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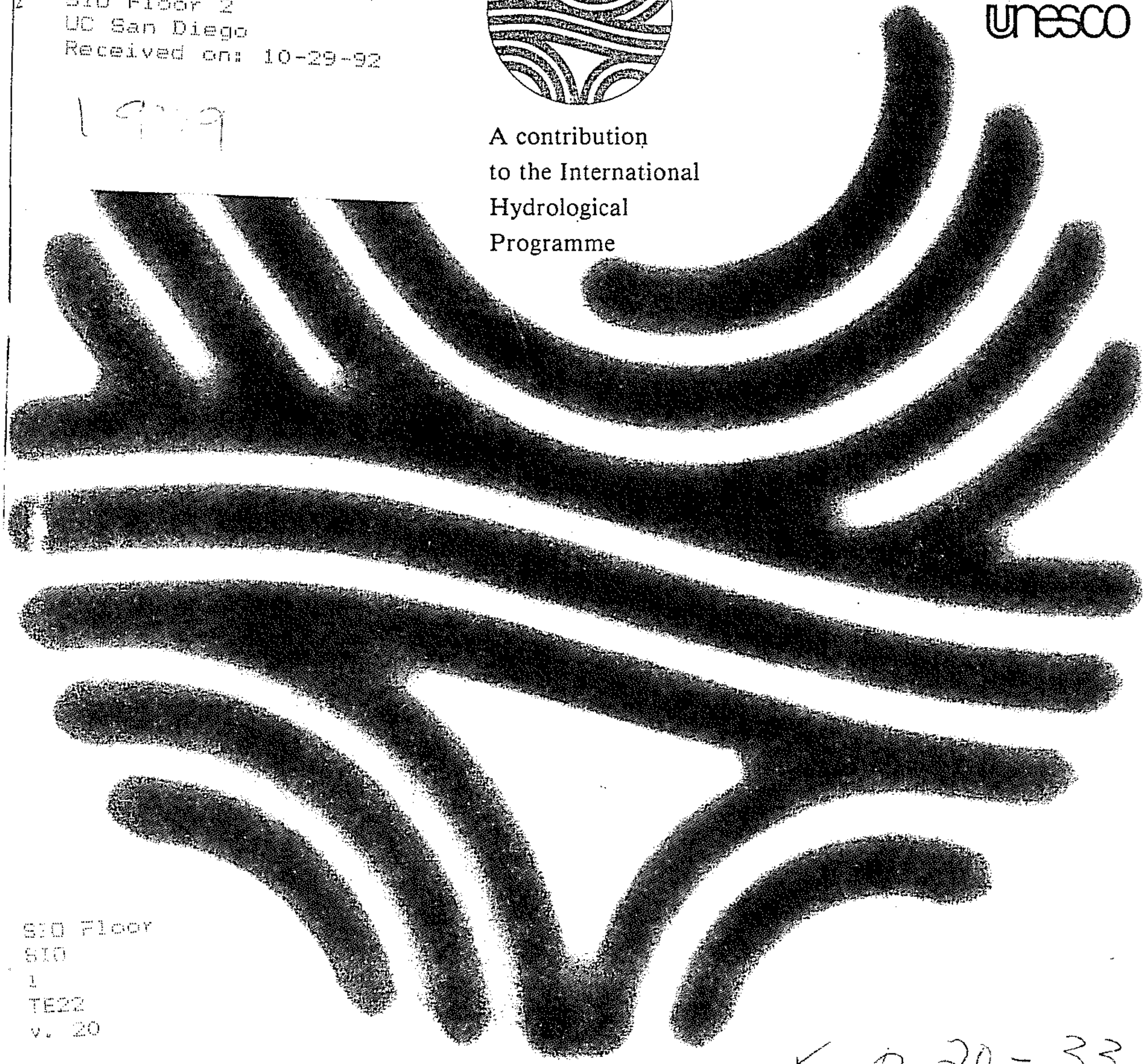

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A contribution to the
International Hydrological Programme

Predicting effects of power plant once-through cooling on aquatic systems

A state-of-the-art report
of IHP Working Group 6.2
on the effects
of thermal discharges

Chief editors:
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II. Potential and observed ecological effects of once-through cooling systems

SUMMARY

This chapter surveys the sources of potential ecological damage for power plant cooling systems, with emphasis on once-through cooling. Heated discharges are only one of several sources of potential adverse ecological changes, others being physical damages of entrapment and impingement at intakes, physical, chemical as well as thermal stresses of entrainment through the cooling circuit, discharge chlorination toxicity, and hydrological and habitat changes with the construction new intake and discharge structures. Observed evidence for these changes are detailed to illustrate that many of the impacts can be real, and not merely hypothesized.

II.1 Sources of Potential Ecological Changes

The heated discharge of a power station is only one of several sources of potential ecological change from the cooling system operations. In addition, researchers also now recognize the potential for ecological effects associated with power plant intakes and processes associated with passage of water through the power plant. Large fish and invertebrates are often impinged and killed on intake screens which are designed to keep debris out of the condenser tubes; small organisms, particularly larval stages of fish, can be mutilated or thermally killed during their transit (entrainment) through pumps, heat exchange condensers and piping. Whether screened out or entrained with the cooling water, organisms entrained in the water at the intake may fare worse than organisms which only encounter the discharge plume.

The shift in attention to include intake entrainment problems as well as discharge stresses occurred with utility industry development of estuarine sites for steam electric stations. Small, freshwater rivers generally have only limited amounts of planktonic organisms that would be susceptible to entrainment. Estuaries, however, are important spawning grounds and nursery areas for large numbers of aquatic species. Here, recirculating hydraulic patterns of fresh and salt water have encouraged evolution of drifting larvae. These drifting larvae generally do not distinguish between the patterns of water flow that provide recirculation and nourishment for them within the estuary and those that draw them into power station intakes. As power stations grow both in size of individual units and in numbers of units on a given estuary, and cooling water use increases, the probability increases that a larval fish will be entrained in a power station cooling system before it leaves the estuary (see for example, US Nuclear Regulatory Commission, 1975).

The principal engineering alternative to the traditional "once-through" cooling system is the cooling tower. Yet we now recognize that this alternative has its own potential to adversely influence the environment, with impact on the terrestrial as well as the aquatic system. These impacts are briefly addressed in this chapter to indicate some of the potential environmental problems of this alternative.

The sources of potential ecological change from once-through (open cycle) power station cooling systems are currently recognized to include the following (Fig. 11.1).

1. Change in the physical (structural) features of the intake and discharge areas by dredging, filling, change of substrate (such as placing rock jetties on sand and gravel beaches), or construction of inlet and outlet works (such as intake pumphouses

or discharge pipes);

2. Changed current patterns near the intake and discharge which may extend to areas far removed from the power station and result in altered patterns of such factors as estuarine salinity, lake thermal (and biochemical) stratification or movement of nutrient and plankton concentrations;
3. Entrapment and impingement of larger organisms, principally fish, on intake screens;
4. Entrainment of phytoplankton, zooplankton, and larvae and juveniles of fish and invertebrates with the pumped cooling water during which these organisms are exposed to:
   - (a) physical damage from mechanical contacts with pumps and piping, and physical effects of pressure changes and shear;
   - (b) temperature shock in the condenser tubes followed by a period of exposure to that elevated temperature until the discharge reaches cooler receiving waters;
   - (c) chemical exposures, principally to chlorine which is added periodically to the circulating water as a biocide to prevent accumulations of fouling materials on heat exchange surfaces and other parts of the piping system, but also including in some plants such varied materials as laundry wastes or radionuclides;
5. Plume entrainment of organisms in the discharge area through dilution of the effluent where organisms receive thermal and chemical exposures which vary according to their location in the mixing zone (physical damages may also result if pumps are used to augment effluent mixing);
6. Temperature elevation for resident and thermally attracted organisms, which is greatest in the vicinity of the discharge and is less at more remote locations, and may influence to some degree the whole of small water bodies;
7. Unnatural temperature changes, often rapid, which may occur in the vicinity of the discharge due to plant operations (e.g. sudden shutdown or start-ups) or due to environmental changes which affect rates of mixing and dispersion of the effluent;
8. Chemical (biocides and condenser metals) exposure for plume biota;
9. Changes in dissolved gas concentrations in the intake and effluent areas due to increased biochemical oxygen demand of warmed waters, to pumping of oxygen-poor hypolimnetic waters, or to gas supersaturation of discharge waters in winter;
10. Increase in nutrients in the effluent area due to kill of plant-entrained plankton;
11. Combinations of the above, which may cause effects greater than the sum of individual effects (synergism).

Cooling towers have several sources of potential ecological change, some of which are unique while some are similar to, but of lesser magnitude, than those for once-through systems. They include:

1. Impingement and entrainment of aquatic organisms at the intake where makeup water is added to the cooling loop to compensate for evaporation and dilution ("blow-down") flows;
2. Chemicals released to water bodies as the "blow-down" or dilution flow released from the "closed-cycle" loop to prevent build-up of dissolved solids, which contains materials added to the cooling loop to prevent corrosion (e.g. chromates, zinc, organophosphorous complexes) or to eliminate biological fouling (e.g. chlorine);
3. Chemical "drift" in the form of small droplets and aerosols which emerge to the terrestrial environment from the top of the tower and contain, in addition to water, chemicals used in the circulatory water system;
4. Temperature elevations or other changes due to heat in the "blow-down" releases;
5. Meteorological effects, including fogging, which affect the terrestrial environment, including man;
6. Combinations of the above (synergism).

II.2 Potential for Adverse Physical/Chemical Changes

Pollution has often been defined as a change in water or air quality that adversely affects other uses. Power plant cooling can change the physical and chemical characteristics of water and air so that other direct uses are impaired. Accelerating trends in many countries toward closed-cycle cooling, especially cooling towers, has encouraged broadened consideration of impacts on air as well as water.

II.2.1 Physical effects on water

Temperature changes are known to affect every physical property of concern in water quality management, including water density, state, viscosity, vapour pressure, surface tension gas solubility and diffusion (Appendix IV). Some of these changes are of importance in their subsequent effects on aquatic life. For example, temperature-induced gas solubility changes affect dissolved gas content and can create supersaturated gas conditions that cause gas bubble disease in aquatic orgniamss (Wolke et al., 1975). (This phenomenon is discusssed further in Section II.3e.4 on biological effects.)

Vapour pressure changes can influence rates of evaporation and thus water consumption. In many regions the quantity of water which is transferred through evaporation by various types of cooling systems is becoming a major factor in the siting and design of large steam-electric power generating facilities. The amounts of water evaporated depends on the specific environmental and plant condition involved. For example, for open water surfaces such as lakes, ponds, rivers, reservoirs and estuaries, about half of the heat is dissipated by evaporation, whereas with wet cooling towers over 75% of the heat is transferred by evaporation during the summer. The magnitude of the potential water consumption problem for large power plant facilities can be considerable. Approximately 2000 m^3/h of water are transferred to the atmosphere through evaporation in a wet cooling tower for a 1000 MW(e) nuclear power plant (IAEA, 1974). A power station can, therefore, be in direct competition for water resources with other uses such as agricultural irrigation and drinking water supplies.

Increased temperature and the resulting decreased viscosity may also result in increased sedimentation in water bodies. This could lead to potential sludge problems, changed sediment carrying capacity of rivers or changes in riverbed.

Temperature-induced density stratification of lakes is a principal regulator of chemical water quality in deep hypolimnetic waters. Changes in thermal structure by power stations can alter the normal annual cyclic pattern. Municipal or industrial water works could be affected by such changes. Problems of a similar nature can arise in stratified estuaries where the ecosystem depends upon the complex stratification and mixing patterns of saline and freshwater.

Temperature increases in winter can reduce ice-cover. This may prolong navigation in rivers and affect biota, such as attracting overwintering waterfowl. Dingman et al., (1968) estimated that a 600 M (e) nuclear power station could keep 18-25 km of the St. Lawrence River (Canada) ice free.

II.2.2 Chemical effects on water

Power stations can influence the chemistry of natural waters by changing reaction rates through temperature changes and by direct addition of chemicals to the cooling water. Altered chemical reaction rates affect the assimilation of other wastes in water bodies, the efficacy of water treatment systems, corrosion of materials, and biological processes.

Certain chemicals are added in the operation of power plant cooling systems for protection against corrosion, scale and biogenic slime build-up on heat transfer surfaces, and biofouling of the cooling water piping or other surfaces. Chlorination of once-through cooling water is the accepted practice at most power stations, either as periodic slugs or as continuous, low-level additions. The discharge of chemical-laden "blow-down" water from cooling tower systems, used to avoid excessive concentrations of dissolved solids within the cooling system, is an essential part of plant operation. The use of oxidizing biocides such as chlorine is also periodically required in a recirculating cooling system to minimize the growth of algae.

Chlorination has recently been questioned because recent studies show formation of chlorinated organics in both polluted and natural waters (Jolley, 1975). These chlorinated organic materials can remain toxic for aquatic life for long periods, as well as being of concern to municipal water users (Gehrs et al., 1974). Chlorinated organic compounds have recently been identified from several municipal water supplies in the USA (Morris and McKay, 1975). These are believed to derived from chlorinated waste effluents.

II.2.3 Atmospheric effects

Large cooling towers, either mechanical or natural draft, and large arrays of cooling ponds represent much more rapid means of releasing heat and moisture to the atmosphere than the discharge through large natural water bodies. Accordingly, cooling towers and cooling ponds, hold the greatest potential atmospheric effects. These include:

(a) ground level fog and icing;
(b) clouds and precipitation;
(c) severe weather effects;
(d) plume length and shadowing; (and)
(e) drift.

(Peterson, 1973; IAEA, 1974; Hanna and Pell, 1975).

More attention has been given to fog and ice associated with plumes from evaporative cooling towers than to any other effects. Many cooling tower reports from the United States contain statements that mechanical and natural draft cooling towers have the 'potential' to cause or increase the frequency of ground level fog or icing. Theoretical analyses (e.g. McVehil, 1970; EG & G; Inc., 1971) all predict tower-induced ground level fog for various periods of time with a greater fog persistence existing in cold weather. In these theoretical studies maximum fog frequencies would result from mechanical draft cooling towers. Available physical observations near towers and extensive European observations indicate that the plumes usually do not cause surface fog (e.g. Decker, 1969; Aynsley, 1970; IAEA, 1974; Hanna and Pell, 1975). In these field observations the warm, moist plume enters the atmosphere at heights of 100 metres or more and evaporates before it reaches ground level. These observations indicate that theoretical models may be too pessimistic in their assumptions.

In the operation of cooling lakes or ponds local climatological changes are to be expected, such as changes in the intensity, frequency, and inland penetration of induced fog, including the creation of freezing fog near the water's edge. Observations at cooling ponds indicate that the fog over the pond is usually thin, wispy, and does not penetrate inland more than 100 to 300 metres. Because the water vapour is released slowly over large areas, ponds are not a major source of fog despite the release at ground level (IAEA, 1974). However, in weather situations producing natural fog over large areas, ponds would act to intensify and prolong fog conditions. Cooling pond site selection is important in order to assure that induced fogs (and freezing fogs) do not affect roads and bridges. Spray units, with which the effective evaporation area is greatly increased by spraying the heated water over the pond or through a canal, will also increase the frequency and intensity of dew, fog, frost, and icing conditions along the banks or downwind of the pond or canal.

Quantitative data on the effects of moist plumes from cooling towers on clouds and precipitation are very limited. Occasional observations of light drizzle or snow have been reported in the vicinity of towers (e.g. Culkowski, 1962; Federal Water Pollution Control Administration, 1968). Additional heat and/or moisture fed into a developing storm cloud might conceivably produce an imbalance that would result in intensification into a severe weather state. In view of the paucity of data available in this area, any effects are only conjecture at this time.

The psychological aspects of the shadowing effect of atmospheric plumes from cooling towers have been considered in nuclear power plant studies in Western Europe. Calculations performed in Switzerland for two natural draft cooling towers (Broehl, 1868) indicates that even if visible cooling tower plumes are assumed to be fully opaque, the reduction of sunlight in nerby areas would be insignificant (the average reduction was one minute per day corresponding to 0.35% of sunshine.) The shadowing effect of mechanical draft towers is smaller than from natural draft towers because the vapour plume, through the several ejection points, obtains a more rapid atmospheric dilution.

A problem in the operation of wet cooling towers involves a small portion of the total water circulated in the tower which enters the atmosphere without being evaporated. This physical water loss is due to droplets entrained in the air leaving the tower and is often referred to as 'drift'. Drift fall-out which occurs near the tower may cause problems such as highway icing in the winter and transmission line flash-over. Drift contains all the salts and impurities in the intake cooling water. When deposited in the area surrounding the plant site, the drift droplets evaporate and leave a solid or salt residue behind. This residue can cause vegetation to accumulate chemicals present in drift (Taylor et al., 1975; Hanna and Pell, 1975). Of particular concern is long-term build-up of potential toxicants such as chromium.

Published test data indicate drift loss rates of 0.005 to 0.0076% for mechanical draft towers and 0.0012 to 0.0025% for natural draft towers (Böhm et al., 1971). Additional operating test data are needed to validate the drift loss rates in this study. Tower equipment companies now guarantee drift rates to be limited between 0.002 and 0.005% of the circulating water flow. For a 100 MW(e) power plant, water loss due to drift can be less than one litre per second.

Up to the present time, wet cooling towers at power stations have been limited to non-saline make-up water. Evaporative towers are, however, now being installed in estuarine or coastal locations of power plants. The first hyperbolic natural draft cooling tower in the United States using brackish water is installed in conjunction with a 630 MW(e) oil-fired power plant at Chalk Point, Maryland (Pell, 1975). A comprehensive soil and vegetation research programme is planned for this site, in order to determine the potential effects that brackish water cooling towers may have on the surrounding vegetation. Additional field studies of this type will be required as salt or brackish water cooling towers are introduced into common power plant use in other regions.

In addition to the meteorological effects, cooling towers may impact the environment in other ways. For example, the synergistic effects of cooling tower plumes mixing with industrial stack effluents which contain oxides of sulphur and nitrogen require further study and evaluation. In some instances, acid rains may result due to the mixing of the cooling tower plumes with fossil-fueled power plant stack effluents. The environmental impact of noise and the aesthetic effects of large cooling tower arrays also deserve consideration.

II.3 Potential and Observed Biological Changes From Once-Through Cooling

It would be impossible for this section to comprehensively report the biological data and observations from laboratory and field studies that bear on cooling system damages to organisms and ecological systems. The scientific literature is simply too vast, and the particular locations and species too diverse. Emphasis has been placed on once-through cooling systems, as these biological problems are better documented. For each source of potential biological damage, knowledge of the circumstances and probability of damage, especially as indicated by operational power plants, is useful for estimating damages or their lack at a given site. This section will emphasize this type of information and indicate where additional information can be obtained. This section also provides an update of an early review on effects of thermal discharges (US Senate Committee on Public Works, 1973) plus the additional consideration of the broader, non-thermal effects of cooling water use.

II.3.1 Change due to physical alterations: Construction of intakes and discharges

Construction activities required to build intake and discharge structures may involve dredging, cutting or tunnelling. Normally, these impacts are of a temporary nature. Increased siltation, for example, may act only as a temporary stress through reducing oxygen content of a water body segment or reducing food uptake by filter feeding invertebrates. Yet heavy or continued siltation (e.g. from scouring by high velocity discharge) can lead to long term loss of benthic communities where suspension feeders dominate, such as oyster bars or coral reefs. Construction which cuts across a barrier dune system of an open beach could also result in long-term ecological alteration. If dune revegetation is not quickly accomplished, storms can breach the beach at this point, resulting in a new ocean inlet which may persist for some time. In inter-tidal areas, slumping of dredged canal banks will result if side slope design exceeds one on fifteen. In cut and fill operations across marsh lands, top soil will be required to achieve revegetation, as the pH of marsh soil drops greatly upon exposure to the atmosphere. These effects are not unique to power plant construction.

Submerged cooling system structures will be colonized by sessile organisms, to which fish and other motile organisms will be attracted to feed. These mobile forms will be exposed to potential intake impingement or entrapment, or at the discharge, possible entrainment in the effluent. Attraction of fish to submerged intake structures has been found to be greater at some off-shore locations e.g. off the California coast) than within large estuarine channels.

New patterns of water circulation, will develop around any cooling system structure which extends into a waterway. These may also serve to attract motile organisms. When intakes are indented into the original shore-line, currents develop which retain fish within intake bays; if protruding, end bay sections may experience greater impingement due to eddy currents which develop (e.g. Lake Norman in the USA, Edwards et al., 1976). Intake structures built flush with the shore-line can minimize these problems. Circulation will also be altered upon operation of the cooling system. Benthic scouring immediate to the intake and discharge is common. This can cause additional benthic loss downstream due to increased sedimentation (Merriman, 1976).

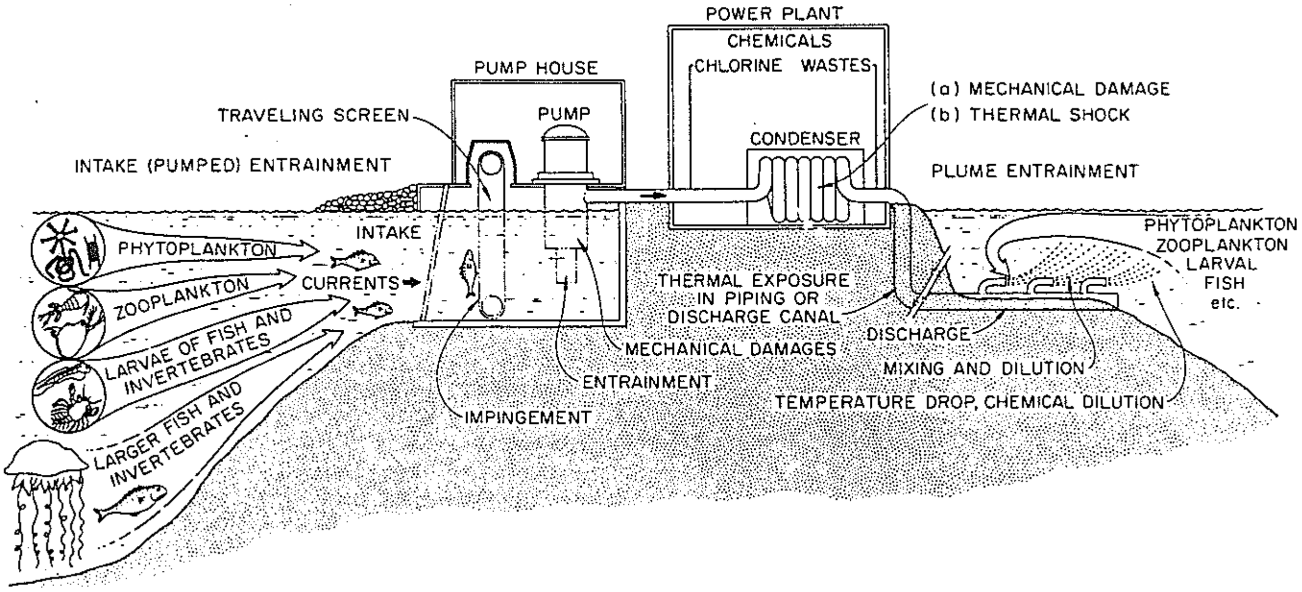

Figure II.1 Sources of potential ecological changes at a power plant cooling system

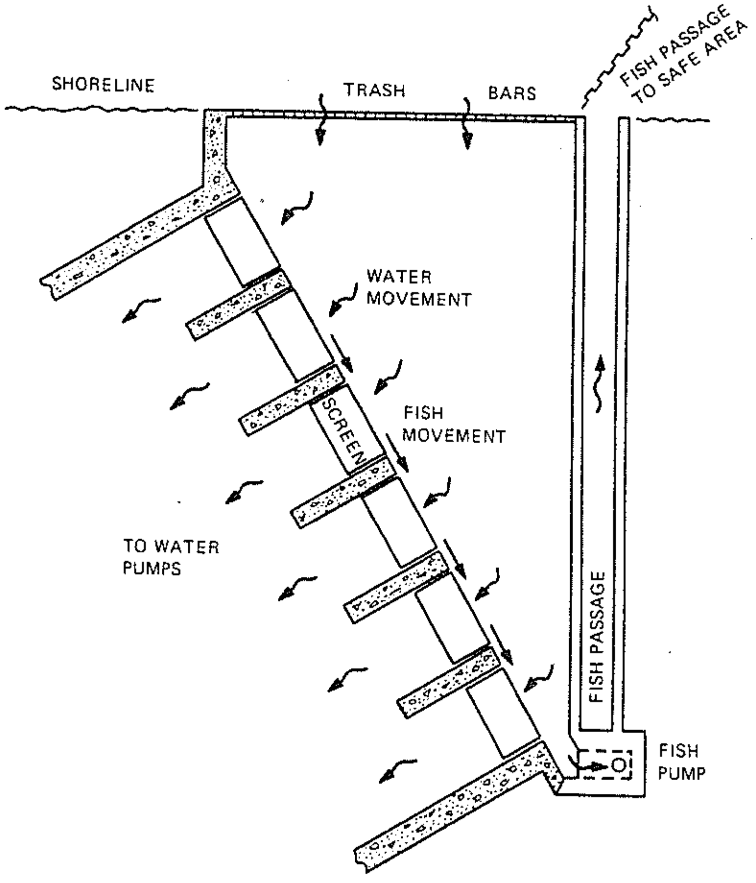

Figure II.2 Fish passage

In estuarine areas, problems can arise if cooling water use creates new circulation patterns which appreciably alter the local salinity regime. At the minimum, a shift in the indigenous community would occur, with a loss of intolerant species and the addition of others. Establishment of intake and discharge structures in small marsh creeks at the Oyster Creek (N.J.) Nuclear Plant resulted in movement of more saline bay water into the marsh system. Wood-boring marine shipworms moved into the creek, which resulted in loss of a local economic and recreational resource, as pilings, docks and boats at several marinas were destroyed (Turner, 1973). Knowledge of the salinity tolerance of potential deleterious species should make predictable.

Concern has been expressed whether jet discharges might affect free movement of animals past a power plant. While the thermal component of discharges may cause problems for some species, discharge currents per se have not been found to block movement of animals (Leggett, 1970; Maryland Power Plant Siting Programme, 1977).

II.3.2 Changes from entrapment and impingement at intakes

Field experience at operating power station intakes has shown clearly that large numbers of fish and invertebrates can be killed on the screens (USEPA, 1976a). The underlying causes of entrapment and impingement remain little understood and predictability is therefore poor. Reasonably quantitative laboratory data on fish swim speeds have been obtained (e.g. Blaxter, 1969) with the initial assumption that physical stamina could be compared directly with water flow rates through screens to determine likelihood of impingement. This has not proved reliable. Current thinking relates susceptibility to impingement to:

(1) behaviour patterns of fish, many completely unknown, in the vicinity of velocity accelerations and intake structures;
(2) physical characteristics of the intake area which many or may not allow routes for easy return to open water following withdrawal to an intake channel;
(3) environmental factors such as low water temperature, Griffith and Tomljanovich, (1975); Griffith, (1978) and high turbidity which influence normal behaviour such that an organism cannot (or does not) escape;
(and) (4) attractants, such as recirculating warm water (often for ice control) in winter, lights, shade, or presence of food organisms.

It seems that some species, particularly in the family Clupeidae, are especially susceptible to impingement.

Reports of chronic intake impingement problems experienced at operating plants can be useful to evaluating site-specific aspects contributing to the problems as well as to identify susceptible species and environmental correlates with impingement. In the USA, this information is available for nuclear plants in reports made to the Nuclear Regulatory Commission and Environmental Protection Agency. (see, for example, the Millstone Nuclear Power Station, 316b report, (Northeast Utilities Service Co., 1976); and reviews by Loar et al., 1977 and Uziel, 1978). At some power plants a direct correlation is apparent between species impinged and the abundance and seasonality of fishes present in the water body segment (Maryland Power Plant Siting Programme, 1977; Landry and Strawn, 1974; Benda and Guluas, 1976). In these cases, the power plant intake represents a non-discriminate stress on the nekton community. The same can also apply to pelagic marine macro-invertebrates, such as squid and swimming crabs. Yet scavengers, such as blue crabs, may be differentially attracted to intake bays in such abundance as to achieve nuisance proportions. Impingement susceptibility can also be species dependent and not representative of the local icthyio-fauna at large, as seen by Edwards et al., 1976, at four freshwater sites in North Carolina, USA. Thread-fin shad was the major species impacted, especially during the late fall and winter, when this species is typically highly stressed or killed by low temperatures. With this species, there was no relationship between intake velocity and impingement, nor was a skimmer wall effective in reducing impingement. A survey of freshwater power plants in the southeast of the USA (Loar et al., 1977), pinpointed the cold sensitivity of threadfin shad as the major cause of impingement in this large region. Additional correlates between impingement and season, river flow, and water level are cited by Grimes (1975 and Mathur et al. (1977).

The numbers of animals lost due to impingement is difficult to assess. (At US nuclear plants, counts of dead animals are summarized and reported to the US Nuclear Regulatory Commission). Some impinged fish are returned alive to the waterway although the chances of their ultimate survival is questionable. For certain fragile fishes, such as the family Clupeidae, there is little probability of survival once the screens are impacted. For other species, survival estimates should be made conservatively. The ability of impinged fish to withstand disease following loss of surface mucus or scales, or to withstand normal predator pressure following impingement stress is doubtless markedly reduced following impingement. Benda and Gulvas (1976) report signs of physical damage in over 50% of the impinged fish at a Lake Michigan power plant. They rate the chances of survival of these fish as minimal. Field or laboratory assessment of injury rate and survival suggest post-impingement survival to be better during cold winter months, with loss increasing proportionally with seasonal temperature increase (Maryland Power Plant Siting Programme, 1977; Landry and Strawn, 1974; Northeast Utilities Service Co., 1976). Relative sensitivity of eight estuarine fish species to impingement stress is reported by Burton Maryland Power Plant Siting Programme, 1977).

The significance of fish mortalities at intake structures can be placed in some perspective by distinguishing between repeated chronic kill and the very conspicuous, but infrequent major kills. Heavy fish kills due to entrapment or impingement are dramatic in nature and are always a serious problem for the power plant, possibly resulting in plant shutdown as the screens become clogged. Yet if major kills occur only infrequently, their ecological significance may be minimal. For example, loss of 50 million juvenile menhaden and blueback herring at Millstone, Conn., (Northeast Utilities Service Co., 1976) occured as atmospheric cooling began in late summer, 1971, and the fish were leaving the estuarine nursery grounds to move south in the fall. Such an event has not reoccurred. In contrast, impingement of winter flounder occurs chronically at the same plant, with this species comprising 28% of all fish impinged over a four year period. The number of flounder impinged represented 0.3, 0.7 and 0.9% of the estimated local population during three of these years. If none of the impinged flounder survived, which is probably unlikely for this species, it is projected that the local population would be reduced by 12% after 35 years operation of three generating units, assuming present impingement experience continued. Fish troughs will be added to the screens at this plant in an attempt to minimize impingement mortalities in this species.

State-of-the-art intake technology to minimize impingement is summarized by Ray et al., 1976, and USEPA, 1976a. These reports address intake orientation (off-shore conduit, shore-line or bankside, and intake approach channel), behavioral and physical intake barriers and fish removal systems. Also discussed are potential approaches to minimize entrainment of larval fish. An extensive bibliography on fish protection at intake structures, with abstracts, has been compiled by Huber (1974). Impingement problems and intake design have also been the topics of several workshops (Jensen, 1974, 1976, 1978). At present, it is difficult to generalize on best technologies for intake structures which minimize environmental impact due to the site specific nature of the problem. For example, the louver bypass system described by Schuler and Larson (1975), was a site specific design and may not be the best technology elsewhere (Schuler, personal communication). At present, there is considerable research underway to develop better phsyical barriers and to perfect fish removal systems. Until new designs are proven in field tests, modifications of the standard rotating vertical screen barrier cited by Ray et al., 1976, should be explored to mitigate impingement loss. These modifications include provision for escape routes inside the intake suction pit or screen well, equipping the screens with fish troughs and continuously rotating the screens to minimize impingement time for a fish. It is usually recommended that intake velocities not exceed 0.15 m sec^{-1} at the trash rack to permit fish to escape the screen wall (Boreman, 1977).

II.3.3 Change due to entrainment of small organisms with pumped cooling water

All small aquatic organisms capable of passing through the intake screens (usually 1 cm^2 openings) are entrainable and potentially subject to passage through power plant cooling systems. This would include planktonic and weakly swimming pelagic organisms ranging from microalgae to copepods and eggs and larvae of fish and invertebrates. Organisms entrained through the cooling system experience a combination of thermal and mechanical stress, plus exposure to a chemical biocide during periods of application. Survival of entrained organisms following plant passage is problematic and will depend on cooling system design, plant operating characteristics as well as overall tolerance of the species and life stages entrained. The larger copepods and fish eggs and larvae tend to be the more sensitive, with entrainment losses reported as ranging from 70 to 100% at several plants. More typical losses average about 30% over an annual cycle (Lawler et al., in press). Of primary environmental concern is entrainment loss of meroplankton (i.e. the eggs or larvae of fish and macroinvertebrates) since these species can have generation times on the order of one to several years. In contrast, phytoplankton or copepods can produce replacement generations in a matter of hours to days, respectively, during favourable seasons.

The significance of meroplankton entrainment loss for the adult populations can be evaluated by biological models, such as developed for the winter flounder by Hess et al. (1975). (For additional details, see also Northeast Utilities Services Co., 1976; Van Winkle, 1977.) Similar modelling efforts are currently being conducted in the USA for the striped bass in the Hudson (discussed in chapter III), and Potomac Rivers. Enright (1977) has proposed a simple first order approach to assess maximum probable impact of larval entrainment mortalities upon the adult populations.

Our present understanding of the pumped entrainment problem is summarized in a volume edited by Schubel and Marcy (1978). Some early papers which document aspects of pumped entrainment stress include Coutant (1970, 1971) and Hoss et al. (1974) (thermal shock); Marcy, (1973) and Carpenter et al., (1974b) (mechanical); and Hamilton et al. (1970) (chemical). Marcy, (1975) has written a succinct overview of the interaction of these three entrainment stresses, particularly as they pertain to fishes. Additional articles can be found in a review by Carrier, (1978) and in the proceedings of several entrainment workshops (Jensen, 1974, 1976, 1978).

The magnitude and the nature of pumped entrainment damage is plant specific, depending on the entrainable biota and hydrodynamic characteristics of the site, cooling system design and operating conditions. Beck et al. (1978) have tabulated, by species, reports of entrainment damage at 14 power plants with once-through cooling systems in the USA. Some of these studies also evaluated the relative influence of single stressors. They suggest that physical stress can be a major contributor of larval and juvenile fish mortality, accounting for 80-100% of observed losses in almost every study where cause of mortality has been partitioned. Thermal stress often contributes to mortalities during the summer, when water temperature is naturally high. Chemical biocides, most frequently used during spring and summer, can dominate as a stress during any period of application, and more so during periods of maximum thermal elevation (Hoss et al., 1977). Zooplankton appear to be more affected by chlorine than are fish larvae.

Yet it must be emphasized that the above conclusions are tenuous at best, as the data from operating power plants are sparse, and often of limited use due to sampling problems, variation in study techniques between workers, and potentially unique differences in power plant cooling systems. Illogical results such as greater numbers of live organisms in the discharge than in the intake or opposite results from day to day or year to year at a single plant emphasize the inadequacy of many of the study methods used.

Problems of sampling entrained plankton have been discussed by Heinle (1976a, 1976b) and Copeland et al. (1976) (estuarine), Jude (1976) (large lake, Kind and Mancini (1976) (river) and Bowles et al. (1978) (ichthyoplankton). Numbers of larvae or zooplankton captured tend to be highly variable due to their patchy distribution (both horizontally and vertically) in the water column, diel variation in activity, possible day-time avoidance of nets, watermass changes at the intake due to tidal (in estuaries) and/or weather changes (especially in large lakes), plus unknown changes in distribution of plankton in the forebay of the intake. At the discharge sampling is usually complicated by high water velocities and changes in organism buoyancy and swimming behaviour due to the heat, which affect vertical distribution. Another problem centres around ability to detect entrainment damage. Problems of net collection, including the potential for net damage, have been addressed by some workers by use of a pump for sampling. Vital stains have been used to distinguish living from dead animals when it is not practical to make counts immediately following collection (Heinle, 1976a). The potential for latent effects, such as reduced ecological fitness and subsequent loss from the system due to differential predation, for example, has been little considered to date. Studies which simply tally mortalities immediately following discharge from the power plant are probably describing only a portion of the ultimate loss.

It is also difficult to directly document the consequences of through-plant entrainment damages on populations on the whole ecosystem in water bodies used for cooling. Many examples exist of field sampling programmes at power stations where no decreases of plankton have been seen that can be attributed to the inplant losses. (e.g. Carpenter et al., 1974 a). "Natural" variability of plankton is typically so great that only very large impacts would be directly discernable if they did exist.

<u>Components of Plant Entrainment Stress : Physical</u>. Physical damage to entrained biota can

result from four stresses operating in power plant cooling systems: pressure; acceleration forces, shear and abrasion or collision (Ulanowitz, 1975; Schubel and Marcy, 1978). At the pumps, organisms are exposed to sudden fluctuations in pressure and velocity shear forces, physical buffeting and abrasion. Once in the pump, there is rapid positive and negative charges in hydrostatic pressure, ranging from 0.29 to 1.6 atm. There is potential for contact with impeller blades (2-5%) or pump walls and pump shear stresses can be up to 10 times that existing near the walls of the condenser tubes. Accordingly, the pump is considered the most likely site of physical damage within an open cooling system. In the condenser water box, physical stress takes the form of negative pressures and high flow rates, which are maximum at this point. Negative pressure is considered particularly damaging to entrained fish. In the condenser tubes, shear and pressure changes also occur, but may pose a minimal physical stress at this point (Marcy et al., 1978).

The consequences for entrained biota of the combined physical forces of a power plant cooling system can be most realistically studied at an operating power plant. Plants pumping water through a cooling system without any thermal load have provided this opportunity at a few sites (e.g. Marcy, 1973). Pressure is one force that can be studied separately (Beck et al., 1975). However it is difficult at present to relate the findings of such studies to an operating power plant, where the magnitude of any one stress is hard to define and its consequences problematical due to interaction with multiple additional stresses. A scaled total cooling system simulator constructed at Oak Ridge National Laboratory (US) may permit detailed cause and effects experimental studies of the whole compliment of cooling system physical stresses which to date have not been possible at operating power plants (Coutant, personal communication).

Studies to date suggest great differences exist between power plants with regards to physical entrainment damage. The primary variables regarding physical damage to entrained biota fall into two categories. Those which are a function of cooling system design and operation, and those which are dependent on the specific biota entrained. Location and design of the intake has the potential to enhance or minimize entrainment of planktonic organisms, as will volume of water pumped. Pump design and the efficiency of their operation (inefficient operation results in excessive cavitation and high biological damage) is another variable. The practice of augmentation pumping to reduce discharge water temperature, for example, is now recognized as clearly counterproductive and not a wise approach to mitigate discharge temperatures, as it increases the number of planktonic organisms exposed to damage. Indeed, the design option of a higher operating Δt to reduce the volume of cooling water required might be considered, should plant entrainment of meroplankton pose a potentially serious environmental problem.

The biological variables influencing the probability of physical entrainment damage focus on the relative fragility of the entrained species, which is often a function of size and lifestage. Fish eggs and larvae appear to be more sensitive, among which the most fragile are larvae of the clupeids, menhaden, Atlantic silverside, sea robin, tautog, cunner and anchovy, (Marcy et al., 1978). Species with the larger respiratory apparatus have also been reported as highly susceptible to physical entrainment stress, possibly since the head area in fish is especially vulnerable to damage (Nawrocki, 1977). For other species, the yolk and post-yolk sac embryonic stages have been seen to be highly sensitive. There has also been a good correlation, for both ichythoplankton (Marcy, 1973) and invertebrates, of increasing physical damage with organism size. Marcy et al. (1978) have proposed a generalized model which relates per cent mortality to size of the entrained biota. For some fishes however there is an upper size limit for this generalization, with the largest entrainable individuals of some species showing increased tolerance to physical stress (Teleki, 1976; Nawrocki, 1977).

Thermal. The contribution of high temperature per se as a dominant entrainment stress has been best illustrated at operating plants by increased loss during the summer, the period that nature water temperature is at its maximum. If the discharge canal is long, considerable kill due to temperature will occur. (For further discussion of this topic, see Sec. 3.4). But otherwise, good estimates of the specific contribution of temperature alone to observed entrainment damage are difficult to make. This would require a good assessment be first made of the extent of damage from mechanical forces alone (i.e. damage without any heat load not chlore added). Only a few workers, such as Marcy (1973), Carpenter et al (1974b), Alden et al. (1976) and Lauer et al. (1974) have had such an opportunity. Reviewing this literature, Schubel et al. (1978), conclude that high temperature can be the dominant entrainment stress at plants where mechanical stress is minimal and biocides are used only infrequently or not at all.

Laboratory studies can be useful in identifying entrainable organisms which clearly cannot tolerate the thermal regime of a given power plant. Yet organisms surviving a laboratory simulation of the entrainment thermal experience will not necessarily survive cooling system passage where mechanical and possibly chemical stresses are also present and probably will act in a synergistic fashion. A laboratory thermal stress simulation typically uses entrainable organisms common to the site in question and acclimation temperatures typical of the site during the seasons of occurrence of the test organism. The thermal dose (i.e. duration and magnitude of heating) should be comparable to that expected in the power plant, including the manner the elevation is experienced, i.e. as an initial instantaneous heat shock. Cooling may occur as a gradual decay if there is a surface discharge (Schubel, 1975), or a sudden drop in the case of a jet discharge (Hoss et al., 1974).

Schubel et al. (1978) suggests that fish eggs and larvae are usually significantly more sensitive to simulated thermal plant entrainment stress than are zooplankton or macroinvertebrates. Early fish embryos (i.e. early clearage to blastopore closure) are more sensitive than later embryonic stages. Mortality is usually complete with a $20^{\circ}\Delta t$, while hatching success may be reduced at a $15^{\circ}\Delta t$, depending on species. Larval deformities can also result when the eggs are thermally stressed ($\Delta t = 10^{\circ}$and 15°), markedly affecting their ability to swim normally and doubtless reducing their ability to avoid predation or feed (Koo and Johnson, 1978).

Working with larval fish, Hoss et al. (1974) found entrainment simulated temperature shock to be potentially very damaging in itself, especially the second (cooling) shock. The larvae exhibit marked initial deviations in behaviour, including complete immobilization. Those which might survive the direct effects of heat are clearly rendered more susceptible to predation at the time of discharge. Species differences in larval thermal resistance times and thermal shock effects should be considered when evaluating the potential for thermal entrainment damage at a given site. Hoss et al. (1974) observed a range in thermal tolerance among the six larval species he tested, with the flounders the more tolerant and menhaden the least. Striped bass larvae appear to have a relatively high thermal tolerance (Laurer et al., 1974).

Juvenile fish may have greater thermal tolerances than either the larval or adult stages (Otto, 1976; Brett, 1956). (The same phenomenon is frequently seen among invertebrates as well). Accordingly, Schuble et al. (1978) suggest that any damage experienced by entrained juvenile fish is probably more due to physical forces than from temperature alone.

Many of the generalizations of thermal entrainment stress made above for larval fish have also been observed with zooplankton and macroinvertebrates at operating power plants or in laboratory simulation studies. Schubel et al. (1978) cite several papers showing temperature can be the dominant entrainment stress in the summer at plants with a large Δt (e.g. in excess of 13^{o}-15^{o}), or when the excess temperature exposure is prolonged by discharge into a long canal or if plume dispersion is slow. (Fig. II.3)

Chemical Biocide Stress. Biocides, such as chlorine, are employed for the purpose of killing bacteria and algae which can build up on condenser walls, and to prevent settlement of growth of fouling invertebrates, such as mussels or barnacles. Accordingly, it is to be expected that many other entrained organisms will be similarly killed when biocides are used. Application practices vary. In the United States, application is usually intermittent. In contrast, continuous low-level chlorination is employed at estuarine and coastal plants in England between April and November in order to prevent fouling by *Mytilus edulis* (Coughlan and Whitehouse, 1977). Considerable research on power plant biocide use and its effects on aquatic orgnisms has been conducted at the Central Electricity Research Laboratories in Great Britain. (See Couglan and Whitehouse (1977) and the review of Whitehouse (1975) for citations). Residual chlorine may persist in the cooling water following discharge from the cooling system, depending on dose level and application techniques.

Morgan and Carpenter (1978) have summarized some observed effects of chlorination at operating power plants in the USA. It appears that most entrained organisms are adversely affected by concentrations in excess of 0.5 ppm residual chlorine. Adverse effects have been most apparent with phytoplankton, perhaps due to the ease of measuring productivity. Yet Coughlan and Whitehouse (1977) also report increased sensitivity to chlorine as orgnism size decreases. Morgan and Carpenter cite numerous reports of 50 to 90% reduction in productivity (probably irreversible) following injection dose levels ranging from 0.1 to 2.7 ppm chlorine. Reports of zooplankton biocide entrainment damage include 50% kill in the presence of 0.25-0.75ppm chlorine residue (Davis and Jensen, 1975), 40-80% copepod kill at two Chesapeake Bay (USA) plants in August, but negligible chlorine kill in May (Heinle, 1976b); 90% mortality of

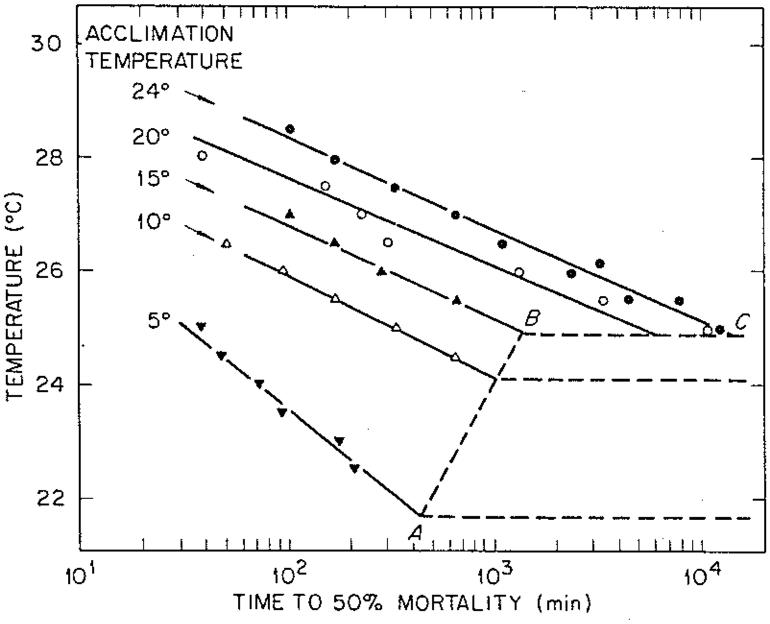

Figure II.3 Mortality rate

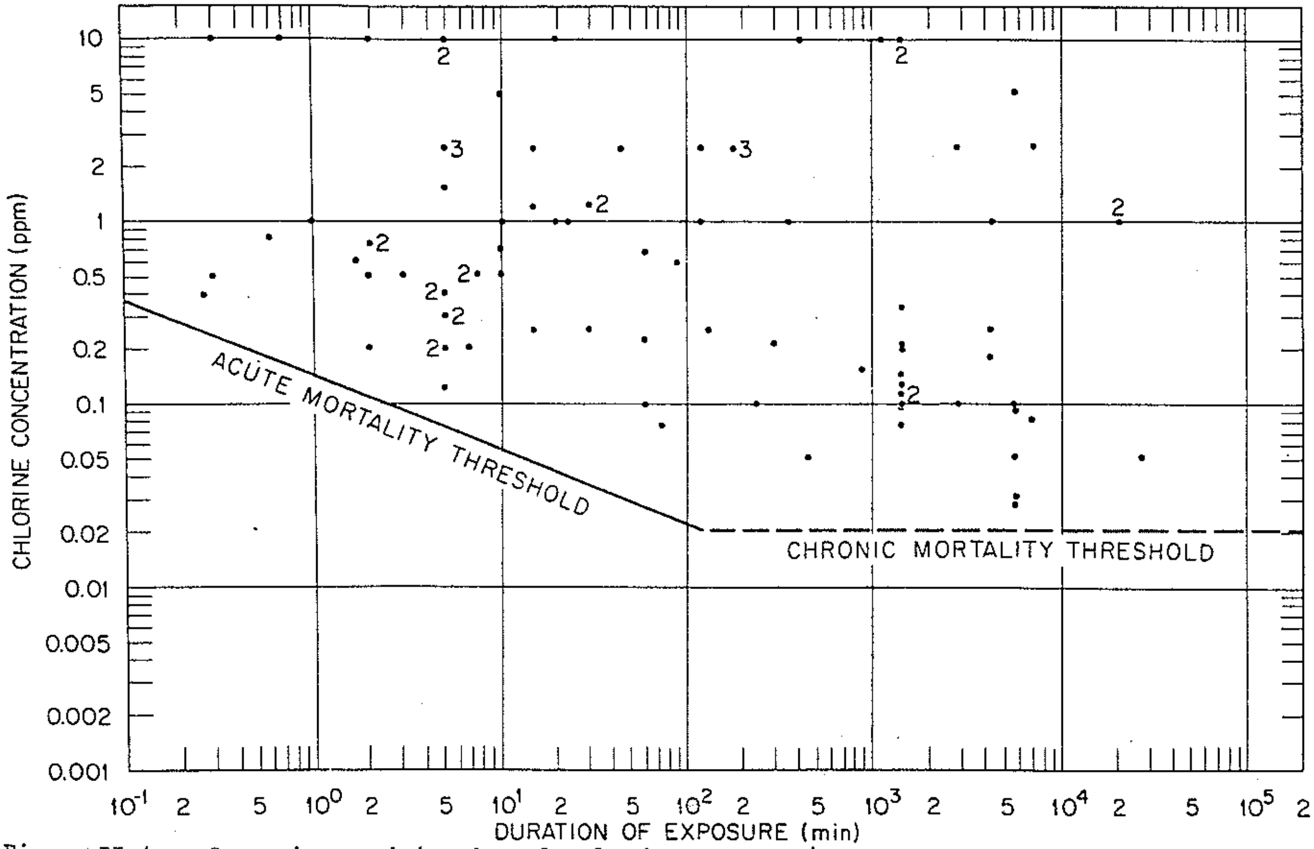

Figure II.4 Composite toxicity data for freshwater organisms

Acartia tonsa at another Chesapeake Bay plant (McLean, 1973); and 50% direct or delayed kill of Gammarus sp. at the Indian Point (N.Y.) plant (Ginn et al., 1974). Loss of approximately 25% of entrained larval Morone spp. was reported by Laurer et al. (1974) during chlorination at the same plant. In contrast, Carpenter et al. (1974b) was unable to see effects on copepod survival, with or without chlorine, above that caused by mechanical stress at the Millstone (CT.) plant; likewise for larval fishes at another plant (Marcy, 1973), although there only one-fourth of the cooling system was chlorinated at one time. Potential and observed effects of chlorine in power plant discharges are discussed further in section 3.6.

Several approaches to minimize chlorination stress are suggested by Morgan and Carpenter (1978). These include intermittent low-level chlorination in preference to continuous dosage (entrained plankton are damaged even at very low-levels); investigation of site specific antifouling needs in order to minimize duration and frequency of chlorination (e.g. during some seasons, little or none may be required); use split condenser chlorination; use a higher Δt, coupled with reduced cooling water volume; and finally, consider alternative biofouling control techniques, such as outlined by Yu, H.H.S. (1977).

II.3.4 Changes from exposure to the cooling system discharge: Near field thermal and physical effects

Cooling system effluent includes primarily heated water, biocides used to prevent slime build-up and fouling in the condensers, and metals leached from the condensers. The potential and observed consequences of exposure to these stresses will be treated and in sections 3.5 and 3.6, primarily as single entitles. It should be recognized that effects observed can also result from synergistic interaction of these stress factors, The thermal component is particularly dominant due to the large colume of heated water continually discharged during plant operation. Biocides are usually used only on an intermittent basis; at some plants mechanical cleaning methods make them unnecessary. Leaching of condenser metals usually do not achieve levels of biological concern.

Temperature Related Problems of Cooling System Discharges

1. Discharge Canal and Near-field Thermal Plume.

Organisms pumped through the condensers experience continued exposure to thermal stress in the discharge canal and discharge plume as the effluent enters the receiving water body. A plume which contacts the bottom will also adversely impact benthic life. Planktonic and weakly swimming pelagic organisms entrained into the discharge plume will experience thermal shock. During non-summer months, fish will move in and out of the plume and canal, attracted by the warmer water. Fish remaining in these waters for some period risk exposure to discharged biocides, and during the winter, could experience gas bubble disease, loss of physiological condition, or in the event of plant shutdown, cold kill.

A long discharge canal may be an attractive engineering design option to limit recirulation of cooling waters to the intake or reduce discharge temperatures a few degrees. Yet these long canals serve to prolong exposure of plant entrained plankton to elevated temperatures and potentially toxic biocides. Thermal death is a function of both the temperature and duration of exposure (Coutant, 1970). Ginn et al. (1974) have documented this fact in the context of the discharge canal for two species of entrainable Gammarid amphipods. Discharge canal stress on both entrained biota nad organisms entering from the mixing zone can be reduced to the extent that the discharge canal is shortened or eliminated altogether. Thermal plume problems can be minimized by enhancing dilution of the discharge, potentially achieved by siting the discharge in an area with strong currents (e.g. Millstone, CT.) or by employing a jet discharge (e.g. Calvert Cliffs, MD., San Onofre, CA.)

Markedly elevated effluent temperatures close to the operating Δt of the power plant usually occur only in discharge canals and in the immediate discharge area. It is here that the most striking biological changes are seen. The consequences and underlying causes are highly site-specific. Important early reviews of power plant thermal effects include Alabaster (1963); Parker and Krenkel (1969); Krenkel and Parker (1969); Coutant (1970), Clark and Brownell (1973). Published case studies on thermal discharge effects include Hedgpath (1973) for two California (estuarine and coastal) power plants and Merriman and Thorp (1976) for a temperate zone riverine site. Coles (1977) has summarized some environmental effects of power plants in the tropics. Recent symposia include those sponsored by IAEA (1975a, 1975b), and the Savannah River Ecology Laboratory (Gibbons and Sharitz, 1974; Esch and McFarlane, 1976; Thorp and Gibbons, in press). Bibliographic information is catalogued and indexed in Raney, Menzel and Weller (1974) (for literature through 1971) and in annual indexed bibliographies by Coutant and his associates (1972-1977).

Early reports of biological change within discharge canals *per se* were reviewed by Coutant (1970). At that date, topics being addressed were canal water column productivity and the potential for "self pollution", benthic productivity, succession to thermally tolerant algae, such as the blue greens, and fish kills. Reports describing the annual dynamics of the biota in discharge canals (Merriman and Thorp, 1976; Miller et al., 1976) show the discharge canal and immediate mixing zone to be areas of extreme ecological instability at plants experiencing occasional shutdown or which operate in a peaking mode. Also, during the summer, thermal conditions in the canals typically reach stress or lethal levels for the indigenous biota. Depending on the absolute temperatures which prevail, productivity is reduced or inhibited, benthic diversity drops and the fishes move out of canal. During non-summer months, productivity of each compartment usually exceeds that of the open water body, although the persistence of this community is limited due to the instability of the canal environment.

Today we recognize two general problem areas regarding discharge canals: continued thermal and chemical stress on plant-entrained plankton (an extension of the entainment problem) and those resulting from the attraction of fishes and other motile organisms during non-summer months. Coutant (1974) suggested that fish will be attracted to heated effluent until discharge temperatures exceed the final preferendum by 2°C (± 1). Then fish tend to avoid the discharge. Field and laboratory studies cited by Coutant, plus major studies by Spigarelli et al. (1974) and Neil and Magnuson (1974) show that many fish do behaviorally thermoregulate, swimming in and out of the plume to maintain body temperature close to the known preferendum. These findings correspond with the frequent literature reports of fish attraction to power plant discharge plumes during non-summer months, followed by avoidance in the summer. The behavioral and physiological aspects of temperature preference in fishes was the subject of a recent symposium (Richards et al., 1977). Included is an update of Coutant's temperature preference data summary. It is generally agreed that the final preferenda in fish closely approximates the optimum temperature for many physiological processes.

Problems associated with fish aggregations in thermal discharges are most conspicuous during the winter. Major problems include loss of condition of fish due to malnutrition (Merriman and Thorp, 1976); occurrence of gas bubble disease as cold water is heated and dissolved gases exceed 115% of saturation (Miller, 1974); and cold kill of fishes aggregated in the thermal plume when there is abrupt winter plant shutdown (Coutant, 1977). Marked loss of weight and condition of brown bullheads (*Ictalurus nebulosus*) overwintering in a discharge canal was reported by Massengill (1973). Three possible reasons are suggested: higher metabolic rate for fish in the warmer discharge canal water; higher metabolic requirements due to the greater swimming activity required to maintain fish in canal flows of 0.3-0.9m s^{-1}; high population densities in the canal which result in overcrowding and increased competition for food. In white crappie, thermal stress during the winter was evidenced by reduced tissue lipid levels and gonosmatic indices in the hot water arm of a reservoir (Knox, 1973). Yet there are exceptions, suggesting that the effects of near-field thermal discharge on fish condition must be considered on a species specific or site-specific basis. Bennett (1972) found no effect on condition of bluefill fingerlings, while condition was enhanced for adult bluegills and black crappies in a heated discharge pond in St. Caroline. (This might be expected if food in not limiting

Extreme temperature stress near the discharge ($\Delta t = 12^{\circ}-15^{\circ}C$) can result in increased incidence of vertebral abnormalities in fish (Mitton and Koehn, 1976). Spring and fall collections of a marine fish (*Fundulus heteroclitus*) at a power plant on Long Island Sound, N.Y., had a 11.5 to 20% increase in vertebral abnormalities relative to two adjacent control populations. Such abnormalities would reduce fitness of the power plant population.

Animals attracted to a thermal discharge may experience increased incidence of parasitism and disease. General loss of physiological condition and crowding would contribute to this. A direct relationship between incidence of parasitism and disease and temperature *per se* is well documented for aquatic animals (Sinderman, 1966). DeSylva (1969) observed high incidence of fungus disease in a marine fish species at a Florida power plant, although the actual relationship between this disease and the power plant discharge was not determined. Sankurathri and Holmes (1976) found the prevalence of certain snail parasites, especially the metacercaria, to be enhanced as thermal effluent eliminated a commensal oligochaet, which normally feeds on the digenia larvae. Additional reports of parasitism and disease incidence are cited in the annual thermal effects reviews by Coutant et al. (1972-1978).

Gas bubble disease (GBD) has also occurred in fish attracted to thermal effluents.

GBD, is not caused ny a pathogen, but rather is the rsult of gas emboli which develop in fish inhabiting waters that have experienced rapid heating or pressure changes and become supersaturated. The same problem occurs with cascading water from hydroelectric dams which entrap air. In supersaturated waters, gas emboli can develop within the fish vascular system and impede normal blood flow to capillary beds, or develop within body tissues, exclusive of the vascular system. Gas "blisters" or emboli may be visible externally between fin rays, along the posterior edge of the operculum, and subcutaneously in the oral cavity and gill arches. The emboli, if persistent, will lead to pathological conditions, disorientation and death. Fish which move out of supersaturated water do commonly recover.

Reports of GBD at power plants are not numerous. Where it has occurred, it appears to be limited primarily to the discharge canal and near field plume. Occurrence of GBD at a power generating station was first reported at Marshall Steam Station on Lake Norman, a North Carolina reservoir. Thirteen species inhabiting the discharge canal and an adjacent cove exhibited symptoms during the winter. Yet the maximum one-month incidence observed did not exceed 4.7% of all fishes captured (Miller, 1974). Adair and Hains (1974) suggest GBD is more prevalent in a discharge cove at the Marshall Station than at other stations in the same locality due to greater water usage and a deeper discharge, where gas solubility would be higher. At a Great Lakes power plant, Otto (1976) found supersaturated conditions (130%) in the discharge water, yet CBD only occurred in carp, which resided in the canal permanently during the winter. Otto suggests other species may not be exposed to these conditions long enough to develop GBD. A large kill of menhaden occurred at the Pilgrim Plant in Plymouth, MA, in April, 1973 (Marcello and Fairbanks, 1976). A school entered the discharge canal and plume shortly after plant start-up. Some 90% of the fish in the discharge canal had external signs of gas bubble disease. The resulting kill was estimated at 43,000 fish. Dissolved oxygen levels in the canal and plume were found to range between 120 and 140% of saturation. A net was subsequently placed across the mouth of the discharge canal which has effectively minimized subsequent exposure of fishes to these conditions.

Excellent recent papers on gas bubble disease in fish include Wolke et al. (1975, Bouck et al. (1976) and a workshop edited by Fickeisen and Schneider (1976). Wolke et al., reviews the early literature, describes the physical and environmental factors contributing to gas supersaturation and details the pathological signs and general lesions common to GBD. Field diagnosis is discussed, as are rapid methods for in situ measurement of total dissolved gas pressure. Bouck et al., summarizes findings of a laboratory study on tolerance of several life stages of two salmonids and large-mouth bass to total dissolved gas pressures ranging from 110-140% barometric pressure. Times to 20% and 50% mortality were determined, with biological variables influencing time to death examined. Few papers of the gas bubble disease workshop deal with temperature or power plants, yet this publication provides an excellent summary of our current knowledge of GBD in fish, the physiological consequences of sublethal exposures, and techniques to monitor total dissolved gases. Included is a working group recommendation of 115% as the gas supersaturation criteria to protect juvenile salmonids fishes migrating through Columbia-Snake River systems (US Pacific Northwest). A lower level (110%) is suggested if shallow water invertebrate benthic food organisms are also to be protected. It is important to note that sensitivity to supersaturation varies with species, as well as life history stages (Bouck et al., 1976). Otto (1976) documents additional species differences. In the laboratory, yellow perch were unaffected at 115% total gas saturation, and the 8-day TLM occurred with 126% saturation. Trout were more sensitive. The maximum no effect level was 110% and the 8-day TLM occurred at 119% gas saturation.

From Cold Shock. Cold kills have occurred with abrupt temperature decreases after fish were attracted to warmed discharge water in winter (usually in canals) (e.g. Ash et al., 1974; USAEC, 1972; Coutant, 1977). At least one cold kill has been reported due to abrupt wind changes causing a shift in a lake thermocline that introduced cold, hypolimnetic water to the power plant intake and thus a rapid decrease in effluent temperatures (Coutant, 1977). The biological basis for cold chock is well understood and death is a function of particular species, the recent thermal history or acclimation temperature, and the new cold temperature. Exposure duration is important, but generally termination of the power plant heat lasts for sufficient time that cold death results. Recent studies on susceptibility of young fish to predation following cold shock indicate that some warm-water species can tolerate only 5°-6°C drop in temperature before they experience increased predation by unstressed prey (Coutant et al., 1974). It had earlier been shown (Brett 1956) that loss of equilibrium occurred long before actual death in cold resistance studies of salmon.

Heat Shock. There is also the potential for loss of organisms due to heat shock following sudden exposure to thermal plumes (Hoss et al., 1974). This could occur if planktonic or weakly swimming pelagic organisms are carried by currents into a discharge plume, or if fish fail to perceive acutely lethal temperatures. The latter situation was observed by Young and Gibson (1973) while SCUBA diving. A migrating school of juvenile menhaden encountered a thermal plume (maximum $\Delta t = 15^{o}C$), experienced immediate thermal shock and sank, with most fish dying. Those which did recover swam back into the plume to subsequently sink again, die and be carried away by bottom currents. It is unknown how prevalent such fish kills may be, as they are not evident at the surface. This behaviour of menhaden is perplexing in light of many studies showing avoidance by fish of temperatures exceeding the final preferendum (Neil and Magneson, 1974); Coutant, 1974; Richards et al., 1977; Gray et al., 1977). Yet at very high temperatures, Gift and Westman (1971) have found breakdown of the avoidance response in fish. This could have occurred in the above field situation.

Organisms encountering a discharge plume can also be lost if they are thermally stunned and temporarily debilitated, becoming more susceptible to predation by fish and birds as they drift out of the plume. This is suggested by field observations and supported by laboratory studies with a number of juvenile fish species following sublethal shock (Coutant, 1973; Stober et al., 1971; Sylvester, 1972; Yocum and Edsall, 1974). Deacutis (1978) has also seen some differential predation in thermally shocked larvae of the Atlantic silversides (Menidia menidia). The potential loss of planktonic organisms, especially larval stages, which drift into thermal plumes has received little consideration to date. The potential for thermal shock can be minimized by designing and siting the discharge to assure rapid dilution in the receiving water.

Blockage of migration routes by thermal plumes has also been suggested as a potential problem at power plants. Yet extensive monitoring of salmon and trout on the Columbia River (Templeton and Coutant, 1970) and American Shad on the Connecticut River (Merriman and Thorp, 1976) have detected no modification of migratory patterns by thermal discharges. High temperatures of some natural tributaries have caused migrations into them to be delayed, however. (Major and Mighell, 1966).

Fish Harvest at Thermal Discharges. Thermal effluents, through their attraction of fish during non-summer seasons, are well known among fishermen for their ability to increase catchability. Several studies in the USA have analyzed this phenomenon in some detail. Coutant (1975) reviewed several studies on black basses. Landry and Strawn (1973) followed the annual activity of sport fishing at a thermal discharge into Galveston Bay, Texas. Marcy and Galvin (1973) characterized the intense winter fishery in the discharge canal of the Connecticut Yankee Atomic Power Plant on the Connecticut River. Moore et al. (1973) surveyed fishing near the Chalk Point plant on the Patuxent Estuary, Maryland. Allen et al. (1970) used experimental angling as a means of identifying fish movements at a marine coastal plant. The Tennessee Valley Authority publicizes the excellent fishing near its steam plants in winter (TVA, 1969).

Results of Elser (1965) have largely been upheld in these later studies. Elser identified seasonal changes in percentages of fish catch among his three test locations, two in unheated sections of the Potomac River, and one in a heated zone. The heated area had disproportionately high catches (greater than 35%) from October through mid-June. Catches declined to essentially zero in August, the warmest month of the year. Yet, the summer is the more popular fishing period, both in the Potomac and elsewhere. Catchability should be balanced with time of recreational demand in any analysis. Also, if food becomes inadequate for large aggregations of fish, a reduced quality of catch for the anglers can result.

Benthic Impact. The impact of thermal discharges on the benthos is largely dependent on the extent that the plume comes in close proximity to the bottom. If the discharge is located in shallow waters, appreciable loss of benthic life can occur, (Coutant, 1962). Limited circulation to rapidly dilute and disperse the plume can make this more acute. At semi-tropical and tropical sites, benthic kill can be particularly extensive with shallow water discharges, as summer temperatures are normally only a few degrees below the upper thermal limites for the biota. (Thorhaug et al., 1978). At all latitudes, where benthic impact has occurred, it is greatest during the summer. Some recovery of sublethally stressed populations may occur during winter and peripheral benthos can experience enhanced productivity.

The Turkey Point Power Plant in Florida provides one example of consequences of a surface discharge into shallow water ($<1m$) at a low latitude site. A total of 1.2 km^2 of benthic and epibenthic community experienced a statistically measurable decline in abundance for at least part of the year. Damage was perceptible between the $+2^{o}$ and $+3^{o}C$ above ambient isotherms during the summer months (Roessler and Tabb, 1974).

The prevalence of summer sublethal stress and thermal kill at other low latitude power plant sites has been reported by Blake et al. (1976) at Tampa Bay, Florida; Jokiel and Coles, (1974) at Hawaii and Kolehmainen et al. (1975) at Puerto Rico. These papers show the relationship which exists between extent of benthic impact and receiving water depth, circulation pattern and volume of thermal effluent.

Benthic impact at shallow water discharges has not been limited to the tropics. Warinner and Brehmer (1966) reported a general depression of benthic community diversity during the summer at a York River, Virginia, station some 200 to 300 m from the discharge. Depth was only 1 m at MLW. Damage further off-shore was minimal, as increased depth permitted the plume to rise off the bottom.

Discharges located in deeper water or where good circulation prevails, usually only impact the benthos immediate to the discharge. Water depth at the discharge at the Pilgrim Plant (Plymouth, Mass., Lat. 42°) ranges from 3 m near shore to 9.7 m within 0.8. km off-shore. While currents at this site are modest, water depth is sufficient to minimize benthic contact by the plume. Thermal discharge from the existing 655 MW(e) unit has eliminated Irish moss (Chondrus crispus) only within a 15 m radius of the canal; with an additional 1180 MW(e) unit operating, the projected impact is a 33 m radius devoid of moss, and moss reproduction thermally excluded over a total area of 0.8 hectares. Likewise, Mytilus edulis is projected to experience some summer mortality over a 0.8 hectare area during periods of natural maximum summer water temperature. The two-unit discharge plume is predicted to rarely contact the bottom below 6 m MLW (Boston Edison Co., 1975).

Epibenthic communities dominated by one or several sessile species are particularly sensitive to thermal discharges (North, 1969; Thorhaug et al., 1978). Loss of major sessile organisms (coral, sea grasses, macroalgae, or bivalves) will also result in loss of associated motile organisms (e.g. fishes and crustacea) which depend on the dominants for food or habitat. At Turkey Point, Florida, Roessler and Tabb (1974) found a direct correlation between loss of the sea grass, Thallasia, and macroalgae (Laurencia and Digenia) and reduction in both the kinds and numbers of benthic animals and fishes collected at the thermally impacted stations. The same phenomenon was observed by Blake et al. (1976) in Tampa Bay, Florida. The animals appeared dependent on this vegetation for food and shelter. The dependence of the coral community on survival of the coral is well established. The relative susceptibility of coral to thermal discharges and thermal stress has been reported by Jokiel and Coles (1974, 1977); Coles (1975) and Jokiel et al. (in press).

Effects of thermal discharges on the mussel (Mytilus edulis) has been reported by Gonzalez and Yevich (1976). A mussel bed had become established in the discharge canal of a Massachusetts power plant during the spring. In June, as discharge temperatures reached 27°C, the mussel bed was killed. Loss of mussels near the intake subsequently also occurred in August, as ambient water temperatures reached 27°. Laboratory studies demonstrated sublethal stress at 25° as feeding ceased.

Reproduction and survival in oysters (Crassostrea virginica) was examined at the mouth of a long discharge canal at a Delaware power plant (Tinsman et al., 1976). Average Δt was 5°C during the seasons that an appreciable temperature differential did occur. During the first year of plant operation the primary effect was precocious gonad development in the spring, but in the second year there was high mortality and reduced gonadel development in survivng oysters, suggesting an overall loss in condition. This also occurred with the scallop, Argopectin irradians concentricus at two southern Florida power plants (Studt and Blake, 1976). One month exposure to an effluent station with 4.5°C Δt resulted in mortality and resorption of oocytes; the same was seen after five months at a 1°-2°C Δt station. These effects were not considered exclusively due to the thermal component of the discharge, but also from an increase in total suspended solids at these stations. It should be noted that this scallop is at the southern extreme of its geographic range at these two study sites, so thermal sensitivity is to be expected.

Thermal Loading of Embayments and Wetlands. At lake, estuarine and coastal sites there is the potential of wind or tidal transport of heated effluent into small bays or wetlands. Two papers document the consequences of thermal loading on the fish community in marsh creeks in Florida. Carr and Giesel (1975) found a 3-10 fold drop in summer abundance and biomass of juveniles of economic species. Thermal effects appeared as pronounced in a creek which received heated discharge for only 1 to 2 hours during flood tide periods as an adjacent creek receiving effluent directly from the power plant. Homer (1976) recorded 93% reduced abundance and 76% less fish biomass during summer in a marsh creek 360 m from a discharge canal which experienced maximum thermal addition of 6°C. The impact of heated water on marsh systems in the summer is of concern in light of their important nursery function during this season.

II.3.5 Changes from small temperature elevations over wide areas

Direct effects of small temperature elevation on single species can potentially occur in the tropics, or during the summer, in species near the lower latitudinal extreme of their geographic range. Crossman (1969) has noted the loss of fish species near the southern limit of their range in Lake Erie due to a general increase in average water temperature of 1.1°C. The commercial clam, Mya arenaria is another case in point. This species reaches the southern extreme of its range in the Chesapeake Bay. A small temperature elevation (e.g. 1°C) above the natural summer maxima due to cooling system discharge can adversely impact Mya populations in this region. North and Adams (1969) cite examples of natural loss of cold water species in coastal bays in Southern California, and deterioration of kelp bed canopies in this region when temperatures 1.1°C or more above normal summer levels persist for several weeks.

In the tropics, coastal marine organisms commonly experience a thermal regime with maxima close to their upper thermal tolerance limit. A compilation of thermal effects data by the US EPA (1976b) suggests the range between optimum and exclusion temperatures in the tropics is around 5°C for many species, while sublethal thermal stress may be evident 2°C above optimum levels (Table II.1). It should be noted that almost all these data represent studies on semi-tropical populations (the larval fish study excepted). Optimum and upper limiting temperatures for tropical populations may be 1° to 2°C higher, according to comparative studies by Coles et al. (1976) in the Pacific and Kolehmainen, et al. (1975) at Puerto Rico. Nonetheless, the narrow range between thermal optima and upper limiting temperatures persists, (Thorhaug et al., 1978).

It is well recognized that small temperature elevations over wide areas, if persistent, can exert sublethal stresses which can become limiting, even if lethal temperatures do not occur. Temperature has a major influence on growth and reproduction in organisms, two phenomena crucial to population success. Such thermal effects are detailed for some percid fish by Hakanson (1977).

The influence of temperature on bioenergetics in aquatic organisms is well studied for fish (Brett, 1970; Warren, 1971). and certain marine bivalves (Thompson and Bayne, 1974; Widdows, 1978). If food supply is limited and the efficiency of any step of food procurement or energy conversion is reduced, energy available for growth and reproduction will also be reduced Bisson and Davis (1976) reports reduced growth in fish exposed to 4°C elevation in experimental streams. Food was clearly limiting in these studies, a condition which can occur in some rivers and deep lakes. Tropical waters are also often characterized by low productivity. Modest thermal additions could reduce growth in this biome if increased metabolic demands could not be met. In estuarine environments, food ususally would not be limiting in a quantitative sense, so normal or even enhanced growth may occur with low-level thermal addition. This has been reported in snails at a coastal site (Barnett, 1971), and in an artificial reservoir lake (McMahon, 1975). Yet even in a highly productive environment, such as an estuary, growth could be reduced should food become limiting in a qualititative sense for animals which have highly specific food requirements (Briand, 1975).

Reproduction is commonly one of the most sensitive life-cycle processes to elevated temperature, Production of eggs is energy expensive, especially when eggs are heavily yolked. Obviously if food is not sufficient to meet basic metabolic requirements, fecundity is reduced. Brungs (1971) found egg production to be the most sensitive parameter in response to experimentally increased temperature in the fathead minnow. He recorded a 29% reduction in fecundity with a 2.5°C elevation, while egg hatchability and fish growth was only affected by 6.5° and 8.5°C elevations. Reduced fecundity was also seen by McMahon, (1975) in the snails at a power plant site. In an experimental study with corals, Jokiel et al. (in press) saw a very narrow (1°-2°C) optimum for reproductive success; a 1°C elevation above this resulted in reduced success by 10 to 100 times. In contrast, growth of these corals was only reduced 10 to 20% by 1°C elevation above optimum. Timing of the reproductive season can also be altered by thermal additions, Barnett (1971, 1972) observed the breeding season begin and end some two months earlier in a few marine molluscs living near a thermal effluent. For species having planktonic larvae, probability for successful development to metamorphosis would be low with an early spawn, as water temperatures distant from the plant would be too low.

Low-level thermal addition also has the potential of altering community structure and important species interactions. The most conspicuous examples of this, involve loss of habitat forming species at semi-tropical and tropical sites (see section 3.4). This potential is also

TABLE II.1

A COMPARISON BETWEEN OPTIMUM AND SUBLETHAL AND LETHAL UPPER LIMITING TEMPERATURES FOR SEMI-TROPICAL AND TROPICAL BIOTA

(After EPA 1976b)

Biotic group	Optimum temperature (°C)	Thermal stress/ limiting temperature (°C)
Molluscs	26.7[a]	31.4 50% species exclusion
Echinoderms	27.2[a]	31.8 50% species exclusion
Coelenterates	25.9[a]	29.5 50% species exclusion
Porifera	24.0[a]	31.2 50% species exclusion
Fouling community larval settlement	25.4-27.8	28 50% reduction
Lytechinus variegatus growth and gonadal development	27	29.9 50% reduction gonadal vol. 32.0 86 hr. TL_{50}
Thallasia testudinum productivity	30	31 (daily av.) long term decreased growth
Survival of larval fish to 12 hrs. post-hatch	28.30	30.32 tolerance limit

[a] Temperature for high species diversity.

illustrated by changes in structure of a fouling community experiencing 3°C elevation above ambient (McCain, 1975). Community changes would be expected during the summer when maximum ambient temperatures were 29°C, yet shifts also occurred in February (ambient 24°C), with a dominant bryozoan sparse or lacking on the +3°C panels.

II.3.6 Changes from power plant chemicals

The effects of cooling water chlorination on aquatic life has received considerable attention recently. Yet laboratory studies predominate, with field assessments of the problem limited in number. Important literature reviews on this topic include those by Brungs (1973, 1976), Whitehouse (1975), and Mattice and Zittel (1976). An abstracted bibliography has been prepared by Mattice and Pfuderer (1976). Recent conference proceedings which detail our current understanding of chlorine related problems in aquatic systems include those edited by Jensen (1977), Jolley (1976, 1978), Block and Helz (1977).

The biocidal effects of chlorine on productivity within discharge canals have been quantified at a few plants. Hamilton et al. (1970) observed a 91% reduction of primary productivity during periods of chlorination at the Chalk Point (Md.) plant. Considering chlorination practices at that time and the magnitude of cooling water usage from the river, these authors projected a maximum loss of 6.6% of the primary productivity of the affected section of the river due to chlorination. Fox and Mayer (1975) reported a 57% drop in productivity in cooling water following chlorination at Crystal River, Fla. The reduction following plant passage was only 13% during periods of no chlorination. Reduced productivity of periphytic algae was observed downstream of a power station at a riverine site (Eiler and Defiro, 1974).

The response of caged fish exposed to chlorinated power plant discharge water has been examined by several workers. Trucken (1977) reported mortalities only for brown trout, non for brown bullheads or various sunfish. He reports ILC_{50} (intermittent median lethal concentration) values of 0.14 - 0.19 mg/l for fish held for 48 hours following either two or four intermittent chlorinations of 30 min. duration each. The 96-hr. ILC_{50} values were more variable, ranging from 0.02 - 0.05 mg/l (3 intermittent chlorinations) to 0.17 - 0.18 mg/l (6 chlorinations). In contrast to these findings, Liden and Burton (1977) failed to see significant effects of discharge canal water on survival of caged juvenile Atlantic menhaden and spot at the Morgantown (Md.) plant. Salinity here ranged from 0.5 to 10% varying with river flow. These fish experienced halogen concentrations of 0.020 - 0.080 mg/l total residual bromine chloride and 0.014 - 0.062 mg/l chlorine during 19 and 20 day study periods. Similarly, Marcy (1973) observed no adverse chlorination effects on fish at the Connecticut Yankee plant, possibly since only 1/4 of the condensers were chlorinated at a time, assuring good dilution at the discharge.

Drawing on existing chlorine toxicity data, Mattice and Zittel (1976) proposed zero mortality curves for acute chlorine exposures (0.0015 mg/l chlorine for freshwater; 0.02 mg/l for marine) (Fig. II.4). Certain sublethal effects of chlorination may fall within these suggested protection levels and some may not. For example, the lowest level of total residual oxidant found by Meldrim and Fava (1977) to elicit behvioral avoidance response in a marine fish was 0.03 mg/l. Cherry et al. (1977) determined the avoidance threshold for two freshwater fishes to be 0.1mg/l TRC or 0.03 mg/l FRC in a complimentary set of field and laboratory studies. Avoidance of near-threshold concentrations was most pronounced in winter. Low temperature (e.g. ≤12°C) contributes to a high concentration of the more toxic free chlorine radicles. Capuzzo et al. (1977) observed significant metabolic stress in stage I lobster larvae at 0.01 mg/l, the lowest total chlorine residual concentration tested. Clearly additional research is necessary to more fully assess the consequences of chlorination on aquatic systems. There are several unknowns concerning potential toxicity of bromate formed when sea water is chlorinated (Macalady et al., 1977); potential health problems regarding chlorinated fresh waters which may be reused for drinking has been addressed by Morris and McKay (1975) and Stevens et al. (1976).

Metals leaching from condenser tubes can also comprise a detectable chemical addition to discharged cooling waters. Highest levels of metals release have been reported at marine sites after a period of plant shutdown, during which non-circulating sea water was in contact with copper-nickel tubing. At a California plant, Martin et al. (1977) report 1,800 μg Cu/l in this water upon initial discharge, with rapid dilution following with flushing. Yet even after 30 days, copper concentration in the effluent water was 20 μg/l while intake water only contained 1 μg/l. These authors report 1500 abalone killed at this plant following testing of the cooling system. Their laboratory study shows 50 to 65 μg/l Cu is lethal to adults after 96 hours for the two species tested. Copper accumulation by the American oyster (<u>Crassostrea virginica</u>), in the vicinity of a power plant has been documented by Roosenburg (1969). Copper body burdens measured as high as 1.28 mg/g dry wt. within the effluent canal. Oyster condition index corresponded inversely with oyster copper body burden over a four year period. At high levels of copper accumulation, greening of the meats and a bitter taste results. At a Florida power plant, Grimes (1971) reported summer high values of 482 ppm zinc and 80 ppm copper in oysters from the discharge. Intake canal body oyster burdens were 138 ppm Zn and 9 ppm Cu. Zinc is high here as zinc ingots had been placed at all metal structures of the intake and discharge to retard electrolysis. Additional metals data are provided by Gilmore et al. (1975) for oysters cultured in power plant cooling ponds. In contrast, metals accumulation was not conspicuous in eels raised in power plant discharge water and cooling ponds (Romeril and Davis, 1976). Indeed, growth was so rapid, due to thermal addition and supplementary feeding, that no increase in metal was seen on a weight specific basis, except for iron, which accumulated in the livers. It is probable that metals body burden data obtained from filter feeding bivalves would represent the higher levels of bioaccumulation likely to occur in discharge waters due to their propensity to concentrate and retain metals.

II.3.7 Changes from total combined stresses

It is well recognized that a once-through cooling system poses a number of potential stresses and perhaps certain enhancing effects on the natural system. Circulation of an appreciable volume of water and addition of heat can enhance productivity. Yet the biota associated with this water mass may be altered as certain populations experience immediate or latent mortalities due to impingement, through-plant entrainment or discharge canal and plume stresses (Briand, 1975). The impact of the total cooling system on the biota can be assessed only in the field. Laboratory multivariate studies simply cannot include the full spectrum of positive and negative conditions, nor attain a scale adequate to realistically simulate cooling water use by a power plant. The consequences of total cooling system operation of a single power plant may be apparent on a waterway which is devoid of other pollution inputs and semi-enclosed, such as a coastal embayment, or on a small river where appreciable river flow is used for cooling. A shift from a benthic of planktonic dominated ecosystem has been seen by McKellar (1977) in the outer portion of an estuarine bay at the Crystal River (Florida) plant. The thermal component of the discharge was fairly dilute in this portion of the bay, but phosphorous levels were elevated. Plankton production increased 4 to 6 fold during the summer relative to a control bay, with turnover time decreasing from 6 to 5 days. In addition to this change in system structure, total system biomass was 15-20% lower in the outer discharge bay, even though total community gross primary production and metabolism was very similar to the coastal bays. This study illustrates the simultaneous stimulating and degrading effects of cooling water use.

Changes in a large oligotrophic lake whch received cooling water pumped from an eutrophic lake for 10 years was described by Koschel and Mothes (1976). The consequences of nutrient enrichment are apparent: phytoplankton productivity increased and water transparency decreased, resulting in a decrease of macrophyte depth limit from 20 to 12 m. Macrozoobenthos productivity increased 3 fold. Much of the observed changes can be explained simply by the pumping of cooling water from the eutrophic lake. Damage to entrained plankton and release of orthophosphate was apparent above 27°, although this would contribute to lake enrichment only in a minor way.

The impact of power plants on the fish community has been much studied. If the plant is small or is located on a large open water body, such as the Great Lakes, the primary consequences are local shifts, in the spatial distribtuion of fish fairly close to the discharge (Nugent, 1970; Neill and Magnunsen, 1974; Stauffer et al., 1976; Yoder and Gammon, 1976; White et al., 1977). The fish community of waterways receiving discharges from one large or multiple power plants has been assessed on the Patuxent River (Maryland) (O'Conner and McErlean, 1975) and a segment of the Wabash River in Indiana (Tepper and Gammon, 1976; Gammon, 1976). O'Conner and McErlean found a downward trend in diversity, number of species caught, species richness and evenness over the study period; biomass remained relatively stable. They concluded that a general degradation of the Patuxent River had occurred, but could not relate this specifically to power plant operations. In the Wabash River, Tepper and Gammon found primarily only local shifts in distribtuion of several species around power plants. There was a 4°C increase in temperature over the 161 km river section studied, but chemical and municipal discharges into the river precluded identification of any community effects of this thermal rise or of cooling water use in general.

Should a power plant be sited on a polluted waterway, one potential negative consequences is further deterioration of water quality, particularly dissolved oxygen content.

Considerable study of this effect has been undertaken in Europe (Wunderlich and Müller, 1976). A study in Poland showed only local elevation of nitrite levels and depression of dissolved oxygen at a 1600 MW(e) power plant on the Vistual River, a waterway which also received industrial and municipal wastes (Dojlido, 1977). However, during this study, no extreme or "worst case" conditions occurred (such as full generating load or low river flow). A formula was developed to predict the consequences of river water temperature rise and pollution loading on dissolved oxygen concentration. At a proposed power plant, this problem would have to be evaluated on a site specific basis, considering the nature and extent of waterway pollution and volume and Δt of cooling water used.

II.4 Potential Changes Due to Cooling Towers From Entrainment in Cooling Tower Makeup Water

Entrainment in cooling tower ("closed-cycle") makeup water can be assumed to cause 100% mortality. Repeated cycles through both tower and condensers, with biocides and anticorrosion chemicals added periodically, make any survival doubtful. While cooling towers may reduce water use at a power station, negligible survival in the "closed system" may cause more plant entrainment deaths than would a well-designed once-through system that allows nearly complete survival. Detailed comparisons are needed, however.

From Chemical Releases in Cooling Tower Blow-down. Toxic effects of chromates and chlorine are reasonably well established in the literature for many species (see Becker and Thatcher, 1973). Effects of newer cooling tower chemicals, often mixtures of complex organic and inorganic compounds for which the composition is held proprietary by the manufacturer, are not well understood. Phosphate compounds may contirbute to eutrophication. Phosphate released from one cooling tower system for a 715 MW(e) nuclear station was estimated to be equivalent to sewage phosphate from a city of 75,000 population (USAEC, 1972). In arid areas, the concentrated dissolved solids may affect aquatic life detrimentally.

From Cooling Tower Drift. Drift affects mostly the terrestrial environment, where cooling tower chemicals can accumulate in foliage and soils and damage may ensue (Taylor et al., 1975). Runoff of accumulated drift chemicals in the vicinity of towers may also create toxic levels in small streams for short periods. Little work has been done in this area.

From Blow-Down Heat. Heat of blow-down is generally too small in magnitude to create problems, unless discharged to very small streams.

From Meteorological Changes. Available evidence from existing cooling towers and ponds suggests that biological effects of meteorological changes would be minimal. There has been little critical study, however.

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