THE DECREASE IN AQUATIC VEGETATION IN EUROPE AND ITS CONSEQUENCES FOR FISH POPULATIONS

EUROPEAN INLAND FISHERIES ADVISORY COMMISSION
FOOD AND AGRICULTURE ORGANIZATION OF THE UNITED NATIONS

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by

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ABSTRACT

A diverse aquatic vegetation is essential to maintain a diverse fish fauna. The fish is an important part of a complex network of relations between nutrients, phytoplankton, epiphytes, herbivorous invertebrates, the aquatic vegetation and fish.

In Northwest Europe and North America and probably in the rest of the industrialized world, the (submersed) aquatic vegetation (macrophytes) is rapidly disappearing from eutrophicated waters. The decrease is well documented.

As a consequence of abundant growth of epiphytes, which are better competitors for inorganic carbon and light in highly eutrophicated waters than submersed aquatic macrophytes are, the condition of the aquatic vegetation becomes worse. Shallow, eutrophic, relatively clear water that is rich in water plants, can change to phytoplankton dominated turbid water, within short time. This change may occur without a remarkable increase in the actual nutrient loading.

Invertebrate grazers like snails, macrocrustaceans and cladoceran zooplankters are able to protect aquatic macrophytes against the negative effects of this competition by removing epiphytes and phytoplanktonic algae. As a man predator on invertebrates, the fish indirectly influences the well-being of the aquatic vegetation.

There is evidence that aquatic macrophytes are the source of biochemical compounds that negatively affect the growth of algae (allelopathy) and attract grazers. These processes are mainly found in model systems and under semi-natural conditions. Their ecological significance still has to be tested in the field.

A situation with turbid, phytoplankton dominated, water without aquatic vegetation can continue after removing nutrients from effluents because: (i) blue-green algae (phytoplankters) may excrete toxic substances, negatively affecting the growth of aquatic macrophytes; (ii) abundantly occurring young fish, but also invertebrate animals like mysids, prey on the bigger (phytoplankton grazing) cladocerans; (iii) acid rain, polluted bottom sediments and/or bird flocks contribute to the nutrient loading of a water body.

Restoration techniques are: lowering the nutrient loading in combination with protection of the remaining stands of reed, replanting of aquatic plants, creation of artificial refugia for zooplankton and manipulation of young-of-the-year fish populations.

Chemical and mechanical control of "nuisance" growth and heavy stocking with herbivorous fish including the common carp *(Cyprinus carpio)* have to be omitted or executed very carefully to avoid phytoplankton-dominated turbid water. In small systems with "nuisance" growth, stocking (50-150 kg/ha, max. 250 kg/ha) with grass carp *(Ctenopharyngodon idella)* can improve the water quality.

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1. INTRODUCTION

The European Inland Fisheries Advisory Commission (EIFAC) felt the need to pay more attention to impacts on fishery resources by other users of water, such as traffic of pleasure boats and intensive recreation on lake shores. Recreational activities and the increased input of nitrogen and phosphorus were identified as the main cause of the reduction of the aquatic vegetation (EIFAC/CECPI 1984). Because the aquatic vegetation is an important part of the habitat of many fish species, EIFAC recommended a compilation of bibliography on the causes of the destruction of reeds and other aquatic vegetation and remedial measures for their restoration (recommendation 69.2 of Thirteenth Session in Aarhus, 1984).

This paper documents the decrease in aquatic plants in many water bodies in the temperate zone of Europe and North America. It also pays attention to control and management techniques to protect or restore the lake shore vegetation.

Although the relationship between the presence of aquatic vegetation and the well-being of fish is obvious, causal relations are poorly documented. The same is true for the damage to fish populations caused by the decline in aquatic vegetation. Moss et al. (1979) produced some evidence for a considerable decline in catches in the Norfolk Broads and Rivers (England) after a strong decrease in biomass and species composition of aquatic plants. Grosch (1978, 1980) reported strong negative effects on the species composition of the fish fauna in the Havel lakes (West-Berlin) because of the destruction of the aquatic vegetation. Oster (1980) (in Lake Ontario, Canada) and Price et al. (1985) (in the Chesapeake Bayt U.S.A.), mentioned dramatic effects on fish stocks because of eutrophication and changes in the aquatic vegetation.

During the last 10 years many studies have been carried out on the ecological significance of aquatic macrophytes for the whole ecosystem with emphasis on nutrient demand and release, and the relations with invertebrates. This paper reviews a number of conceptual models by which the effect of eutrophication on the aquatic vegetation, and indirectly the effect on fish, can be understood. These mechanisms of interaction are of great significance for habitat improvement techniques, which will be very important for the fish fauna.

2. PLANTS IN THE AQUATIC HABITAT

2.1. MACROPHYTES

Aquatic plants do not belong to one distinct taxonomic group but rather form a collection of many plant taxa. The term "aquatic macrophytes" is commonly used for all macroscopic forms of aquatic vegetation; it includes macroscopic algae (stoneworts and the alga *Cladophora)*, some ferns and mosses (pteridophytes) and many flowering plants (angiosperms). On the basis of their emergence or submergence and the manner of attachment or rooting in the bottom sediment, two main groups, with 3 subdivisions are commonly distinguished (Wetzel, 1975).

A. Aquatic macrophytes rooting in sediment

i. Emergent aquatic macrophytes

Reed is often found in monospecific stands, but also mixed with Typha spp., *Scirpus lacustris* , *Acorus calamus, Dseudacorus, Butomus umbellatus* and *Sagittaria sagittifolia.* Emergent macrophytes are rooted in the sediment and may grow to a water depth of ca. 1 m. During the growing season all members of this group produce aerial leaves and flowers.

ii. Floating-leaved aquatic macrophytes

The floating-leaved plant communities are often predominated by *Nymphaea* spp., *Nuphar lutea* and *Nymphoides peltata. Potamogeton natans and Polygonum hydropiper* also belong to this group. The floating-leaved plants may root in water depths up to 3 m. and have floating or aerial flowers (reproductive organs).

iii. Submersed macrophytes

This group includes the stoneworts (charophytes) Chara and *Nitella*, a few moss species like Fontinalis antipyretica_and many flowering plants e.g. *Myriophyllum spicatum*, Elodea nuttallii, *Potamogeton pectinatus* and *E. perfoliatus*. The submersed macrophytes complete their life cycle under the water~ surface. Some species cause "nuisance growth". The degree of nuisance depend5 on the pursued management aims (water transport, recreation, fishery management or nature conservation) in the water body.

B. Freely floating macrophytes

These macrophytes are not rooted in the sediment, but live unattached in the water. The life forms within this group range from macrophytes with floating or aerial leaves and well developed submersed roots *(Hydrocharis morsus-ranae)* to very small surface floating or submersed plants with few or no roots (*Lemna trisulca* and *the waterfern Azolla*). Some plants in this group have aerial_flowers *(Utricularia vulgaris)* others complete their life cycle under the water surface *(Ceratophyllum demersum.).*

All aquatic macrophyte species mentioned in this review are listed in Table 1. A few species and some terms are visualized in Fig. 1.

2.2. MICROPHYTES

2.2.1 Phytoplankton

The phytoplankton consists of a large assemblage of microscopic algae. These plants freely move in the water column. Some species become buoyant after dying most algae sink to the water bottom. There are many taxonomic groups of phytoplankton algae and their systematics are very comp1ex. Many species of the freshwater planktonic algae belong to the extremely diverse chlorophytes (including also the macrophytes like *Chara*). Other important taxonomic groups

Table 1. The aquatic macrophytic species as referred in the text. sm = submersed; fl = floating-leaved; em = emergent; ter = terrestial; B = not rooted in the sediment

Charophytes (stoneworts)		-. Hydrocharitaceae	
Chara spec.	sm	Hydrocharis morsus-ranae L.	B/fl
Nitella spec.	sm	Stratiotes aloides L.	B/fl
Nitellopsis spec.	sm	Elodea canadensis Michx.	B/sm
		Elodea nuttallii (Planchon) St. John	B/sm
Bryophytes (mosses)		Hydrilla verticillata (L. f.) Royle	B/fl
Fontinalis antipyretica L.	sm	Vallisneria americana Michx	B/sm
Pteridophyta (ferns and allies)		-. Potamogetonaceae (“pondweeds”)	sm
Azolla spec. (“duckweed”)	B/fl	Potamogeton compressus L.	sm
Isoetes spec.	sm	Potamogeton crispus L.	sm
		Potamogeton densus L.	sm
Spermatophyta (flowring plants)		Potamogeton filiformis pers.	sm
-. Polygonaceae		Potamogeton lucens L.	sm
Polygonum amhibium L.	Ter/fl	Potamogeton natans L.	fl
Polygonum hydropiper L.	fl	Potamogeton obtusifolius L.	sm
		Potamogeton pectinatus L.	sm
-. Nymphaeceae (“water-lilies”)		Potamogeton perfoliatus L.	sm
Nuphar lutea (L.)	fl	Ruppia spec.	sm
Nymphea alba L.	fl	Zannichellia palustris L.	sm
Nymphea candida presl.	fl	Zostera marina L.	sm
-. Ceratophyllaceae (“Hornworts”)		-. Najadaceae	
Ceratophyllum demersum L.	B/sm	Najas marina L.	sm
-. Ranunculaceae		-. Iridaceae	
Ranunculus circinatus Sibth.	sm	Iris pseudacorus L.	em
-. Onagraceae		-. Gramineae	
Epilobium hirsutum L.	Ter.	Glyceria maxima (Hartmann) Holmberg	em
		Phragmites australis (Cav.) Trin. Ex steudel	em
-. Haloragaceae		-. Araceae	
Miriophyllum spicatum L. (“water milfoil”)	sm	Acorus calamus L.	em
Hippuridaceae		-. Lemnaceae (duckweeds”)	
Hippuris vulgaris L.	sm	Lemna trisulca L.	B/fl
		Lemna minor L.	B/fl
-. Primulaceae			
Hottonia palustris L.	em	-. Typhaceae (“cattails”)	
		Typha latifolia L.	em
-. Menyanthaceae		Typha angustifolia L.	em
Nymphoides peltata (Gmel.) O.Kuntze	fl		
		-. Cyperaceae	
-. Lentibulariaceae		Scirpus lacustris L.	em
Utricularia vulgaris L.	B/sm	Carex acuta L.	em
-. Alismataceae			
Sagittaria sagittofolia L.	em		
-. Butomaceae			
Butomus umbellatus L.	em		

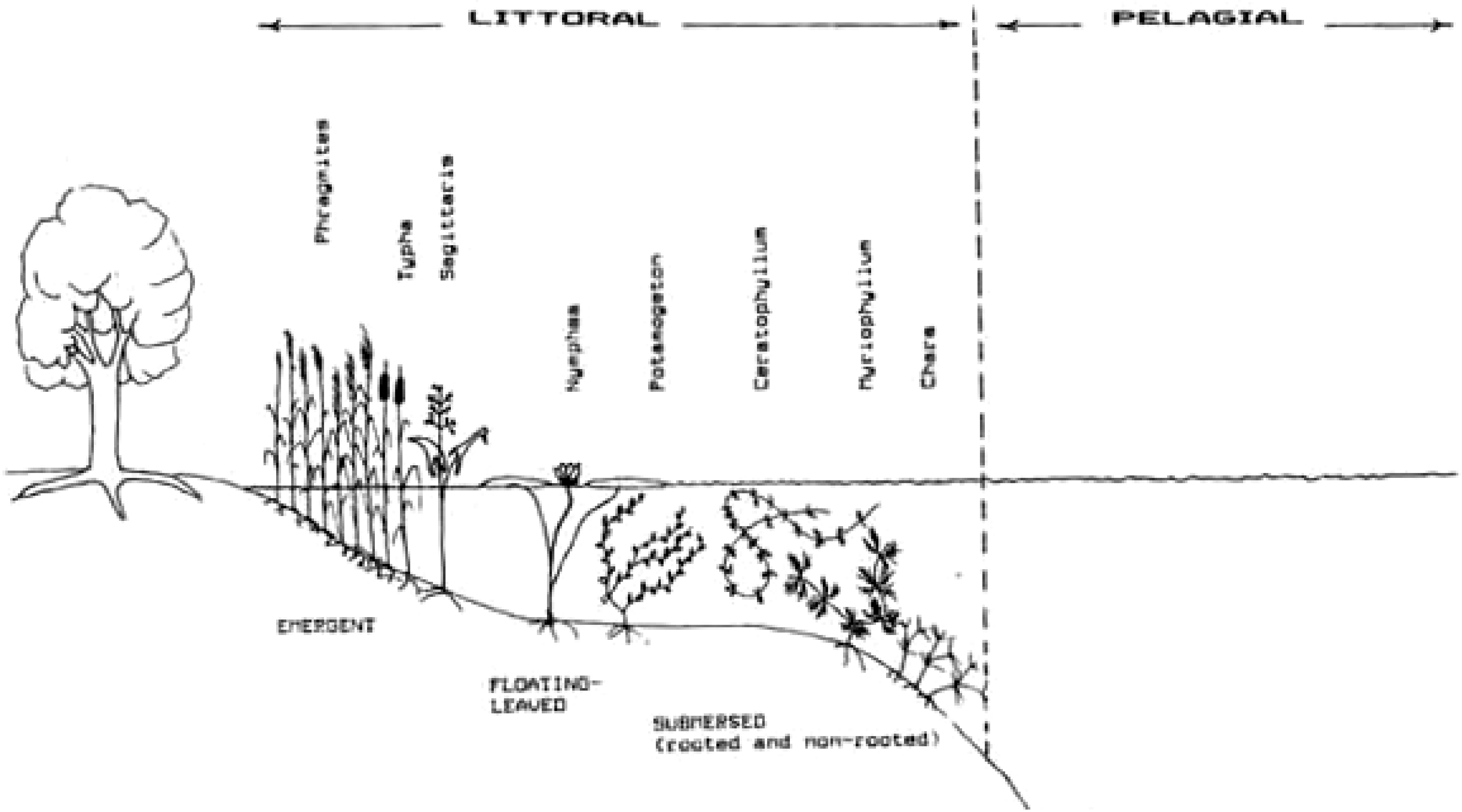

Fig 1. Schematic representation of the zonation in the aquatic vegetation and major terms used in this report.

that contain phytoplankton algae are (among others) the diatoms (Bacillariophyceae) and the blue-green algae (cyanophytes or Myxophyceae). Blue-green algae usually occur in colonies; many species can fix atmospheric nitrogen, like some bacterial species. They taxonomically differ much from the other groups of algae, like green algae and diatoms. Many detailed works deal with the identification of phytoplankton species; for an introduction to functional and ecological aspects see Golterman (1975) and Wetzel (1975).

2.2.2. Periphyton

The microphytes growing on the bottom sediments, stones, submersed macrophytes and other objects in the water column are called periphyton. If only growing on the submerged parts of the aquatic macrophytes, this periphyton is also referred as "epiphytes". Often the entire community of microscopic algae (with many diatom species), bacteria, protozoa and detritus on the submerged surfaces of the leaves is meant. The macrophytes benefit the epiphytes by providing organic carbon and nutrients (Allen, 1971). The presence of epiphytes may protect the leaves from grazing by invertebrates like snails, crustaceans and insect larvae, because not the plant itself is consumed but the epiphytes are scraped off (Carpenter & Lodge, 1986).

2.3. THE PHYSICAL AND CHEMICAL HABITAT CHARACTERISTICS

The aquatic macrophytes strongly affect the physical environment in the water. Within stands of aquatic vegetation the light intensity quickly decreases with depth, although great differences exist in the degree of light attenuation between specs (Titus & Adams, 1979). Not only the light regime but also the temperature in plant stands differs from open water sites. The vertical temperature gradient within vegetation may be very steep. So there are great differences between the surface temperature (where locally tropical temperatures can be reached) and the water layer just above the sediment (Grosch, 1978; Carpenter & Lodge, 1986)

Aquatic macrophytes influence the water movement and the sedimentation of particulate mineral and organic matter because the water plants suppress water turbulence. Suspended small particles settle faster to the bottom sediment in quiet water.

Aquatic macrophytes change the chemical properties of the water. Because green plants produce oxygen they contribute to the oxygen concentration in the water. Pokorny & Rejmankova (1983) measured a net oxygen production up to 5.7 mg/l daily in dense *Ceratophyllum demersum* stands in small fish ponds. Oxygen is not only released in the water column. Many plant species have air spaces in their tissue, in which photosynthetically produced oxygen is transported by diffusion. In this way the plants transport oxygen to their roots. Subsequently the oxygen is often released in the sediment (Carpenter & Lodge, 1986).

On the other hand, aquatic macrophytes may also indirectly cause oxygen depletion. Pokorny & Rejmankova (1983) found decreased oxygen concentrations in duckweed-covered water, because the plants reduce the light penetration into the water. Respiration, not compensated by oxygen production during night time. and the decay of macrophytes will directly take oxygen from the water. Usually the main part of the decay will take place in autumn, when the water temperature is low and the oxygen demand of invertebrates and fish is slight. However. in abnormal circumstances. especially in eutrophic, warm, shallow, poorly circulated waters, high amounts of oxygen will be derived from the water column (Godshalk & Wetzel, 1978; Carpenter & Greenlee, 1981).

Rooted submersed and floating leaved macrophytes form a living link between sediment and water column. Nutrients can be transported from the sediments to the water: the f1oating-leaved plants like *Nymphaea* spp. and *Nuphar lutea* for instance potentially can function as an important nitrogen and phosphorus "pump" (Brock et al. 1983). However, all (aquatic) macrophytes may also act as a sink and can immobilize nutrients. During periods of active growth e.g. *Potamogeton pectinatus* and the attached epiphytes can act as a sink for phosphorus (PO4-P) (Howard-Williams, 1981)

This internal nutrient cycle is basically a continuous process during the growing season. On the one hand aquatic macrophytes actively excrete nutrients and organic substances; on the other hand nutrients and organic substances are passively released in the water (Pomogyi et al., 1984). Nutrients are released when living parts of the plants are damaged by animals, or by means of autolysis, leaching and microbial breakdown. The role of muskrats (*Ondatra zibethicus*), birds (e.g. coots, *Fulica atra*), insects and crustaceans in decomposition of *Nymphoides peltata* is extensively studied (Lammens & van der Ve1de. 1978; van der Velde et al., 1982; Wallace & O'Hop, 1985). In laboratory investigations the loss rates of nutrients from *N. peltata* showed the following order: K>Na>P>Mg>C>N>Ca>Fe (Brock et al., 1983; Brock, 1984). Very local environmental circumstances. for instance the grazing activity of small animals. may influence this decomposition process, but field studies on *Nuphar lutea* revealed that the decompostion of the leaves was also strongly dependent on the alkalinity of the water body (Brock et al., 1985).

The residence time of nutrients in the biomass of living organisms depends on the length of their life cycle. This is several months for aquatic macrophytes. which is nearly always intermediate between the residence time in plankton (several days or weeks) and in fish (several years) (Carpenter & Lodge, 1986).

The emergent aquatic macrophytes are an interface between the surrounding land and the water. They are, to a greater degree than the rooted submersed and floating leaved aquatic macrophytes. influenced by the quality of the groundwater and the composition of the soil of the surrounding land (Verhoeven, 1983). Emergent aquatic macrophytes can remove nutrients from the water column. For instance reed *(Phragmites)* beds may remove nitrogen and (PO4)-phosphorus from the water. The phosphorus is adsorbed to the sediments and the nitrogen is released from the sediment as ammonia or nitrogen gas due to denitrification by bacteria (Kickuth. 1978). During the decompostion of *Phragmites* leaves, the released nutrients occur initially in the water but are successively adsorbed to the sediment (Best et al., 1982).

Summary: the aquatic vegetation strongly influences the light conditions. temperature. oxygen concentration. sedimentation rate and turbulence in the water body. The submersed and floating leaved macrophytes have a central position in the process of internal recycling of materials in the aquatic ecosystem. Whereas the emergent aquatic macrophyte can reduce the external nutrient loading.

2.4. AQUATIC VEGETATION AND INVERTEBRATES

2.4.1. Macrofauna

Invertebrates are the main food source for most fish species. The quantity of invertebrate organisms is related to the amount of aquatic vegetation (Murphy & Eaton. 1981; Dvorak & Best. 1982; Whitefield. 1984). The invertebrates use the macrophytes as substrate. shelter and habitat for feeding. The (American) crayfish. *Orconectes limosus* occurs mainly in the aquatic vegetation. Juvenile animals feed on filamentous_algae. but also vascular plants. Larger individuals (>25 mm) are omnivorous and also consume molluscs. Grosch (1978) reported a dramatic decline in the catches of this crayfish. when the aquatic vegetation in the Heiligensee (West-Berlin)

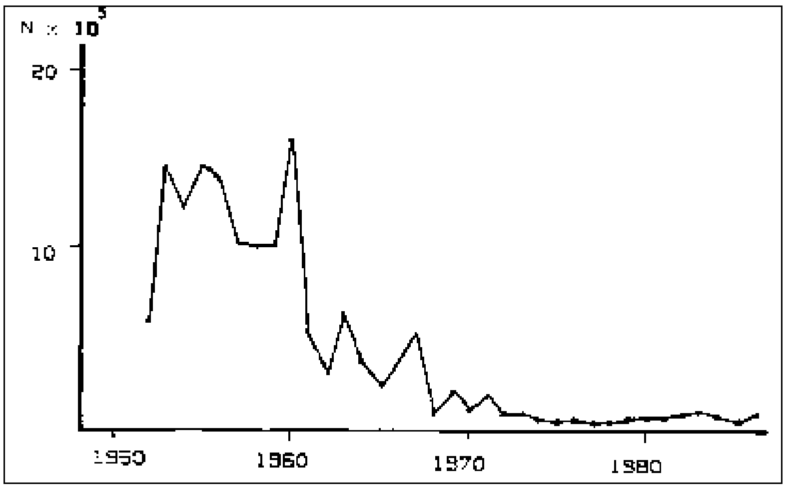

Fig.2. Annual catch (in numbers) in the Berlin area of the American crayfisch (*Orconectes limosus* Raff.) (Unpublished data by courtesy of dr. Ulrich Grosch)

decreased (Fig. 2). The macrophyte itself is rarely consumed. but the epiphytes on the surface of the leaves and other submerged parts of the plants are consumed. Excessive growth of epiphytes is harmful to the macrophyte. because insufficient quantities of light and inorganic carbon (HCO_3^- and CO_2*)* can reach the plant and consequently the macrophyte photosynthesis is hampered (Sand-Jensen. 1977; Sand-Jensen & Sondergaard. 1981). Therefore the amount of epiphytes is an ecologically important factor which. in its turn. is strongly dependent on invertebrate grazing (Orth & van Montfrans. 1984). Hootsmans & Vermaat (1985) found under experimental conditions a five fold increase in growth of *Zostera marina* plants in the presence of grazing snails in comparison with non-grazed plants.

Recently the relations between snails and aquatic macrophytes have been investigated in more detail. Bronmark (1985b) found high correlations between the species diversity of plants and gastropods (snails) within a number of eutrophic ponds. Lodge & Kelly (1985) showed a rapid recovery of the *Lymnae peregra* population concomitant with the come-back of submersed macrophytes after their sudden disappearance.

2.4.2. Zooplankton

The aquatic macrophytes provide shelter to "large" cladoceran zooplankton species (Leah et al. 1980; Timms & Moss. 1984). The grazing activity of zooplankton benefits the smaller edible phytoplankton species that grow faster because of more efficient nutrient uptake. If the larger zooplankton species become scarce (for whatever reason) the phytoplankton community will change to slow-growing bigger, mainly unedible blue-green species. These bigger. often colonial blue-green phytoplankters compete more easily in a nutrient-rich environment with the small edible species. They form a high population biomass with chlorophyll-a concentrations of several hundreds µg/l. Because high chlorophyll concentrations hamper the light penetration. the conditions for submersed aquatic macrophytes will become worse (Moss. 1976; Moss & Leah. 1982; Moss et al. 1985).

2.5. AQUATIC MACROPHYTES AND FISH

2.5.1. Spawning and foraging

Many fish species of the temperate zone need sites with macrophyte stands in the spawning season to deposit their eggs. For successful hatching the oxygen concentrations have to be sufficient. while after emerging from the eggs. the larvae need substrate and oxygen. The oxygen supply for the larvae is maximal close to the surface of the living leaves. Young fish also need the plants to protect themselves from predation by piscivorous species (Grosch. 1978) or to avoid cannibalism (Grimm. 1981). Several indigenous adult fish species of , the Northwestern and Middle Europe like tench (*Tinca tinca*), rudd, (*Scardinius eryhtrophthalmus*) and pike (*Esox lucius*) live in stands of emergent and floating-leaved aquatic macrophytes. The submersed aquatic macrophytes form important feeding habitats for perch (*Perca* *fluviatilis*), roach (*Rutilus rutilus)*, bleak (*Alburnus alburnus*) and eel (*Anguilla anguilla*), while the ruff (*Gymnocephalus cernua*) prefers the *Chara* beds (Table 2). Some species need relatively soft submerged plants to eat: especially rudd and roach consume considerable amounts of macrophytes. But also in the diet of tench, bream (*Abramis brama*), white bream (*Blicca bjoerkna*), carp (*Cyprinus carpio*), chub (*Leuciscus cephalus*), ide (*l. idus*), crucian carp (*Carassius carassius*), gudgeon (*Gobio gobio*) and stickleback (*Gasterosteus aculeatus*) small or minimal amounts of plant material was found (Prejs, 1984).

Table 2. Preferred zones in North-German lakes by adult fish species. 1: Eulittoral (zone with emergent aquatic macrophytes). 2: Littoral (zone with floating-leaved and submersed aquatic macrophytes). 3: Lower littoral (zone with scattered *Chara*-beds). 4: Littori-profundal (no or very few living aquatic macrophytes). 5: Pelagial (open water zone) (after Grosch, 1978)

fish species	**zone in the lake**				
	1	**2**	**3**	**4**	**5**
Anguilla anguilla (eel)		+	+	+	
Osmerus eperlanus (smelt)					+
Esox lucius (pike)		+			
Abramis brama (bream)				+	
Alburnus-alburnus (bleak)					+
Blicca bjoerkna (white bream)		+			
Carassius auratus gibelio (gibel carp)		+			
Carassius carassius (crucian carp)		+			
Cyprinus carpio (carp)		+			
Gobio gobio (gudgeon)	+				
Leucaspius delineatus (German: Moderlieschen)	+				
Leuciscus cephalus (chub)				+	
Rhodeus sericeus (bitterling)		+			
Rutilus rutilus (roach)		+			
Scardinius erythrophthalmus (rudd)		+			
Tinca tinca (tench)		+			
Misgurnus fossilis (weatherfish)		+			
Silurus glanis (wels or European catfish)		+		+	
Lota lota (burbot)					
Gasterosteus aculeatus (stickleback)	+				
Gymnocephalus cernua (ruffe)			+		
Perca fluviatilis (perch)	+	+	+	+	+
Stizostedion lucioperca (pikeperch)					+

Traditionally. the abundant occurrence of reed is not considered as a desirable situation for the exploitation of fish populations (Grosch, 1978). However, Deufel (1978) stressed the importance of the reed swamps and other emergent vegetation for fish populations in lake Constance (Bodensee) (Table 3). There are seven species in his list which are of commercial importance.

2.5.2. Influence of fish on aquatic macrophytes

The fish can directly influence the dispersion of the plants. Carpenter & McCreary (1985) showed that the nesting behaviour of three sunfish species (the American Perciformes: *Lepomis gibosus*, *Micropterus sa1moides* and *M. dolomieui*) strongly influenced the zonation of aquatic macrophytes. The nest sites of these fishes are firstly cleared of vegetation by the fish. This vegetation spreads vegetatively via horizontal stems and mainly consists of a Myriophyllum-species. In August, after use, the nest sites are colonized by diaspore-propagated plant species (among others an Isoetes- species) which do not spread vegetatively. Every following spawning Season about 20% of the original nest sites are not occupied.

Table 3. List of fish species in Lake Constance, temporarily living in the zone of emergent vegetation. 1. Feeding habitat; 2. Spawning area; *. Commercially important species (Deufel. 1978)

Fish species	**function of emergent vegetation**	
	1	**2**
Anguilla anguilla (eel) *	+	
Esox lucius (pike) *		+
Abramis brama (bream) *		+
Blicca bjoerkna (white bream)		+
Carassius carassius (crucian carp)		+
Cyprinus carpio (carp) *		+
Gobio gobio (gudgeon)	+	+
Leuciscus cephalus (chub)		+
Leuciscus leuciscus (dace)		+
Phoxinus phoxinus (minnow)	+	
Rhodeus sericeus (bitterling)		+
Rutilus rutilus (roach) *	+	+
Scardinius erythrophtalmus (rudd) *	+	+
Tinca tinca (tench) *	+	+
Misgurnus-fossilis (weatherfish)	+	+
Noemacheilus barbatulus (stone loach)	+	+
Silurus glanis (wels or European catfish)		+
Gasterosteus aculeatus (stickleback)		+
Cottus gobio(bullhead)	+	

Patches of a mixed vegetation arises on these abandoned nest sites of *Myriophyllum* (recolonizing the site from the edges) and the diaspore propagated plant species. These patches can persist several years. Thust the basses actively contributed to the creation of a mosaic pattern of different aquatic plant species.

2.5.3. Influence of macrophytes on piscivorous predation

Among others Savino & Stein (1982) have shown that moderately complex habitat structures are important for young fish to hide and feed and that they will produce more stable prey-predator relations between fry and piscivorous fish. Crowder & Cooper (1982) proved that for bluegill sunfishes (*Lepomis macrochirus*) in experimental ponds, a habitat structure with intermediate macrophyte density is favourable for the fish as well for the prey organisms. In this situation they found the highest growth rates for the fish and more variation in its diet.

Grimm (1981, 1983) studied the effect of stocking artificially propagated young pike (*Esox lucius*) on the natural pike population in a number of water bodies in the Netherlands. He concluded that the presence of aquatic plants and not stocking with young piket determined the abundance of adult pike. In vegetated areas the survival of young pikes was higher than in unvegetated waters. Intraspecific predation (cannibalism) was the most important factor that regulated the abundance of adult pike. Stocking had no effect on the composition of the adult pike population. In a Florida lake in the presence of a coverage by 75% of *Hydrilla verticillata* also a higher survival was also found for the young of the year of *Esox niger* and some other important American sportfishes (Shireman et al., 1983).

2.5.4. Indirect effects of predation

Stein & Magnuson (1976) found that the feeding behaviour of the crayfish *Orconectes propinquus* in the U.S.A. is influenced by the presence of predators like the smallmouth bass, *Micropterus dolomieui*. The crayfishes will hide more often in the substrate and spend less time on grazing. So the basses indirectly affect the feeding relations between plants and crayfishes.

Predation by young of the year fish of smelt (*Osmerus eperlanus*), perch (*Perca fluviatilis*), roach (*Rutilus rutilus*) and bream (*Abramis brama*) can affect the body size and biomass of the zooplankton community (Hrbàcek et al., 1961; van Densen, 1985 and many others). This predation on the "larger" zooplankton can affect the phytoplankton community because the most important phytoplankton grazers disappear. The unedible biggert colonial algal species start to dominate (see Section 2.4.2.). Therefore the removal of fish from an eutrophic lake may cause clearer water and the return of rapidly-growing small phytoplanktonic species (Andersson et al. 1978; Leah et al. 1980; Reinertsen & Olsen, 1984; Brabrand et al. 1986).

2.5.5. Effects of changes in macrophyte abundance on fish

Moss et al. (1979) mentioned a decline in catches in the Norfolk Broads and Rivers after a dramatic decrease of the aquatic vegetation. The perch (*Perca fluviatilis*) and gudgeon (*Gobio gobio*) declined, probably due to decreased habitat diversity.

At the beginning of the 20 th century about 28 truly indigenous species occurred in the Havel lakes (West-Berlin). In 1965 these species were still present. However, in 1979 6 species became extinct among which barbel (*Barbus barbus*), dace (*Leuciscus leuciscus*) and weatherfish (*Misgurnis fossilis*). About 9 species were almost extinct or sporadically found; among them were: smelt (*Osmerus eperlanus*), chub *(Leuciscus cephalus)*, rudd (*Scardinius erythrophthalmus*), gudgeon (*Gobio gobio*), wels (*Silurus glanis*) and burbot (*Lota lota*). Among the 5 species that became rare, pike (*Esox lucius*), crucian carp (*Carassius carassius*) and tench (*Tinca tinca*) were mentioned. These changes probably were connected with the decrease in aquatic plants (see also Section 2.5.1. and Table 1). On the other hand bream (*Abramis brama*), white bream (*Blicca bjoerkna*), perch (*Perca fuviatilis*), ruff (*Gymnocephalus cernua*) and pikeperch (*Stizostedion lucioperca*) increased. Only the eel (*Anguilla anguilla*) -due to intensive stocking- and the bleak (Alburnus alburnus) remained constant since 1965. The pikeperch catches increased considerably (Fig. 3). This species usually stays in the pelagial zone and requires bare sediment for spawning (Grosch, 1978, 1980).

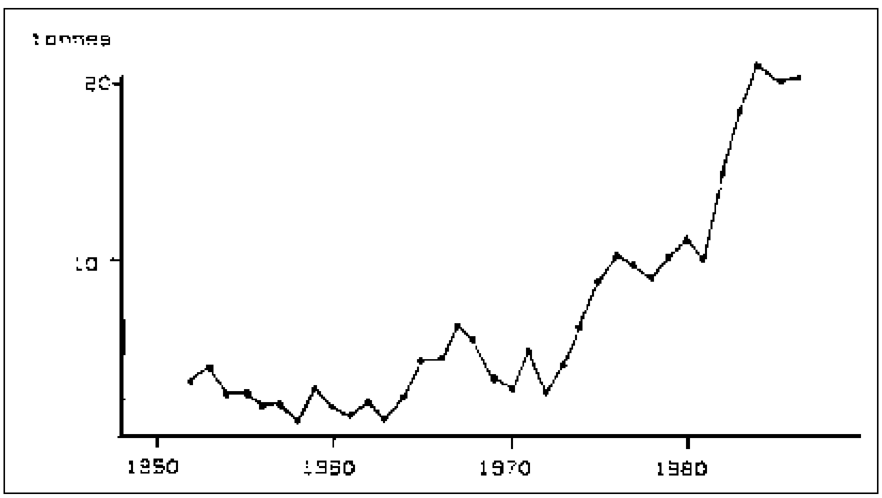

Fig. 3. Annual yield (in tonnes) of pikeperch (Stizostedion *lucioperca*) in the Berlin area (Unpublished data. by courtesy of dr. Ulrich Grosch, 1978)

Large areas with submersed *Potamogeton* disappeared from the hypertrophic Tjeukemeer between 1971 and 1981. De Nie (1987) produced evidence for a strong decrease in the availabi1ity of gammarids and worsened feeding conditions for eel (*Anguilla anguilla*) since that time. Lammens (1986) observed an increased abundance of pikeperch and a decrease in biomass of roach (*Rutilus rutilus*). perch (*Perca fluviatilis*) and ruff (*Gymnocephalus cernua*) in the Tjeukemeer during 1977-85 in comparison with the previous six years (1971-77). Lammens et al., (1986) showed that the three decreased fish species are more adapted to foraging in a 1ittora1 habitat (see a1so Tab1e 2. Section 2.5.1.).

Eutrophication caused in Lake Ontario (Canada) an expklosive growth of the filamentous algae *Cladophora* sp. and a collapse of the walleye (*Stizostedion vitreum*) population. Adverse effects on egg hatching of salmonids and coregonids were reported too. The (introduced) carp (*Cyprinus carpio*) and (native) yellow perch (*Perca flavescens*) populations increased because of the effects of eutrophication (Oster, 1980).

Price et al. (1985) reported a negative effect of eutrophication on the striped bass (*Moron*e *saxatilis*) population In the Chesapeake Bay area (U.S.A.). Although presented as a "speculative hypothesis", they argue that the surface area with sufficient oxygen has dramaticly decreased and so the hatching area of this fish species. This consequently endangers the well-being of the entire population.

Summary: Fish species need oxygen and most species also need substrate for their eggs and larvae. Young and adult fish need oxygen, food and shelter. The aquatic vegetation must be present to satisfy these needs, which differ in space and time. Therefore not only the presence of a few water plant species, but a diverse aquatic vegetation is essential to maintain a diverse fish fauna. Fish are an important part of a complex network of relations between nutrients, phytoplankton, epiphytes, grazers and aquatic macrophytes: directly because of their predation on herbivorous zooplankton and larger invertebrate organisms in aquatic vegetation, and indirectly because of piscivorous predation on planktivorous (small) fish.

3. THE DECLINE IN AQUATIC PLANTS

The decline in aquatic macrophytic vegetation in a number of water bodies is well-documented. The eutrophication is always indicated as a main cause.

3.1 THE DECLINE IN EMERGENT VEGETATION

(mostly reed, *Phragmites australis*)

In many parts of Europe pollution and eutrophication of water bodies started at the end of the 19th century. Slight eutrophication benefits the vegetation growth. Although reed can endure relatively high nutrient loading, already during the 1940s Hurlimann (1951) found declines of the reed beds due to eutrophication in the Zurcher

See and the Vierwaldstattersee. Since the 1960s very severe declines were reported from quite different localities (Table 4).

Table 4. Rates of decrease in the surface area of reed (*Phragmites australis*) at different localities. The surface area at the beginning of the period is fixed on 100%. The emphasized numbers are the rates of decrease per year (comparable with inflation rates). (1): changes due to natural succession

LOCALITY	**PERIOD**	**% DECREASE**		**AUTHOR (S)**
		In period	Per year	
Norfolk Broads	1880 - 1905	100 - 113	0.5	Boorman & Fuller (1981)
	1905 - 1946	113 - 56	-1.7	
	1946 - 1958	56 - 44	-2.1	
	1958 - 1963	44 - 31	-6.7	
	1963 - 1970	31 - 29	-1.1	
	1970 - 1977	29 - 23	-3.2	
Lake Constanz (Bodensee)	1926 - 1974	100 - 65	-0.9	Schroeder (1979)
Pfaffikersee	1954 - 1966	100 - 70	-3.0	Burgermeister & Lachavanne (1980)
Havel lakes	1962 - 1967	100 - 69	-7.2	Sukopp & Markstein (1981)
	1967 – 1972	69 - 50	-6.3	
	1972 - 1977	50 - 39	-4.8	
	1977 - 1982	39 - 28	-6.3	Markstein & Sukopp (1983)
NW. Dummer	1968 - 1974	100 - 83	-3.0	Akkermann (1978)
Dgal Wielky	1966 - 1978	100 - 33	-8.7	Krzwosz et al. (1980)
Vechten	1973 - 1980	100 - 13	-25.7	Best (1982)

The rate of decrease ranged from 0.9% to 24.7% per year. The rates were not constant during the whole period. In the Pfaffikersee the main decrease occurred between 1954 and 1966, while in Lake Vechten the reed-covered surface area decreased by 83% in one year (Best, 1982). In Lake Mikolajkie (Ozimek & Kowalczewki, 1984) and Hickling Broad (Boorman & Fuller, 1981), where rapid changes in submersed aquatic macrophytes occurred, the reed beds virtually stayed unaffected.

In the Northbrandenburgian lakes a relatively slight decrease in emergent plants was found compared to the strong decrease in submersed aquatic macrophytes (Jeschke & Müther, 1978)

Most losses of reed progress from the water side of the reed bed to the shore, sometimes followed by expansion of the emergent *Typha* sp. (Sukopp, 1971; Dinka et al. 1979) or floating-leaved *Nuphar lutea* (Cragg et al. 1980).

3.2. THE DECLINE IN SUBMERSED AND FLOATING-LEAVED AQUATIC VEGETATION

Denmark (Randers Fjord)
Sand-Jensen (1977) mentioned Danish investigations in the Randers Fjord where large areas with richly developed submersed macrophytes, described in the 1950s, had disappeared in 1975 from the innermost 17 km of the Fjord.

Finland
Toivonen (1985) compared data from the late 1940s with records of non-rooted, floating-leaved and submersed vascular plants made between 1976 and 1979 in 54 different lakes. They found that five lakes had become hypertrophic. The submersed vegetation had greatly decreased or was restricted to small lagoons. *Lemna minor* and *Ceratophyllum demersum* appeared to be the only species highly resistant to eutrophication.

German Democratic Republic (Northbrandenburgian lakes)
Jeschke & Müther (1978) describe the changes in the vegetation of two lakes (Grienerick-See, 96 ha and Rheinsberger See, 269 ha) between 1964 and 1974. They distinguish 25 types of aquatic vegetation. One vegetation type, characterized by large leaved *Potamogeton* species *P lucens* and *P. perfoliatus)*, became extinct while 8 types were in a regression phase. Communities with Characeae nearly became extinct while communities with *Stratiotes aloides* and *Hydrocharis morsus-ranae* decreased.

Vegetation types, in which *Myriophyllu spicatum* and Potamogeton pectinatus together dominated, had expanded since 1966. The community dominated by *Myriophyllum spicatum* mixed with *Nuphar lutea* was the most stable, while the monospecific vegetation of *M. spicatum* considerably decreased.

The eutrophication stronger influenced the submersed than the emergent macrophytes and degeneration of the reed beds was also reported.

The Netherlands
(Loenderveen- and Loosdrecht Lakes)
Between 1949 and 1980/82 23% of the submersed, 40% of the floating-leaved and 48% of emergent plant species disappeared in the slightly eutrophicated Loenderveen Lakes. Between 1961 and 1980 the Characeae became almost extinct. Close to these lakes are the more polluted.

Loosdrecht Lakes, which are strongly affected by recreational activities. On the average the decline in submersed aquatic macrophytes was far greater in these lakes than in the Loenderveen Lakes (Best et al.. 1984).

(Lake Vechten and Lake Maarsseveen) In two small Dutch meso- to eutrophic sand pits the decline of submersed macrophytes has been accurately documented. Within 6 (or 10 in Vechten) years of the study the area covered by submersed vegetation declined by a factor 6 to 570. In lake Vechten (4.7 ha) the total area of submersed macrophytes decreased by 16.7% per year (*Elodea canadensis* alone by 23.1%) from 1973 to 1983. In lake Maarsseveen-I (70 ha) the total area covered by submersed macrophytes nearly halved (49.1%) each year between 1977 to 1983. (*Elodea canadensis* decreased by 60.8% per year, *Potamogeton lucens* by 34.2%, the Characeae by 65.3%) (Best & Meulemans, 1984)

Poland (Mazurian lakes)
The decline in aquatic vegetation of lake Mikolajskie (460 ha) is well-documented (Ozimek & Kowalczewki. 1984). Although these authors did not found a decrease in the number of plant species, they recorded a strong decrease in biomass by 16.6% per year in the period 1963-1980 (= 94% over the entire period).

The area covered with submersed macrophytes decreased by 1.1% per year from 1963 to 1971 and further to 4.2% per year from 1971 to 1980. The Characeae, *Fontinalis antipyretica, Ceratophyllum demersum* nearly disappeared. The total area covered by *Utricularia vulgaris*, *Sagittaria sagittifolia* and *Stratiotes aloides* remained constant while that of *Potamogeton obtusifolius, P. lucens, P. perfoliatus, P. compressus, P. pectinatus,* and *Lemna trisulca* increased.

In polluted areas the submersed macrophytes obviously established relatively stable communities composed by *Potamogeton perfoliatus, P. pectinatus* and *Ceratophyllum demersum,* while more distant from sources of pollution the changes were greater

Switzerland and the German Federal Republic (Perialpine lakes)
Several Swiss lakes are changing from the meso-eutrophic to the hypertrophic phase. The meso-eutrophic phase is characterized by abundant growth of submersed macrophytes down to 5 to 6 m depth. The Characeae are still present and *Potamogeton pectinatus* and *P. erfoliatus* as the dominant submersed species. Well-developed reed beds fringe the lake.

In the hypertrophic stage the submersed macrophytes (like *Najas marina* and *Potamogeton crispus)* become scarce or are completely absent at depths greater than 2 m (Lachavanne, 1985). The stands of *Phragmites* decrease while floating-leaved species like *Nuphar lutea*, are the only common aquatic macrophytes outside the reed beds. The Characeae disappeared from the Pfaffikersee (Burgermeister & Lachavanne. 1980), Lake Morat (Murtensee) and the Burgaschsee (Lachavanne 1979a&b). In Lake Constance the depth of occurrence of the

Characeae changed from a maximum depth of 25 m in the 1950s to 5-7 m in 1978 (Deufel, 1978).

United Kingdom
(Norfolk Broads, England)
Moss (1980) extensively documented the decline of aquatic macrophytes in the Norfolk Broadlands, an area with ca. 50 mostly small (1- 20 ha) shallow lakes, interconnected by channels with a few rivers. From three Broads the pollen and algal remains in sediment cores were analysed. These remains were dated by the Pb-210 and Cs-137 method or by means of estimation of the sedimentation rate. It became apparent

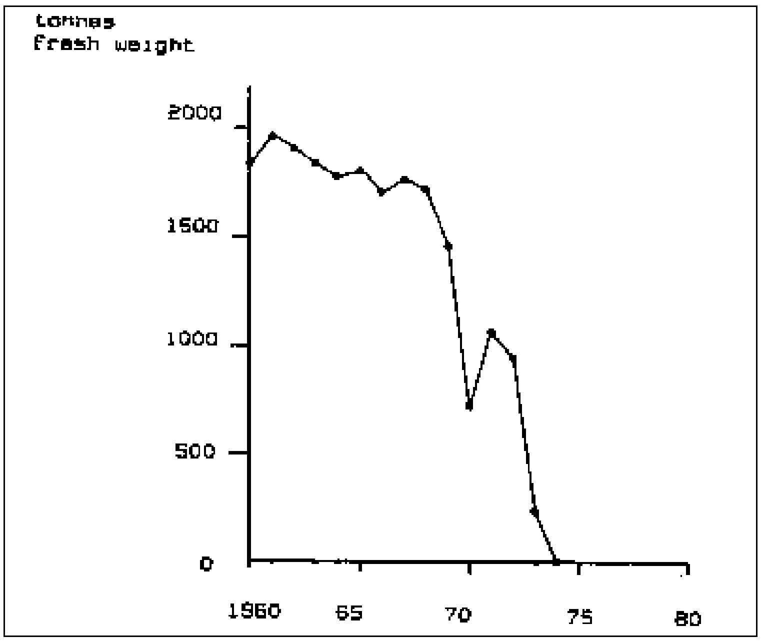

Fig. 4. Decline of submersed aquatic plant populations in Hickling Broad, reflected in the quantities removed each summer to maintain open the navigation channel through the Broad. Weights are approximate and based on the average weight of each truck load removed (Moss & Leah, 1982; with permission of the authors)

from this paleolimnological study that before 1800 AD Strumpshaw Broad (ca. 12 km from the city of Norwich) was clear with submersed macrophytes rooted in the bottom and virtually no phytoplankton. From 1800 to 1900 the epiphytes and Characeae increased. Moss et al. (1985) call this situation Phase I. In this situation the fisheries are productive and the area is rich in wildlife. In the beginning of the 20th century the Characeae are replaced by taller macrophytes like *Potamogeton pectinatis, Myriophyllum spicatum* and *Ceratophyllum demersum.* During that period the eutrophication started by sewage effluents from the nearby city and cultivation and fertilization of the surrounding land. This is called Phase II, the phase with abundant growth of taller submersed aquatic angiosperms together with epiphytic and filamentous algae. The fisheries are still productive and the area is considered important for nature conservation.

Phase II gradually ended in Strumpshaw Broad between 1912 and 1950, while Broads without sewage discharges retained submersed vegetation (Characeae and *Najas marina*) until the early 1970s. Since the 1960s in much of the Broads system the aquatic macrophytes disappeared (Fig. 4) and phytoplankton started to predominate. This is Phase III, in which also the fisheries became less productive. Increased sedimentation (mud flats) and erosion of the banks became problematic. Some isolated patches of *Potamogeton pectinatis, Hippurus vulgaris* and *Myriophyllum spicatum* in the least affected lakes were the poor remains of Phase II between 1974 and 1980.

Hickling Broad (the largest lake; 120 ha)is not affected by human sewage effluent input. However this lake is "guanotrophic" because large flocks of gulls, *Larus ridibundus* but also starlings, *Sturnus vulgaris* roost in or near this lake during nighttime. Fish-kills occur almost annually since 1969. These kills are caused by substances, secreted by the phytoplanktonic species *Prymnesiurn parvum,* which are toxic for fish (Moss, 1980).

(Loch Leven, Scotland) A comparison could be made between data from 1910 and field studies during 1972-74 in Loch Leven (shallow eutrophic, 1330 ha). Jupp & Spence (1977a) recovered only 12 species in 1974 of the 23 aquatic macrophytes documented in 1910. Four species were still common, 8 were scarce. Submersed vegetation occurred to a depth of 5 m in 1910, while in 1974 the maximum depth of plant distribution was 1 m, a five fold decrease! There was a continuous decline in the the number of *Chara aspera* beds, while *Nitella* sp. slightly increased. *Potamogeton filiformis* and f. *pectinatus* became more abundant. The turbidity of the lake strongly increased with the number of blue-green phytoplanktonic (esp. *Anabaena*) and filamentous algae *(Cladophora, Oedogonium* and *Enteromorpha*).

(Llangorse Lake, Wales) Only one submersed plant species *(Zannichella palustris)* was found in 1977 at Lake Llangorse (150 ha), whereas in 1972 still 10, and in 1964,12 species were documented. (Cragg et al. 1980). *Potamogeton crispus, P. pectinatus, Elodea canadensis, Ranunculus circinatus* and the Characeae disappeared between 1972 and 1977.

United States of America (Chesapeake Bay)
Davis (1985) reported a paleolimnological study of fossil seed assemblages in the sediment of the Upper Chesapeake Bay, an estuary with a gradient in salinity. The major changes in the aquatic macro- phytic community were related to human impacts, because immediately after the colonist settlements in 1730 the vegetation started to change in species composition. According to observations performed between 1968 and 1975 the area covered by submerged aquatic plants in the Upper Chesapeake Bay decreased by 30% per year Only two *(Myriophyllum spicatum* and *Vallisneria Americana)* of the 11 species recorded in 1971 still remained in 1978 (Orth & Moore, 1984).

For the lower (brackish) bay region Orth & Moore (1983) reported an overall decline in eel grass, *Zostera marina* in the 1930s. After this pandemic decline (also occurring in Europe the vegetation recovered within about five years. The area covered with *Z. marina* increased on the average by 1.3% per year between 1937 and 1965. A far more drastic decline started at the end of the 1960s and accelerated during the 1970s. The mean rate of decrease was 9% per year but this could go up to 21% per year at sates that became completely bare. In contrast with the 1930s this decline occurred regionally (was not pandemic) and there was no recovery.

Summary:
There is a characteristic pattern in the events preceding the complete disappearance of aquatic macrophytes:

(i) Optimal conditions for growth occur when the light penetration .is still high in mesotrophic to moderately eutrophic conditions. In this situation (phase II) the water is clear (light penetration down to the bottom or Secchi disc readings more than -: 2 metres). There is a high diversity in species composition and variation in habitat structure in stands of aquatic macrophytes. Dependent on management goals, some waters may show "nuisance growth".

(ii) In a translation phase the species diversity decreases, but some submersed aquatic species can stand high nutrient loadings and still may be a "nuisance".

(iii) Very often a rather sudden change to phase III occurs. The water becomes very turbid (Secchi reading 0.20 to 0.40 m) because the phytoplankton starts to dominate while the submersed aquatic macrophytes become virtually absent.

(iv) In many hyper- or eutrophic lakes the reed vegetation is, decreasing by a median rate of 3% year (range 0.9 to 26%).

(v) Phytoplankton, especially blue-green algae and some macroscopic filamentous algae, will predominate.

(vi) Only some types of vegetation with floating-leaved plants, especially *Nuphar lutea* and/or *lemna* spp. remain or are not affected in the same degree as the submersed and emergent are.

4. CAUSES OF THE DECLINE IN AQUATIC PLANTS

4.1. THE CAUSES OF THE DECLINE IN EMERGENT VEGETATION
(exemplified by data on reed)

The nitrogen concentration For optimum growth of reed *(Phragmites australis*) the nitrogen (NO_3^-N) concentration should be lower than 5 mg/l in the water and between 0.3 and 8.0 mg/100 9 (dry weight) in the sediment (Rodewald-Rudescu, 1974). Reed growing in litter-mats which contain 20-25 mg NO_3^-N per 100 g (dry weight) showed a reduction in sclerenchyma of the stems (Klotzli, 1971). In the Havel lakes the total dissolved nitrogen concentrations in the water in 1977 ranged between 2 and 5 mg/l. The flexibility of reed shoots increased in this area while the sclerenchyma ring in the reed stems decreased. Thus, the mechanical stability of reed stands is negatively influenced by high nitrogen concentrations (Sukopp et al. 1975; Bornkamm & Raghi-Atri, 1978). Boar & Crook (1984) found a correlation between the NO_3^-N concentration (more than 2 mg/l) of the water and the degree of deterioration of reed swamps in different Norfolk Broads. The partitioning of biomass between rhizomes and shoots changed with higher NO_3-N concentrations. The growth of above ground plant parts increased disproportionally with increasing nitrate concentration.

Filamentous algae
Increasing nutrient concentrations in the water enhances the growth of algae. Filamentous algae, especially *Cladophora*, form floating mats in which decaying (blue-green) phytoplanktonic algae are caught. These “carpets” become entangled; n the reed swamps at spec; al weather conditions. They fill up the space between bottom sediment and water surface. Wind induced wave action exercises severe stress on these carpets, so the reed will break. The remaining stems are filled with water and suffocate. Decaying stems also will become part of the mat, like all other kind of floating litter (Klotzli, 1971, 1973 & 1982; Schroder, 1979; Sukopp & Markstein, 1981).

Apart from the mechanical damage, these mats prevent light and oxygen to reach the shoots. The light is intercepted by the carpet, whereas oxygen is kept from penetrating the mat by free exchange of water. The oxygen is largely consumed in bacterial processes within the mat. So anaerobic conditions may occur between the reed stems. Schroder (1979) presented a model in which he showed that these processes lead to the release of phosphorus from the sediment; so the nutrient loading will increase. Moreover, the concentration of ammonia and hydrosulphide ions will increase in the organic substances until hydrogen sulphide gas is released in toxic concentrations (5 mg/l) that kill the rhizomes. In the Grosser Ploner See (Utermohl, 1982). Lake Constance (Schroder, 1979) and Tegelersee (Sukopp & Markstein, 1981) these *Cladophora* carpets were the main cause of the disappearance of reed.

Floating litter
The floating carpets of *Cladophora* are not the only cause of mechanical damage. In the Havel lakes without pronounced algal development, the highest decrease in reed stands (up to 27.7% per year)

Occurred at sites with a high volume of floating litter, up to more than cubic metre per metre shore line. The undamaged stands had no or less yhan average (= 0.33 cubic metre per metre; no weight mentioned) floating litter (Sukopp & Markstein, 1981). According to Klotzli (1982) more than 50 kg litter per metre is a serious threat, resulting in "reed death"on the lake front. A number of processes are summarized in his model (Fig. 5).

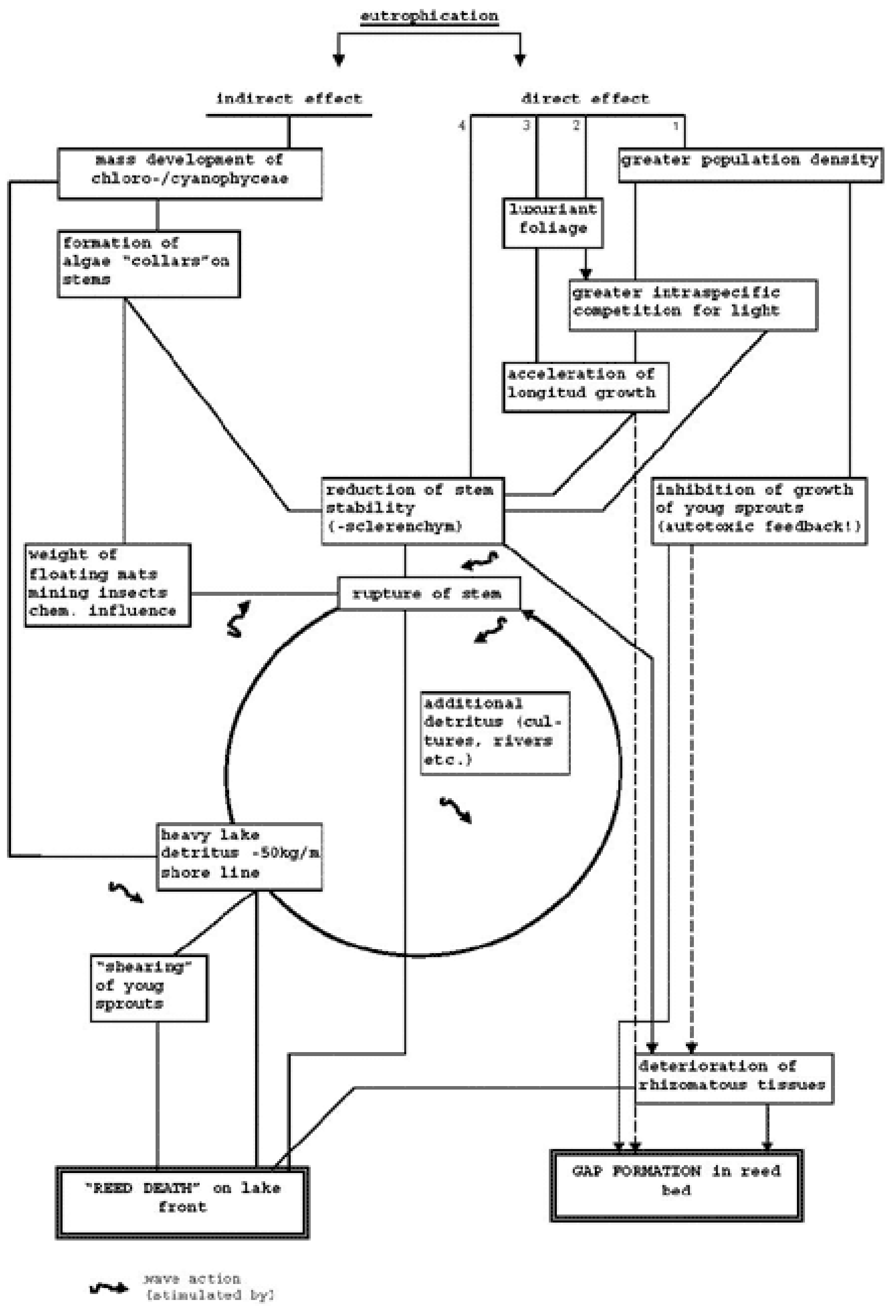

Fig. 5 The influence of eutrophication on a reed belt.
(Klotzli, 1982; with permission of the author)

Sedimentation
The terms (soft) mud, silt, bottom substrate and sediment are often interchanged. Because of the complex chemical nature of these categories, the terms are difficult to define. The term sediment is most often used; its dry matter part consists of three basic components:

1) dead organic matter;
2) particulate mineral matter, especially silicate;
3) inorganic components of biogenic origin like calcium carbonate (Wetzel, 1975).

High phytoplankton concentrations due to eutrophication will raise the amount of dead organic matter in the water and subsequently the sedimentation rate. Ripl (1984) reported a sedimentation rate of 44 tonnes per ha per year in 1982 in the hypertrophic Dummer lake (1240 ha; Niedersachsen, West-Germany). One tonne (1000 kg) sediment contained 80 kg (8%) dry matter, subdivided in 51 kg biogenic precipitated calcium carbonate, 20 kg organic material and 9 kg silicate. According to Akkermann (1978) the combination of rapid sedimentation with the occurrence of floating\ litter and high nitrate concentrations in the water, was the main cause of "reed death" in the Dummer (Table 4; Section 3.1).

Boorman & Fuller (1981) found a negative correlation between reed stem height and depth of the sediment ("soft mud") layer. The dry matter proportion of the sediment ranged between 12 and 35 %, the organic ("loss on ignition") part ranged between 13 and 48% of the dry matter. In this soft mud anaerobic conditions occurred during summer.

Disturbance and recreational activities
From the landside reed beds are threatened by the invasion of terrestrial plant species like *Epilobium hirsutum.* Land drainage, resulting in lowering the water table, will favour the introduction of these plants. The reed stands also become vulnerable after disturbance because of human recreation and tracks made by hares (*Lepus capensis*), roedeer (*Capreolus capreolus*) and pheasants (*Phasianus colchicus*) (van der Toorn et al. 1983).

In the 1960s bathing, trampling along the shore line and boating in the Havel area caused considerable damage to the reed beds. Damage by recreational boats has also been reported from Wales (Cragg et al. 1980), while Boorman & Fuller (1981) did not find any relation between decline in reeds and the quantity of pleasure boats in the Norfolk Broads.

Grazing by birds and mammals
Grazing of young reed shoots by mute swans (*Cygnus olor*), coots (*Fulica atra*) and muskrats (*Ondatra zibethicus*) badly influenced the reed beds (Sukopp & Markstein, 1981). In the Norfolk Broads the highest rate of decrease (7%) was found when the population of the exotic rodent coypus (*Myocaster coypus*) was very high. After the severe winter of 1963 the population declined 14 fold, and the rate of decrease in reed swamp surface slowed down too (Boorman & Fuller, 1981) (Table 4; Section 3.1). Akkermann (1975) studied the muskrats in the Dummer and calculated that a population of ca. 3000 individuals consumed (or damaged otherwise) 0.27 ha *Typha* spp., 0.15 ha *Phragmites australis*, 0.86 ha *Glyceria* sp. and 1.58 ha *Scirpus* sp.

The damaged area of *Scirpus* and *Phragmites* did not recover, even when the muskrats significantly decreased in number.

Young reed shoots in soft mud are easier to find by grazing animals (i.e. geese, *Anser anser* and *Branta canadensis*) than reed on more solid sediments. The birds tear off the shoots and rhizomes depriving the roots from their oxygen supply. This will eventually cause their death. The excessive amount of sediment, polluted with organic matter, made the reed stems more vulnerable to grazing. This is a consequence of the increased nutrient loading of the water body, thus the damage to reed swamps by waterfowl is in fact an indirect effect of eutrophication (Boorman & Fuller, 1981). Under normal conditions reed stands are able to sustain grazing by wildfowl.

Grazing by cows alongside water bodies is an important cause of reed death in many parts of the Netherlands (Best, 1982), and is also mentioned as a cause in the Havel lakes area (Sukopp & Markstein, 1981). This pertains to areas with intensified agricultural practise with a high density of cattle per ha. Cragg et al. (1980) argue that mild grazing does not harm the aquatic vegetation.

Summary: the emergent vegetation (mainly reed, *Phragmites australis*), is directly influenced by eutrophication. The increased nitrogen concentration adversely affects the mechanical properties of reed, making them less resistant against damage. Indirectly the reed and other emergent vegetation are threatened by carpet-forming filamentous algae. Under these conditions other activities like human recreation, grazing by cattle, other mammals and waterfowl haye a much stronger negative impact, while under normal conditions reed stands can sustain grazing by wildlife.

4.2. CAUSES OF THE DECLINE IN SUBMERSED AND FLOATING-LEAVED MACROPHYTES

4.2.1. The effect of high nutrient levels

Main factor?

Eutrophication, i.e. increasing concentrations of phosphorus (-P) and nitrogen (-N) containing dissolved compounds, is most often mentioned as the main factor of the decline in submersed macrophytes. These nutrients stimulate phytoplankton growth. Jupp & Spence (1977b), Moss (1977), Ozimek & Kowalczewki (1984) and Toivonen (1985) called the increased light attenuation due to water turbidity (phytoplankton blooms, high concentrations of particulate dead organic or mineral matter) an important factor of the decline in submersed macrophytes. Shading by trees, recommended as a management technique to control nuisance growth (Dawson & Kern-Hansen, 1979), is also reported as a cause of the decline in submersed macrophytes in a small lake (Best, 1982). Jupp & Spence (1977a) observed a relationship between P- concentration and the decline in the dominant submersed macrophyte *Potamogeton filiformis.* Moss & Leah (1982) and Best et al. (1984) stressed that raised P-concentrations are not immediately followed by the decline in aquatic macrophytes.

Normally the nutrient loading increases gradually or even remains constant while the decline in aquatic macrophytes is often rapid (Fig. 6).

In general the largest decrease is shown by the submersed aquatic macrophytes while floating-leaved vegetation types with *Nuphar lutea* (Jeschke & Muther, 1978; Cragg et al. 1980; Moss & Leah. 1982; Ozimek & Kowalczewki. 1984) or *Lemna minor* (Toivonen. 1985) decreased to a lesser extend, remained constant or even expanded. Whenever the leaves of these plants reach the water surface, they will receive full light, and the growth is no longer hampered by water turbidity.

4.2.2. The role of epiphytes

If the turbidity due to phytoplankton abundance is the main cause of the decline, one ought to find a clear relation between the turbidity (the degree of light attenuation) and the presence of submersed macrophytes. Phillips et al. (1978) did not find such a simple relation. The measured extension of the zone with net photosynthesis did not correlate with the distribution range of submersed macrophytes in different Norfolk Broads. In artificially fertilized experimental tanks the epiphytic and filamentous algae increased more rapidly than the aquatic macrophytes. In the field a higher biomass of epiphytic algae per dry weight of aquatic plant was found on sites with higher concentrations of total P and N in the sediment. The species diversity of the epiphytic community was lower on these enriched sites (Eminson. 1978). Thus, a high nutrient loading of the water in the Norfolk Broads favoured in the first place the amount of biomass (not the number of species) in epiphytic algae (Fig. 6).

Epiphytic algae may form several layers which hamper the irradiance of the macrophytic leaves and the exchange of inorganic carbon (HCO_3- and CO_2). Consequently, the growth of the macrophyte becomes slower or even stops (Sand-Jensen, 1977; Sand-Jensen & Sondergaard, 1981). Under non-eutrophic circumstances the macrophyte may release allelo-pathic substances to suppress the growth of phytoplankton (Wium-Andersen et al., 1982). or to attract selectively epiphytes (Carpenter & Lodge, 1986) and grazing macro-invertebrates (Orth & van Montfrans, 1984; Bronmark, 1985a; Lodge, 1985). Many of these relations are discovered in laboratory conditions; their ecological instance the ability of *Chara* to suppress phytoplankton growth in fish ponds is well established (Crawford, 1979).

Hence, increased nutrient loading favours few epiphytic species. The epiphytic species diversity decreases, while the biomass increases (Eminson, 1978). The diversity in grazing species like snails decreases. Probably the remaining few epiphytic species are less attractive food for many herbivorous invertebrates (Den Hartog, pers comm.). Maybe the eutrophication disturbs complex but subtle mechanisms (by means of released substances) to suppress the phytoplankton growth and to attract invertebrate grazers, which once existed between the diverse species of epiphytes, macrophytes and grazing invertebrates.

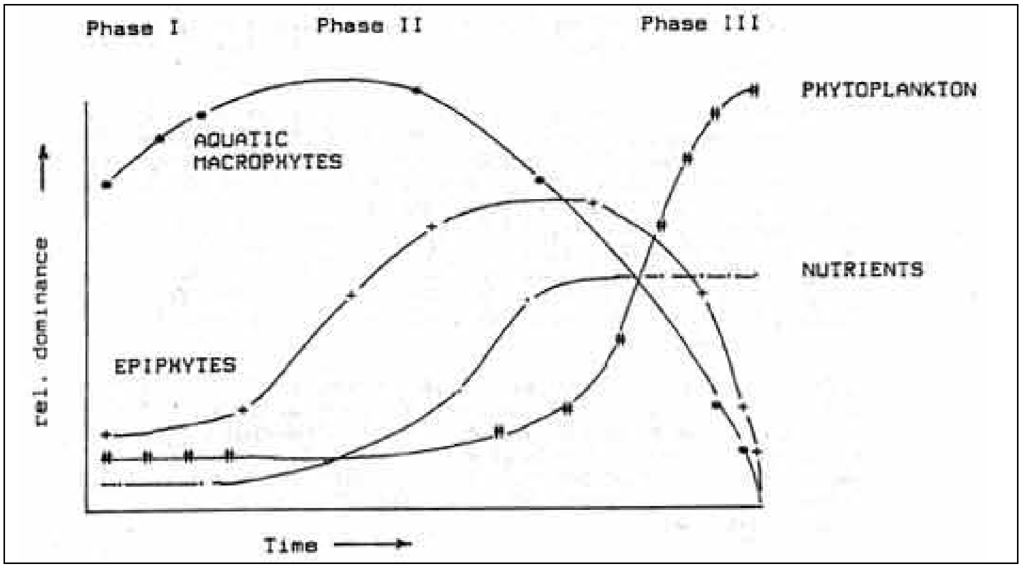

Fig. 6. The relative dominance of the amount of biomass of submersed macrophytes, phytoplankton and epiphytes and the availability of nutrients in an aquatic ecosystem with gradually increasing nutrient levels (Phase I to III as described by Moss et al, 1985).

4.2.3. Factors stabilizing the situation without aquatic plants

A decrease in nutrient loading of the water is probably the panacea to re-establish clear water with abundant aquatic macrophytes and subsequently waters rich in fish. Moss & Leah (1982) warn their readers not to be overoptimistic: after a decrease in the nutrient loading there are several processes maintaining the deteriorated situation.

Blue-green algae may remain if large-bodied cladoceran zooplankters stay away because of intensive predation by small (young-of-the-year) fish (see Section 2.5.4.). The slow-growing, colony-forming, blue- green algae like *Aphanothece* are more difficult to handle by the small remaining zooplanktonic species. In the absence of large-bodied cladocerans this alga may predominate even at lower than preferred nutrient concentrations because it has a low death rate, a low sinking rate and it is able to use the nutrients more efficiently by internal recycling (Moss & Leah, 1982). Excretion of allelopathic substances by blue-green algae that suppress the growth of aquatic macrophytes is proved in the laboratory (van Vierssen & Prins, 1985) and may help to preserve the situation with predominance of phytoplankton.

Reduction of zooplankton eating fish in an eutrophicated lake may reduce the symptoms of eutrophication (Hrbacek et al. 1961; Andersson et al. 1978; Leah et al. 1980; Reinertsen & Olsen, 1985; Braband et al., 1986). In that case the phytoplankton will be grazed more effectively by the larger (mainly cladoceran) zooplankters and the water will become clear. However, if the fish population has considerably decreased, invertebrates (e.g. mysids) may occupy part of the niche of the young fish and prey on the cladocerans. The introduction of *Mysis relicta* will significantly decrease zooplankton densities (Langeland, 1981; among others) and therefore alter the trophic state of the water body.

Moss et al. (1986) reported the reactivation of PO_4-P from the sediment after reduction of the nutrient loading of the water in a small lake in the Norfolk Broads. At first the water became clear and submersed water plants successfully recolonized the lake within four years. Six years after the reduction, activated phosphorus from the sediment caused a comeback of the phytoplankton and the growth of aquatic plants became negligible.

In the Norfolk Broads large flocks of birds bring guano (nutrients) in some lakes (Section 3.2) (Moss & Leah, 1982).

Especially in soft (non-alkaline) waters acid precipitation causes dramatic changes in the species composition and biomass of aquatic macrophytes and fish populations (Hendrey, 1982; Roelofs, 1983; Roelofs et al., 1984). These phenomena have been excluded from the scope of this review because most attention has been paid to productive, hard, nutrient-rich waters. However, the increased nitrogenous deposition in some areas also may influence these waters and will help to maintain undesirably high blue-green algae concentrations.

4.2.4. Other factors responsible for the decline

i. **Wave action** The submersed macrophytes rooted in the sediment become more and more restricted to the shallowest parts of the water, where they can receive enough light for growth. They also become more vulnerable to wave action at these shallow parts. From shallow, wind- exposed sites the nutrients will be flushed out. Direct mechanical destruction and hampered growth because of nutrient-poor sediments were the causes of the deterioration of the whole Potamogeton vegetation in Loch Leven (Jupp & Spence, 1977b).

ii. **Grazing by birds. mammals and fish** Mute swans *(Cygnus olor*)and coots (*Fulica atra*) in the Havel lakes (Sukopp & Markstein, 1981) and the Mazurian and Northbrandenburgian lakes (Ozimek & Kowalczewki, 1984; Jeschke & Muther, 1978), whooper swans (*Cygnus cygnus*), geese and ducks in Loch Leven and the Norfolk Broads (Jupp & Spence, 1977b; Boorman & Fuller, 1981) intensively grazed on submerged and floating-leaved macrophytes.

These animals also contributed to the decrease in l. aquatic macrophytes. However, Kiorboe (1980) proved that the effect of grazing by waterfowl on the abundance of submersed macrophytes in spring and summer is small, although the birds consume quantities up to 60% of the annual plant biomass production. Because grazing by waterfowl takes place in autumn and winter, when the aquatic plant population already produced its survivals organs (diaspores), the macrophyte populations are not significantly damaged.

Because the decrease of submersed aquatic vegetation is multicausal, the grazing by waterfowl may become harmful when these birds have neither decreased at all, nor lessened their population in the proportion to the amount of aquatic macrophytes. Their grazing pressure in relation to the remaining amounts of aquatic plants is stronger.

Van der Velde et al. (1982) reported grazing and damage to *Nymphoides peltata* stands by coots (*Fulica atra*), but also by snails, insects, muskrats (*Ondatra zibethicus*) and cattle (mainly *Bos taurus*). Animals were responsible for the disappearance of 22% of the total leaf area produced during the growing season, being the combined effect of consumption and microbial decay after damage.

Grazing by carp (*Cyprinus carpio*) and other cyprinids, often stocked and managed to improve fisheries, can contribute to the decline of *Chara* weedbeds (Jeschke & Muther, 1978; ten Winkel & Meulemans, 1984).

iii. **Effects of pleasure-boats** Turbidity is also caused by pleasure- boat traffic. However, Hilton & Phillips (1982) did not find a long-term build-up turbidity in the River Ant as a consequence of frequent pleasure-boat traffic. Like Moss (1977), they state that turbidity is mainly caused by phytoplankton. Yousef et al. (1980) found evidence for increased phosphorus content due to water mixing by recreational motorboats in Florida. In all cases the direct mechanical damage to macrophytes by motorboats is evident. Strong correlations have been found between the intensity of boat traffic and the decrease in submersed and floating-leaved macrophytes, especially *Potamogeton natans* and *Nuphar lutea* (Cragg et al. 1980; Jeschke & Muther, 1978; Murphy & Eaton, 1983). Increased pleasure-boat traffic is often accompanied by the construction of jetties and ports which have a considerable effect on the reed vegetation and may have caused the rapid decline in *Hydrocharis morsus-ranae* dominated (aquatic) vegetation in the Northbrandenburgian lakes (Jeschke & Muther, 1978).

4.3. A GENERAL HYPOTHESIS

All processes leading to the decline in submersed aquatic macrophytes can be summarized in a model (Fig. 6 & 7). This model is mainly based on the ideas of Phillips et al. (1978) and Moss et al. (1985), with modifications added by van Vierssen et al. (1985a).

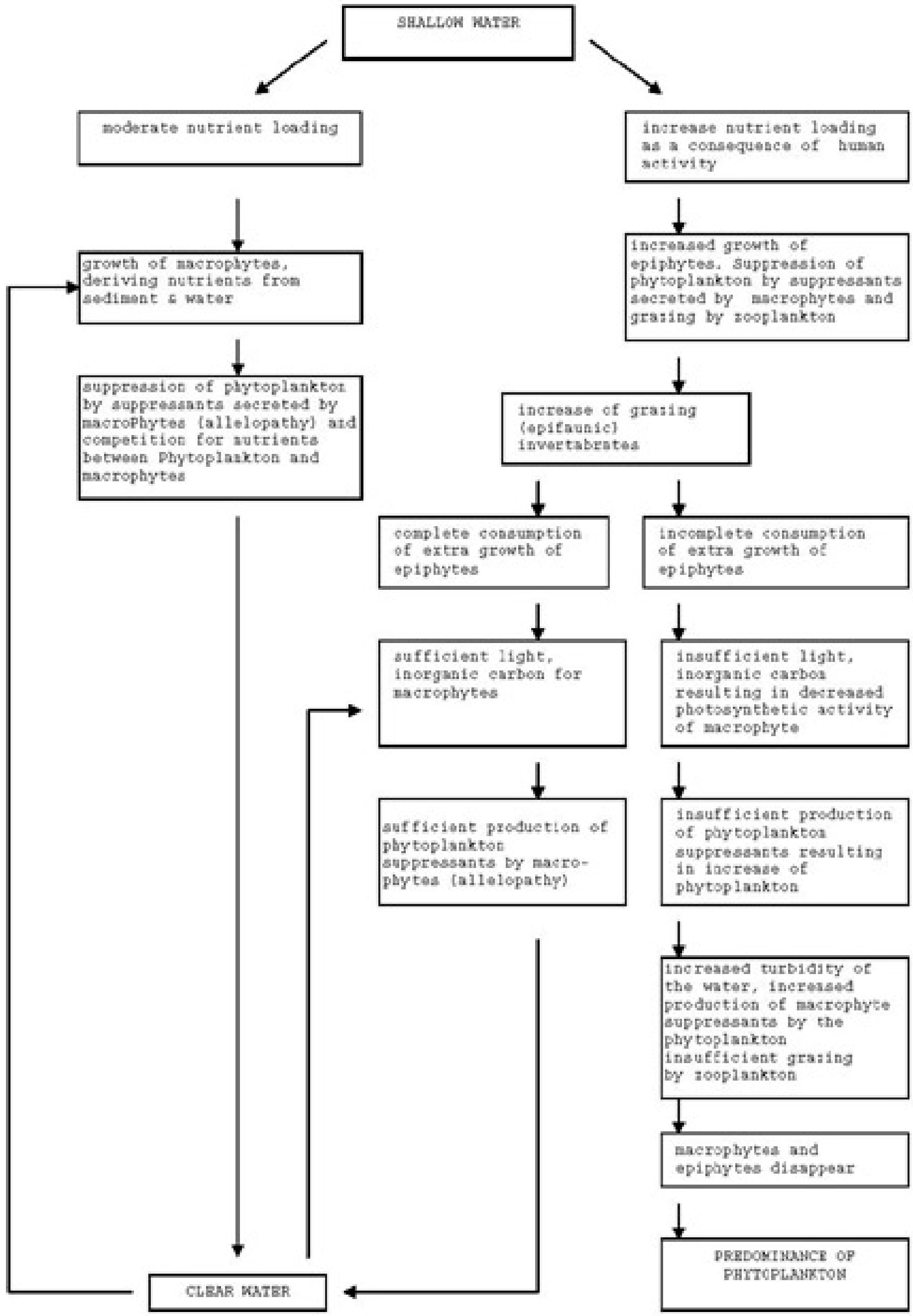

Fig. 7 Relationships within different component parts of the shallow water ecosystem at increasing nutrient loading (after Phillips et al., 1978; modified by Van Vierssen)

Increased inputs of nutrients will cause initially an increase in aquatic macrophytes (Phase II in Fig. 6). Later on the epiphytic algae start to increase. If the macrophytes are successful in the production of diaspores, they will build up sufficient biomass in the next season. Thus a dynamic balance between macrophytes and epiphytes exists in the mildly eutrophicated situation. The phytoplankton growth may be inhibited by the excretion of suppressants. In mildly eutrophic situations the macrophytes can also compete successfully with phytoplankton for nutrients (Goulder, 1969, van Vierssen et al. 1985a) and light (Fig. 7).

When nutrient levels increase the total biomass of epiphytes increases, hence there is more food available to the grazer population, which will increase too. Subsequently there is enough food for fish, substrate (=aquatic macrophytes) to deposit their eggs and sufficient shelter for fish larvae and fry (Phase II in Fig. 6). These eutrophicated water bodies are more productive because the extra input of nutrients will cause an energy flow to the adult (commercially important or game) fish through the aquatic macrophytes, epiphytes or phytoplankton, via herbivorous invertebrates and small fish. This positive relationship between trophic state and fish productivity is well-established (Grosch, 1980, Hoyer et al., 1985).

As a consequence of increased abundance of the few epiphytic species, the macrophytes become gradually hampered in their growth if the consumption of the extra growth of epiphytes becomes insufficient. As long as they produce enough diaspores (survival organs), the aquatic macrophytes will appear the next season. When the production of diaspores diminishes, fewer plants will start to grow the following year. The few aquatic macrophytes (which are then strongly overgrown with epiphytic algae) no longer produce suppressants to inhibit phytoplanktonic growth. More phytoplankton will be produced at the end of Phase II consequently the light does not reach the young plants. A catastrophic decrease of submersed aquatic macrophytes will occur, while the phytoplankton predominates (Fig. 6 & 7). Some macrophytes, especially species which will rapidly grow to the water surface, like *P. pectinatus* endure this situation for a relatively long time. However, sometimes within one season the whole aquatic vegetation disappears (Phase III in Fig. 6). et al. (1985a)

Summary:

(i) At moderate nutrient levels the submersed aquatic macrophytes can compete successfully for inorganic carbon and light with phytoplankton and epiphytes. The water is clear.

(ii) The grazing of epiphytes by macro fauna, the grazing of phytoplankton by zooplankton and the excretion of inhibiting or attracting compounds by aquatic macrophytes are of importance for the survival of the aquatic vegetation.

(iii) Because of (ii) the predation of fish on zooplankton and macro-fauna, and subsequently also the predation of piscivorous fish on small fish, indirectly affects the growth of the aquatic vegetation.

(iv) At higher nutrient levels the epiphytes increase. Incomplete consumption of the extra growth causes hampered growth of the submersed macrophytes and insufficient diaspore production. The survival of the macrophyte in the next season will become uncertain.

(v) At about the same concentration level of nutrients. the macrophytes disappear. while the phytoplankton start to predominate. The light inhibition caused by the phytoplankton and probably the excretion of inhibiting compounds by blue-green algae. prevent the resettlement of submersed plants.

(vi) The decline in aquatic macrophytes is a multicausal process. not only caused by an increased concentration of NO_3-N or PO_4-P. Therefore growth of macrophytes will be possible at different levels of nutrient concentrations.

5. MANAGEMENT. CONTROL AND RESTORATION MEASURES

> “Where shallow, previously aquatic plant-dominated lakes are to be restored, (...) the problems may be likened to those of restoring a tropical forest after clear-felling ...” (Moss & Leah, 1982)

Eutrophication, i.e. the increased inputs of PO4-P and NO3-N is the main cause of the deterioration of the aquatic habitat. Therefore, the lowering of the nutrient levels seems to be the panacea for the problem. For reed (*Phragmites australis*) a direct influence of high dissolved NO_3-N concentration on the condition was established (see Section 4.1.). However, the growth of submersed aquatic macrophytes appeared to be possible at quite different levels of nutrient concentrations. While under moderately high eutrophicated conditions even “nuisance” growth may become a problem under certain management regimes (water transport, irrigation, angling, swimming purposes).

The need of aquatic vegetation for good water quality and as a habitat for fish populations raises two questions:

i. if water plants are present: how do we keep them and how do we control them to maintain optimal water quality?
ii. if the water plants have disappeared. how do we restore the aquatic habitat with aquatic vegetation?

The following Section will describe a number of control measures for submersed water plants within the framework of the conceptual model, as given in Section 4. For reed and other emergent aquatic vegetation; this model is not applicable. The most important for the conservation of reed are the control of nutrient regime (especially the nitrate concentration), development of fence systems and replanting.

5.1. THE EMERGENT VEGETATION

Fencing

In the Havel lakes fences are constructed since 1971 to prevent losses in the reed stands alongside the shoreline (Markstein & Sukopp, 1983). These measures are very expensive. Depending on the construction (logs, palisades, wooden boxes) the prices range between 110 and 340 DM per metre. It proved not to be effective, since the rate of decline of the proportion of the shoreline covered with reed stands did not decrease (Fig. 8). The construction of special swimming sites did not prevent further losses either. Experiments are going on with the underwater construction of wattle fences for protection. Klotzli (1971) also reports the use of wattle fences composed of willow twigs or plastic laths and plastic nets to keep floating litter less than 15-30 kg per metre shore line.

Recently Grosch (per. comm.) reported the end of the decline in emergent vegetation alongside the Havel lakes. The total costs of a number of restoration measurements (including dephosphatization) amounted 4000 DM per ha within lo years.

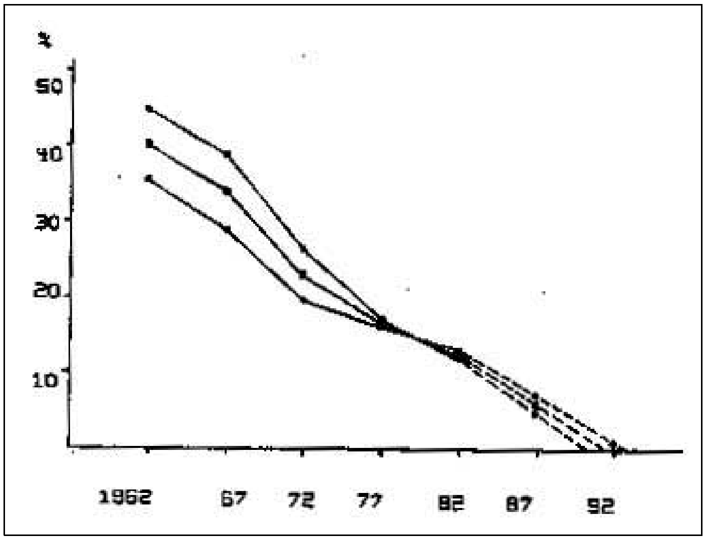

Fig. 8. Reed stands along the Berlin Havel (1962-'82) o: Proportion of the shoreline covered in the Upper Havel lakes; .: Proportion of the shoreline covered in the Lower Havel lakes; D: Both; --: Extrapolation (Markstein & Sukopp. 1983. with permission of the authors).

Intensive agricultural practise results in more cattle per ha and an early start of the grazing. In these cases. fences are needed to keep the cattle out of the zone of emergent aquatic plants.

Removal of old reed stems by mowing, burning and "raking"

There are no general applicable management techniques to stimulate reed growth which are effective in all situations. The effect depends on several factors:

(i) the probability of ground frost in spring;
(ii) the degree of infestation by herbivorous insects;
(iii) the fluctuations in water level at the reed stands. The degree of exposure to wind and wave action is important too. Reed bordering a deep lake has to be managed differently compared to reed stands in a swampy lowland area with shallow lakes.

Burning during winter or early spring positively influences the bud, and shoot formation and growth of the shoots. The bud development : determines the shoot growth and also the density later in the season.
Burning in springtime will increase the shoot density, but decreases the shoot biomass because the damaged shoots are replaced by more shoots from other. but smaller buds (Haslam. 1971).

The removal of litter causes more rapid growth, but makes the stand .more vulnerable to ground-frost and the invasion of other emergent plants especially at non-flooded sites (Mook & van der Toorn. 1982; van der Toorn & Mook, 1982). The larvae of stem- and rhizome-boring insects also can cause considerable damage. In case of the latter burning is an effective measure, provided that it is done every winter and not incidentally (van der Toorn et al. 1983).

In the Netherlands a machine, especially designed for reed management along the shores of canals, is successful. After mowing the reed, the other vegetation and sol between the remaining reed stems and rhizomes is scraped away by a 1.5 m wide mechanical rake. This rake is mounted on a movable machine, which can be tended from the landside of the reed bed. The effect is a lowering of the ground level and breaking the rhizomes in the reed beds. In the next season, in the "raked" reed beds the stem diameter and height of reed increased more than after mowing or burning. This treatment is repeated after 5, sometimes 7 or 8 years (Baart & Ross, 1984).
However, "raking" is risky at shores with weakened reed stands. Klotzli (1971, 1973) advised not to mow the 3-4 m wide zone, bordering the open water of deep lakes. Weakened stands and stands endangered by algal mats also should not be mowed (for the peri-alpine lakes see also Schroder. 1979).

Cultivation of reed swamps for the removal of nutrients
Reed can remove considerable amounts of phosphorus and nitrogen due .to the activity of bacteria in the rhizosphere of the reed bed. In the Dummer (shallow, hypertrophic lake; open water area 1240 ha, West Germany) a reed swamp area (200 ha) has been constructed in the mouth .of the river Hunte to “trap” the nutrients from this eutrophicated stream. About 90% of the annual sedimentation and 5 -10 tonnes phosphorus will be trapped in this swamp (Ripl, 1984). However, harvesting during winter time of the above ground parts of reed may reduce the annual p and N loading of the water in a Dutch hypertrophy lake by 1%, which is very small in comparison with the cost (Loenen & Koridon, 1978).

Replanting
In the Havel area the planting of *Typha angustifolia* rhizomes behind palisades, the fencing of remaining reed and protected plantings of *Acorus calamus*, *Carex acuta* and *T. angustifolia* was successful. The replanting of *Phragmites* and *Scirpus lacustris* was problematic (Markstein & Sukopp, 1983). In the Netherlands a combination of "hard" structures by stones and loam and plantations of reed sods are successfully applied alongside lakes and canals (Acht & Sessink, 1982). These plantings are used to protect the shore because reed and several other emergent species are able to subdue the wave action: 2- 3 m wide stands of *Phragmites* but also of *A. calamus* or *angustifolia* can absorb 60-75% of the wave energy (Bonham, 1980). However, reed can resist only a limited amount of mechanical stress. Klotzli (1973) indicated 20 m as the minimum distance for passing motorboats with a maximum speed of 5 km/h.

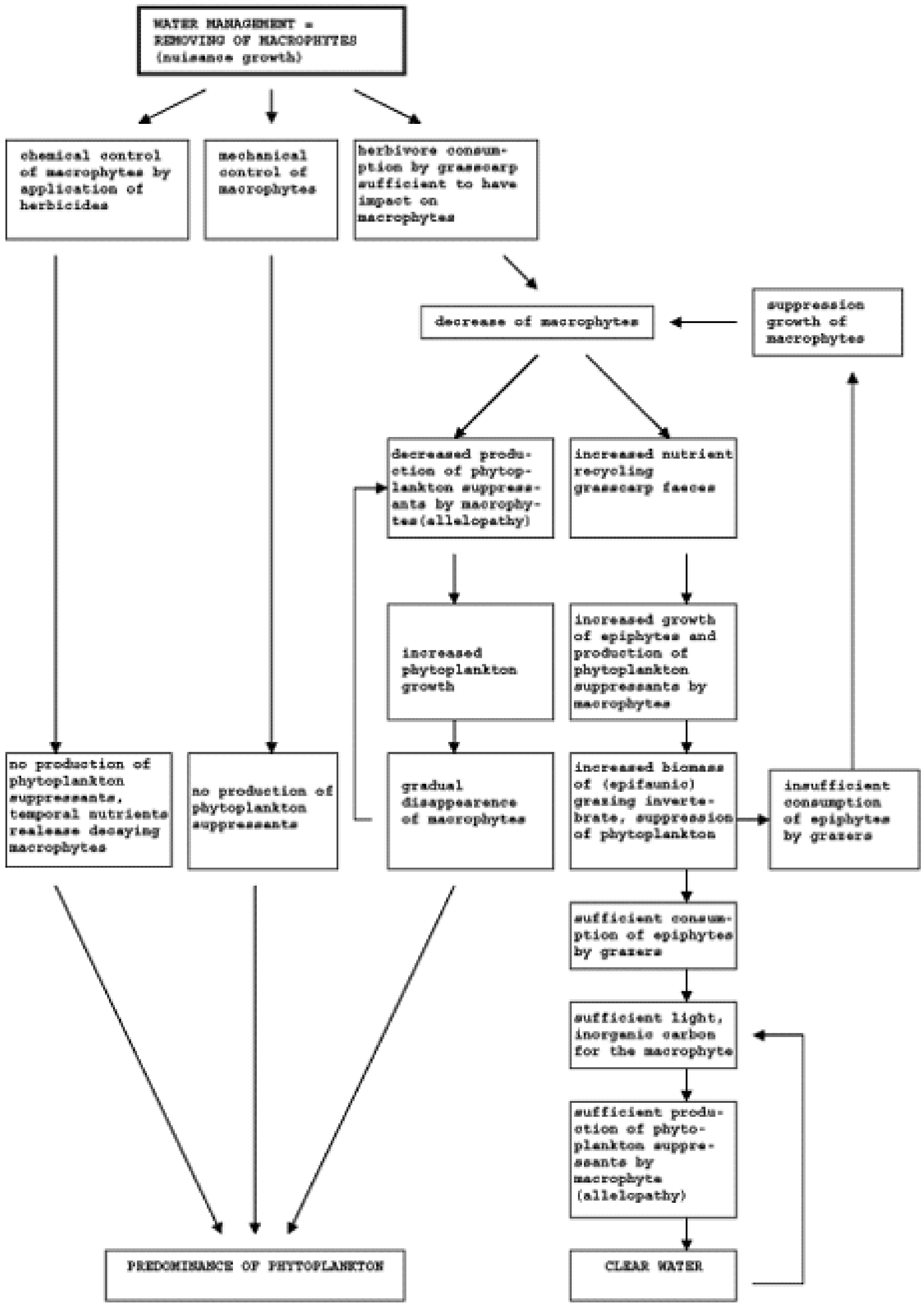

Fig. 9 The effects on the submersed aquatic macrophytes of three control measurements, predicted within the frame work of the conceptual model of Phillips et al. 1978. (Van Vierssen, 1985a)

5.2. THE SUBMERSED VEGETATION .

5.2.1. Control

The effects of three common management techniques used to control aquatic weeds are discussed within the framework of the conceptual model (see Section 4.3. and Fig. 9.).

(i) **Mechanical and chemical control**. Mechanical control of waterweeds is an expensive method. It may also stimulate the predominance of *Elodea* sp. and *Potamogeton pectinatus* and thus impoverish the aquatic macrophytic community. Chemical control is 50% cheaper (van Zon, 1977), but the side-effects are worse. Effective removal of all aquatic macrophytes may lead to phytoplankton dominance (Fig. 9). The contribution to the rapid decline of aquatic macrophytes by large-scale application of chemicals is discussed by Nicholson (1981), Johnstone (1982) and Jones & Winchell (1984). The implications of the application of herbicides are extensively reviewed by Brooker & Edwards (1975) and Hellawell & Bryan (1982), of whom the latter also give recommendations for use. Within the conceptual model these control measurements are risky because one will probably end up in a phase III situation (Fig. 6, 7 and 9).

(ii) **Biological control by grass carp (*Ctenopharyngdon idella* Val.).** Stocking with grass carp is often recommended as an acceptable solution (Fig. 9).This method is cheap as it represents 50% of the cost for chemical control and 20-30% of the cost inherent to mechanical control (van Zon. 1977). The grass carp can have a beneficial effect upon the diversity of the aquatic vegetation if stocked at densities under 250 kg/ha. Small grass carps do not eat parts of emergent plants. An increase in *Hottonia palustris* has been reported after 2 years in an experiment with 250 kg/ha grass carp in fish ponds (Provoost et al., 1984). Rough-leaved plants like *Stratiotes aloides*, floating-leaved plants (*Nuphar lutea* and *Nymphea* spp.) and plants with a strong taste like *Polygonum hydropiper* and *Ranunculus* spp., are not consumed by fingerlings (van Zon, 1977). Stocking over a period of three years in a small fishpond diminished the coverage of the vegetation and increased the number of macrophytic species from 9 to 20 (Provoost et al. 1984). Ahling & Jernelov (1971) reported the return of *chara* and improved dissolved oxygen concentrations in a small Swedish lake, after stocking with grass carp. Better growth, production and survival of other fishes have been repeatedly reported (van Zon, 1977; Pierce, 1983; Provoost et al, 1984). However, the presence of grazing macro-invertebrates on the remaining aquatic macrophytes is extraordinarily important to maintain clear water. If the grass carp stocking (>250 kg/ha) results in complete removal of the aquatic vegetation the herbivorous macro-invertebrates will also disappear. Bottom dwelling detritivores like chironomids may compensate the feeding conditions for other fish species (van der Zweerde, 1982. 1983; Provoost et al., 1984), but then the pathway to phytoplankton dominated waters (Phase III) becomes inevitable (Fig. 9).

Krzywosz et al., (1980) carefully documented negative effects of a long-lasting, large stocking project in Dgal Wielki (95 ha, Mazurian Lakes, Poland). In 1966 about 6 kg/ha fingerlings were stocked in the lake. More and bigger fish (up to 6 kg per individual) were added between 1970 and 1977. The average grass carp biomass over 1966 to 1978 in the open water zone was 84 kg/ha. Between 1966 and 1974 the macrophytic production remained on the same level and no essential quantitative changes were observed until the warm summer of 1976. In that year the vegetation decreased 9 to 17 fold. *Chara* sp., *Potamogeton natans*, *P. pectinatus*, *P. perfoliatus*, *Stratiotes aloides* and *Elodea canadensis* disappeared. Only *Fontinalis antipyretica*, *Nuphar lutea* and small traces of *Myriophyllum* sp. remained. The avifauna was pushed out because of a reduction of the reed swamps by 66%. The native rudd (*Scardinius erythrophthalmus*) and tench (*Tinca tinca*) considerably decreased in number, while the condition of bream (*Abramis brama*) became poor. The weight increase of the grass carp also became lower than under natural conditions in the Amur. It is unclear whether other factors (like the decreased water volume in the lake during the warm summer of 1976) influenced the decrease of aquatic macrophytes, as was reported for other Mazurian lakes (Ozimek & Kowalczewski, 1984).

The direct effect of moderate stocking with grass carp and other non- native phytophageous fish on phytoplankton remains unclear (Provoost et al., 1984). Heavy stocking causes turbidity because of disturbance of the bottom substrate and faeces excretion, which will enhance the growth of bluegreen algae (Januszko, 1974). Heavy stocking (more than 250 kg/ha) of the common carp (*Cyprinus carpio*) also affects the vegetation (Crivelli, 1983; ten Winkel & Meulemans, 1984).

(iii) **Shading.** Dawson & Kern-Hansen (1979) give many helpful suggestions for weed management in streams. They plead for moderate shading of the stream water surface by careful management of the (emergent) bank vegetation. In the absence of such vegetation they recommend shading of small streams by an artificial canopy made of plastic mesh or blind cloth. However, shading by fast growing willow trees around a small lake may cause a strong decrease in submersed aquatic macrophytes (Best, 1982).

Summary: Control measures against “nuisance” growth are risky because an irreversible situation without any aquatic vegetation may be created. Beneficial effects of grass carp stocking can be expected with stocking densities below 250 kg/ha and strictly maintained regulations and management. Artificial shading also is a mean to control nuisance growth. If conservation interests are involved, these methods must be considered very carefully.

5.2.2. Restoration

Replanting

Artificial recolonization of aquatic plants by replanting will often be interfered by waterfowl grazing. Therefore, young plantations should be protected by nettings (Moss & Leah, 1982); although in the normal situation healthy plant populations can sustain high grazing rates (Kiorboe, 1980). Crawford (1979, 1981) successfully inoculated *Chara* oogonia in artificial fish ponds. The water quality significantly improved in these ponds.

Furtherance of grazing on phytoplankton

Turbidity, caused by abundant phytoplankton growth, may be lessened by stimulating the grazing of zooplankton (see Section 2.5.4.). Timms J & Moss (1984) found during summer in one of the Norfolk Broads t rapidly growing small phytoplanktonic algae in sparse densities (<10 µg/l chlorophyll-a) and “normal” densities of fish. They showed that, the period of clear water coincided with the presence of a large stand of *Nuphar lutea* and plant-associated large-bodied cladocerans. The nutrient level of the water was very high (186 µg/l total-P and 6.5 mg/l total-N) and could support great phytoplankton growth and did so in spring and autumn. Thus, aquatic macrophytes may be important to maintain a good water quality in coexistence with fish at high nutrient concentrations. In the Norfolk Broads experiments are going on with bundles of alder tree (*Alnus glutinosa*) twigs. These submerged bundles harboured (in 1986) good populations of large cladocerans (Moss, pers. comm.).

The manipulation of young fish populations is an acknowledged management tool to decrease the predation on zooplanktonic grazers and thus reducing the phytoplankton. Ripl (1984) mentioned the stocking of 1-2 year old pike (*Esox lucens*) to reduce the abundant but badly growing population of zooplanktivorous roach (*Rutilus rutilus*) and bream (*Abramis brama*). This measure is part of the lake restoration project in the Dummer. The Dutch Organisation of Improvement of the Inland Fisheries (OVB) is trying to control small planktivorous fish by stocking them with artificially propagated pikeperch (*Stizostedion lucioperca*), pike (*Esox lucius*) and wels (*Silurus glanis*).

Manipulating the water level

Below 30 µE $.cm^{-2}$ $.sec^{-1}$ (i.e. photosynthetically active radiation in micro-Einstein per square cm per sec) no growth of aquatic macrophytes is possible, whereas between 30 and 400 µE $.cm^{-2}$ $.sec^{-1}$ the production of aquatic macrophytes is mainly influenced by the light factor (van Vierssen, 1985b). If the radiation on the bottom sediment is slightly above 30 µE $.cm^{-2}$ $.sec^{-1}$ the light penetration of shallow waters can be improved in spring by temporary lowering the water level. If there is sufficient light, the seedlings of aquatic macrophytes can grow more quickly during springtime. Hence the final seasonal product; on increases considerably, while the zone extends in which the macrophytes can develop (van Vierssen pers. comm.)

Sediment removal

Sediment removal is sometimes practised in small lakes:

1. to prevent sediment/water exchange of nutrients;
2. to remove (or reduce) nuisance plant growth;
3. to reduce the effect of toxicants; 4. to deepen the water body for fishing, boating and other recreational activities.

Deepening a lake, without increasing the transparency in the water column after the execution of the work, and without lowering the water level, will worsen the light conditions near the bottom and hamper the growth of submersed aquatic macrophytes. Therefore it is a risky management tool to reduce nuisance growth according to the conceptual model, because one may create a phase III situation.

Peterson (1982) reviewed 60 projects of sediment removal and five case histories. He does not give explicit recommendations for sediment removal to control plant growth because conclusive information on macrophytic regrowth is lacking. Sediment removal projects fail if only a limited part of the polluted sediment is removed and/or external loadings of nutrients and pollutants can not be controlled. The cost may range between 1 to 16 US$ per cubic metre; an average unit cost can not be calculated because the determining variables are numerous. Moss et al. (1986) estimated the cost of removing the sediment from a 63 ha large lake in the Norfolk Broads at least 1.5 million UK£ (3.4 US$ per square metre of the lake surface). Peterson (1982) stressed that every project should be preceded by a thorough study on the mass balance of nutrients and/or pollutants in the sediments and the water column.

In the Dummer lake (1240 ha) 800,000 tonnes of sediment (cost 7 US$ per cubic metre) have to be removed. To prevent the discharge of nutrient rich water from the intensively cultured surrounding arable land, the lake has been isolated from a part of the watershed (Anon., 1984). The total cost of sediment removal and other restoration measures is 10 million DM (0.4 US$ per square metre of the lake) and annual management cost is 400,000 DM (0.16 US$ per square metre).

A restoration project in Cockshoot Broad (3.3 ha; Norfolk Broads) started in 1982 by removing the surface sediments by pumps and isolating the lake from the River Bure (cost 80,000 UK£, 3.4 US$ per square metre). As a result the total phosphorus concentration decreased to 50-60 µg/l and the phytoplankton concentration to less than 15 µg chlorophyll-a/l within 2 years, while a diverse collection of aquatic macrophytes is developing. Another Broad (Alderfen) has also been isolated, but no sediment has been removed. A macrophytic community characteristic for phase II was re-established, and this mobilized much phosphorus from the sediments. After four years a large phytoplankton population appeared.

Dephosphatization of effluents

In Barton Broad (Norfolk Broads) the phytoplankton concentration was decreased by 50% after dephosphatization of effluents to the River Ant, discharging this lake. The total phosphorus concentration decreased from 350 µg/l to the designated 100 µg/l, but no recolonization of aquatic macrophytes occurred (Moss et al. 1985).

In the Dutch Loosdrecht Lakes the total phosphorus concentration ranged until 1984 between 60- 520 µg/l. A large restoration project then started. The discharge of water from the polluted River Vecht has been diverted. The lakes are flushed with dephosphatized water from the less polluted Amsterdam-Rhine Canal since 1984. The phosphorus absorption and release from the sediments and the inherent effects on the biota are to be studied (van Liere 1984, 1985; Best et al., 1984)

Landscaping
Many man-made lakes and ponds only serve for recreational purposes like windsurfing. swimming and catching sport fish. To meet these demands, usually uniformly shaped waters with uninteresting, deep water zones and rather steep profiles are made. Larger, well protected, bank zones of near-natural forms are necessary to create hatching areas for fish and amphibians and refugia for a greater variety of emergent plant species, macro-invertebrates, fish and amphibians. Akkermann (1985) presented a number of design possibilities and examples to improve existing objects or to construct new ones after digging sand and gravel.

Limiting recreational activities
Since 1969 a conservation law protects the aquatic ecosystem of the Havel Lakes area. This law prohibits treading the shore and the penetration of the reed stands by boats and sets a speed limit for motor boats (Sukopp & Kunick. 1969). In some peri-alpine lakes only a fixed number of motorboats are allowed (Klotzki 1973). Special spawning areas, where boats and other activities are strictly forbidden, have been created in the Havel lakes for protection of the fish population (Grosch. 1978).

In the Cruising and Remainder canals (England) more than 4000 mhy (boat movements per ha per year) damaged the submersed vegetation to an unacceptable extent for angling purposes. It was proposed to divert the boats from heavily used canals to small, hardly used canals. It was expected that boat traffic between 2000 and 4000 mhy will regulate the amount of vegetation within limits acceptable for sport fishing activities (Murphy & Eaton. 1981). The protection of the reed and other emergent vegetation against the negative effects of pleasure boat traffic is also important for the conservation of the submersed vegetation. Klotzli (1973) fixed 20 m as the minimum distance for passing motorboats with a maximum speed of 5 km/h. These findings may be useful elsewhere to assess maximum limits to human recreational activities where protection of the aquatic vegetation is necessary.

6. CONCLUSIONS

Changes in the species composition and biomass of aquatic macrophytes in lakes and streams have dramatic impact on the physio-chemical and biological conditions. They strongly influence the plant-associated invertebrate fauna. The aquatic macrophytes have a central position in the network of ecological relations between nutrients, plankton and macro-invertebrates. Therefore, they also determine the carrying capacity for the population of fish species like pike, most cyprinids and eel. The negative effects of eutrophication on the fishery conditions are mediated through these aquatic plants. Research directed towards quantifying the effect of habitat change on fisheries should be aware of the direct and indirect relations between fish and aquatic plants.

Research on fisheries often is carried out independently and it is rarely fitted within the framework of limnological studies covering also the lower trophic levels (phytoplankton, epiphytes, plants and macro-invertebrates). Research aimed at improving habitats for fish should link up with ecological research engaged in "ecosystem restoration".

Nowadays there is a certain consensus about the models of interaction between the invertebrate grazers, macrophytes, phytoplankton and nutrients derived from isolated process studies and small scale experiments. In cases where long-term base line studies are available on larger water bodies, the impact of these mainly hypothetic interactions between fish, invertebrates and plants have to be tested on the ecosystem level by whole lake manipulations. These ecosystem experiments should be designed carefully and ought to include meaningful control or reference objects (Likens, 1985).

Removing large parts of the fish population may improve water quality. Although for the short term this type of biomanipulation appears of no importance for fisheries, these experiments may result in the long term in the restoration of fish habitats. Stocking with piscivorous fish and other means to manipulate the young-of-the-year fish populations should be implemented in these large-scale ecosystem experiments. Besides these large-scale ecosystem experiments, the painstaking study of relations between fish, grazers and macrophytes is important.

This research should be carried, out by teams of scientists investigating different trophic levels (fish, macro-invertebrates, aquatic plants, plankton and their physico-chemical environment). Because restoration will serve the multiple purpose of water bodies, these teams should be financially supported by the different organizations (recreational and commercial fishery, Water Boards and other governmental organizations) involved in water management. International cooperation is needed, because problems in the different industrialized countries have much in common. The exchange of know-how by means of small workshops is probably the most rapid and effective way to steer this research.

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