive periods of desiccation better than many other invertebrates (Sect. 3.1). Nilsen and Larimore (1973) also observed that chironomids, oligochaetes, hydropsychid larvae, and elmid larvae survive for long periods in logs deposited on the floodplains of streams. Although the survival of chironomids and oligochaetes in fluctuating zones may moderate the reduction of benthic biomass in streams, the overall food resource for benthic feeding fish may be reduced as a result of reductions in larger prey organisms (mayflies and stoneflies).

Reduced population numbers, biomass, and diversity are often reported in streams where fluctuating conditions result in considerable habitat exposure. Both Fisher and LaVoy (1972) and Covich et al. (1978) report that areas subject to the greatest exposure are characterized by lower diversity, density, and biomass of benthic organisms. The diversity of benthic organisms is almost always lower in tailwaters below dams that experience water level fluctuations than in unregulated stream reaches. However, the total density of benthic, organisms is sometimes increased in fluctuating tailwaters (Pfitzer 1954, Ward 1976b, Ward and Stanford 1976b). These appear to be situations where regulation reduces the high flows associated with spring runoff in unregulated streams and daily flow fluctuations are relatively moderate. Based on the available literature, however, modification of a nonfluctuating, or moderately fluctuating, tailwater to one experiencing large daily fluctuations would be expected to result in both reduced density and diversity of benthic invertebrates.

### 5.2.3 Variable Food Resource

Fluctuating discharges from hydroelectric dams create an unstable food supply in tailwater areas (Armitage 1976, Ward 1975). Rapid flow fluctuations eliminate accumulations of leaf litter and provide an intermittent source of plankton for filter-feeding organisms. The persistence of reservoir zooplankton in tailwaters has been found to vary with the discharge rates of a dam (Ward 1975). Fluctuating conditions would be expected to eliminate most species that use leaves as food and reduce populations of species that use organic particulates as well. Ward (1976b) noted that, in all cases where standing crop increased below dams, the flow had become constant with regulation. Decreases in standing crop were associated with all cases in which fluctuating or low flow conditions were reported (Ward 1976b). Fluctuation in the food resource for plankton feeders may adversely affect some species, even under conditions where velocity and habitat changes are relatively mild.

### 5.2.4 Recovery Potential

The recovery potential of tailwater areas is limited if fluctuations occur daily. However, if fluctuations are stopped for periods of time and if extreme fluctuations that cause scouring or desiccation are reduced or eliminated, some recovery can occur. The fastest source for normal stream recovery is downstream drift. Waters (1966) reported downstream drift of up to 22 g/d during the summer. However, downstream drift is not likely to quickly repopulate tailwater areas because stream-adapted organisms are an unlikely or rare component of reservoir discharge water. Repopulation would occur quickly by drift only if tributary streams enter the tailwater area directly. Upstream mating flights by adults would be the most likely source of insect fauna recruitment to the tailwater areas. Upstream migration of insect nymphs may occur in some cases. Hayden and

Clifford (1974) reported seasonal upstream migrations of a mayfly nymph (Leptophlebia cupida) toward marsh areas. The particular species discussed was not well adapted to lotic conditions and would not be expected in tailwaters, but this behavior may also occur in other species. Organisms (such as crayfish) would migrate upstream. Mussels may slowly repopulate upstream areas by transport of larval stages on host fish species if the fish migrate upstream.

In summary, the effect of subjecting a nonfluctuating tailwater to fluctuating conditions will be closely correlated to the amplitude and rapidity of the fluctuations. The greater the amplitude and the more quickly flow increases and reductions occur, the more extreme the reductions in benthic and fish biomass, diversity, and density will be. Moderately fluctuating tailwaters or tailwaters further downstream will develop a community characterized by species that can tolerate exposure to air. If water flow fluctuations result in decreased siltation, habitat diversity will possibly increase and offset some effects of unstable habitat and food supply. Our review of the literature suggests that operation of a small-scale hydroelectric facility in a peaking mode may decrease the productivity of a tailwater for a limited distance downstream.

# 6.0 SUMMARY AND RECOMMENDATIONS

Potential environmental impacts that can occur because of water level fluctuation at small scale hydroelectric sites are presented in this report. These potential impacts are not of concern at run-of-river projects. The potential significance of these impacts is a direct function of the mode of operation of the small-scale hydroelectric facility (store and release, peaking), the magnitude and timing of fluctuation, and the site-specific environmental setting. Also, most of the information reviewed in this report comes from studies of dam and impoundment systems that are much larger than typical small-scale hydroelectric projects. However, because hydroelectric projects are classified as "small-scale" based on a capacity rating (≦25 MW), it is difficult to generalize about size. Because capacity of a hydroelectric facility is a function of available flow and available head, a considerable size range is possible for a project rated at ≦25 MW. Acknowledging these constraints, potentially significant environmental issues resulting from water level fluctuation are summarized below.

## 6.1 Physical and Chemical Impacts

The potentially significant physical and chemical impacts of water level fluctuations in reservoirs not exposed to such fluctuations before, or exposed to different temporal patterns and ranges of fluctuations, are expected to involve shoreline substrate stability and water quality. These effects may be specifically summarized as:

1. Resuspension and redistribution of bed and bank sediment.
2. Leaching of soluble matter from sediment in the littoral zone as water moves into and out of the interstices (bank storage) in response to water level fluctuations.

3. Changes in (a) sediment and nutrient retention (trap efficiency) of the impoundment because of hydro-imposed changes in circulation patterns and hydraulic efficiency and (b) water quality, which is coupled to circulation pattern and hydraulic efficiency.

When water level fluctuates because of hydro-peaking operations, poorly consolidated bank and substrate materials will be continually resuspended and redistributed. The net result of this resuspension is the preferential removal of the finer, less dense particles (silt, clay, organic matter) from the littoral zone and redeposition in deeper and more quiescent water or complete loss from the reservoir in the outflow. The material remaining in the littoral zone after extended exposure to a fluctuating water level is, therefore, of coarser particle size than material remaining after exposure to a stable water level.

Alluvium, when exposed periodically to a fluctuating water level, will occasionally contain very little coarse material that can withstand the erosion forces associated with waves and currents. Therefore, banks exposed at low water levels may be subjected to continuous undercutting and collapse. The eroded material can contribute to increased turbidity of the water column and accelerated sedimentation in deeper water.

Regular reservoir drawdowns can also have profound effects on the longitudinal and lateral distribution of sediment. These effects may sometimes be beneficial in the context of prolonged reservoir life because sediments deposited previously might be sluiced out of the reservoir, which ordinarily requires a deep outlet, or the effects may be deleterious in the context of the excessive resuspension of sediments into the water column and the potential clogging of outlet structures.

The alternate inundation and drainage to which soils and sediments in the littoral zone are exposed when water levels fluctuate may lead to increased leaching of soluble matter from the exposed substrate. In shallow water, where aerobic (chemically oxidized) conditions are likely to prevail in the water overlying flooded soils and sediments, a sharp gradient in the concentrations of nutrients and metals is likely to exist across the substrate-water interface. When the water level is lowered, some part of the interstitial water containing higher concentrations of nutrients and metals may drain onto the surface of the substrate or enter the adjacent surface water body. If this drainage water is highly enriched in soluble plant nutrients or deleterious substances such as heavy metals, the near-shore water quality may be adversely affected.

The reservoir's ability to trap and retain sediment is known as the "trap efficiency" and is expressed as the percentage of the total inflowing sediment that is deposited in the basin. The degree to which a reservoir traps inflowing sediment is the prime determinant of the useful lifespan of the reservoir and is an important factor affecting reservoir and downstream water quality (especially turbidity). Generally, sediment trap efficiency decreases with (1) a decrease in water retention time, (2) a decrease in sediment particle size, and (3) a change from surface-water withdrawal to deep-water withdrawal. Imposing water level fluctuations on a reservoir would directly affect the water retention time as a result of the changes in average storage volume. The overall effect expected would be a probable reduction in sediment trap efficiency. Also, the periodic lowering of water level may allow coarser-grained sediments to reach, and be transported farther into, the reservoir basin, thereby initially increasing the trap efficiency. However, the latter effect would also lead to a faster loss of reservoir storage capacity. Thus, any apparent increase in trap efficiency resulting from the input of coarser sediment may ultimately be offset by the decrease in retention time caused by the decrease in storage volume.

The term "trap efficiency" has also been used in the context of nutrient retention by reservoirs. The efficiency with which impoundments trap inflowing nutrients is an important environmental issue because water quality and productivity, within and downstream from impoundments, is partially determined by trap efficiency.

Nutrient trap efficiency appears to vary in response to the same factors that affect the trap efficiency for sediment. Thus, any significant change in water retention time, which is the primary determinant of sediment trap efficiency, will also affect nutrient trap efficiency.

The primary physical effects of water level fluctuation downstream from impoundments are streambed and bank erosion and water quality problems related to resuspension and redistribution of these materials. Erosion and downstream transport of these materials may increase, resulting in increased turbidity, decreased bank stability, and alterations to channel geometry. Potential indirect effects of increased erosion include the undermining of piers and abutments and a decrease in biological productivity.

## 6.2 Biological Impacts

The effects of water level fluctuation on reservoir biota are, in part, a result of changes in the physical and chemical environment discussed above. General effects of water level fluctuations on reservoir biota include (1) habitat destruction resulting in partial or total loss of organisms, (2) changes in habitat quality resulting in reduced standing crop and production, and (3) shifts in species diversity.

The importance of the littoral and shore zone to biological production in lakes is well established. In general, a littoral zone with aquatic macrophytes may have higher productivity and species diversity than a shore zone without aquatic macrophytes.

Water level fluctuation imposed on the littoral or shore zone of a reservoir can change species diversity, eliminating species that are unable to migrate, aestivate, or rapidly recolonize the drawdown zone. Littoral benthic invertebrates stranded above receding water for more than a few hours may be lost. However, chironomid larvae and oligochaetes are able to retreat deeper into the substrate and survive for periods ranging from days to months. Benthic invertebrate populations that recolonize the littoral zone on rising water levels may be less diverse but possibly more abundant than populations of undisturbed systems.

With few exceptions, most reservoir fish are closely associated with the littoral and shore zones of reservoirs for some part of their life cycle. Specifically, spawning, incubation, and hatching of eggs, and development of larvae, postlarvae, and juveniles occur in shallow littoral and shore zones. The basic requirements for successful spawning, development, and growth are adequate spawning habitat, stable water level, and adequate food supply. Water level fluctuation can significantly affect all these requirements.

Water level fluctuation probably will not directly affect aquatic species in the deep, open-water areas of reservoirs. However, drops in water level are ultimately reflected in increased discharge from the reservoir. Thus, planktonic species (algae and zooplankton), early life history stages of fish, and some benthic invertebrate species could potentially be lost from the reservoir ecosystem as a result of water level fluctuation. Such losses could adversely affect reservoir food webs and productivity.

Potential impacts on biological communities in tailwaters below dams because of water level fluctuation are the result of changes in current velocity, habitat reduction, and alteration in food supply. Current velocity is one of the most important factors regulating occurrence of stream-dwelling invertebrates. Changes in current velocity because of water level fluctuation can cause changes in abundance and community structure of benthic invertebrates. Sudden increases in current velocity can physically scour benthic invertebrates from the substrate, causing direct mortality or downstream displacement.

Water level fluctuation and flow variation can reduce abundance and diversity of the fish community in tailwaters. Migration of fish species can be affected by flow fluctuation. Discharge regime can initiate migration, pose a barrier to migration, delay migration, or alter the speed of migration.

Reductions in current velocity and flow are normally associated with a reduction in the wetted perimeter of a tailwater. Reductions in wetted perimeter (available habitat) can directly reduce density and diversity of biota and cause mortality by stranding organisms in an unfavorable habitat. Fluctuating discharges from hydroelectric installations may also create an unstable food supply below dams, causing indirect effects on overall biological productivity.

The effect of fluctuating water levels on biological communities of tailwaters will likely be most severe for high amplitude fluctuations that occur rapidly. Moderately fluctuating tailwaters will be characterized by the presence of species that can tolerate these conditions.

## 6.3 Recommendations

The information reviewed in this report provides background material that can be used to evaluate the potential significance of impacts resulting from water level fluctuation at small-scale hydroelectric sites. The following progression of site-specific analysis of this issue is recommended at existing dams that are being considered for small-scale hydroelectric development and that will include water level fluctuation as part of the operation of the hydroelectric facility:

1. Estimate the magnitude and frequency of fluctuations resulting from facility operation.

2. Consult with state natural resource agencies, regional offices of the U.S. Fish and Wildlife Service, the Federal Energy Regulatory Commission, and local agencies with expertise or interest relating to environmental impacts of water level fluctuation. Individuals in these groups who have specific knowledge concerning the particular site should be contacted.

3. Determine, through these initial contacts, the level of effort and specific expertise required to assess the potential adverse environmental impacts resulting from the proposed water level fluctuation.

4. Initiate appropriate site-specific studies to determine the significance of impacts resulting from proposed water level fluctuations.

5. Include the results of site-specific studies and plans for minimizing or mitigating adverse environmental impacts as part of the license application to the Federal Energy Regulatory Commission.

The fundamental premise embodied in these recommendations is that the potential adverse impacts resulting from water level fluctuation are site-specific. Therefore, it is crucial to first determine whether water level fluctuation is potentially a significant issue at the site. The appropriate effort and expertise can then be secured to quantify the adverse impact. This suggested approach is specifically recommended in the Council on Environmental Quality guidelines for implementation of the NEPA (National Environmental Policy Act) process (Council on Environmental Quality 1978).

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# APPENDIX A

# RESERVOIR SHORE AREA CHANGE IN RESPONSE TO WATER LEVEL FLUCTUATIONS: A QUANTITATIVE METHODOLOGY TO PREDICT LOSS OF SHORE ZONE HABITAT

B. A. Tschantz
Simon Tam

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## 1. BACKGROUND AND INTRODUCTION

This appendix was prepared based on a subcontract report to Oak Ridge National Laboratory (ORNL/UT 7647) by Dr. Bruce Tschantz and Mr. Simon Tam, Civil Engineering Department, University of Tennessee, Knoxville, Tennessee.

The construction of impoundments converts free-flowing streams to reservoir systems for flood control, hydroelectric generation, recreation, flow augmentation, and other uses. The change from free-flowing streams to reservoirs can cause major changes in the aquatic environment. Water level fluctuation in reservoirs can also result from hydroelectric power generation. The various operating modes of hydroelectric dams result in a variety of potential water level fluctuation regimes. Fluctuations may sometimes be small if the power facility operates in a run-of-the-river mode. However, if the facility is operated in a peaking mode, water release and power generation may concentrate in certain periods resulting in a significant drop of water level. Between periods of generation, water level may be restored by pondage. Shore areas exposed and inundated in this sequence may reduce the potential for biological productivity in the reservoir ecosystem.

Impacts of water level fluctuations resulting from various management and use practices can cause adverse impacts on the aquatic communities of reservoirs that depend on shore zone habitat for some part of their life cycle. The growing demand for water allocations for different uses will increase the impacts of water level fluctuations. To achieve a balance between reservoir uses that cause water level fluctuations and optimum biological production in reservoirs, the proportional loss of shore zone habitats resulting from water level fluctuation must be determined.

## 2. PURPOSE AND APPLICATION OF WORK

The purpose of the mathematical models developed in this appendix is to estimate the bottom area that becomes exposed (or inundated) with a certain drop (or rise) in reservoir surface level.

Because the extent of primary production in the littoral or shore zone is partly a function of light penetration, the extent of this littoral zone can be estimated by depth information. By simple calculations, parts of the littoral aquatic habitats lost by water level fluctuations can be easily determined. Because the range of water level fluctuations depends on the type of water allocation and use, its extent can be projected, and mitigation plans can be included in reservoir management schemes if the predicted impacts on the biological communities are judged unacceptable.

Although the conception and development of this methodology was the result of an effort to quantify the impacts of water level fluctuations associated with hydroelectric generation at existing dams, use of this model is not restricted to reservoirs used only for hydroelectric generation.

To analyze the biological impacts in the littoral zone resulting from water level fluctuations in reservoirs, the following information must be determined.

1. Extent of water level fluctuation.
2. Extent of the littoral or shore zone of the reservoir impacted by fluctuation.
3. Measure of relative biological importance of the littoral or shore zones affected to the overall reservoir ecosystem.

This information would provide a reference point to establish the magnitude, frequency, and time of year of water level fluctuations that are most compatible with competing uses of the reservoir.

## 3. BASIS FOR MODEL DEVELOPMENT

Two principal model approaches have been developed for estimating shoreline area exposure (or inundation) in response to water level fluctuations that could be encountered on reservoirs during small hydroelectric plant operation. The type and availability of topographic and reservoir data can vary significantly from site to site and, therefore, two basic approaches are needed. Ideally, to estimate shoreline area increase or decrease resulting from reservoir level fluctuation, accurate shoreline (bank) slope and shoreline length inputs are required for all expected pool level conditions. Usually, standard 1:24000 scale 20-ft contour USGS Quadrangle maps adequately serve this need if the reservoir shoreline is also shown.

For classification and model development purposes in this appendix, an array of data conditions, arranged in descending order of data availability, may actually occur at reservoir sites (Table 1).

The two model approaches considered include (1) a simple estimate method and (2) a three-dimensional analytical geometric method. Both models will produce estimates of shoreline area change with inputs of water level fluctuation and specific reservoir characteristics.

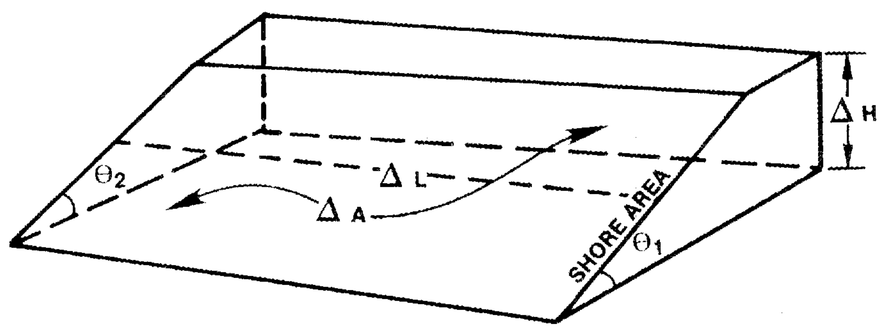
### 3.1 Simple Model

The simple model concept is based on summing successive rectangular strips around the reservoir shoreline. Each successive strip area ($\Delta A$) is defined in terms of the average local shoreline slope ($\bar{S} \cong \tan \bar{\theta}$) and water level fluctuation ($\Delta H$) along a uniform shoreline ($\Delta L$), as shown in Fig. 1.

The total change in shoreline area resulting from water level fluctuation is determined by summing $\Delta A$ for n shore length increments around the reservoir perimeter, or

Table 1. Possible reservoir data conditions

| Case | Topographic contour map | Reservoir surface area | Comments |
|---|---|---|---|
| I (Ideal) | Available for determining representative shoreline slopes at reservoir pool level. Contours shown above and below water surface. | Pool surface shown on topographic map for determining shoreline length. | Two data alternatives to reservoir area map are equally acceptable:<br>1. Reservoir photograph of known scale, showing pool surface area.<br>2. Surface elevation data along longitudinal axis of reservoir, upstream from dam (continuous profile or intermittent elevations). |
| II | Contour map available, but shown only (A) above or (B) below water surface. | Pool surface shown on topographic map for determining shoreline length (same as Case I). | Shoreline slopes determined from known contour information would need to be extrapolated (A) below or (B) above water surface. |
| III | Intermittent valley or transverse cross sections available at known distances upstream from dam. | Pool surface shown on topographic map for determining shoreline length (same as Case I). | A. Shoreline slope determination limited to cross-section points only.<br>B. A less desirable, but acceptable, alternative would be the situation where the depth of water and reservoir width are known at intermittent distances upstream from dam. |
| IV | One valley or transverse cross section available (e.g., at dam section only). | Pool surface shown on topographic map for determining shoreline length (same as Case I). | |
| V | Cross section of valley at dam only. | None. | General approximation results for regular shoreline only. |




$$\text{WHERE} \quad \bar{S} = \frac{S_1 + S_2}{2},$$

$$S_1 = \tan \Theta_1 ,$$

$$S_2 = \tan \Theta_2 ,$$

$$\Delta A = \Delta L \frac{\Delta H}{3} \sqrt{1 + \bar{S}^2}$$

Fig. 1. Simple model development configuration of a reservoir to estimate shore area change resulting from water level fluctuation.

$$\text{Total Area} = \Delta H \sum_{i=1}^{n} \frac{\Delta L_i}{\bar{S}_i^{\,2}} \sqrt{1 + \bar{S}_i^{\,2}} \quad ,$$

where

$$(1 + \bar{S}_i)^{\frac{1}{2}} \cong (1 + \tan^2 \bar{\theta})^{\frac{1}{2}} = \frac{1}{\cos \bar{\theta}} \quad .$$

This approach assumes that $\Theta_1 \cong \Theta_2$ (Fig. 1). Note that shore area is exposed when the water level lowers, but is inundated when the level rises.

A computer program has been developed for this simple model. Based on this approach, Table 2 has been developed, which gives shore area change (hectares) per unit shore length (kilometers) for representative ranges of average shore slope and water level fluctuation (meters). Section IV of this appendix explains the use of the table and model.

### 3.2 Modified Simple Model

Reservoir shorelines are often highly irregular and vary significantly in slope from one section to another, causing a distortion in the assumed rectangular strip in the simple model. For this reason, the incremental shore area $\Delta A$ is represented by a trapezoid configuration as shown in Fig. 2.

For this case the incremental shore area change $\Delta A$, resulting from a water level fluctuation, is

$$\Delta A \cong \frac{(\Delta H)(\Delta L)}{2} \left( \frac{\sqrt{1 + S_1^{\,2}}}{S_1} + \frac{\sqrt{1 + S_2^{\,2}}}{S_2} \right) \quad ,$$

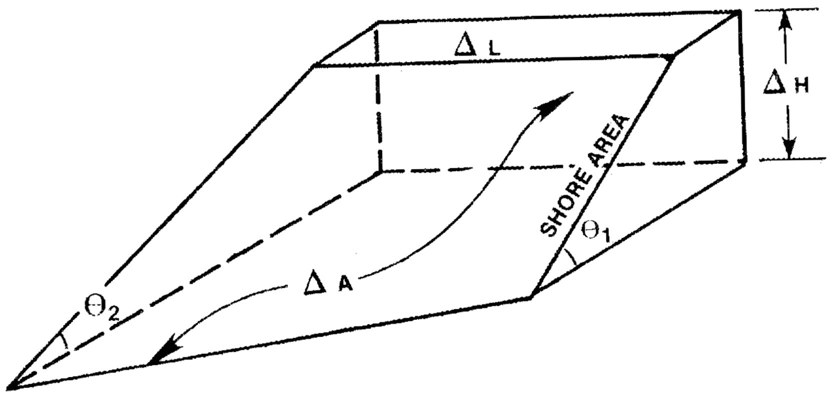



Fig. 2. Modified simple model development configuration of a reservoir to estimate shore area change resulting from water level fluctuation.

and the total change for the reservoir is

$$\text{Total Area} \cong \sum_{i=1}^{n} \Delta A_i \ .$$

$\Theta_1$ and $\Theta_2$ in this approach are not assumed equal. Inputs to this model include average (mid-depth) incremental shore lengths $\Delta L$, section slopes $S_1$ and $S_2$, and water level fluctuation $\Delta H$. A computer program has been developed for the modified simple model. Sections 3 and 4 of this appendix describe application and use of the modified simple model.

### 3.3 Geometrical Model

Situations may arise where topographic contour maps to determine shore slopes are unavailable for the reservoir. In these cases, shore area changes cannot be estimated from the simple or modified simple models. However, if reservoir or river cross-section area information is available (1) at the dam or (2) at the dam and other upstream points, a geometrical model can be developed for approximating the actual reservoir conditions, allowing certain assumptions. The following assumptions are made:

1. The reservoir represents a single stream (or stem) having little or no shoreline irregularity as shown in Fig. 3.
2. The lateral reservoir cross-section configuration of the reservoir bed (e.g., section x-x), as shown in Fig. 3, is parabolic.
3. The reservoir cross-section configuration is changing gradually and linearly upstream along the bed.
4. The longitudinal stream bed slope $S_b$ is constant.
5. The water surface slope is always less than the river bed slope, as is the case in most reservoirs.

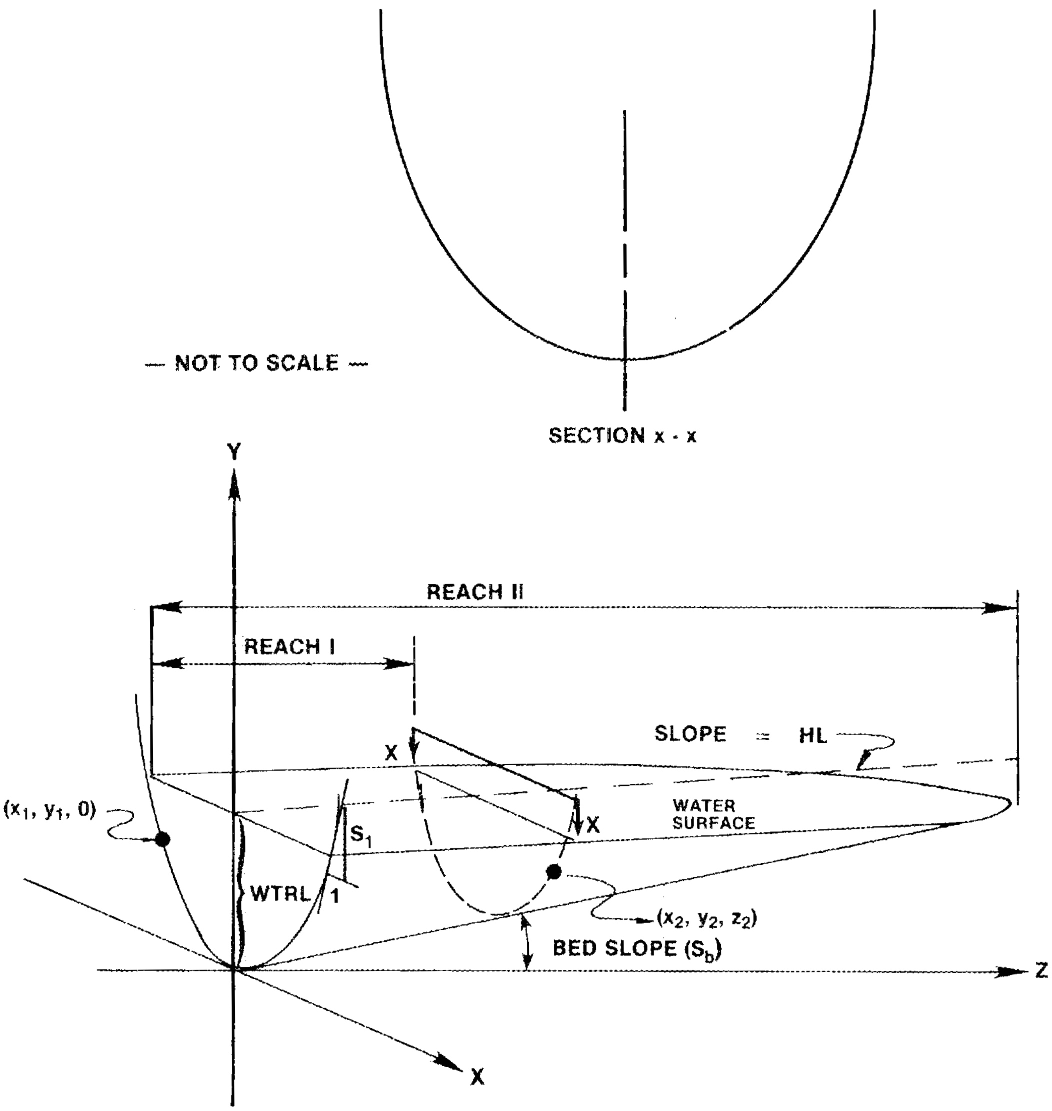



Fig. 3. Geometrical model development configuration of a reservoir to estimate shore area change resulting from water level fluctuation.

A three-dimensional equation used to describe the reservoir channel is:

$$y = Ax^2 + Bzx^2 + S_b z \quad ,$$

where coordinates x, y, and z are defined in Fig. 3; $S_b$ is the channel bed slope; and A and B are characteristic input parameters that describe the reservoir bed configuration. A and B are defined as follows:

$$A = y_1/x_1{}^2$$

$$B = \frac{y_2 - Ax_2{}^2 - S_b z_2}{z_2 \; x_2{}^2}$$

where $x_1$, $y_1$, and 0 is a reservoir coordinate point at the vertical channel cross-section along the z-axis, taken at the dam, and $x_2$, $y_2$, and $z_2$ is a second point, located $z_2$ distance upstream from $z_1 = 0$.

An integral is developed for defining the submerged portion of the reservoir bottom area:

$$\text{Submerged Bottom Area} = \int_x \int_z \sqrt{1 + \left(\frac{\partial y}{\partial z}\right)^2 + \left(\frac{\partial y}{\partial x}\right)^2} \quad dzdx$$

$$= 2 \int_0^{\sqrt{\frac{WTRL}{A}}} \int_0^{\frac{WTRL - Ax^2}{Bx^2 + S_b - HL}} \sqrt{1 + (Bx^2 + S_b - HL)^2 + [2(A + Bz)x]^2} \quad dzdx$$

$$= \int_0^{\sqrt{\frac{WTRL}{A}}} 4Bx \left\{ \frac{1}{2}\left(\frac{A}{B} + z\right) \sqrt{[G(x)]^2 + \left(\frac{A}{B} + z\right)^2} \right.$$

$$\left. + \frac{[G(x)]^2}{2} \cdot \log \left[ \left(\frac{A}{B} + z\right) + \sqrt{[G(x)]^2 + \left(\frac{A}{B} + z\right)^2} \right] \right\} \Bigg|_0^{\frac{WTRL - Ax^2}{Bx^2 + S_b - HL}} dx \quad ,$$

where the Z-integration was performed using Formula 132, page 317 of the CRC tables (14th ed.). WTRL represents the water level elevation at the downstream section, and HL is water surface slope (may be zero for "flat" reservoir surface). G(x) is defined according to:

$$[G(x)]^2 = \frac{1 + (Bx^2 + S_b - HL)}{4B^2 X^2} \quad .$$

The upper and lower bounds in the double integral correspond to the edges of the water surface in the X- and Z-directions. Since the lower bound in the X-direction is $-\sqrt{WTRL/A}$ instead of 0, a factor of 2 is introduced.

The equation can be solved by numerical analysis using a digital computer. If the water level fluctuates from an initial elevation, $(WTRL)_1$, to another elevation, $(WTRL)_2$, shore area change can be defined by computing the difference between the submerged bottom areas.

The above model can be used to determine shore area change for two possible reservoir conditions:

1. Run-of-river.
2. Reservoir pool.

A run-of-river condition is characterized by reach I in Fig. 3. A reservoir pool condition is represented by reach II, where the opposite shorelines converge upstream.

A geometrical computer model is developed to solve either reservoir condition. The geometrical model produces a parabolic or regular shoreline configuration, from which shoreline area change is determined. However, if the actual shoreline is known to be significantly irregular, and therefore longer, the computer model would underestimate the shore area change. To compensate for this known condition, an adjustment is recommended. The computer model will automatically adjust the shore area change when the actual total shoreline length is known and is included as input data.

## 3.4 Model Choice

The choice of model for determining change of shore area from water level fluctuation depends on the data conditions, level of precision needed, and available time and money for computation. Referring to the five reservoir data cases in Table 1, the following model uses are recommended for best results:

| Reservoir data condition | Recommended model | Comments |
|---|---|---|
| Case I<br>Regular or irregular shoreline | 1. Modified simple | Most precise, but requires computer solution. |
| | 2. Simple | Produces slightly less precision (lower areas), but Table II or computer can be used for solution. |
| Case II<br>Regular or irregular shoreline | 1. Modified simple | (see Case I) |
| | 2. Simple | (see Case I) |
| Case III<br>Regular or irregular shoreline | 1. Modified simple | (see Case I) |
| | 2. Simple | (see Case I) |
| Case IV<br>Regular or irregular shoreline | 1. Simple | Assume single constant cross-section for run-of-river conditions only; water surface slope is approximately parallel to river bed slope. |
| | 2. Geometrical | Recommended for reservoir condition only, but also feasible for run-of-river. |
| Case V<br>Regular or irregular shoreline | 1. Geometrical | Run-of-river or reservoir condition where little geometric data is available; approximation only. |

## 4. GENERAL PROCEDURES AND LIMITATIONS FOR MODEL USES

This section introduces the procedure for model use. All computer models developed are written in Fortran. To make the models compatible to metric and English units, units in the models are left to the discretion of the analyst. However, units of inputs must be consistent when using the models. The unit of output corresponds to the unit of input. For example, if the input of length is in feet, all other inputs of length must be in feet. Corresponding output will then be feet for length and square feet for area.

### 4.1 Simple Model

The simple and the modified simple models require an initial common procedure. A reservoir shoreline boundary must be defined from a contour map (a 1:24000 scale quad map is sufficient). The defined shoreline is divided into equal intervals of a maximum of 2000 ft (or 600 m). The last interval section length may, however, be shorter than the other lengths. Shore bank slopes are measured at each interval mark [note that there must be (n+1) interval marks for n intervals or sections].

Table 2, designed for metric lengths, can be used for a fast estimation of the shore area resulting from water level fluctuation. An overall average shore bank slope is computed from the measured bank slopes at interval marks. A corresponding shoreline area value in hectares per kilometer shoreline length can be obtained from Table 2 with inputs of the average shore slope and the expected water level fluctuation in meters. The product of the Table 2 unit shoreline area value and the actual shoreline length in kilometers is the estimation of the total shore area resulting from the water fluctuation.

A more precise, but time-consuming, alternative to using Table 2 would be to sum up incremental shore slopes and resultant areas. A computer model, using this incremental method, is set up for a more precise estimation (Fig. 4). Section 3 describes the basis of the

Table 2. Estimated reservoir shore area affected by water level fluctuations based on simple model discussed in text.

Shoreline area (ha)/Shoreline length (km)

| Water Level Fluct. (m) | Average reservoir shore slope | | | | | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | 0.01 | 0.05 | 0.10 | 0.15 | 0.20 | 0.25 | 0.30 | 0.35 | 0.40 | 0.45 | 0.50 | 0.60 | 0.70 | 0.80 | 0.90 | 1.00 | 2.00 | 4.00 | 6.00 | 8.00 | 10.00 |
| 0.1 | 1.000 | 0.200 | 0.100 | 0.067 | 0.051 | 0.041 | 0.035 | 0.030 | 0.027 | 0.024 | 0.022 | 0.019 | 0.017 | 0.016 | 0.015 | 0.014 | 0.011 | 0.010 | 0.010 | 0.010 | 0.010 |
| 0.2 | 2.000 | 0.400 | 0.201 | 0.135 | 0.102 | 0.082 | 0.070 | 0.061 | 0.054 | 0.049 | 0.045 | 0.039 | 0.035 | 0.032 | 0.030 | 0.028 | 0.022 | 0.021 | 0.020 | 0.020 | 0.020 |
| 0.3 | 3.000 | 0.601 | 0.301 | 0.202 | 0.153 | 0.124 | 0.104 | 0.091 | 0.081 | 0.073 | 0.067 | 0.058 | 0.052 | 0.048 | 0.045 | 0.042 | 0.034 | 0.031 | 0.030 | 0.030 | 0.030 |
| 0.4 | 4.000 | 0.801 | 0.402 | 0.270 | 0.204 | 0.165 | 0.139 | 0.121 | 0.108 | 0.097 | 0.089 | 0.078 | 0.070 | 0.064 | 0.060 | 0.057 | 0.045 | 0.041 | 0.041 | 0.040 | 0.040 |
| 0.5 | 5.000 | 1.001 | 0.502 | 0.337 | 0.255 | 0.206 | 0.174 | 0.151 | 0.135 | 0.122 | 0.112 | 0.097 | 0.087 | 0.080 | 0.075 | 0.071 | 0.056 | 0.052 | 0.051 | 0.050 | 0.050 |
| 0.6 | 6.000 | 1.201 | 0.603 | 0.404 | 0.306 | 0.247 | 0.209 | 0.182 | 0.162 | 0.146 | 0.134 | 0.117 | 0.105 | 0.096 | 0.090 | 0.085 | 0.067 | 0.062 | 0.061 | 0.060 | 0.060 |
| 0.7 | 7.000 | 1.402 | 0.703 | 0.472 | 0.357 | 0.289 | 0.244 | 0.212 | 0.188 | 0.171 | 0.157 | 0.136 | 0.122 | 0.112 | 0.105 | 0.099 | 0.078 | 0.072 | 0.071 | 0.071 | 0.070 |
| 0.8 | 8.000 | 1.602 | 0.804 | 0.539 | 0.408 | 0.330 | 0.278 | 0.242 | 0.215 | 0.195 | 0.179 | 0.155 | 0.140 | 0.128 | 0.120 | 0.113 | 0.089 | 0.082 | 0.081 | 0.081 | 0.080 |
| 0.9 | 9.000 | 1.802 | 0.904 | 0.607 | 0.459 | 0.371 | 0.313 | 0.272 | 0.242 | 0.219 | 0.201 | 0.175 | 0.157 | 0.144 | 0.135 | 0.127 | 0.101 | 0.093 | 0.091 | 0.091 | 0.090 |
| 1.0 | 10.000 | 2.002 | 1.005 | 0.674 | 0.510 | 0.412 | 0.348 | 0.303 | 0.269 | 0.244 | 0.224 | 0.194 | 0.174 | 0.160 | 0.149 | 0.141 | 0.112 | 0.103 | 0.101 | 0.101 | 0.100 |
| 1.1 | 11.001 | 2.203 | 1.105 | 0.742 | 0.561 | 0.454 | 0.383 | 0.333 | 0.296 | 0.268 | 0.246 | 0.214 | 0.192 | 0.176 | 0.164 | 0.156 | 0.123 | 0.113 | 0.112 | 0.111 | 0.111 |
| 1.2 | 12.001 | 2.403 | 1.206 | 0.809 | 0.612 | 0.495 | 0.418 | 0.363 | 0.323 | 0.292 | 0.268 | 0.233 | 0.209 | 0.192 | 0.179 | 0.170 | 0.134 | 0.124 | 0.122 | 0.121 | 0.121 |
| 1.3 | 13.001 | 2.603 | 1.306 | 0.876 | 0.663 | 0.536 | 0.452 | 0.394 | 0.350 | 0.317 | 0.291 | 0.253 | 0.227 | 0.208 | 0.194 | 0.184 | 0.145 | 0.134 | 0.132 | 0.131 | 0.131 |
| 1.4 | 14.001 | 2.803 | 1.407 | 0.944 | 0.714 | 0.577 | 0.487 | 0.424 | 0.377 | 0.341 | 0.313 | 0.272 | 0.244 | 0.224 | 0.209 | 0.198 | 0.157 | 0.144 | 0.142 | 0.141 | 0.141 |
| 1.5 | 15.001 | 3.004 | 1.507 | 1.011 | 0.765 | 0.618 | 0.522 | 0.454 | 0.404 | 0.366 | 0.335 | 0.292 | 0.262 | 0.240 | 0.224 | 0.212 | 0.168 | 0.155 | 0.152 | 0.151 | 0.151 |
| 1.6 | 16.001 | 3.204 | 1.608 | 1.079 | 0.816 | 0.660 | 0.557 | 0.484 | 0.431 | 0.390 | 0.358 | 0.311 | 0.279 | 0.256 | 0.239 | 0.226 | 0.179 | 0.165 | 0.162 | 0.161 | 0.161 |
| 1.7 | 17.001 | 3.404 | 1.708 | 1.146 | 0.867 | 0.701 | 0.592 | 0.515 | 0.458 | 0.414 | 0.380 | 0.330 | 0.296 | 0.272 | 0.254 | 0.240 | 0.190 | 0.175 | 0.172 | 0.171 | 0.171 |
| 1.8 | 18.001 | 3.604 | 1.809 | 1.213 | 0.918 | 0.742 | 0.626 | 0.545 | 0.485 | 0.439 | 0.402 | 0.350 | 0.314 | 0.288 | 0.269 | 0.255 | 0.201 | 0.186 | 0.182 | 0.181 | 0.181 |
| 1.9 | 19.001 | 3.805 | 1.909 | 1.281 | 0.969 | 0.783 | 0.661 | 0.575 | 0.512 | 0.463 | 0.425 | 0.369 | 0.331 | 0.304 | 0.284 | 0.269 | 0.212 | 0.196 | 0.193 | 0.191 | 0.191 |
| 2.0 | 20.001 | 4.005 | 2.010 | 1.348 | 1.020 | 0.825 | 0.696 | 0.605 | 0.539 | 0.487 | 0.447 | 0.389 | 0.349 | 0.320 | 0.299 | 0.283 | 0.224 | 0.206 | 0.203 | 0.202 | 0.201 |
| 2.1 | 21.001 | 4.205 | 2.110 | 1.416 | 1.071 | 0.866 | 0.731 | 0.636 | 0.565 | 0.512 | 0.470 | 0.408 | 0.366 | 0.336 | 0.314 | 0.297 | 0.235 | 0.216 | 0.213 | 0.212 | 0.211 |
| 2.2 | 22.001 | 4.405 | 2.211 | 1.483 | 1.122 | 0.907 | 0.766 | 0.666 | 0.592 | 0.536 | 0.492 | 0.428 | 0.384 | 0.352 | 0.329 | 0.311 | 0.246 | 0.227 | 0.223 | 0.222 | 0.221 |
| 2.3 | 23.001 | 4.606 | 2.311 | 1.550 | 1.173 | 0.948 | 0.800 | 0.696 | 0.619 | 0.560 | 0.514 | 0.447 | 0.401 | 0.368 | 0.344 | 0.325 | 0.257 | 0.237 | 0.233 | 0.232 | 0.231 |
| 2.4 | 24.001 | 4.806 | 2.412 | 1.618 | 1.224 | 0.990 | 0.835 | 0.727 | 0.646 | 0.585 | 0.537 | 0.466 | 0.419 | 0.384 | 0.359 | 0.339 | 0.268 | 0.247 | 0.243 | 0.242 | 0.241 |
| 2.5 | 25.001 | 5.006 | 2.512 | 1.685 | 1.275 | 1.031 | 0.870 | 0.757 | 0.673 | 0.609 | 0.559 | 0.486 | 0.436 | 0.400 | 0.374 | 0.354 | 0.280 | 0.258 | 0.253 | 0.252 | 0.251 |
| 2.6 | 26.001 | 5.206 | 2.613 | 1.753 | 1.326 | 1.072 | 0.905 | 0.787 | 0.700 | 0.634 | 0.581 | 0.505 | 0.453 | 0.416 | 0.389 | 0.368 | 0.291 | 0.268 | 0.264 | 0.262 | 0.261 |
| 2.7 | 27.001 | 5.407 | 2.713 | 1.820 | 1.377 | 1.113 | 0.940 | 0.817 | 0.727 | 0.658 | 0.604 | 0.525 | 0.471 | 0.432 | 0.404 | 0.382 | 0.302 | 0.278 | 0.274 | 0.272 | 0.271 |
| 2.8 | 28.001 | 5.607 | 2.814 | 1.888 | 1.428 | 1.154 | 0.974 | 0.848 | 0.754 | 0.682 | 0.626 | 0.544 | 0.488 | 0.448 | 0.419 | 0.396 | 0.313 | 0.289 | 0.284 | 0.282 | 0.281 |
| 2.9 | 29.001 | 5.807 | 2.914 | 1.955 | 1.479 | 1.196 | 1.009 | 0.878 | 0.781 | 0.707 | 0.648 | 0.564 | 0.505 | 0.464 | 0.434 | 0.410 | 0.324 | 0.299 | 0.294 | 0.292 | 0.291 |
| 3.0 | 30.001 | 6.007 | 3.015 | 2.022 | 1.530 | 1.237 | 1.044 | 0.908 | 0.808 | 0.731 | 0.671 | 0.583 | 0.523 | 0.480 | 0.448 | 0.424 | 0.335 | 0.309 | 0.304 | 0.302 | 0.301 |
| 3.1 | 31.002 | 6.208 | 3.115 | 2.090 | 1.581 | 1.278 | 1.079 | 0.938 | 0.835 | 0.755 | 0.693 | 0.603 | 0.541 | 0.496 | 0.463 | 0.438 | 0.347 | 0.320 | 0.314 | 0.312 | 0.312 |
| 3.2 | 32.002 | 6.408 | 3.216 | 2.157 | 1.632 | 1.319 | 1.114 | 0.969 | 0.862 | 0.780 | 0.716 | 0.622 | 0.558 | 0.512 | 0.478 | 0.453 | 0.358 | 0.330 | 0.324 | 0.322 | 0.322 |
| 3.3 | 33.002 | 6.608 | 3.316 | 2.225 | 1.683 | 1.361 | 1.148 | 0.999 | 0.889 | 0.804 | 0.738 | 0.641 | 0.575 | 0.528 | 0.493 | 0.467 | 0.369 | 0.340 | 0.335 | 0.333 | 0.332 |
| 3.4 | 34.002 | 6.808 | 3.417 | 2.292 | 1.734 | 1.402 | 1.183 | 1.029 | 0.915 | 0.829 | 0.760 | 0.661 | 0.593 | 0.544 | 0.508 | 0.481 | 0.380 | 0.350 | 0.345 | 0.343 | 0.342 |
| 3.5 | 35.002 | 7.009 | 3.517 | 2.359 | 1.785 | 1.443 | 1.218 | 1.059 | 0.942 | 0.853 | 0.783 | 0.680 | 0.610 | 0.560 | 0.523 | 0.495 | 0.391 | 0.361 | 0.355 | 0.353 | 0.352 |
| 3.6 | 36.002 | 7.209 | 3.618 | 2.427 | 1.836 | 1.484 | 1.253 | 1.090 | 0.969 | 0.877 | 0.805 | 0.700 | 0.628 | 0.576 | 0.538 | 0.509 | 0.402 | 0.371 | 0.365 | 0.363 | 0.362 |
| 3.7 | 37.002 | 7.409 | 3.718 | 2.494 | 1.887 | 1.526 | 1.288 | 1.120 | 0.996 | 0.902 | 0.827 | 0.719 | 0.645 | 0.592 | 0.553 | 0.523 | 0.414 | 0.381 | 0.375 | 0.373 | 0.372 |
| 3.8 | 38.002 | 7.609 | 3.819 | 2.562 | 1.938 | 1.567 | 1.322 | 1.150 | 1.023 | 0.926 | 0.850 | 0.739 | 0.663 | 0.608 | 0.568 | 0.537 | 0.425 | 0.392 | 0.385 | 0.383 | 0.382 |
| 3.9 | 39.002 | 7.810 | 3.919 | 2.629 | 1.989 | 1.608 | 1.357 | 1.181 | 1.050 | 0.950 | 0.872 | 0.758 | 0.680 | 0.624 | 0.583 | 0.552 | 0.436 | 0.402 | 0.395 | 0.393 | 0.392 |
| 4.0 | 40.002 | 8.101 | 4.020 | 2.696 | 2.040 | 1.649 | 1.392 | 1.211 | 1.077 | 0.975 | 0.894 | 0.777 | 0.698 | 0.640 | 0.598 | 0.566 | 0.447 | 0.412 | 0.406 | 0.403 | 0.402 |
| 4.1 | 41.002 | 8.010 | 4.120 | 2.764 | 2.091 | 1.690 | 1.427 | 1.241 | 1.104 | 0.999 | 0.917 | 0.797 | 0.715 | 0.656 | 0.613 | 0.580 | 0.458 | 0.423 | 0.416 | 0.413 | 0.412 |
| 4.2 | 42.002 | 8.410 | 4.221 | 2.831 | 2.142 | 1.732 | 1.462 | 1.271 | 1.131 | 1.023 | 0.939 | 0.816 | 0.732 | 0.672 | 0.628 | 0.594 | 0.470 | 0.433 | 0.426 | 0.423 | 0.422 |
| 4.3 | 43.002 | 8.611 | 4.321 | 2.899 | 2.193 | 1.773 | 1.496 | 1.302 | 1.158 | 1.048 | 0.962 | 0.836 | 0.750 | 0.688 | 0.643 | 0.608 | 0.481 | 0.443 | 0.436 | 0.433 | 0.432 |
| 4.4 | 44.002 | 8.811 | 4.422 | 2.966 | 2.244 | 1.814 | 1.531 | 1.332 | 1.185 | 1.072 | 0.984 | 0.855 | 0.767 | 0.704 | 0.658 | 0.622 | 0.492 | 0.454 | 0.446 | 0.443 | 0.442 |
| 4.5 | 45.002 | 9.011 | 4.522 | 3.034 | 2.295 | 1.855 | 1.566 | 1.362 | 1.212 | 1.097 | 1.006 | 0.875 | 0.785 | 0.720 | 0.673 | 0.636 | 0.503 | 0.464 | 0.456 | 0.454 | 0.452 |
| 4.6 | 46.002 | 9.211 | 4.623 | 3.101 | 2.346 | 1.897 | 1.601 | 1.392 | 1.239 | 1.121 | 1.029 | 0.894 | 0.802 | 0.736 | 0.688 | 0.651 | 0.514 | 0.474 | 0.466 | 0.464 | 0.462 |
| 4.7 | 47.002 | 9.412 | 4.723 | 3.168 | 2.397 | 1.938 | 1.636 | 1.423 | 1.266 | 1.145 | 1.051 | 0.914 | 0.820 | 0.752 | 0.703 | 0.665 | 0.525 | 0.484 | 0.476 | 0.474 | 0.472 |
| 4.8 | 48.002 | 9.612 | 4.824 | 3.286 | 2.448 | 1.979 | 1.670 | 1.453 | 1.292 | 1.170 | 1.073 | 0.933 | 0.837 | 0.768 | 0.718 | 0.679 | 0.537 | 0.495 | 0.487 | 0.484 | 0.482 |
| 4.9 | 49.002 | 9.812 | 4.924 | 3.303 | 2.499 | 2.020 | 1.705 | 1.483 | 1.319 | 1.194 | 1.096 | 0.952 | 0.854 | 0.784 | 0.732 | 0.693 | 0.548 | 0.505 | 0.497 | 0.494 | 0.492 |
| 5.0 | 50.003 | 10.012 | 5.025 | 3.371 | 2.550 | 2.062 | 1.740 | 1.514 | 1.346 | 1.218 | 1.118 | 0.972 | 0.872 | 0.800 | 0.747 | 0.707 | 0.559 | 0.515 | 0.507 | 0.504 | 0.502 |

SIMPLE MODEL

```
      DIMENSION SLOPE(501),SLENG(500)
      READ(5,510) WTRL
      READ(5,520) NUM,SLEN,SLNL
      DO 10 I=1,NUM+1
      READ (5,510)SLOPE(I)
      SLENG(I)=SLEN
10    CONTINUE
      WRITE(6,610)WTRL
      IF(SLNL.NE.0.)SLENG(NUM)=SLNL
      SHLEN=0.0
      TAREA=0.0
      DO 50 I=1,NUM
      AS=(SLOPE(I)+SLOPE(I+1))/2.
      AREA=SLENG(I)*WTRL*((1+AS**2.)**.5)/AS
      SHLEN=SLENG(I)+SHLEN
      TAREA=AREA+TAREA
      WRITE(6,620)I,SLENG(I),AS,AREA
50    CONTINUE
      WRITE(6,630)SHLEN,TAREA
510   FORMAT(F10.6)
520   FORMAT(I3,2X,2F10.2)
610   FORMAT(' WATER LEVEL FLUCTUATION =',F6.2,//,
     1 4X,'SECTION',9X,'LENGTH',7X,'AVERAGE SLOPE',12X,
     2 'AREA',/)
620   FORMAT(6X,I3,7X,F10.2,9X,F10.6,6X,F15.2)
630   FORMAT(//,2X,'SHORE LENGTH=',F10.2,15X,
     1 'SHORE AREA =',F15.2)
      STOP
      END
```

Fig. 4. Computer program for simple model to estimate shore area change in a reservoir resulting from water level fluctuation.

simple model. The input format for the computer method solution of the simple model is shown in Table 3.

Table 3. Input format for simple (and modified simple) model

| Card No. | Variables | Format |
|---|---|---|
| 1 | Expected water level fluctuation | F10.6 |
| 2 | Number of sections or intervals, section length, last section length | I3, 2X, F10.2, F10.2 |
| 3 to (n+3) | Shore (bank) slopes at interval marks: note that there are n+1 shore slope input cards for n sections | F10.6 |

### 4.2 Modified Simple Model

Section 3 describes the basis of the modified simple model. This model uses the same basic approach as the simple model except for the different method of computing the shore area. Shoreline boundary intervals for obtaining shore slopes are determined as the initial procedure of the simple model. Figure 5 presents the computer program of the modified simple model. The input format is the same as for the simple model (Table 3).

The modified simple model is the most precise method to estimate the shore area exposure. However, more detailed information is required.

### 4.3 Geometrical Model

As indicated in the development and description of the geometrical model in Section 3, two cross-sections are required in the computer model. A computer model program of the geometrical model is shown in Fig. 6. For run-of-river conditions, these two input

MODIFIED SIMPLE MODEL

```
         DIMENSION SLOPE(501),SLENG(500),AS(501)
         READ(5,510) WTRL
         READ(5,520) NUM,SLEN,SLNL
         DO 10 I=1,NUM+1
         READ(5,510)SLOPE(I)
         AS(I)=WTRL*((1.+SLOPE(I)**2.)**.5)/SLOPE(I)
         SLENG(I)=SLEN
10       CONTINUE
         WRITE(6,610)WTRL
         IF(SLNL.NE.0.)SLENG(NUM)=SLNL
         SHLEN=0.0
         TAREA=0.0
         DO 50 I=1,NUM
         ASL=(AS(I)+AS(I+1))/2.
         AREA=SLENG(I)*ASL
         SHLEN=SLENG(I)+SHLEN
         TAREA=AREA+TAREA
         WRITE(6,620)I,SLENG(I),ASL,AREA
50       CONTINUE
         WRITE(6,630)SHLEN,TAREA
510      FORMAT(F10.6)
520      FORMAT(I3,2X,2F10.2)
610      FORMAT(' WATER LEVEL FLUCTUATION =',F6.2,//,
      1 4X,'SECTION',9X,'LENGTH',7X,'AVERAGE WIDTH',12X,
      2 'AREA',/)
620      FORMAT(6X,I3,7X,F10.2,9X,F10.6,6X,F15.2)
630      FORMAT(//,2X,'SHORE LENGTH=',F10.2,15X,
      1 'SHORE AREA=',F15.2)
         STOP
         END
```

Fig. 5. Computer program for modified simple model used to estimate shore area change in a reservoir resulting from water level fluctuation.

GEOMETRICAL MODEL

```
        REAL LEVEL1,LEVEL2
        REAL L,K
        READ(5,5) TYPE
5       FORMAT(A3)
        READ (5,10) DATUM
10      FORMAT(F10.4)
        READ (5,20) ELEV1,W1,S1
        READ (5,20) BS,L,HL
        READ (5,20) ELEV2,W2,S2
        READ (5,30) LEVEL1, LEVEL2
        READ (5,10) SLINE
20      FORMAT(3F10.4)
30      FORMAT(2F10.4)
        X1=W1/2.
        X2-W2/2.
        WTRL1=LEVEL1-DATUM
        WTRL2=LEVEL2-DATUM
        IF(TYPE.EQ.'RUN')GO TO 50
        WRITE (6,40)
40      FORMAT(29X,'RESERVOIR SHORLINE',///)
        GO TO 200
50      WRITE(6,60)
60      FORMAT(27X,'RUN-OF-RIVER SHORELINE',///)
        GO TO 300
*************************************************************
200     CALL CHARA (ELEV1,DATUM,X1,S1,0,0,A)
        CALL CHARR (ELEV2,DATUM,X2,S2,L,BS,A,B)
        WRITE(6,70) A,B,BS,HL
70      FORMAT(27X,'CHANNEL CHARACTERISTICS:',/,
     1 17X,'A =',F15.12,5X,'B =',F15.12,/,
     2 6X,'STREAM SLOPE =',F6.4,4X,
     3 'WATER SLOPE =',F6.4)
        CALL SHOLIN (A,B,BS,HL,WTRL1,WTRL2,SHLENG)
        IF(B.EQ.o.) GO TO 210
        CALL AREA 1(A,B,BS,HL,WTRL1,WTRL2,AREA)
        GO TO 500
210     CALL AREA2(A,BS,HL,WTRL1,WTRL1,AREA)
        GO TO 500
```

Fig. 6. Computer program for geometric model used to estimate shore area change in a reservoir resulting from water level fluctuation.

```
300     CALL CHARA(ELEV1,DATUM,X1,S1,0,0,A)
        CALL CHARR(ELEV2,DATUM,X2,S2,L,BS,A,B)
        WRITE(6,70) A,B,BS,HL
        Y1=(HL*L)+WTRL1-(BS*L)
        Y2=(HL*L)+WTRL2-(BS*L)
        ADA=A+(B*L)
        CALL SHOLIN(A,B,BS,HL,WTRL1,WTRL2,DSHO1)
        CALL SHOLIN(ADA,B,BS,HL,Y1,Y2,DSHO2)
        SHLENG=DSHO1-DSHO2
        IF(B.EQ.0) GO TO 310
        CALL AREA1(A,B,BS,HL,WTRL1,WTRL2,DSA1)
        CALL AREA1(ADA,B,BS,HL,Y1,Y2,DSA2)
        AREA=DSA1-DSA2
        GO TO 500
310     CALL AREA2(A,BS,HL,WTRL1,WTRL2,DSA1)
        CALL AREA2(ADA,BS,HL,Y1,Y2,DSA2)
        AREA=DSA1-DSA2
        GO TO 500
*****************************************************************
500     WRITE(6,80) WTRL1,WTRL2
80      FORMAT(/,27X,'WATER LEVEL FLUCTUATION:',/,
     1 5X, 'MINIMUM WATER LEVEL =',F10.4,5X,
     2 'MAXIMUM WATER LEVEL =',F10.4,//)
        WRITE(6,520)AREA
520     FORMAT(7X,'SIMULATED SHORELINE AREA',
     1 ' EXPOSURE =',F15.2)
        IF(SLINE.NE.0.) GO TO 550
        K=1.0
510     WRITE(6,540)SHLENG,SLINE,K,AREA
540     FORMAT(14X,'SIMULATED SHORELINE LENGTH =',F15.2,/,
     1 17X,'ACTUAL SHORELINE LENGTH =',F15.2,/,
     2 24X,'ADJUSTING FACTOR =',5X,F10.5,///,
     3 8X,'ADJUSTED SHORELINE AREA EXPOSURE =',F15.2)
        GO TO 999
550     K=SLINE/SHLENG
        AREA=AREA*K
        GO TO 510
999     STOP
        END
*****************************************************************
        SUBROUTINE CHARA(ELEV,DATUM,X,S,L,BS,A)
        REAL L
        IF(ELEV.EQ.0.) GO TO 10
        IF(X.EQ.0.) GO TO 20
        A=(ELEV-DATUM-(BS*L))/(X**2.)
        RETURN
10      A=(S/X)/2
        RETURN
20      A=(S**2.)/(4.*(ELEV-DATUM-(BS*L)))
        RETURN
        END
```

Fig. 6 (continued)

```
      SUBROUTINE CHARB(ELEV,DATUM,X,S,L,BS,A,B)
      REAL L
      IF(ELEV.EQ.0) GO TO 10
      IF(X.EQ.0.) GO TO 20
      B=(ELEV-DATUM-(BS*L)-(A*(X**2.)))/(L*(X**2.))
      RETURN
10    B=((S/(2.*X))-A)/L
      RETURN
20    B=(((S**2.)/(4.*(ELEV-DATUM-(BS*L))))
     1 -A)/L
      RETURN
      END
**************************************************************
      SUBROUTINE AREA1(A,B,S,HL,Y1,Y2,AREA)
      DIMENSION X1(501),X2(501),FN1(501),FN2(501)
      CALL FUNCT (FN1,X1,A,B,S,HL,Y1)
      CALL INTEGR (FN1,X1,AR1)
      CALL FUNCT (FN2,X2,A,B,S,HL,Y2)
      CALL INTEGR (FN2,X2,AR2)
      AREA=AR2-AR1
      RETURN
      END
**************************************************************
      SUBROUTINE FUNCT (Y,X,A,B,S,HL,Y1)
      DIMENSION X(501),Y(501)
      DIMENSION G2(501),G(501),H(501),H2(501),SQR(501)
      DIMENSION SQRO(501),FN1(501),FN2(501),FN3(501),FN4(501)
      X(501)=Y1/A)**0.5
      D=X(501)/500.
      DO 10 J=500,1,-1
      X(J)=X(J+1)-D
10    CONTINUE
      DO 20 I=1.501
      G2(I)=(1.+(B*(X(I)**2.)+S)**2.)/
     1 (4.*(B**2.)*(X(I)**2.))
      G(I)=G2(I)**0.5
      H(I)=((Y1A*(X(I)**2.))/(B*(X(I)**2.)+S-HL)
     1 +(A+B)
      H2(I)=H(I)**2,
      SQR(I)=(G2(I)+H2(I))**0.5
      SQRO(I)=(G2(I)+(A/B)**2.)**0.5
      FN1(I)=H(I)*SQR(I)
      FN2(I)=G2(I)*ALOG((H(I)+SQR(I))/
     1 G(I))
      FN3(I)=A*SQRO(I)/B
      FN4(I)=G2(I)*ALOG(((A/B)+SQRO(I))/G(I))
      Y(I)=ABS(B)*X(I)*(FN1(I)+FN2(I)-FN3(I)-FN4(I))
20    CONTINUE
      RETURN
      END
```

Fig. 6 (continued)

```
************************************************************
      SUBROUTINE INTEGR (Y,X,AREA)
      DIMENSION Y(501),X(501)
      H=X(501)/500.0
      ODD=0.0
      EVEN=0.0
      DO10 I=2.498,2
      EVEN=EVEN+Y(I)
      ODD=ODD+Y(I+1)
10    CONTINUE
      AREA=(H/3.)*(Y(1)+4.*(EVEN+Y(500))+2.*ODD+Y(501))
      AREA=2.*AREA
      RETURN
      END
************************************************************
      SUBROUTINE AREA2(A,S,HL,Y1,Y2,AREA)
      DIMENSION FN1(501),FN2(501),X1(501),X2(501)
      X1(501)=(Y1/A)**0.5
      X2(501)=(Y2/A)**0.5
      H1=X1(501)/500.0
      H2=X2(501)/500.0
      DO 50 I=500,1,-1
      X1(I)=X1(I+1)-H1
      X2(I)=X2(I+1)-H2
50    CONTINUE
      DO 60 J=1.501
      FN1(J)=((1.+S**2.+(2.*A*X1(J))**2.)**.5)
     1 *(Y1-A*(X1(J)**2.))/(S-HL)
      FN2(J)=((1.+S**2.+(2.*A*X2(J))**2.)**.5)
     1 *(Y2-A*(X2(J)**2.))/(S-HL)
60    CONTINUE
      ODD1=0.0
      ODD2=0.0
      EVEN1=0.0
      EVEN2=0.0
      DO 70 K=2.498,2
      EVEN1=EVEN1+FN1(K)
      EVEN2=EVEN2+FN2(K)
      ODD1=ODD1+FN1(K+1)
      ODD2=ODD2+FN2(K+1)
70    CONTINUE
      AR1=(H1/3.0)*(FN1(1)+4.*(EVEN1
     1 +FN1(500))+2,*ODD1+FN1(501))
      AR2=(H2/3.0)*(FN2(1)+4.*(EVEN2
     1 +FN2(500))+2.*ODD2+FN2(501))
      AREA=(AR2-AR1)*2.
      RETURN
      END
```

Fig. 6 (continued)

```
************************************************************
      SUBROUTINE SHOLIN(A,B,S,HL,Y1,Y2,SHOLEN)
      DIMENSION Y(501),X(501),F1(501),F2(501),Z(501)
      Y0=(Y2+Y1)/2.
      X(501)=(Y0/A)**0.5
      D=X(501)/500.
      DO 10 I=500,1,-1
      X(I)=X(I+1)-D
10    CONTINUE
      DO 20 I=1.501
      F1(I)=B*(X(I)**2.)+S-HL
      F2(I)=Y0-(A*(X(I)**2.))
      Z(I)=2.*(A*X(I)*F1(I)+B*X(I)*F2(I))/F1(I)**2.)
      Y(I)=(1.+7(I)**2.)**0.5
20    CONTINUE
      CALL INTEGR(Y,X,SHOLEN)
      RETURN
      END
```

Fig. 6 (continued)

cross sections should be at both ends of the river. For a single stream reservoir (pool condition), one input cross section should be determined at the dam site of the stream, and another cross section should be defined at a distance about one-half the reservoir length upstream from the dam, but never more than two-thirds of the reservoir length. In some cases, if only a single cross section at the dam is available, use of the input format given in step 5b of Table 4 automatically adjusts for this condition.

The geometrical model has an advantage in the ability to estimate shore area even though limited data are available. However, this model is constructed for reservoirs with no or only slight shoreline irregularity, and the model is not recommended for reservoirs with "iregularity factors" greater than 3.0. Irregularity factor F is defined as the ratio of the actual shoreline length to the main stream length in a reservoir.

## 5. APPLICATION EXAMPLES

This section demonstrates the use of each model and compares results for four selected reservoirs. Although all four reservoirs represent a Case I condition (Table 1), applications of the geometrical model are performed to demonstrate the use of the geometrical model. The four reservoirs selected for analysis and their general characteristics are as follows:

| Reservoir | Type | Shoreline configuration | Contour map scale |
|---|---|---|---|
| Derwent | Reservoir pool | Regular | 1" = 1 km |
| Normandy (part of the reservoir) | Reservoir pool | Slightly irregular | 1:12000; (1" = 1000 ft) |
| Nottely | Reservoir pool | Irregular | 1:24000; (1" = 2000 ft) |
| Ocoee #2 | Run-of-river | Regular | 1:24000 (1" = 2000 ft) |

Table 4. Input format for geometric model

| Card No. | | Variables | Format |
|---|---|---|---|
| 1 | | Reservoir type | Run-of-river or reservoir |
| 2 | | Elevation of dam base, DATUM | F10.4 |
| 3 | | Cross-section data at the dam: elevation of a point, ELEV1; width of the reservoir at the point, W1; shore slope at the point, $S_1$ (input only if elevation or width is not available) | F10.4, F10.4, F10.4 |
| 4 | | Stream bed slope, $S_b$ or BS;[a] distance between the cross section at the dam and the other cross section upstream, L (input 1.0 if only cross section data at the dam is available); water surface slope, HL (less than the stream bed slope) | F10.4, F10.4, F10.4 |
| 5 | a | Two cross sections Elevation of a point, ELEV2; width of the reservoir at the point W2; shore slope at the point, S2, (input only if elevation of width is not available) | F10.4, F10.4, F10.4 |
| | b | Only one cross section ELEV1 + $S_b$ (or GS), W1, S1 (input only if elevation or width is not available) | |
| 6 | | Expected minimum water level elevation, WTRL1; expected maximum water level elevation, WTRL2 | F10.4, F10.4, |
| 7 | | Actual shoreline length, SLINE (input 0.0 if not available) | F10.4 |

[a]BS used to represent bed slope term, $S_b$, in computer model.

Example 1: Derwent (Reservoir) (Fig. 7)

Data condition: Case I (Table 1)
Method: Simple model using Table 2 solution.

40 sections

Water fluctuation = 3.3 m (down)
Average shore slope = 0.10
Shore length = 12,000 m

From Table 2, $\dfrac{\text{shore area}}{\text{km shore length}} = 3.316$

Total shore area $= 3.316 \times \dfrac{12000}{1000} = 39.792$ hectares $= 397{,}920\ m^2$

Example 2: Normandy Lake (Reservoir) (Fig. 8)

Data condition: Case I (Table 1)

Method: Simple model and modified simple model (Figs. 1 and 2)

Input data (lengths in ft):

| | | Section intervals | Last Section length |
|---|---|---|---|
| Water level fluc. | 5.0 | 1000.0 | 1000.0 |
| No. of sections | 41 | | |
| Shore slopes: | 0.3 | | |
| | 0.3 | | |
| | 0.3 | | |
| | 0.3 | | |
| | 0.2 | | |
| | 0.25 | | |
| | 0.3 | | |
| | 0.3 | | |
| | 0.01 | | |
| | 0.20 | | |
| | 0.25 | | |
| | 0.3 | | |
| | 0.3 | | |
| | 0.3 | | |
| | 0.25 | | |
| | 0.3 | | |
| | 0.25 | | |
| | 0.2 | | |
| | 0.2 | | |

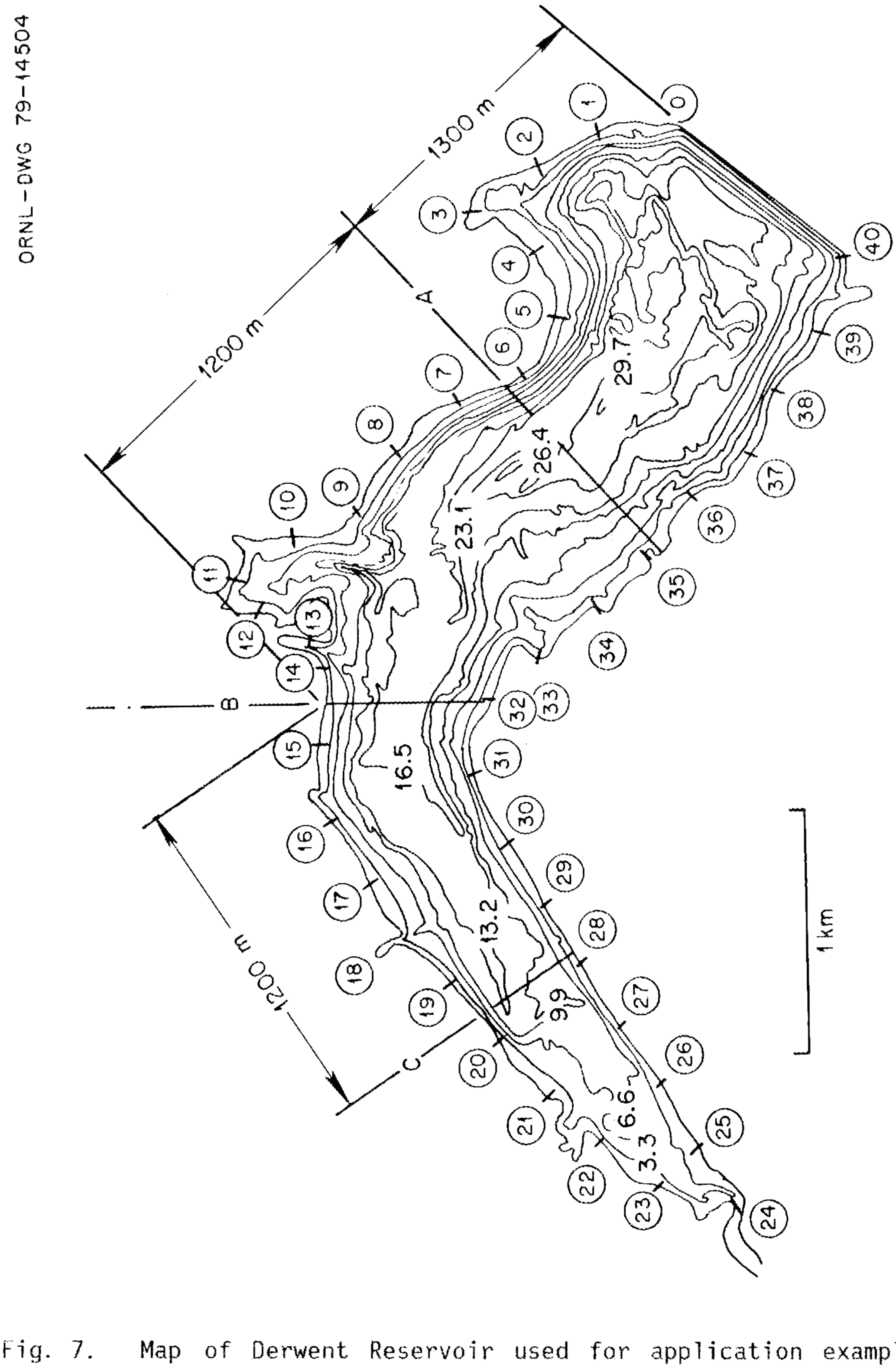



Fig. 7. Map of Derwent Reservoir used for application examples (Section 5) for predicting shore area change.

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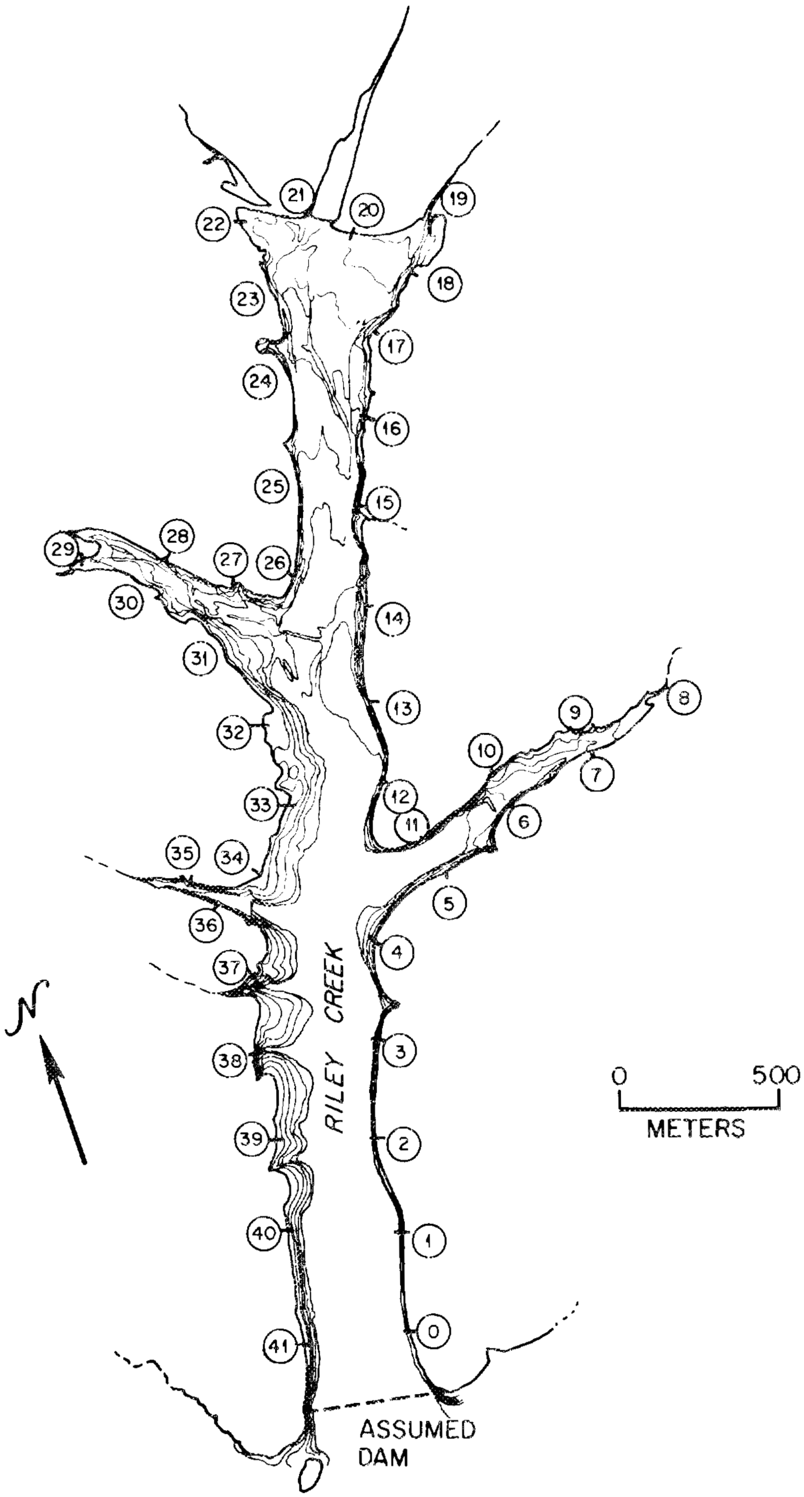


Fig. 8. Map of Normandy Reservoir used for application examples (Section 5) for predicting shore area change.

0.01
0.02
0.01
0.03
0.1
0.3
0.3
0.25
0.25
0.1
0.03
0.1
0.15
0.05
0.08
0.06
0.02
0.3
0.15
0.25
0.1
0.15
0.15

Example 2: Simple model method output (shore length in ft, shore area in $ft^2$):

| Section | Length | Average slope | Area |
|---|---|---|---|
| 1 | 1000.00 | 0.300000 | 17400.51 |
| 2 | 1000.00 | 0.300000 | 17400.51 |
| 3 | 1000.00 | 0.300000 | 17400.51 |
| 4 | 1000.00 | 0.250000 | 20615.53 |
| 5 | 1000.00 | 0.225000 | 22777.78 |
| 6 | 1000.00 | 0.275000 | 18856.79 |
| 7 | 1000.00 | 0.300000 | 17400.51 |
| 8 | 1000.00 | 0.155000 | 32643.26 |
| 9 | 1000.00 | 0.105000 | 47880.83 |
| 10 | 1000.00 | 0.225000 | 22777.78 |
| 11 | 1000.00 | 0.275000 | 18856.79 |
| 12 | 1000.00 | 0.300000 | 17400.51 |
| 13 | 1000.00 | 0.300000 | 17400.51 |
| 14 | 1000.00 | 0.275000 | 18856.79 |
| 15 | 1000.00 | 0.275000 | 18856.79 |
| 16 | 1000.00 | 0.275000 | 18856.79 |
| 17 | 1000.00 | 0.225000 | 22777.78 |
| 18 | 1000.00 | 0.200000 | 25495.10 |

| Section | Length | Average slope | Area |
|---|---|---|---|
| 19 | 1000.00 | 0.105000 | 47880.83 |
| 20 | 1000.00 | 0.015000 | 333370.82 |
| 21 | 1000.00 | 0.015000 | 333370.82 |
| 22 | 1000.00 | 0.020000 | 250049.99 |
| 23 | 1000.00 | 0.065000 | 77085.40 |
| 24 | 1000.00 | 0.200000 | 25495.10 |
| 25 | 1000.00 | 0.300000 | 17400.51 |
| 26 | 1000.00 | 0.275000 | 18856.79 |
| 27 | 1000.00 | 0.250000 | 20615.53 |
| 28 | 1000.00 | 0.175000 | 29005.63 |
| 29 | 1000.00 | 0.065000 | 77085.40 |
| 30 | 1000.00 | 0.065000 | 77085.40 |
| 31 | 1000.00 | 0.125000 | 40311.29 |
| 32 | 1000.00 | 0.100000 | 50249.38 |
| 33 | 1000.00 | 0.065000 | 77085.40 |
| 34 | 1000.00 | 0.070000 | 71603.36 |
| 35 | 1000.00 | 0.040000 | 125099.96 |
| 36 | 1000.00 | 0.160000 | 31647.47 |
| 37 | 1000.00 | 0.225000 | 22777.78 |
| 38 | 1000.00 | 0.200000 | 25495.10 |
| 39 | 1000.00 | 0.175000 | 29005.63 |
| 40 | 1000.00 | 0.125000 | 40311.29 |
| 41 | 1000.00 | 0.150000 | 33706.25 |

Shore length = 41000.00 Shore area = 2248250.20

Water level fluctuation = 5.00

Example 2: (Continued)

Modified simple model output (shore length in ft, shore area in $ft^2$):

| Section | Length | Average width | Area |
|---|---|---|---|
| 1 | 1000.00 | 17.400511 | 17400.51 |
| 2 | 1000.00 | 17.400511 | 17400.51 |
| 3 | 1000.00 | 17.400511 | 17400.51 |
| 4 | 1000.00 | 21.447804 | 21447.80 |
| 5 | 1000.00 | 23.055313 | 23055.31 |
| 6 | 1000.00 | 19.008019 | 19008.02 |
| 7 | 1000.00 | 17.400511 | 17400.51 |
| 8 | 1000.00 | 258.712750 | 258712.75 |
| 9 | 1000.00 | 262.760050 | 262760.05 |
| 10 | 1000.00 | 23.055313 | 23055.31 |
| 11 | 1000.00 | 19.008019 | 19008.02 |
| 12 | 1000.00 | 17.400511 | 17400.51 |

| Section | Length | Average width | Area |
|---|---|---|---|
| 13 | 1000.00 | 17.400511 | 17400.51 |
| 14 | 1000.00 | 19.008019 | 19008.02 |
| 15 | 1000.00 | 19.008019 | 19008.02 |
| 16 | 1000.00 | 19.008019 | 19008.02 |
| 17 | 1000.00 | 23.055313 | 23055.31 |
| 18 | 1000.00 | 25.495097 | 25495.10 |
| 19 | 1000.00 | 262.760050 | 262760.05 |
| 20 | 1000.00 | 375.037490 | 375037.50 |
| 21 | 1000.00 | 375.037490 | 375037.50 |
| 22 | 1000.00 | 333.383320 | 333383.32 |
| 23 | 1000.00 | 108.495510 | 108495.51 |
| 24 | 1000.00 | 33.824944 | 33824.94 |
| 25 | 1000.00 | 17.400511 | 17400.51 |
| 26 | 1000.00 | 19.008019 | 19008.02 |
| 27 | 1000.00 | 20.615528 | 20615.53 |
| 28 | 1000.00 | 35.432453 | 35432.53 |
| 29 | 1000.00 | 108.495510 | 108495.51 |
| 30 | 1000.00 | 108.495510 | 108495.51 |
| 31 | 1000.00 | 41.977812 | 41977.81 |
| 32 | 1000.00 | 66.915584 | 66915.58 |
| 33 | 1000.00 | 81.412300 | 81412.30 |
| 34 | 1000.00 | 73.091439 | 73091.44 |
| 35 | 1000.00 | 166.766600 | 166766.60 |
| 36 | 1000.00 | 133.725250 | 133725.25 |
| 37 | 1000.00 | 25.553379 | 25553.38 |
| 38 | 1000.00 | 27.160887 | 27160.89 |
| 39 | 1000.00 | 35.432453 | 35432.45 |
| 40 | 1000.00 | 41.977812 | 41977.81 |
| 41 | 1000.00 | 33.706247 | 33706.25 |

Shore length = 41000.00 Shore area = 3362730.80

Water level fluctuation = 5.00

Example 3: Ocoee #2 Dam (run-of-river) (Fig. 9)

Data condition: Case I (used as Case IV here, Table 1)
Method: Geometrical model (Fig. 6)
Input data (length in ft), Table 4:

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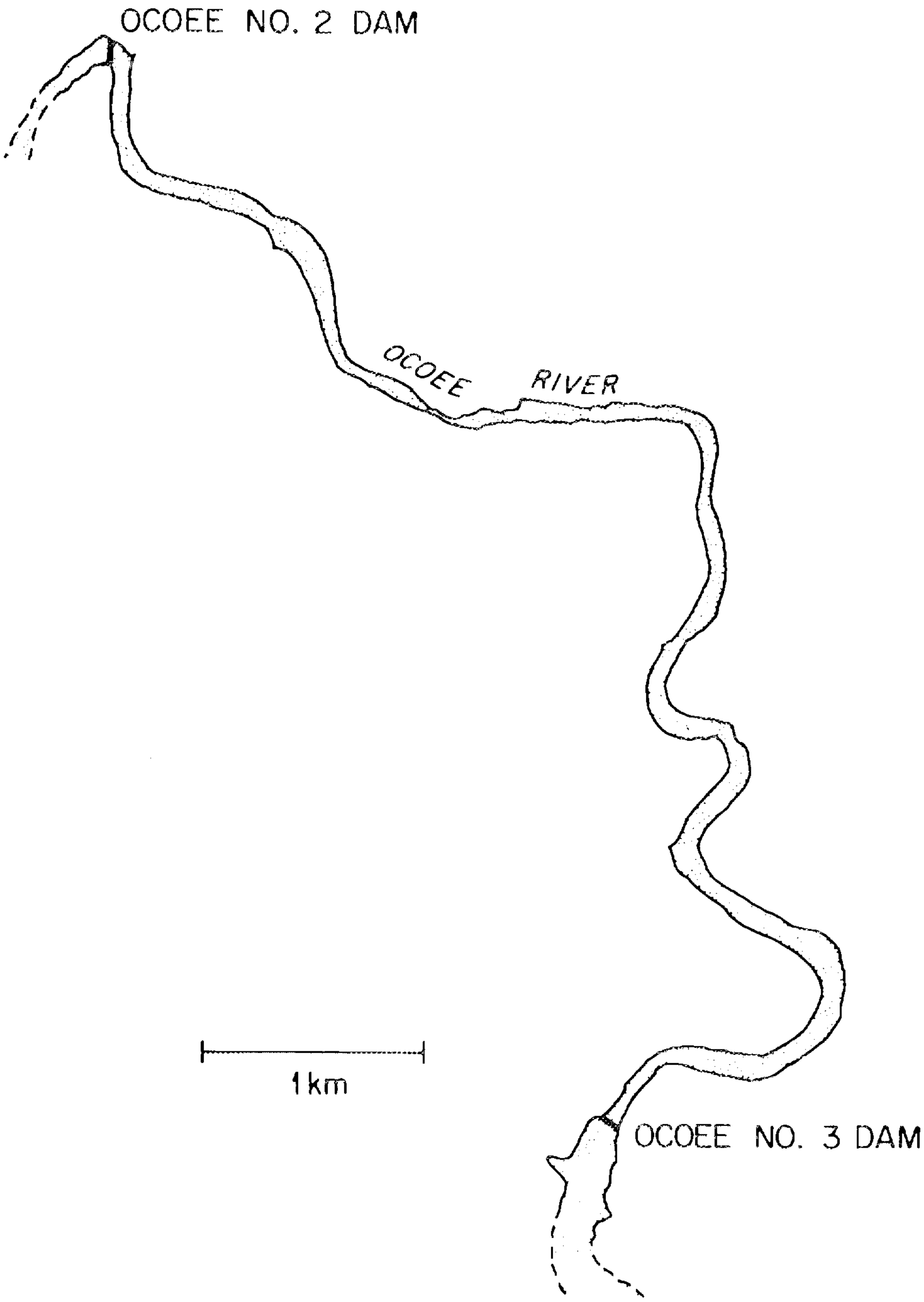


Fig. 9. Map of Ocoee No. 2 Reservoir used in application examples (Section 5) for predicting shore area change.

Run-Of-River

| | | |
|---|---|---|
| 0.00 | | |
| | 600.0 | .5333 |
| 0.0155 | 26400.0 | .0150 |
| | 50.0 | .8636 |
| 45.00 | 50.00 | |
| 54800.0 | | |

Geometrical model output (length in ft, area in $ft^2$)

Run-of-River Shoreline

Channel characteristics:

A = 0.000888833330          B = 0.000000620574

Stream slope = 0.0155          Water slope = 0.0150

Water level fluctuation:

Minimum water level = 45.0000          Maximum water level = 50.0000

Simulated shoreline area exposure = 417130.91

Simulated shoreline length = 52805.09

Actual shoreline length = 54800.00

Adjusting factor = 1.03778

Adjusted shoreline area exposure = 432889.56

Example 4: Derwent (Reservoir) (Fig. 7)

Data condition: Case I (used as Case IV here, Table 1)
Method: Geometrical model (Fig. 6)
Input data (length in m) (Table 4):

Reservoir

| | | |
|---|---|---|
| 0.00 | | |
| 30.0 | 900.0 | |
| 0.0057 | 2500.0 | 0.0 |
| 30.0 | 600.0 | |
| 26.7 | 30.0 | |
| 12000.0 | | |

Geometrical model output (length in m, area in $m^2$)

Reservoir shoreline

Channel characteristics:

A = 0.000148148150 B = 0.000000010741

Stream slope = 0.0057 Water slope = 0.0000

Water level fluctuation:

Minimum water level = 26.7000 Maximum water level = 30.0000

Simulated shoreline area exposure = 457215.78

Simulated shoreline length = 10018.28

Actual shoreline length = 12000.00

Adjusting factor = 1.19781

Adjusted Shoreline Area Exposure = 547657.62

In addition to the example work shown above, simple model, modified simple model, and geometrical models were applied to all four selected reservoirs except Nottely Lake (Fig. 10), where the geometrical model cannot be applied because of the high shoreline irregularity configuration. The results of different models applied to these four selected reservoirs are tabulated in Table 5 for comparison.

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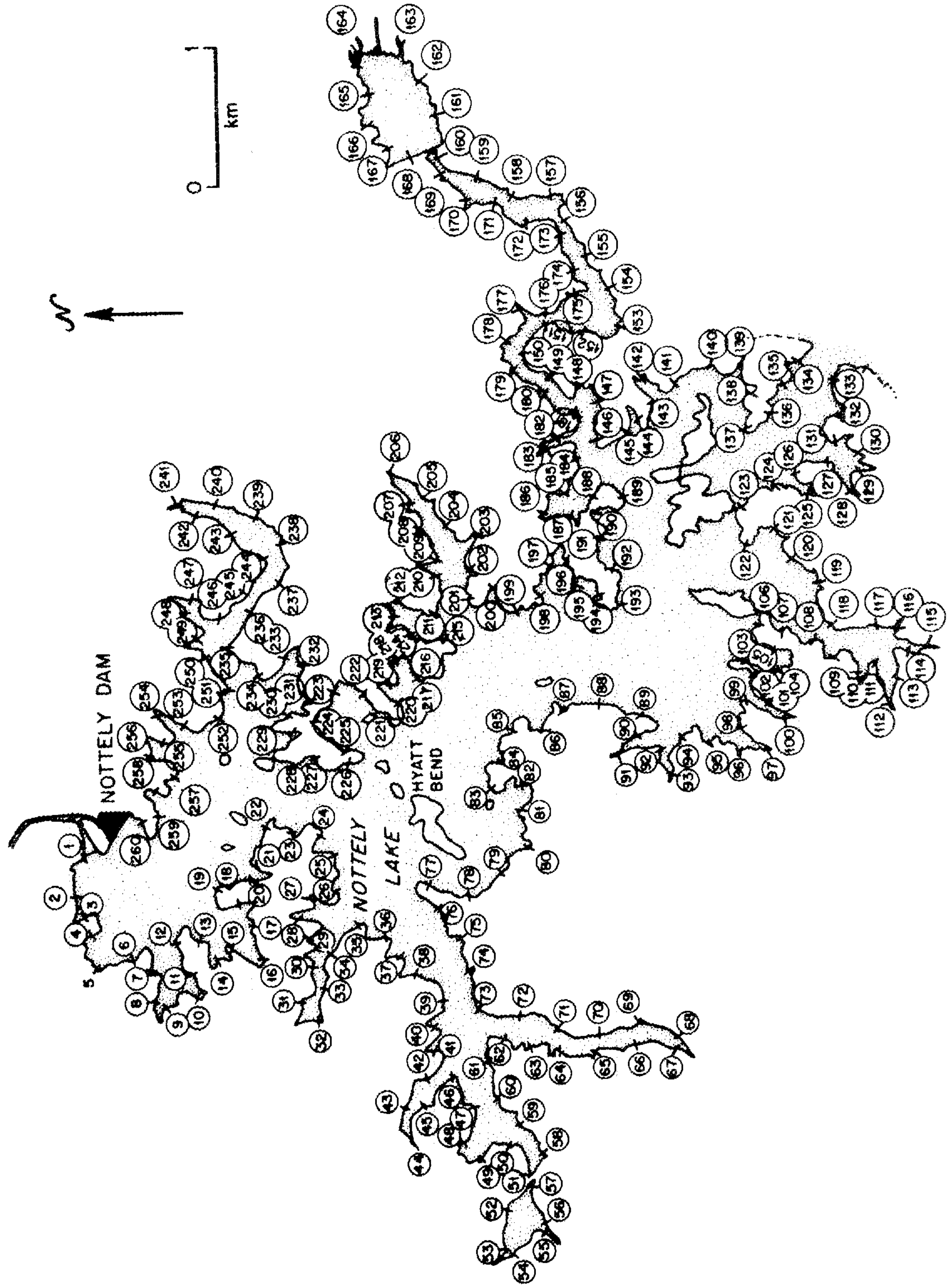


Fig. 10. Map of Nottely Reservoir used in application examples (Section 5) for predicting shore area change.

Table 5. Comparison of model use results for four reservoirs

| Reservoir | Model used | | |
|---|---|---|---|
| | Simple | Modified simple | Geometrical |
| Derwent | 476,913 $m^2$(a) | 525,487 $m^2$ | 547,658 $m^2$ |
| Normandy | 2,248,250 $ft^2$ | 3,362,731 $ft^2$ | 2,497,488 $ft^2$ |
| Nottely | 13,618,715 $ft^2$ | 15,622,420 $ft^2$ | b |
| Ocoee No. 2 | 625,425 $ft^2$ | 651,266 $ft^2$ | 432,890 $ft^2$ |

[a] 397,920 $m^2$ using Table 2.

[b] Geometrical model not applied since irregularity factor, F, is greater than 3.0.

Based on the results, the modified simple model generally gives higher estimates on shore area exposure. However, from the discussion presented in Section 3, this is to be expected when compared with the "simple" model approximation. Note that the geometrical model predicts shore area change within about one-third of the "true" value as assumed from the modified simple approach.

# 6. GLOSSARY OF TERMS

| Symbol | Meaning |
|---|---|
| A | Reservoir characteristic parameter used in geometrical model |
| B | Reservoir characteristic parameter used in geometrical model |
| $\Delta A$ | Incremental shore area, for incremental shoreline distance $\Delta L$, and resulting from water level fluctuation, $\Delta H$. |
| $\Delta L$ | Increment of shore length |
| $S_1$, $S_2$, $S_i$ | Local shore or bank slope, dy/dx |
| $\bar{S}$ | Average shore slope, either within $\Delta L$ or for entire reservoir shore |
| n | Number of $\Delta L$ increments |
| BS, $S_b$ | Longitudinal or streambed slope, dy/dx; interchangeable |
| WTRL | Water level elevation, usually taken to be at dam. Subscripts 1, 2 refer to initial and final conditions, respectively. |
| HL | Hydraulic grade line or longitudinal water surface slope of reservoir (assumed $< S_b$) |
| x, y, z | Cartesian coordinates used in developing geometrical model referenced at bottom centerline of dam; x = width; y = depth; z = upstream distance, subscript i refers to downstream cross section |
| L | Distance (reach) along z, between dam and upstream cross sections |

W . . . . . . . . . . . . . . . . . . Width of reservoir at prescribed point or cross section

F . . . . . . . . . . . . . . . . . . Reservoir shoreline irregularity factor, ratio of measured shoreline length of main channel reservoir length

$\Theta_1,\Theta_2$ . . . . . . . . . . . . . . . The angle of rise of the shoreline from the horizontal

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