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POTENTIAL COAL MINE WASTEWATER TREATMENT OPTIONS

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ABSTRACT

Surface coal mine operations have been shown to release significant amounts of nitrogen to receiving waters. This loading of nitrogen to the aquatic environment could have detrimental effects on the aquatic environment. Current technologies to treat nitrogen enriched waters are not applicable to coal mines. These considerations have led to the evaluation of potential alternative treatment techniques involving the use of aquatic plants and upland irrigation.

The literature shows that aquatic plants have been widely used to treat municipal wastewaters and have had some recent use in the treatment of industrial applicable wastewaters. The technique has been shown to be applicable in temperate climates. Aquatic plants have the capability to remove nutrients and a wide variety of other chemicals from water, including heavy metals and organic chemicals. They also assist in the flocculation and settling of solids. The literature review indicated that aquatic plants have the potential to remove elevated nitrogen from wastewaters and could be useful in treating coal mine wastewaters in Canada.

Upland irrigation literature is related almost entirely to the treatment of municipal wastewater with limited reference to industrial applications. The technique was shown to be useful for the treatment of municipal wastewaters and is widely used for that purpose.

Depending on conditions, nitrogen can be quite mobile in the soils. Application rates will also depend upon soil conditions. Upland irrigation with coal mine wastewater remains a potential treatment method but will depend on the specific site situation.

Experimental work on duckweed, a floating aquatic plant, was carried out. It was found that duckweed was able to reduce nitrogen concentrations in coal mine wastewaters from 12 mg N/l to <1 mg N/l within ten days. Nitrogen removal efficiencies were as high as 98.6% in static tests and 91.2% in flow-through tests. Uncropped populations of duckweed were more efficient in removing nitrogen from the wastewater. It was also found that tissue nutrient (nitrogen and phosphorus) concentrations decreased with time but decreased more rapidly in the cropped populations of the plants.

RESUME

Il a été dèmontré que les mines de charbon à ciel ouvert libèrent des quantités significatives d'azote dans les cours d'eaux récepteurs. Cet apport d'azote peut causer des effets négatifs sur l'environnement aquatique. Les techniques existantes pour le traitement des eaux riches en azote ne peuvent être appliquées aux mines de charbon. Ces constatations ont donné lieu à une évaluation des méthodes de traitement utilisant les plantes aquatiques et l'épandage.

La littérature rapporte que les plantes àquatiques on été largement utilisées pour le traitement des effluents municipaux. Elles ont aussi servi récomment pour le traitement des eaux résiduaires industrielles. Celle technologie s'est avérée applicable dans les climats tempérés. Les plantes aquatiques sont capables d'enlever de l'eau les éléments nutritifs ainsi qu'une grande variété de substances, incluant métaux lourds et substances organiques. Elles favorisent aussi la flocculation et la sédimentation des solides. Un examen de la littérature a montré que les plantes aquatiques peuvent enlever des quantités élevées d'azote des eaux usées et pouraient servir au traitement des eaux résiduaires des mines de charbon au Canada.

La littérature sur l'épandage traite presqu'entiérement des eaux usées municipales et ne fait référence aux applications industrielles que de façon limitée. Cette technique s'est avérée utile dans le traitement des

eaux municipales et on l'a largement utilisée à cet effet. Sous certaines conditions, l'azote peut être mobile dans les sols. Le taux d'épandage varie également selon les conditions du sol. L'épandage des eaux des mines de charbon demeure une méthode possible de traitement, mais son application dépend des conditions spécifiques à la situation.

Des travaux expérimentaux ont été effectués sur des plantes aquatiques flottantes (Spirodela, Lemna). On a observé que ces plantes étaient capables de réduire les concentrations d'azote dans les eaux des mines de charbon, de 12 mg N/L à moins de 1 mg N/L, et ce en moins de 10 jours. L'efficacité d'enlèvement de l'azote fut de 98.6% dans les essais statiques et de 91.2% dans les essais en continu. Les populations non-récoltées de plantes flottantes furent plus efficaces pour enlever l'azote de l'eau usée. On a aussi trouvé que la concentration en éléments nutritifs (azote et phosphore) des tissus diminuait avec le temps mais qu'elle diminuait plus rapidement dans les populations où on faisait une récolte périodique des plantes.

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1 Duckweed Experiment Data

1.0 INTRODUCTION

1.1 Background

The Canadian coal mining industry as well as other types of mining (e.g. base metal and uranium) are faced with resolving the release of large amounts of nutrients, especially nitrogen, from mining/milling operations to receiving water environments. These nutrients when added to the aquatic environment have been shown to cause nitrate, nitrite and ammonia to exceed Canadian drinking water standards in some receiving waters and, by excessive algal growth, could cause significant degradation of aquatic environments. Each of these problems have not necessarily occurred at all operating surface mines and depends on the local environment and mining conditions. The consideration that elevated nitrogen could have detrimental effects on the local aquatic environment have led to delays and difficulties in obtaining environmental approval for developing mines as well as additional monitoring and permit requirements for existing operations.

A series of studies were conducted in British Columbia at an operating surface coal mine and information from several other mining operations were reviewed by the B.C. Ministry of Environment (Pommen, 1983). Findings of these studies were that large quantities of nitrogen (1 to 6% of nitrogen in annual explosive consumption) entered receiving waters from the mine site. Most of the nitrogen was derived from nitrogen based explosives and high levels of inorganic nitrogen were found in the spoil (waste rock) materials. Most of the nitrogen leaving the mine site was in the form of nitrate. However, mining operations were found to cause each of the three forms of nitrogen (nitrate, nitrite and ammonia) to exceed the maximum acceptable concentration drinking water limits in receiving waters during some parts of the year. It was also felt that wet mining conditions caused a greater nitrogen loss from the mine site than would occur under dry conditions, even though nitrogen loss from drier mines was also considered as a problem.

Aquatic systems over broad geographic areas are thought to exhibit nitrogen limitation and could experience significant detrimental environmental effects caused by excessive algal growth with the addition of large quantities of nitrogen. Examples of aquatic systems that could be affected are coastal systems, high mountain systems, oligotrophic lakes and mesotrophic and eutrophic systems; acute toxicity of ammonia to aquatic organisms is also of concern.

There is currently no recognized cost effective method of treating large amounts of water to remove nitrogen even though there are a number of possible treatment techniques (Pommen, 1983). Mines may generate water volumes requiring treatment ranging from 1,500 m^3/d to far in excess of 5,000 m^3/d. The cost of constructing and operating a treatment plant using the available methods would be excessive. None of the more conventional engineered treatment techniques have been employed in actual operating treatment plants for coal mine wastewater.

Several mitigative measures have been suggested to reduce the amount of nitrogen released as a result of mining activities. These are related to handling of blasting agents (dewatering the blasting area, short loading to blast time, spill control and good house-keeping) and preventing water contact with waste rock (diversion of surface water away from waste rock and covering the rock with material of lower permeability as the waste rock dump is developed). No tests have been carried out to determine the effectiveness or practicality of these measures although changes in handling practices at one surface coal mine are thought to have reduced nitrogen losses. The institution of these measures where feasible will probably reduce nitrogen losses but thus far high nitrogen losses from operating coal mines have not been eliminated.

Two possible new approaches to removing nitrogen and phosphorus from coal mine wastewater involve the employment of biological techniques which have proven capabilities in treating sewage and other industrial effluents. They also have the potential to remove other deleterious components of wastewaters, including base metals and radionuclides. The two techniques are the use of aquatic vascular plants and the disposal of excess water on vegetated land by irrigation. These two methods could be put in place for use, at least, during the critical spring and summer growing season at many mine locations in Canada. Further examinations of these two potential methods were undertaken in this study by reviewing the available literature and conducting bench scale tests using aquatic plants to treat coal mine wastewater.

1.2 Objectives

The objectives of the work were to evaluate the potential for the use of aquatic plants and/or irrigation techniques to treat coal mine wastewaters, particularly with regard to the removal of inorganic nitrogen. The specific study objectives were to:

- conduct a literature review to determine the feasibility of using aquatic plants and irrigation techniques to treat coal mine wastewater for the removal of nitrogen and other potential contaminants, and

- carry out a bench scale test with an aquatic plant (*Lemna minor* and *Spirodela polyrhiza*) to determine its effectiveness in removing nitrogen from coal mine wastewater.

2.0 WASTEWATER TREATMENT CONCEPTS

The loss of nutrients, particularly nitrogen, from blasting materials used at Canadian mines has emerged as a significant environmental concern. The concern over the loss of this material is with regard to the possibility of causing deterioration in receiving water environments. The addition of nitrogen to receiving waters can result in nitrogen compounds exceeding drinking water standards and algae growth becoming excessive thereby causing taste and odor problems in drinking water, reduced aesthetic values and/or reducing the capacity of waterbodies to support fish and other aquatic organisms if dissolved oxygen depletion occurs. In addition, elevated ammonia levels in receiving waters are of concern with respect to toxicity to aquatic organisms.

The methods being evaluated in this report are the use of aquatic vascular plants and land irrigation. These techniques were evolved primarily for treatment of wastewaters during the warmer periods when aquatic plants are most active and when soils can accept additional water. This period corresponds to the usual warm, longer day-length growing season. Normally drier periods are coincident with the growing season thus volumes of surface runoff and thus wastewater to be treated are less. The treatment of wastewater during these drier growing periods may be accomplished without the need for large facilities and in an economic manner by utilizing aquatic plant treatment facilities or upland irrigation techniques. This would prevent these contaminants from entering nearby natural surface water courses and causing environmental degradation, especially during the critical biologically active periods. As materials other than nutrients are also known to be removed in such biological processes, these techniques may prove useful for removal of contaminants other than nutrients.

In most systems employing aquatic plants, the plant growth is harvested regularly or at the end of the growing season. Harvesting is the process of cutting and removing from the water a substantial part of the plant growth that has occurred during the growing period. This removal of bound nutrients and contaminants from the aquatic system prevents their re-entry to the system and the plant material would be useful in mine reclamation. Plant systems have proven to be useful as a mulch/compost material in the removal of nutrients and organic and inorganic contaminants from other wastewaters and may prove suitable for the treatment of coal mine wastewaters.

Application of wastewater to upland areas by irrigation is a widely used method for the treatment of municipal sewage. These systems rely on the biological and chemical activity within the soil to remove nutrients and other substances from the water as it passes through the soil profile. The same principles apply in the treatment of coal mine wastewater except that municipal wastewaters are most often applied to agricultural crops whereas mine wastewaters would most likely be applied to ecosystems near a mine; usually natural ecosystems although these waters could also be useful in mine reclamation.

3.0 AQUATIC MACROPHYTES

The use of aquatic vegetation in the treatment of diverse wastewaters is becoming widely recognized as a realistic alternative to traditional treatment methods (Boyd, 1963; National Academy of Sciences, 1976; Environmental Protection Service, 1979; Tourbier and Pierson, 1976; Stephenson et al. 1980). The plants contribute to treatment through both the direct uptake of pollutants and through indirect contaminant removal. The plant biomass has a variety of realized and potential uses, including the production of fiber (Rudescu, 1976), animal feed (Burton et al. 1977; Wolverton and McDonald, 1978; Hillman and Culley, 1978; Limpkin and Plucknett, 1980), fertilizer (Limpkin and Plucknett, 1980; Edwards, 1980), fuel gas (Wolverton and McDonald, 1971; Chynoweth et al. 1982), and human food (Hillman and Culley, 1978; Bhantumnavin and McGarry, 1971). Most of the published information centers around species from tropical and sub-tropical regions, where climatic factors favour high-rate biological productivity. A considerable amount of information is emerging more recently, however, from temperate latitudes (Environmental Protection Service, 1979; Tourbier and Pierson, 1976; Hammer and Kadlec, 1983). While the goal of waste management is recognized to be the integration of environmental protection with resource regeneration, the scope of the present review is limited to the use and management of aquatic plants for wastewater treatment purposes. (For more detailed information on macrophyte re-use, the reader is referred to the pertinent publications cited above.)

Macrophytes, or vascular aquatic plants, are commonly categorized according to whether they are free-floating or rooted in the bottom soil, and if rooted, whether most of the plant grows above the water surface (emergent) or underwater (submergent) (Hutchinson, 1975). Intermediate forms exist, of course, but the above definitions of floating, emergent and submerged will be used throughout this discussion. As shall be reviewed below, the different plant forms have different implications relative to their use in wastewater treatment. The identification of form-specific environmental constraints can aid in the selection of one form or species over another for any given wastewater type or treatment objective.

The present review attempts to cover both municipal and industrial wastewaters. Within the literature surveyed, there is a predominance of information on the operative mechanisms, performance and, to a lesser extent, design and operation, of macrophyte systems treating municipal effluents. A great deal of this information is probably transferrable to many industrial wastewaters, with slight modification.

Floating macrophytes used in wastewater treatment include the water hyacinth (<u>Eichornia crassipes</u>), the duckweeds (a common name for several species: <u>Lemna</u> spp., <u>Spirodela</u> spp., <u>Wolffia</u> spp. and <u>Wolffiella</u> spp.), and the water ferns (<u>Azolla</u> spp. and <u>Salvinia</u> spp.). Emergent forms include the cattails or bulrushes (Typha spp.), reeds (Phragmites spp.), rushes (Scirpus spp.) and a variety of sedges and grasses. Submergent species of mention in the present context include the pondweeds (Potamogeton spp.), coontails (Ceratophyllum spp.), elodea (Elodea spp.) and others. Duckweed, Azolla, and all the above emergent and submergent species are found in temperate climates. Many additional species with promising characteristics will likely be identified through continuing research.

3.1 Wastewater Treatment Principle

It is evident from the literature that, overall, the treatment of a wastewater in a system containing macrophytes is brought about by both direct and indirect, biotic and abiotic effects. The following discussion is based largely on the excellent publications from the California Water Resources Control Board (Stephenson et al. 1980), the U.S. National Academy of Sciences (National Academy of Sciences, 1976), the U.S. Environmental Protection Agency (Environmental Protection Service, 1979; Hammer and Kadlec, 1983), and the Czechoslovakian International Biological Program Wetland Project (Dykyjova and Kvet, 1978).

3.1.1 Macrophyte ecology in relation to wastewater

The processes by which aquatic plants contribute to the decontamination of wastewaters, while not yet fully explained, can best be understood in terms of the ecology of the macrophytes in relation to the desired results of treatment. Solar energy, carbon dioxide (CO_2), oxygen (O_2), water and nutrient minerals are the basic requirements for plant growth. Macrophyte growth will be affected by wastewater characteristics which interfere with the availability of any of these basic elements. In this context, important wastewater characteristics include temperature, turbidity, rate and seasonality of flow, and chemical composition. Environmental parameters not related to the wastewater, such as illumination, climate and weather also influence the performance of macrophyte-based treatment systems. There is considerable variablility, among species, in the optimum conditions for growth. Table 3-1 summarizes some of the environmental requirements of aquatic plants that have been used in wastewater treatment.

Solar energy, aside from the seasonal cycles which affect all plants, is only limiting to submerged macrophytes (and phytoplankton) due to shading by water depth, snow-covered ice, floating vegetation or turbidity. Emergent and floating species, while affected by shade, are not limited by the transparency of the water except, in the case of emergent species, when turbidity is accompanied by an excessive increase in water depth.

Wastewater temperature is of concern when it is either too hot or too cold for plant growth or for specific metabolic (enzyme) activities involved in the treatment process (Mathis et al. 1979). While the thermal optima vary between species, plant growth can take place over the range of approximately 4-35^{o}C (Stephenson et al.

TABLE 3-1

SUMMARY OF ENVIRONMENTAL REQUIREMENTS OF SELECTED MACROPHYTE SPECIES

SPECIES AND COMMON NAME	PARAMETER				
	Temperature (°C)	Illumination (units as reported)	Salinity ‰	pH	Water Depth (m)
<u>FLOATING</u>:					
Duckweeds:					
<u>Lemna</u> sp.	n/a - 30	300 - 600	0.7-3.25	5 - 7	-
<u>Spirodela</u> sp.	n/a - 37	600 - 1200			-
<u>Wolffia</u> sp.	n/a - 30	(uE/m^2·s)			-
<u>EMERGENT</u>					
Cattails:					
<u>Typha</u> sp.	9 - 31	n/a	0 - 26	4.6 - 10	+0.2 to +1.0
Rushes:					
<u>Scirpus</u> sp.	16 - 26	n/a	7 - 24	4.9 - 8	-0.1 to +0.1
Reeds:					
<u>Phragmites</u> sp.	11 - 32	n/a	0 - 40	2 - 8.5	-0.3 to +1.5
Sedges:					
<u>Carex</u>	n/a	9 - 42 w/m^2	n/a	n/a	n/a
<u>SUBMERGENT</u>					
Pondweed:					
<u>Potamogeton</u> sp.	12 - 33	>5 langley/d >1076 lux	0 - 19	6.3 - 10	1 to 7
Coontail:					
<u>Ceratophyllum</u> sp.	6 - 25	optimum: 16000 lux	0 - 9	7.1 - 8.8	1 to 4
Water milfoil					
<u>Myriophyllum</u> sp.	1 - 42	n/a	0 - 15	5 - 10	1 to 3
Elodea:					
<u>Elodea</u>	4 - 33	optimum: 18000 - 38000 lux	0 - 31.4	6.4 - 10	1 to 12

n/a = not available

1980). Floating and emergent macrophytes tend to have an insulating effect on water temperature, reducing the rates of daytime heat uptake or nighttime heat loss. Temperature control (eg. heated effluents, greenhouse-type covers) can enhance the ability of macrophytes to remove many pollutants from the water (Serfling and Alsten, 1979; Reed and Bouzon, 1981). Seasonal variations in both ambient and water temperatures will affect system performance. Freezing temperatures cause a virtual cessation of biological activity, while extreme summertime heat can also disrupt plant metabolism (Stowell et al. 1980). However, in temperate latitudes, heat stress to aquatic plants is unlikely outdoors, though possible indoors (Whitehead, unpublished). The effects of temperature and light are apparently interlinked, such that an increase in one will affect the plant's tolerance of the other (Porath and Ben-Shaul, 1973).

All plants assimilate carbon (as CO_2 or bicarbonate) and evolve O_2 during photosynthesis; the reverse occurs in the absence of sufficient light. Submerged macrophytes obtain these vital gases from the surrounding water, where their availability may not be constant. For example, if the water receives wastes having a high biochemical oxygen demand (BOD), or through nutrient enrichment develops a dense algal bloom, nocturnal availability of oxygen may not meet the submergent plant's needs. Submergent species are unable to survive in anaerobic waters of polluted lakes (Ozimek, 1978) and wastewater lagoons (McNabb, 1976). Similarly, the pH can affect the solubility of CO_2 and, hence interfere with the daytime rate of carbon fixation under water (Cole, 1979), and consequently the relative abundance of plant species (Shapiro, 1973). Most submerged species can utilize bicarbonate ions as a carbon source; however, certain species can only use dissolved CO_2 (eg. Myriophyllum spicatum) (Hutchinson, 1970). Most floating and all emergent species obtain these gases directly from the atmosphere, where the supply is generally not limiting. Mats of floating vegetation reduce the diffusion rate of O_2 from the atmosphere into the water (Morris and Barker, 1977). Similarly, masses of decomposing vegetation consume dissolved oxygen (DO) and can cause anaerobic conditions (Jewell, 1971). Air channels in the stems of many rooted species facilitate gaseous exchange between the bottom soil and the atmosphere (National Academy of Sciences, 1976; Stephenson et al. 1980) or water (McNabb, 1976). Aerobic micro-organisms in the rhizosphere are essential to the nutrient cycling processes in soils. The presence or absence of DO can drastically alter the flux of dissolved minerals between the water and the hydrosoil (Patrick and Khalid, 1974; Kessel, 1978), which can in turn affect treatment.

There is evidence that buffering of highly acidic or alkaline wastewaters (National Academy of Sciences, 1976; Edwards, 1980) is often associated with macrophyte systems (Seidel, 1976). The mechanisms involved are not yet understood. Hargreaves et al. (1975) report finding the emergents Typha latifolia and Juncus effusus growing in acid drainage waters at pH 2.5-2.9 in England. The latter species was found in flowing water only, while the former occurred in both still and flowing water.

Removal of dissolved contaminants (solutes) by plants involves both adsorption and passive or active absorption (Hutchinson, 1975; Salisbury and Ross, 1978). The assimilation of nutrients and other substances by macrophytes takes place through both the roots and other submerged plant surfaces (Cole, 1979; Hutchinson, 1975; Mayes et al. 1977). In submergent, and in rootless or sparsely rooted floating species the leaves and stems are often primarily involved in sorption (Hillman, 1961). Adsorption appears to involve ion exchange at the cell surface, and can exhibit saturation (Gale and Wixson, 1978; Snyder and Aharrah, 1984; Hammer and Kadlec, 1983). Adsorption of phosphorus only takes place under aerobic conditions (Boyd, 1970; Patrick and Khalid, 1974).

The uptake of macro-nutrients (N,P,K) can be limited by their relative concentrations and, in some instances, by the availability of micronutrients such as iron and manganese. The availability of Fe reportedly limits nitrate uptake by water hyacinth growing in wastewater (Reddy, 1983; Lee, 1980). Hutchinson (1957, in Mutzar et al. 1978) indicates that the growth of aquatic plants may be limited by P when the ratio of total-N to total-P is below 9:1, and by N when the N/P ratio is higher. Similarly, Walrath and Natter (1976) suggest that an N/P ratio of 10:1 to 15:1 is optimum for the uptake of both N and P. Carpenter and Adams (1977), in a study on N and P uptake by water millfoil, report that decaying macrophytes tend to release P at a faster rate than N.

There is a considerable body of literature on the uptake of heavy metals by macrophytes. Campbell et al. (1985) studied copper and zinc accumulation by the floating (rooted) yellow water lily, Nuphar variegatum, and reported that Zn in the plant tissue originated primarily from the sediment, while Cu originated in the water. In the same study, Fe was inferred to play a role in regulating Cu bio-availability. Gellini and Barbolani (1981) reported on copper uptake by Eichornia crassipes, Salvinia natans, Lemna minor and Iris pseudacorus, from synthetic and industrial wastewaters containing up to 5 mg Cu/L. In the presence of Eichornia and Salvinia, Cu concentrations in the water were reduced to 0.49 and 0.69 mg/L in 48 hours. The four test species all survived well in the solutions.

Mayes et al. (1977) studied the uptake of cadmium and lead by the submergent Elodea canadensis, and experimentally determined that both the water and sediments were sources of the metals accumulated in the plant tissue. (Information on the distribution of metals, nutrients and other contaminants within the aquatic plants were not always identified in the literature, however a more detailed assessment of the partitioning of these materials is included in a report now in preparation by Norecol.) The results also suggested that metals could be released from the plants into uncontaminated water. Plant Pb content increased from 2.9 to 160.9 parts per million (ppm) over a six week exposure to contaminated water and sediment containing 9-45 ug Pb/L and 392-487 ug Pb/L, respectively. Tissue Cd content rose from 0.27 to 32.3 ppm, after exposure for a similar period, to water and sediment containing 0.5-6.0 ug Cd/L and 88.4-125.3 ug Cd/L, respectively. The Pb and Cd in this study originated from an electroplating industry. Wolverton and McDonald (1978) reported that water hyacinth grown experimentally on water contaminated with 10 ppm Pb and 1 ppm Cd and Hg was able to absorb 65% of the Pb, 50% of the Cd and 65% of the Hg within one hour. No interactive effects between the metals were observed.

Seidel (1976) studied the absorption of 14 elements by ten species of aquatic plants, including cattail and other emergents, in Germany. The report provides the data in units of mg/kg of biomass dry weight and in mg/m^2 of growth area. However, the conditions under which the data were generated were not described. Rodgers et al. (1978) studied the cycling of 21 elements in a coal ash drainage system, and collected data on the concentrations in the water, sediment and water. The duckweed concentrated all the elements studied, by 5 times (Cs) to 2825 times (Mn) the concentration in the water. Bioconcentration factors of less than 30 times were found as well for Zn, Cu, Cd, Hg, Br, and Se, while factors of greater than 300 times were also found for Al, K, Sr, and Mg. Fe was concentrated by 34-145 times. Duckweed was reported to completely dominate the contaminated wetland.

Wolverton and McKown (1976) reported that water hyacinth grown on water experimentally contaminated with up to 100 ppm of phenol could reduce the concentration of this organic chemical to 0.4-2.8 ppm within 72 hours. No phenol could be recovered from the plant tissue, leading the authors to conclude that the phenol was metabolized to other compounds or partly evapotranspired. Reduction of phenol concentrations in a variety of wastewaters have been reported by Karaseva and Papchekov (1974), Seidel (1976; and in Wolverton and McKown, 1976) with *Scirpus lacustris*, and by Vasigov et al. (1976) with *Phragmites* and *Typha*. Seidel (1976) reported that other phenolics such as chlorophenol, pentachlorophenol, as well as xylol, pyrocatechol, and cyanide could be eliminated in the presence of emergent vegetation. Kordakov (1971) found that cyanide and Cu concentrations in effluents from a metal refinery in the Soviet Union decreased after passing through ponds containing cattails and reeds. The study also reported that decomposition products of macrophytes accelerated the removal of cyanides, Cu, and Zn; maximum removal of these contaminants occurred during maximum leaching of easily dissolved organic substances from the decaying plant tissue (details of the mechanisms involved were not available from the abstract reviewed). Yakubowskii et al. (1971) studied the biochemical removal of lead cyanides from lead smelting wastewaters in the presence of living and dead aquatic plants. Cyanides were found to be removed more actively when the wastewater was exposed to growing plants.

There is generally a direct, sometimes linear relationship between the concentration of any particular solute (whether an essential nutrient or other substance) in the water and in the plant tissues, up to a threshold ambient concentration. Above this critical level, there is little further increase in the plant tissue. Culley et al. (1981)

have provided data for duckweed which suggest that the threshold ambient concentration of nitrogen is in the range of 4-8 mg N/L and of phosphorus in the range of 3-4 mg P/L. McNabb (1976) has reported that the tissue N content of submerged macrophytes was unaffected by ambient inorganic-N concentrations above approximately 1 mg/L. In the case of P the same author has reported that the linear regression equation relating the seasonal mean percent P of dry organic weight and the seasonal mean of ambient soluble P for Ceratophyllum demersum (up to 3 mg/L ambient soluble P) was found to apply as well for Elodea canadensis and E. nutalli. Explicit reports on threshold concentrations for other solutes or plant species have not been located. Many reports dealing with the composition of aquatic plant tissue neglect to report the composition of the water in which the plants were growing. All macrophytes appear to be able to accumulate nutrients, heavy metals and complex organic molecules to higher than ambient levels, often by orders of magnitude (Rodgers et al. 1978; McNabb, 1976; Seidel, 1976 Guthrie and Cherry, 1979; Mudroch and Capobianco, 1979 a,b.). There is mounting evidence that the microbial communities (and their consumers) growing on the submerged surfaces of aquatic plants, or on any other suitable substrate, may play a more important role than the plants in the removal of nutrients and other substances from the water (Serfling and Alsten, 1979; Stowell et al. 1980; de Jong, 1976). The importance of these biofilm communities as an integral part of the treatment potential of aquatic plants must be emphasized. Table 3-2 shows the percent contribution of the epiphytic layer to the total mineral content of two species of

TABLE 3-2

PERCENT CONTRIBUTION BY SURFACE BIOFILM AND MARL TO MACROPHYTE MINERAL CONTENT

Mineral*	*Potamogeton* sp. (%)	*Myriophyllum* sp. (%)
Ash	65.2	15.8
Silica, soluble	- **	6.1
Silica, insoluble	13.8	82.3
Ca	79.1	20.8
Mg	19.4	-
Na	19.1	-
K	-	8.0
Fe	52.2	36.4
Cu	-	6.5
Mn	31.7	0
Zn	-	-
Cl	34.5	10.0
SO_4	8.9	7.8
P	-	-

* Value shown indicates percentage of the mineral content in the plant sample which can be removed by washing the plant surface.

** Indicates that the epiphytic layer had a diluting effect on the mineral concentration of the plant tissue.

Source: Mutzar et al. 1978.

submergent macrophytes. The data were determined by comparing the composition of washed and unwashed plant samples (Mutzar et al. 1978). High concentrations of calcium and magnesium are evidence of marl precipitation on the plant surfaces. Up to 75 percent of the nitrogen cycled through an experimental aquatic ecosystem is reported to take place via the biofilm (periphyton) / detritus / detritivore pathway (Serfling, 1976, in Serfling and Alsten, 1979).

Physical sedimentation of suspended matter can be enhanced by the submerged surfaces of aquatic plants. The plant mass absorbs energy, reducing turbulence and flow velocity, allowing suspended particulates to precipitate. Similarly, adsorption and flocculation have a higher probability of occurring as the ratio of submerged surface area to water volume is increased (Metcalf and Eddy, 1979). This is particularly true in the case of the water hyacinth, which has an extensive root system, and may be true for submergent and emergent macrophytes, since the same principle would appear to be operative. Conversely, the less developed root system of duckweeds contributes little to sedimentation. There is evidence however, that the presence of certain aquatic plants can favour the removal of coliform bacteria and microalgae by zooplankton (Ehrlich, 1966; Dinges, 1974; Dinges, 1982).

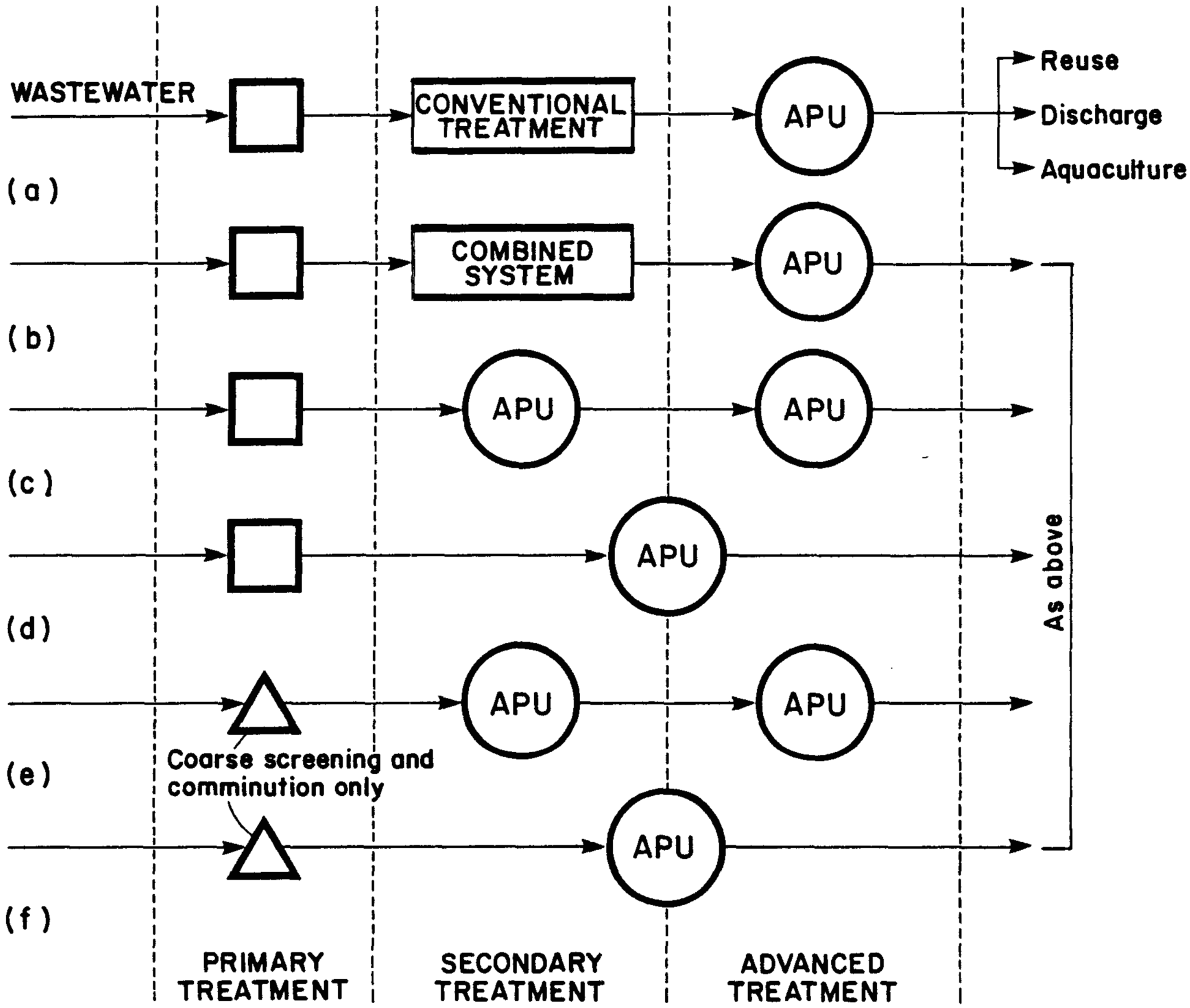
Macrophytes which are rooted in the bottom soil are, within limits, the least affected by wave action, current or wind (Husak and Hejny, 1978; Dykyjova and Ulelhova, 1978). Stands of emergent Scirpus and Typha are often found growing along the exposed shorelines of flowing or impounded water bodies. Similarly, stands of submergent Potamogeton or Elodea can be found in waters which are subject to gentle current or considerable wave action (but are limited by shallow water depth and proximity to the shore)(McNabb, 1976). On the other hand, free floating species, particularly duckweeds, are unable to grow significantly in flowing water, and in unsheltered impoundments are readily blown ashore by winds (DeBusk et al. 1981). Another important aspect of the rate of water flow is the contact time between the plants and the wastewater. The shorter the interval, the less time there is for the macrophytes to, for example, remove the influent solutes, but also, the less likely are the plants situated downstream to be nutrient limited (Stewart, 1979). The converse is true when the exposure time is long: solute removal may be more complete but nutrient limitation is more likely to develop. Zonation of submergent (Ozimek, 1978) and emergent (Small and Gaynor, 1975) species can occur along a concentration gradient resulting from the dilution of wastewater in a receiving water.

3.1.2 Design and operation of macrophyte systems

The designs of existing treatment systems which utilize aquatic plants fall into three main categories: floating, emergent and combined species systems. The research to date has concentrated on the use of natural wetlands rather than constructed aquatic plant systems. The natural wetlands have been described as combined, or multi-species systems dominated by emergent macrophytes. A very thorough review of design principles relative to the use of natural wetlands has been provided by Hammer and Kadlec (1983). Design criteria have been presented by Wolverton (1979) and Gee and Jenson, Inc. (1980), for water hyacinth, and by Wolverton (1979) and Stowell et al. (1980) for duckweed systems, and by Reed et al. (1981) and Lakshman (1979) for emergent systems. Design criteria for reclamation of mine drainage settling ponds, by means of planting selected macrophytes, have been provided by Nawrot and Yaich (1982). A particularly useful conceptual treatment of aquatic treatment system design has been written by Stowell and co-workers (Stowell et al. 1981), who have introduced the concept of the Aquatic Processing Unit (APU).

> "An APU is a wetlands containing an assemblage of plants (and possibly animals) grouped together to create an aquatic environment in which specific wastewater treatment objectives will be achieved. In this regard APU's are similar to conventional processes making up a conventional system but direct comparisons between APU's and specific types of conventional processes cannot be made in general. However, this does not preclude the use of APU's in conjunction with conventional processes." (Stowell et al. 1981)

APU's can be variously located within a treatment train, either as polishing units, retrofitted additions to ailing lagoons, secondary or tertiary treatment compartments, as illustrated in Figure 3-1.



Source: Stowell et. al. 1981

APPLICATIONS OF AQUATIC PROCESSING UNITS
FOR THE TREATMENT OF WASTEWATER

Figure 3-1

3.1.2.1 Management of the water

Management of water chemistry, where feasible, may also enhance the treatment effectiveness of a constructed wetland. The use of lime for pH control of acid mine drainage flowing into a lagoon system is mentioned by Nawrot and Yaich (1982). The importance of maintaining aerobic conditions (at least in certain sections of the system) in order to favour the adsorption and co-precipitation of P is stressed by Boyd (1970). Maximization of nitrogen removal via denitrification in natural or constructed wetlands is limited by the availability of carbon. Gersberg et al. (1983) report that a BOD:NO_3 ratio of 2.3:1 is the optimum for denitrification in domestic sewage. These researchers have shown that added methanol or decaying vegetation can be used as carbon sources to enhance denitrification. Cutting and mulching of *Scirpus*, *Typha*, and *Phragmites* can improve N removal without concommittant increases in effluent BOD or P concentrations. The treatment effectiveness of constructed macrophyte systems is directly related to the degree of control exerted on water flow. Control can be achieved through careful selection of basin geometry in order to provide both the desired depth (Table 3-1) and the optimum contact betweeen the water and the plants. In general, a high length to width ratio (>3:1) is favourable, in order to minimize short circuiting of the flow (Reed et al. 1984; Wile, 1980). Where little or no control is possible over the volumes delivered to the APU, the design of the system may need to accomodate the extremes in flow in such a manner as to not significantly reduce its effectiveness. For example, seasonal peaks in flow must be prevented from scouring rooted species or washing floating species out of the system, by means of equalization basins or distribution of the flow through several parallel streams. Conversely, low flows due to drought or exfiltration may need to be supplemented from equalization reservoirs or through recycling of treated effluent. The use of impermeable basin liners is also a realistic alternative.

The selected water depth will also depend on the macrophyte species. For floating species, the plants' contribution to treatment is increased by decreasing depth (i.e. providing a high ratio of surface area to volume); nevertheless water depth can vary without affecting growth. For emergent forms, depth should be within the range of 5 - 100 cm, depending on the season, species or plant age. Increasing the depth from 10 cm in summer to 30 cm in winter can avoid flow problems related to ice formation (Black, 1983). For submergent plants, depths of up to 6 meters can be used with certain pondweed species (McNabb, 1976), although generally shallower waters (1-3 m) are preferable. Since almost all macrophytes can tolerate some variation in water depth to a lesser or greater degree, the possibility exists of using depth control within macrophyte ponds as a means of equalizing wastewater flows.

It appears from the literature that the hydraulic residence time within a macrophyte system should be on the order of 5 - 30 days, depending on the degree of treatment being sought. In general, the longer the retention period, the more complete will be the treatment.

3.1.2.2 Management of the vegetation

The selection of macrophyte species will be dictated primarily by local availability. Some species, such as Phragmites, grow best in monoculture; others, such as Glyceria, Acorus and Iris, thrive in mixed stands (Martin and Janiak, 1982). Among other factors influencing the initial selection of species, water depth, water chemistry (including degree of pre-treatment) and the potential plant end uses, need to be considered.

Data on recommended stocking densities are available for floating and emergent species, but not for submergent forms. Duckweed initial standing crop of 144 grams wet weight per m^2 is suggested by Said et al. (1979), for achieving maximum growth rate while avoiding crowding. Successful development of stands of emergent species are reportedly achievable by planting anywhere from 1 - 10 plants per m^2 (Reed et al. 1984; Lakshman, 1979; Gersberg et al. 1983). Propagation of emergent species by means of rhizome cuttings, shoot cuttings, layering and direct seeding is described by Veber (1978). Fulton and Bjugstad (1983) compare the use of pots, plugs and sprigs for transplanting emergents onto the slopes of settling basins. Their research shows that plant survival is highest with pots, followed by plugs and sprigs, and that 10-20 cm initial water depth is preferable over deeper conditions. It is also interesting to note that, given sufficient initial species diversity, a constructed wetland will "self design", such that any given species will tend to colonize the habitats to which it is pre-adapted. The possibility of self-design was initially suggested by Odum (1971); evidence for the process is described by Fulton and Bjugstad (1983).

The role of macrophyte cropping or harvesting is an important consideration for two primary reasons: the potential recontamination of the water by decaying plant biomass, and the inhibitory effect of crowding on growth of at least certain floating species (Jewell, 1971; Said et al. 1979). Where the treatment effect sought is the removal of nutrients or other solutes, it is desirable to maximize plant growth rate, since treatment efficiency is related to plant growth rate (McNabb, 1976; Gersberg et al. 1983). It may also be important to maintain the plants in a state whereby the recirculation of nutrients from storage organs is minimized, thereby increasing uptake. Where the plant matter may have re-use potential, harvesting will also be important. Thus, the plant matter accumulated during the growing season may or may not be desirable from a wastewater treatment perspective.

In nature, the plant matter accumulated during the growing season is stored during the winter in either a dormant or dead state. Species in which the individual plants have a brief lifespan, (eg., duckweeds), contribute dead organic matter and therefore exert a constant but low oxygen demand on the water throughtout the growing season (Stowell et al. 1980). Annual species, such as most emergents, pulse-load the ecosystem with organic matter at the end of each growing season. Submergents follow a pattern of almost complete mortality during the winter and rapid regeneration of the standing crop in the spring (McNabb, 1976). With the onset of warmer temperatures, microbial degradation of any dead organic matter is accelerated, re-solubilizing carbon, nutrients and other compounds (Jewell, 1971), and as mentioned above, consuming DO in the process. Plant composition also varies with season or developmental stage (Carpenter and Adams, 1977; Mutzar et al. 1978; Yakubowskii et al. 1975; Ozimek, 1978). Thus, where the removal of non-volatile solutes (eg., P or metals) or the maintenance of aerobic conditions are of primary concern, and where it is not desired to utilize the waterbody as materials sink, harvesting will be advantageous, if not essential.

Maximum harvesting is generally not advantageous for two main reasons: excessive damage to the plants (see following paragraph) and secondary ecological effects. There is evidence, in the case of both submergent (McNabb, 1976) and floating (Ehrlich, 1966) species, that the presence of macrophytes creates a favourable habitat for zooplankton which, in turn, contribute significantly to maintaining water clarity by consuming suspended solids including microalgae. Excessive removal of the zooplankton habitat can therefore result in phytoplankton blooms. Under such conditions, recovery of cropped submergent plants may be inhibited or prevented due to shading by the plankton. Dinges (1982) recommends that, in systems containing floating plants, excessive increases in effluent BOD and suspended solids can be avoided by limiting the area of open water to less than 20 percent of the total pond area.

The key factors controlling harvestability are the tolerance of the plants to the physical damage caused by cropping, and their rate of regeneration. Generally speaking, frequent cropping is possible only in the case of floating macrophytes (DeBusk et al. 1981; Wolverton and McDonald, 1979) in which the remaining unharvested plants are not damaged. There is evidence that total biomass production decreases with the frequency of harvest for certain emergent and submergent species (Spangler et al. 1976). Emergents such as the bulrushes and reeds, however, respond best to harvesting once or twice (Burton and Ulrich, 1983) or, at most, four times per year (Spangler et al. 1976). A similar situation exists with submerged species such as water millfoil (Carpenter and Adams, 1977) and pondweed, coontail and Elodea (McNabb, 1976). The timing of the harvests can be made to coincide with the period of highest elemental content in the harvestable biomass (McNabb, 1976; Carpenter and Adams, 1977).

3.2 Wastewater Types Treated Using Macrophytes

Wastewaters which are amenable to biological treatment in aquatic macrophyte systems will necessarily be non-phytotoxic or physically limiting, and must contain sufficient nutrients (either in solution or in combination with the substrate at the treatment site) to support plant growth.

3.2.1 Municipal wastewater characteristics

Municipal or domestic sewage is referred to herein as that relatively dilute wastewater composed primarily of human excreta and household or commercial liquid wastes, but also including wastewaters from a variety of other urban sources which are not required by law to pre-treat their discharges to the sewerage system. In general, municipal wastewater, or sewage, is characterized by a predominance of particulate and dissolved organic matter, as well as elevated concentrations of dissolved minerals including macro- and micro-nutrients. Microbial or plant growth in municipal wastewaters is unlikely to be limited by imbalances in the relative concentrations of nutrients. Concentrations of pathogens (i.e. bacteria, viruses) are generally high in raw municipal wastewaters. Toxic substances are not normally present in significant quantities. The flow and degree of dilution can vary with locale, and is influenced by such factors as groundwater infiltration to the piping, seasonal precipitation patterns, and whether or not the sewerage system is receiving storm water in addition to wastewater. Table 3-3 presents the composition of typical municipal wastewaters in North America (Metcalf and Eddy, Inc. 1979).

The treatment requirements of municipal sewage include: removal of inorganic and organic suspended solids, oxidation of organic matter (i.e. satisfaction of BOD), disinfection and removal of dissolved nutrients.

TABLE 3-3

TYPICAL COMPOSITION OF UNTREATED DOMESTIC WASTEWATER

Constituent	Concentration[a]		
	Strong	Medium	Weak
Solids, total:	1200	720	350
Dissolved, total	850	500	250
Fixed	525	300	145
Volatile	325	200	105
Suspended, total	350	220	100
Fixed	75	55	20
Volatile	275	165	80
Settleable solids, mL/L	20	10	5
Biochemical oxygen demand, 5-day, 20°C (BOD_5, 20°C)	400	220	110
Total organic carbon (TOC)	290	160	80
Chemical oxygen demand (COD)	1000	500	250
Nitrogen (total as N):	85	40	20
Organic	35	15	8
Free ammonia	50	25	12
Nitrites	0	0	0
Nitrates	0	0	0
Phosphorus (total as P):	15	8	4
Organic	5	3	1
Inorganic	10	5	3
Chlorides[b]	100	50	30
Alkalinity (as $CaCO_3$)[b]	200	100	50
Grease	150	100	50

a All values except settleable solids are expressed in mg/L (mg/L = g/m^3).

b Values should be increased by amount in domestic water supply.

Note: 1.8(°C) + 32 = °F.

3.2.2 Industrial wastewaters characteristics

Industrial wastewaters are as diverse as the industries that generate them. A comprehensive review of wastewaters from all major industries, however, is beyond the scope of the present report. The approach here, therefore, is to examine three broad categories of industry - agro-industry, manufacturing, and mining - grouped according to relatively consistent similarities in their waste water characteristics. The coverage is necessarily very condensed and the reader is advised to consult other sources for more detailed, industry specific information. (For example, the literature review issues of the Journal of the Water Pollution Control Federation [eg. Volume 57, No. 6, June, 1985], contain a wealth of information on the treatment of industrial, as well as other, wastewaters.)

The wastewaters from agriculture and food processing (i.e. agro-industry) are, in a broad sense, similar to municipal wastewaters, in that they typically contain high concentrations of particulate organic matter and dissolved nutrients. The relative concentrations of the nutrients (eg., N:P ratio) are more likely to be less balanced with respect to their suitability for supporting microbial or plant growth. The degree of dilution may be higher or lower than in municipal sewage. For example, the wasterwaters from confined animal production are usually much more concentrated than those from vegetable packaging installations. Many waters from animal-related (eg. dairy, rendering), and from certain plant-related (eg. oil extraction) industries carry elevated concentrations of fats and oils. In addition, agro-industrial wastewaters often contain elevated concentrations of potential pathogens, process chemicals and other substances, such as surfactants, cleaning agents and disinfectants. Some of these may affect the pH or interfere with the bio-degradability of the organic matter. The flows of agro-industrial wastewaters are often generated on an intermittent or seasonal basis, particularly in the case of products based on plant crops.

The treatment requirements of most agro-industrial wastewaters are, generally, similar to those of municipal sewage. For this reason, many food processing industries, for example, are permitted to discharge their effluents into the municipal sewerage system, with or without in-plant pretreatment.

Manufacturing is broadly defined, for present purposes, to include the industries related to wood, textile, metal, plastic, chemical and other products. The wastewaters generated by these industries all characteristically contain low concentrations of readily degradable organic matter and high concentrations of process reagents. Higher concentrations of refractory (resitant to biodegradation) organic matter are common in the case of industries dealing with biological products. The reagents typically include acids, bases, solvents, bleaching and colouring agents, and other chemicals. Concentrations of potential pathogens are variable, although less of a concern than in municipal and agro-industrial wastes. Manufacturing wastewaters are often contaminated as well by various leachates including heavy metals and complex organic and inorganic molecules. Many of the solutes in these waters are toxic when occuring at elevated concentrations. Also, the largely non-biological nature of these wastewaters creates a high likelihood that macrophytes or other treatment organisms exposed to them would exhibit nutrient limitation. Volumes generated are tied to the industrial output, and are generally constant year round.

Manufacturing wastewaters are, undoubtedly, the most diverse within the three industrial categories examined. Consequently, their treatment requirements are nearly always case-specific and are more likely to be met in part, if not totally, on site. The types of treatment required may include solids removal, clarification, neutralization, oxidation, and removal or ecologically deleterious chemicals, including heavy metals.

Mining wastewaters, as with those from manufacturing, necessarily vary with the type of operation. Olem and Betson (1985) provide a review of recent literature on wastewater treatment in the coal mining and processing industry. In general though, mining wastewaters typically may contain very high concentrations of suspended or colloidal inorganic solids, and elevated concentrations of dissolved minerals such as residues from explosives, heavy metals, and others. The pH may be far from neutrality, with excess acidity often being a major concern, particularly in eastern North America. As with manufacturing, the fraction of organic matter is typically very small, if not negligible. Volumes generated generally follow the seasonal precipitation, groundwater and freeze-thaw cycles.

The treatment requirements of mining wastewaters, as with those from manufacturing, are case specific. Nevertheless, removal of suspended solids, heavy metals, and dissolved nutrients, as well as neutralization, are emphasized in the context of the present literature review.

3.3 Effectiveness of Macrophytes in Wastewater Treatment

The use of macrophytes in wastewater treatment is, despite the abundance of published information, still in its infancy. Dr. Kathe Seidel, of the Max Planck Institute in Krefeld, West Germany, and a pioneering investigator of the potential of aquatic plants for wastewater treatment, wrote in the mid 1970's (Seidel, 1976):

> ". . . have plants latent qualities of which we are unaware because they have not been challenged until now?
>
> Nature offers an abundance of plant species; man produces many types of sewage. How plants will react to new chemical wastes produced by man, what latent qualities will emerge enabling them to survive, and what arrangements will have to be made to lighten their way into an unencumbered future are questions of vital importance."

The following sub-sections examine the information available on the performance and limitations of macrophyte systems treating municipal and industrial wastewaters. Reports dealing with experimental as well as full scale operations are reviewed.

3.3.1 Performance of macrophyte systems

Aquatic plant systems have been used, either incidentally or purposefully, for the treatment of municipal sewage, wastes from dairy and swine operations, and from a variety of food processing and manufacturing industries, as well as mine drainage.

Floating macrophytes have been used at temperate latitudes, though only in greenhouse-type enclosures, at Hercules, California (Serfling and Alsten, 1979), and at Eugene, Oregon (Head, pers. comm.), to treat municipal and aquacultural wastewaters, respectively. The full scale system at Hercules (38^{o}N latitude) used water hyacinth and duckweed, while only hyacinth was used experimentally at Eugene (44^{o}N). There is evidence that at the latter latitude the hyacinth is limited in winter by insufficient light (Head, pers. comm.). These two species have been promoted for upgrading secondary treatment lagoons in the southern U.S. and other warm climate regions (Wolverton and McDonald, 1979).

Nutrient uptake by water hyacinth grown experimentally on agricultural drainage water in Florida has been found to account for 41.2% of the ammonia-N, 39.3% of the nitrate-N and 28.6% of the total-P input; the corresponding overall nutrient removal efficiencies were 96.6% of the ammonia-N, 88.5% of the nitrate-N and 36% of the total-P, after 27 days (Reddy, 1983). The main processes involved in N and P loss were denitrification and adsorption/precipitation, respectively. Hyacinth productivity was 4.26 $g/m^2/d$, with nutrient contents of 4.7% N and 0.6% P.

Nitrogen uptake by duckweeds cultured experimentally on raw sewage in Israel at hydraulic retention times (HRT) of 10 and 20 days ranged over 5-70% of the influent N, with the higher removals correlating with the longer retention time (Oron et al. 1984). In the same study, growth of Spirodela polyrrhiza and Lemna gibba was found to be inhibited by ammonium-N concentration of 200 mg/l, though not at 50 mg/L. The two species responded differently to varying COD (chemical oxygen demand) levels of 100, 300 and 600 mg/L. Production of Spirodela showed a positive correlation with COD at the high ammonium level, regardless of HRT, while, at the lower ammonium level, showed a negative correlation with COD at 10 days HRT, but a positive correlation at 20 days. Growth of Lemna showed no correlation with COD at either HRT. There was also evidence of nitrogen fixation in the culture systems. Generally, N removal via duckweed uptake appeared to be affected more by the ammonium concentration than by COD. The study found that, in areas where the evaporation rate from a free water surface is greater than 4.5 mm/d, evaporative water losses could be reduced by about 30% from a duckweed covered pond. Duckweed dry weight productivities of 8-15 $g/m^2/d$ were reported.

Seasonal succession of hyacinth in the summer, followed by duckweed in the cold months has been described in Mississippi (Wolverton and McDonald, 1979) and California (Stowell et al. 1980). Hyacinth has been found to out-perform duckweed in California (Dewante and Stowell, 1981). Full scale hyacinth systems are currently in use at Walt Disney World in Orlando, Florida (Schwegler and McKim, 1984; Lee, 1980), and at the City of San Diego, California. There have been calls for more research into the use of shade- and cold-tolerant species for temperate climates (Duffer and Moyer, 1978; Middlebrooks, 1980; Reed et al. 1981).

Table 3-4, reproduced from Middlebrooks (1980), summarizes the design and performance of a selected floating plant system. While the species (water hyacinth) grows only in warm climates, many of the design parameters would apply to duckweed or waterfern systems in the temperate zone. Floating plant systems are most effective in removing suspended solids and nitrogen, somewhat less effective in reducing the BOD, and least effective at removing phosphorus. Some information is available as well on the removal of heavy metals and organic chemicals by water hyacinth (Wolverton and McDonald, 1978; Wolverton and McKown, 1976), though data from designed systems is lacking for other floating species. It should be noted that, despite the recognized importance of harvesting in maximizing nutrient removal, a large part of the performance data available for floating plant systems was generated under non-harvested conditions (Dewante and Stowell, 1981; Wolverton and McDonald, 1979; Reed et al. 1984).

TABLE 3-4

DESIGN CRITERIA FOR WATER HYACINTH WASTEWATER TREATMENT SYSTEMS BASED UPON BEST AVAILABLE DATA FOR OPERATION IN WARM CLIMATES.

Parameter	Design Value		Expected Effluent Quality
	Metric	English	
A. <u>RAW WASTEWATER SYSTEM (Algae Control)</u>			BOD_5 < 30 mg/L
Hydraulic Residence Time	> 50 days	> 50 days	SS $\lesssim$ 30 mg/L
Hydraulic Loading Rate	200 m^3/ha.d	0.0214 mg ac.d	
Depth, Maximum	< 1.5 meters	< 5 feet	
Area of Individual Basins	0.4 hectare	1 acre	
Organic Loading Rate	≤ 30 kg BOD_5/ha.d	≤ 26.7 lbs BOD_5/ac.d	
Length to Width Ratio of Hyacinth Basin	> 3:1	> 3:1	
Water Temperature	> 10°C	> 50°F	
Mosquito Control	Essential	Essential	
Diffuser at Inlet	Essential	Essential	
Dual Systems, Each Designed to Treat Total Flow	Essential	Essential	
B. <u>SECONDARY EFFLUENT SYSTEM (Nitrogen Removal and Algae Control)</u>			BOD_5 < 10 mg/L
Hydraulic Residence Time	> 6 days	> 6 days	SS $\lesssim$ 10 mg/L
Hydraulic Loading Rate	800 m^3/ha.d	0.0855 mgac.d	TP $\lesssim$ 5 mg/L
Depth, Maximum	0.91 meter	3 feet	TN 5 mg/L
Area of Individual Basins	0.4 hectare	1 acre	
Organic Loading Rate	≤ 50 kg BOD_5/ha.d	≤ 44.5 lbs BOD_5 ac.d	
Length to Width Ratio of Hyacinth Basin	> 3:1	> 3:1	
Water Temperature	> 20°C	> 68°F	
Mosquito Control	Essential	Essential	
Diffuser at Inlet	Essential	Essential	
Dual Systems, Each Designed to Treat Total Flow	Essential	Essential	
Nitrogen Loading Rate	≤ 15 kg TKN/ha.d	≤ 13.4 lbs TKN/ac.d	

Source: Middlebrooks, 1980.

Emergent plant systems have been investigated mostly in temperate climates. There are reports from West Germany (Seidel, 1976), the Netherlands (de Jong, 1976; Greiner and DeJong, 1982), Poland (Ozimek, 1985), the Soviet Union (Chernyskev, 1979; Karaseva and Papchenkov, 1974), the United States (Wolverton et al. 1983; Spangler et al. 1976; Dewante and Stowell, 1981; Tilton and Kadlec, 1979; Helle, 1983) and Canada (Lakshman, 1979; Black, 1983; Wile, 1980). Both natural and constructed wetlands have been used. Generally, natural sites have received pretreated effluents, whereas constructed sites have been designed to receive either raw or pretreated wastewaters. Water quality improvement has been recorded at sites receiving effluents incidentally (Small and Gaynor, 1975; Yonika, 1979; Mudroch and Capobianco, 1979 a,b) and at sites where specialized wastewater delivery systems have been deployed (Kadlec, 1979; Fritz and Helle, 1979; Pope, 1981). It is notable that volunteer duckweed populations have been reported to become established in both natural and constructed emergent plant systems (Small and Gaynor, 1975; Kadlec, 1979; Black, 1983).

Reddy (1983) has reported that an experimental system using a combination of cattail and _Elodea_ was able to remove 43.8% and 23.8% of the labelled influent ammonium-N and nitrate-N, respectively, from agricultural drainage water. Corresponding overall removals of 99.9% were measured for both N forms. Similarly, a system containing water pennywort, _Hydrocotyle umbellata_, (a smaller emergent species) was able to remove 67.3% and 13.0% of the ammonium-N and nitrate-N respectively, with corresponding overall removals of 100% for both N forms. Plant uptake of influent total-P accounted for 4.4% by the Elodea-cattail system and 64.5 by the pennywort system. Biomass productivities were 4.2 $g/m^2/d$ by pennywort, 3.6 $g/m^2/d$ by cattail and 0.67 $g/m^2/d$ by Elodea. Tissue nitrogen and phosphorus content was 4.5% N and 1.2% P for pennywort, 1.7% N and 0.03% P for cattail and 10.5% N and 0.4% P for Elodea. Initial water quality was approximately 10 mg/L for ammonium and nitrate, and 5.1 mg/L for P, with an initial pH of 7-7.4. In the system containing Elodea, the pH rose daily to 9 by sundown, decreasing again overnight, while in the pennywort (and hyacinth) system, the pH remained relatively unchanged.

Spangler et al. (1976), in Wisconsin, quantified the distribution of P in experimental ponds containing Scirpus validus and Typha angustifolia, operating at an HRT of 5 hours, and receiving clarified secondary effluent from an "overloaded" municipal treatment plant. The plants were harvested monthly. The total P in the biomass in mid June, late September and early December increased from 33.7 to 54.2 and 69.8 percent of the total P in the system. The fraction of P removed by harvesting the plants represented between 14% and 10.5% of the P added to the system. The study did not quantify biomass N content, nor the fraction of influent P removed by harvesting. The authors did find that, while repeated cropping resulted in a lesser total production of biomass, the higher P content of the younger shoots, as compared to older unharvested tissue, permitted a higher overall P removal via multiple harvests. Phosphorus content in shoots aged two weeks, one month and five months was found to be 0.48, 0.36 and 0.14 percent in Typha, and 0.62, 0.43, and 0.15 percent in Scirpus, respectively.

Tilton and Kadlec (1979) have studied the feasibility of utilizing a natural freshwater wetland, dominated by Typha and Carex, for tertiary treatment of pretreated municipal effluent in Michigan. A preliminary mass balance, based on tentative system boundaries, showed that 99% of the added (nitrite + nitrate)-N, 77% of the added ammonium, and 95% of the total dissolved phosphorus was immobilized within a one-hectare area around the 200m manifold discharge pipeline. P removal in particular was found to be more effective in shallow (6cm) than in deep (30cm) water when plant biomass was not harvested.

Pope (1981) in California evaluated the use of Phragmites and Scirpus in serial trenches during the treatment of screened raw municipal sewage. The HRT was 6-8.5 hours at flows of 133-96 m^3/d. The wetland was designed after the patented MPI (Max Planck Institute) system developed by Seidel and co-workers, in which water percolating vertically through the Phragmites root zone and underlying sand then flows horizontally through the Scirpus bed. Considerable operational difficulties due to plugging of the percolation bed were reported, and several remedial measures evaluated. Pollutant removal effeciencies and effluent concentrations from operation at 8.5 hours HRT were as follows: BOD - 88% and 27 mg/L, total suspended solids 91% and 20 mg/L, NH_4-N - 33% and 16

TABLE 3-5

DESIGN AND PERFORMANCE OF SELECTED EMERGENT MACROPHYTE SYSTEMS TREATING MUNICIPAL WASTEWATERS

Location	Population or (flow m^3/d)	Area (ha)	HRT (days)	Pre-treat-ment	Effluent Quality (mg/l) or (Percent reduction)				Reference
					BOD	SS	N	Total P	
NATURAL WETLANDS									
							Total:		
Massachusetts	6000	19.4	~57	2°	5.8	-	6.2	0.5	Yonika,
					(67)	-	(38)	(47)	1979
							NO_3:		
Michigan	6 - 8000	700	-	2°	-	-	≤0.02	0.03	Kadlec,
					(100)	-	(100)	(100)	1979
							Total:		
Florida	-	-	-	2°	-	-	-	-	Fritz and
					-	-	(98)	(97)	Helle, 1979
CONSTRUCTED SYSTEMS									
							Total:		
Netherlands	1000	1	10	1°	-	-	-	-	
	-				(>98)	-	(95)	(93)	Ryther, 1979
Ontario	-	20	7	1°	8.3	11	6.3	-	Reed et al.,
	(3785)				(66)	(63)	(54)	(65)	1984
California	-	.02	8.5	1°	26	20	21.2	8	Black, 1983
	(95 - 133)				(88)	(91)	(43)	(12)	Pope, 1981

TABLE 3-6

DESIGN AND PERFORMANCE OF SELECTED EMERGENT MACROPHYTE SYSTEMS TREATING INDUSTRIAL WASTEWATERS

Industry	Species	Design/Operation Information Available	Performance Data			Reference
			Parameter	Final conc. (mg/l)	% Reduction	
Dough Cannery	Schoenoplectus lacustris	HRT ≥ 4 days	pH in = 11.7	7.3	-	Seidel, 1976
Sugar Refinery	Schoenoplectus lacustris	HRT ≥ 4 days	pH in = 4.4	6.7	-	Seidel, 1976
Various (polluted river)	Schoenoplectus lacustris	Depth ~ 2m exfiltration channel	phenols "organics"	0.025 11.9	58 63	Czerwenka and Seidel, 1976
Mining (unspecified)	Phragmites sp. Carex sp. Glyceria sp.	HRT = 36 - 58h Flow = 0.5 - 0.8 m^3/min	TSS	18 - 47	39 - 46	Martin and Janiak, 1982
Mining (unspecified)	"thickets" of emergents	lagoons	TSS SO_4	2334 19	28 89.3	Karaseva and Papchenkov, 1974
(unspecified)	Typha Phragmites	HRT = 8d Flow = 78 L/h	BOD Petroleum Phenols	27.5 - 57 1.2 - 8.5 0	- - -	Vasigov et al., 1976

mg/L and total phosphorus - 8% and 12 mg/L; nitrate values increased from 0.1 mg/L to 0.4 mg/L.

Tables 3-5 and 3-6 summarize the performance of selected emergent macrophytes systems. The highest treatment efficiencies were generally achieved in the reduction of suspended solids, BOD, nitrogen, and pathogenic indicators. Phosphorus removal tended to be less, though greater than in the case of floating species. Heavy metals and organic chemicals were also effectively removed.

Treatment systems employing submerged species have not really evolved beyond the experimental stages. McNabb (1976) has explored the potential of submerged species growing in wastewater lagoons, identifying the species successional stages and environmental preferences, as well as investigating plant tissue composition and the effect of cropping on plant growth and on solute removal. Older lagoons (>12 yr) were observed to be dominated by species such as _Ceratophyllum demersum_, _Potamogeton zosteriformes_, _P. pectinatus_ and _Elodea canadensis_, while younger ponds were dominated by _P. foliosus_, _P. berchtoldi_, and _P. pectinatus_. The succession was attributed to the transition in the quality of the bottom soil from the initially compacted clay to a softer, more organic substrate. The author estimated that, in lagoons operated at an HRT of 27 days, harvesting of the submerged biomass could potentially remove 20-25% of the P, 50-70% of the N, 80-100% of the Mn, 20-30% of the Fe, 5-10% of the Cu and Zn, and 1-3% of the Cd, Co, Cr, and Ni added to the system during the growing season.

Carpenter and Adams (1977) studied the harvesting of _Myriophyllum spicatum_ for nutrient control in a hard water eutrophic lake. Potential nutrient removals were estimated to be 16.4-18.5% of the annual N input and 37.4-15.5% of the annual P input, depending on whether the timing of the harvest was made to coincide with the period of higher P (late August) or N (mid summer) in the plant tissue.

Pondweeds, in combination with a diversity of other macrophytes and aquatic animals, have been proposed in Texas for tertiary treatment of municipal sewage (Dinges, 1976). Marine macrophytes have been used at Woods Hole, Massachusetts, as nutrient removal components of an experimental wastewater mariculture system (Ryther, 1979). Submerged freshwater plants have been planted in fishponds and in a filtration unit of an intensive fish culture system in Colombia (Pati.o, 1973) and the U.S. (McLarney and Todd, 1974), respectively.

In general, it is evident that submerged macrophytes are effective at oxygenation and removal of dissolved nutrients, heavy metals and organic chemicals. It is likely that pathogenic organisms would also be effectively reduced due to the creation by the plants of a chemical and biological environment hostile to such organisms (i.e. high daytime pH and predation by zooplankton) (McNabb, 1976; Ehrlich, 1966).

3.3.2 Summary of macrophyte attributes and limitations

Floating plants are pre-adapted to small surface area, relatively still, nutrient rich waters, and are not affected by organic loading (DO), turbidity or fluctuations in depth. Harvesting of the free floating biomass is relatively uncomplicated, and the plants can tolerate a broad range of cropping frequencies. Their major weakness is wind and current, hence the requirement for smaller (<0.2ha) sheltered or narrow lagoons, or some form of in-pond wind barriers. Floating plants can contribute most to wastewater treatment where odour control, light attenuation, and nutrient removal are desired.

Emergent macrophytes are pre-adapted to relatively shallow, stable depth, still or gently flowing waters, and are unaffected by lagoon surface area, DO or turbidity. Their major weakness is sensitivity to water depth, sensitivity to cropping and difficulty of harvest. Emergents can contribute most where the desired treatment effects include clarification, oxidation, pH neutralization and solute removal.

Submergent macrophytes are generally pre-adapted to relatively clear, still to gently flowing waters, and (within limits) variable depths. Their use is not limited by lagoon surface area, and there may be some flexibility regarding cropping frequency. Their major weakness is sensitivity to turbidity and organic loading, and difficulty of harvest. Submergent macrophytes can contribute most in situations where clear waters require oxygenation, alkalinization and solute removal.

The discharge of wastewater into natural wetlands, while historically more common, can be limited by legislative barriers to their utilization as treatment components in wastewater management schemes (Eichbaum, 1976). This is because, in many areas, current environmental legislation requires that the water discharged into such wetlands be fully treated beforehand. In contrast, constructed wetlands are perceived, in this context, as having the advantage that such regulatory constraints can be by-passed.

3.3.3 Potential for treating coal mine wastewaters

The concept of using aquatic macrophytes to treat coal mine wastewaters appears to be valid, based on the information available on both the ecology of the plants and the characteristics of the wastewater. Table 3-7 presents a conceptualized treatment stream, and summarizes the pertinent design and operational guidelines which could be drawn from the information available. The information therein should be regarded as preliminary and tentative.

The selection of species would depend primarily on the physical characteristics of the water, such as current, depth and transparency. In this context, floating species might be made best use of early in the system, where waters tend to be (at least intermittently) very turbid, but devoid of significant current, along or in combination with emergent species. The function of floating species would be the reduction, via adsorption and assimilation, of nutrients and heavy metals concentrations in the water.

TABLE 3-7

DESIGN CONCEPT FOR A COAL MINE DRAINAGE TREATMENT SYSTEM USING AQUATIC VEGETATION

TREATMENT STAGE	INFLUENT QUALITY	MACROPHYTE			BASIN DESIGN		
		TYPE	SPECIES	FUNCTION	DEPTH (m)	GEOMETRY	HRT (days)
PRIMARY SETTLING AND STORAGE	Often turbid, high SS and solute load, variable pH	Floating	Duckweed _Azolla_	removal of solutes and potential conditioning*	≥ 1	Ponds (< 0.2 ha) or channels	≥ 0.4
INTERMEDIATE TREATMENT	Less turbid, moderate SS; high solute load; variable pH	Emergent and floating	_Scirpus_ _Typha_ _Phragmites_	coagulation of SS, removal of solutes, neutralization, oxidation	≤ 0.5	Ponds or channels L:W $\geq$ 3.1	> 5
TERTIARY TREATMENT AND STORAGE	Clear; moderate solute load; variable pH	Submergent and/or emergent	Pondweeds Coontail _Elodea_	oxygenation, removal of solutes, neutralization	≥ 1 ≤ 0.5	Ponds or Channels	> 5

* through contribution of organic matter and buffering effect.

Emergent species would appear suited to shallow (<50cm), turbid or clear water, in ponds or slow flowing channels. The function of the emergents would be multiple. In the case of turbid waters, this would include precipitation of suspended or colloidal matter, by providing a large surface area for coagulation. Once turbidity is no longer limiting, the development of periphyton on the plant surface would contribute to the removal of nutrients and other solutes through adsorption and absorption. The rooted plants would contribute to the oxygenation of the root zone sediment, creating aerobic conditions which would favour the immobilization of certain metal and nutrient ions, such as Fe and P. In additon, the empirical evidence suggests that the emergent species could effect a partial neutralization of acidic pH, which would favour the precipitation of heavy metals.

Submergent species would appear to be suited to the clear, deeper waters such as might be found at the downstream end of the treatment system, either in ponds or slow-flowing channels. The treatment functions would include: potentially significant reductions in the concentrations of dissolved nutrients and heavy metals through adsorption and assimilation by the plants and associated periphyton; oxygenation of the rhizosphere and water; and further neutralization or even diurnal alkalinization - with benefits similar to those described for emergent species.

It remains to be seen, however, whether the above treatment effects would be achieved under field conditions. Site specific factors - such as the relative concentrations of essential plant nutrients and of potentially phytotoxic conditions, in the wastewater or in the soil; the timing and variability of flows in relation to the growing season of the plants; and the land area available for macrophyte-based treatment - may limit the applicability of the concept. Each of these potential constraints is examined further below.

Nutrient limitation is most likely to be found in the case of floating plants because the supply (with the exception of C) can only come from the water. The degree of limitation will be influenced by a combination of mass loading and concentration. Given the nature of the water (i.e. very high N:P ratio), phosphorus would probably be the limiting element. Phosphorus for rooted species would appear, in most instances, to be sufficiently available from the bottom soil. Carbon is unlikely to be a limiting nutrient to species in which most or all of the leaf surface is above the water. This is because, in such cases, the C supply can be supplemented by atmospheric rather than originating entirely from dissolved inorganic C. Similarly, inorganic C in low nutrient water should provide sufficient carbon to supply submergent plant requirements especially in the shallow, well mixed ponds used at most mines. Carbon limitation in freshwater systems has been associated with high nutrient (eutrophic) systems where plant growth is profuse. Theory would suggest that nutrient accumulation and cycling from the growth and decay of upstream vegetation would tend to mitigate potential nutrient limitations to downstream species.

Some aquatic plants are able to thrive in acid mine drainage water, however phytotoxicity to other species by acid water is most likely to occur in instances of insufficient dilution. This assumption, coupled with the fact that flows will vary, implies that any macrophytes in a minewater treatment system would have the opportunity for acclimation, and therefore might be able to tolerate an intermittently toxic environment. Upstream mortality during toxic intervals may have a buffering effect on downstream vegetation, which may or may not be able to recolonize the upstream reaches once favourable growing conditions resume. The magnitude of the toxicity problem and research into management options can only be addressed on a site-specific basis.

The potentially limiting effects of wastewater flow rate are related to the supply of nutrients and/or the concentration of harmful substances, as discussed above. During the growing season, low nutrient input due to low flows could conceivably limit the macrophytes' heavy metal uptake rate by reducing the growth rate, hence reducing treatment efficiency. The influence of rain and snowfall on the rate of leaching and on pH would also appear to affect the mass of nutrients or heavy metals being delivered to the macrophyte system. The possibility of recycling effluent or pumping water from uncontaminated sources to the head of the system in order to mitigate the potentially harmful effects may be a realistic management option in some instances.

Land area requirements for the relatively shallow macrophyte systems tend to increase with the level of treatment desired. The geometry of existing lagoons will not always be the most desirable for direct conversion to wetlands capable of effective treatment. In many instances, expansion of the area devoted to lagoons or channels, and/or remodelling of the existing impoundments would be the only option available in order to adopt macrophyte-based treatment. This may not be possible at some existing sites due to limitations by topography, nor desirable due to economic considerations. Ideally, the land area required for achieving a given level of treatment would be alloted early in the planning stages of mine development.

In summary, the development of macrophyte system technology for treating coal mine wastewaters appears to be warranted. There is sufficient information currently available on the environmental requirements and tolerances of a wide variety of floating, emergent and submergent aquatic plant species. There is a growing body of empirical data on the effectiveness of macrophytes as components of municipal and, to a lesser extent, industrial wastewater treatment systems, from which preliminary design criteria could be drawn. Similarly, the harvesting requirements for optimizing treatment, as well as the disposal options and re-use potential of the harvested plant biomass appear to be sufficiently documented. However, both field and laboratory research is necessary on the potential effects, on plant growth, of the variations in wastewater flow and chemical quality as affected by season of year. Also, the role of the epiphytic community remains as an important information gap with important implications on the potential designs of biological treatment systems for mine wastewaters.

4.0 UPLAND IRRIGATION

Irrigation is recognized as an effective method for the disposal of municipal and industrial wastewater. Gently sloped uplands or well-drained level areas are generally utilized, although experimentation has been carried out on wetlands in recent years (Williams, 1980). Most of the available literature deals with municipal wastewaters; agricultural and mixed municipal and industrial effluents have also been utilized for land application but specific information pertaining to land application, of mine wastewater is scarce. However, meaningful comparisons can be made between mine and municipal wastewaters.

There are four general methods of wastewater application to land: spray irrigation, flood irrigation, ridge and furrow irrigation, and grass filtration (Webber and Leyshon, 1975). Most of the studies in the literature pertain to spray irrigation on upland areas, but the choice of application method is usually dictated by specific site conditions. For example, flood irrigation requires level, moderately to rapidly permeable sites and may involve ponding; grass filtration is usually used on soils of low permeability (Loehr et al. 1979; Webber and Leyshon, 1975); ridge and furrow irrigation involves the flooding of approximately 60 cm deep furrows that are terraced into natural hillsides or level land (Loehr et al. 1979). Bendixen et al. (1968) concluded that there were no major differences in performance and renovative efficiency of the spray, flood or ridge and furrow methods of irrigation. However, sprinkler irrigation of high altitude hay meadows can be more economical, with respect to the amounts of water used, than flood irrigation (Barbarick et al. 1982). Details of system design and economic considerations are discussed in Myers and Young (1979).

Many factors determine the effectiveness of wastewater applications, including: climate, stability of plant and animal communities, and soil conditions. However, the nature of the effluent is a major concern in this study. Considerable experience has been gained with municipal wastewater irrigation, and findings should be applicable to mine wastewater because the chemical and biological characteristics of the effluents are similar.

4.1 Wastewater Characteristics

4.1.1 Municipal wastewater characteristics

The application of municipal wastewater to land is generally considered to be a tertiary treatment technique. In most cases, raw sewage is subjected to the removal of settleable and suspended solids, and the reduction of biological oxygen demand (BOD), prior to land application. This procedure is termed primary treatment and involves the elimination of settleable solids, suspended solids and BOD. Secondary treatment is the further reduction of suspended solids and BOD. It is after the secondary treatment (less commonly after primary treatment) that municipal wastewater is used for land application.

The actual physical, chemical and biological makeup of sewage effluent depends on the amount and type of industrial waste contribution, the method of treatment, the time of year and other factors. There are, however, some generalizations that can be made. Table 4-1 provides typical values for several chemical parameters for wastewater applied to six study areas near Pennsylvania State University. Each treated plot was irrigated at a rate of 2.5 or 5.0 cm/wk depending on the vegetative cover type; the natural rainfall occurring in the study area is approximately 2.5 cm/wk. Also given in Table 4-1 is the total amount of each constituent that was applied over the test period. The average concentration of the various constituents is typical of many municipal wastewaters, although considerable variation can occur if the industrial contribution to the wastewater is high.

Even though most settleable and suspended solids have been removed before the wastewater is applied to the land, the suspended solids remaining have usually been reduced to less than 1000 mg/L. Such concentrations can represent a considerable amount of solid material when large volumes of effluent are applied to soil systems over a long period of time. In some municipal wastewaters, particularly where there is industrial input, a wide variety of organic chemicals may be present at concentrations in the microgram per litre range. Such things as solvents, sulfides, substituted phenols, benzenes and napthalenes, halomethanes, polynuclear aromatic hydrocarbons, aromatic amines and numerous other compounds may be present (Demirjian et al. 1983).

TABLE 4-1

CHEMICAL COMPOSITION OF TREATED SEWAGE EFFLUENT USED FOR IRRIGATION[a]

Constituent	Average Concentration[b]	Total Annual Amount Applied (kg/ha)
pH	7.4	
Ortho P	4.42	21.14
Total P	5.00	23.79
NO_3-N	11.2	51.3
NH_4-N	3.3	15.0
Organic N	3.2	19.6
Total N	17.7	85.9
Cl	45.1	207.0
K	10.1	46.2
Ca	34.6	158.9
Mg	13.4	59.5
Na	32.6	145.5
Dry solids	360.8	1657.4
Vol. solids	140.8	524.0
Cu	52.0	0.3
Zn	191.9	0.9
Mn	56.4	0.3
Cr	25.3	0.1
Pb	48.5	0.2
Cd	3.8	<0.1
Co	22.6	0.1
Ni	41.7	0.2
Hg	454.4	2.1
Fe	337.3	1.6

a Based on 17 weekly samples collected during the period May 21 - October 30, 1975.

b Metals in ug/L all others as mg/L.

NOTE:

A common value for organic carbon would be approximately 10 mg/L and for total carbon approximately 30 mg/L. An example for sulphate concentration is 10 mg/L. (Breuer et al. 1979).

Source: Sopper and Richenderfer, 1979

A primary chemical component of most municipal wastewaters is nitrogen which occurs in the form of nitrate, nitrite, ammonium and organic N. In many cases the carbon to nitrogen ratios can be as low as 1.5:1 (Breuer et al. 1979). Also, concentrations of phosphates and monovalent and divalent soluble salts can be high and pH tends toward neutrality (i.e. pH 6.5-8). There may be a component of heavy metals (up to 0.5 mg/L) in the effluent water, although most of these remain in the sewage sludge (Sopper and Kerr, 1979; Breuer et al. 1979; Richenderfer and Sopper, 1979). Elements such as boron and manganese may also occur at concentrations up to 0.5 mg/L (Sopper and Kerr, 1979); and, sulfur is often found in relatively high concentrations (up to 10 mg S/L) in the form of sulphate (Breuer et al. 1979).

Numerous complex elemental interactions take place in the soil. Important aspects of these and factors affecting movement of metals are discussed in a later section.

4.1.2 Coal mine wastewater characteristics

The nature of coal mine wastewater depends on local environmental factors (e.g. physiography, climate, hydrology, etc.), coal type, methods of extraction, mine layout, origin of wastewater, and other variables. Table 4-2 shows data from one source of wastewater at a coal mine in southeastern B.C., and is thought to be reasonably representative of most western Canadian coal mines. These data indicate the relatively variable character of mine waste waters.

Many elements, including nickel, cadmium, mercury, lead, copper, cobalt, zinc, iron and aluminum become less soluble with increasing pH, while molybdenum becomes more soluble (especially between pH 7 and 11). Increased pH causes reduced solubility of ammonium; phosphate precipitates with calcium at high pH, and with aluminum, iron or manganese at low pH. Because of these relationships, wastewaters from different mine sites have the greatest similarity at similar pH ranges. In general, acid mine wastewaters will probably contain higher levels of metals than basic wastewater, depending on the nature of the native rock and other factors.

Nitrogen compounds have been shown to be elevated in waters draining surface coal mines. Pommen (1983) estimated that about 95% of the nitrogen discharged in wastewater from the Fording Coal Ltd. mine in Southeast British Columbia was derived from explosives. A combination of either AN/FO (ammonium nitrate/fuel oil mixtures) or slurry/water gels, depending on moisture conditions, are used as explosives at virtually all mines (Pommen, 1983). These preparations contain 20-33% nitrogen, fuel oil and, in the case of slurry/water gels, a number of other materials. According to Pommen's (1983) information, nitrogen concentrations in waters arising from open pit mines tend to be higher than from underground mines, and probably relates to the amount and type of explosives used.

The relative amount of nitrate, nitrite, ammonium and ammonia that is introduced to drainage waters by

TABLE 4-2

FORDING COAL LTD.
ANALYSIS OF NORTH TAILING POND SUPERNATANT,
CARRIED OUT BY THE COMPANY AND THE PROVINCE OF B.C. IN 1975 AND 1976

Parameter / Type of Value	Max.	Min.	Mean	No. of Values	Mining Objectives (range)
Alkalinity, Total mg/L	123	70	101	14	
Aluminum, Dissolved mg/L	0.2	<0.01	0.06	6	0.5 - 1.0
Total mg/L	0.2	<0.1	0.1	3	
Arsenic, Dissolved mg/L	<0.005	<0.005	<0.005	3	0.10 - 1.0
Total mg/L	0.02	<0.005	0.01	3	
Carbon, Total Organic mg/L	14	14	14	1	
Chromium, Dissolved mg/L	<0.005	<0.005	<0.005	3	0.05 - 0.3
Total mg/L	<0.005	<0.005	<0.005	3	
Copper, Dissolved mg/L	0.005	<0.001	0.003	12	0.05 - 0.3
Total mg/L	0.005	<0.005	0.005	5	
Iron, Dissolved mg/L	0.1	<0.05	0.1	12	0.3 - 1.0
Total mg/L	0.1	<0.05	0.1	5	
Lead, Dissolved mg/L	0.01	<0.001	0.005	12	0.05 - 0.2
Total mg/L	0.01	<0.01	0.01	5	
Manganese, Dissolved mg/L	0.09	<0.05	0.06	8	0.1 - 1.0
Total mg/L	0.05	<0.05	0.05	3	
Mercury, Dissolved ug/L	<0.05	<0.05	<0.05	4	
Total ug/L	0.08	<0.05	0.06	6	1.00 - 0.005
Nickel, Dissolved mg/L	<0.01	<0.01	<0.01	3	
Total mg/L	<0.01	<0.01	<0.01	3	0.2 - 1.0
Nitrogen, Ammonia mg/L	0.85	0.27	0.58	3	0.5
Nitrite+Nitrate mg/L	3.3	2.1	2.8	3	10.0 - 25.0
Organic mg/L	0.85	0.34	0.60	2	
Total mg/L	4.19	3.22	3.71	2	
Oil and grease mg/L	5	<0.5	3.17	9	10 - 15
pH	8.4	6.9	7.7	26	6.5 - 10
Phosphorus, Total mg/L	0.042	0.023	0.032	4	
Total Dissolved mg/L	0.005	<0.003	0.004	4	2.0 - 10.0
Solids, Dissolved mg/L	364	160	284	12	2500 - 5000
Suspended mg/L	60	2	31	12	25 - 75
Total mg/L	418	302	357	7	2500 - 5000
Sulphate, Dissolved mg/L	114	72	91	9	
Turbidity J. T. U.	33	1	17	12	
Zinc, Dissolved mg/L	0.03	<0.005	0.011	8	0.2 - 1.0

ource: British Columbia Ministry of Environment, 1979.

explosives is variable (Pommen, 1983). Generally, nitrate concentrations will exceed those of ammonium, which will exceed the concentration of nitrite. As a result, nitrogen leachate from surface mine pits and spoil piles is typically dominated by nitrate with small amounts of ammonium and nitrite (Pommen, 1983). Nitrate nitrogen levels reached 110 mg/L in drainage water from spoil areas at the Fording Coal mine (Pommen, 1983).

Most available data pertaining to phosphorus levels in coal mine wastewater is not definitive (Norecol, 1985); however, it appears that total P values generally range from a few to several hundred micrograms per litre. Levels of orthophosphate for one southeastern B.C. mine ranged from 3 ug P/L to 261 ug P/L; most concentrations were about 16 ug P/L (Norecol, 1985).

Particulate P is frequently the major source of P transported in the aquatic environment, and P is often associated with suspended solids (Norecol, 1985). P levels in most mine wastewaters seem to be at least an order of magnitude lower than those commonly found in municipal wastewaters.

4.2 Effectiveness of Wastewater Irrigation Treatments

4.2.1 Effectiveness of irrigation for treating municipal wastewater

The effectiveness of irrigation for treatment of municipal wastewater is judged not only on the degree of renovation achieved, but also on the degree to which it alters the environment. It is considered by many to be the best method of wastewater disposal (Freshman, 1976). However, because public acceptance lags behind the proven performance of the method, it has not been fully exploited in modern communities (Vela and Eubanks, 1973; Goddard, 1979; Freshman, 1976). The effectiveness of the process does vary. Climate, soil type, vegetation cover, rate, method and duration of application and other factors all have some bearing on the effectiveness of irrigation. To some degree, each situation is unique.

In general, it has been found that land application of wastewater substantially reduces effluent levels of solids, nutrients, organic toxins, heavy metals, and pathogens (Urie, 1979; Lapakko, 1981; Nutter et al., 1979; Barbarick et al., 1982; Menser et al., 1979; Quin and Syers, 1978; Demirjian et al., 1983; Breuer et al., 1979; Sopper and Kerr, 1979; Brockway et al., 1979). Nutter et al. (1979) found that 98% of the calcium and 90% of the phosphorus in wastewater was removed through land application. Results of numerous tests cited by Urie (1979) showed a value of total nitrogen in groundwater that was 2 - 35% of the total nitrogen supplied, although others (Sopper and Kerr, 1979 and Quin and Forsythe, 1978) found that effectiveness of nitrogen removal was dependent on the rate and duration of application, and on the vegetation type. Demirjian et al. (1983) found that virtually all trace organics in wastewater were removed by land application. Webber and Leyshon (1975) document heavy metal accumulation in soil and uptake by plants, that indicates significant removal. Gerba et al. (1975) demonstrated that 92 -

97% of bacteria in wastewater was retained in the first centimeter and that three to five percent were found at depths between one and five centimetres, although viruses have been isolated in groundwater receiving water from treated soil (Wellings et al. 1975).

Considerable recent attention has been given to potential difficulties associated with municipal wastewater land application. Land application programs run some risk of adding excess nutrients and other substances (i.e. nitrogen, monovalent and divalent salts, organic chemicals, phosphates, pathogens, etc.) to the groundwater. The build up of solids in soil pores, alteration of soil structure and the retention of heavy metals, salts, toxic organic compounds and pathogens in the soil can also cause problems (Day et al. 1972). Inadequate infiltration and percolation (Sopper and Richenderfer, 1978), or excessive loading can result in high levels of overland flow with resulting particulate pollution of local, downslope water bodies. Destabilization of plant and animal populations are another concern. These concerns can be divided into those related to vegetation and those related to soils. Most of the concerns cannot be related to coal mine wastewaters but those that are especially related to vegetation and nitrogen in soils, which could arise from mine wastewater application, are discussed in more detail below. The majority of problems associated with land disposal of wastewater have been related to over-loading of the soil system (Sopper and Kerr, 1979), application of inapropriate wastewater (i.e. containing high solids), or inadequate site selection or preparation.

The local climate, stability of plant and animal communities, soil structure and chemical composition of the wastewater are important factors determining the effectiveness of wastewater application. Climate has an important effect because many physical, chemical and biological processes that accomplish the renovation of wastewater are governed by temperature and moisture. An example of this occurs during periods of subzero temperatures, when spray irrigation may result in ice accumulations, frozen soil and reduced renovation of the effluent (DeWalle, 1979). However, Leland (1979) found that frost penetration into the soil could be prevented if irrigation was begun early in the winter season. He also found that groundwater quality was poorer during cold season operations. Reduced renovation may also result during periods of high rainfall due to saturated soil conditions and reduced evapotranspiration rates. If there is concern for NH_3 toxicity at a particular minesite, it may be necessary to pond wastewater or discharge it in another manner during the coldest period of the year and during wet periods to allow sufficient drying of soils.

Spray irrigation can have important effects on microclimate. Spray irrigation below the canopy in forest ecosystems may decrease daily maximum air temperatures by as much as $2^{o}C$ and growing season mean temperatures by $1^{o}C$. During cold weather, freezing of water on soil or vegetation may significantly warm the air. Mean forest soil temperature and daily maximums have been shown to be increased ($1^{o}C$ and $2^{o}C$ respectively) during the growing season as a result of spray irrigation

(DeWalle, 1979). Soil temperature maximums were reduced by up to $6^{o}C$ and soil temperature minimums were increased by as much as $2.5^{o}C$ on strip mine spoil; and decreased by $6^{o}C$ on bare soil during the growing season (DeWalle, 1979). Generally, these effects would be favourable for plant growth.

The effects of wastewater application on plants and animals inhabiting a wastewater disposal site can be complex. While plant canopies are generally unaffected, except over the long term, significant decreases in the amount of shrub cover and increases in herbaceous plant cover have been observed (Lewis and Sampson, 1981; Weinstein, 1976). Hurd and Wolf (1974), Kirchner (1977), Wakefield and Barrett (1979), and Grant et al. (1977) indicate that experimental nutrient enrichment of upland prairies and old fields has destabilized communities of arthropods, plants and small mammals. Lewis and Sampson (1981) found that while bird species increase in diversity they decrease in eveness (i.e. the comparative number of individuals between species), which is an indication of a decline in population stability. Much of the alteration in the natural animal populations utilizing an area irrigated with effluent is due to changes in the vegetative cover, which affect animal food sources and habitat. A graphic example illustrating the interelationship between nutrient application and ecosystem stability occurred in Washington state. A Douglas-fir plantation fertilized with sewage sludge was destroyed by voles, which moved in after a luxuriant growth of grass developed on the treated site (West, 1986).

Reports vary considerably regarding the effect of wastewater irrigation on soil chemical properties. In some cases no detrimental effects were noted (Richenderfer et al. 1975). In other cases, soil nutrients increased to high levels, beyond which the soil could no longer remove nutrients (except P) from the effluent (Quin, 1979; Quin and Forsythe, 1978).

Sewage effluent application has been shown to increase forest floor decomposition, primarily because of the addition of nutrients (particularly nitrogen) and moisture. The stability of the forest floor is important because it acts as a protective covering against the destructive impact of spray droplets, and may even provide some organic matter for cementation of soil aggregates. Reduction in the depth of the litter layer can result in deterioration of soil structure (Richenderfer and Sopper, 1979). Some positive effects may be reduced bulk density, higher levels of soil respiration, increased soil organic matter and plant nutrient content, and stimulation of plant growth (Richenderfer and Sopper, 1979; Sopper and Richenderfer, 1978; Quin and Syers, 1978; Palazzo, 1976).

Of the numerous elemental components that may occur in wastewater, nitrogen (N) and phosphorus (P) are often most important because they can induce excessive growths of algae and rooted aquatic vegetation, and present a threat to fish and other aquatic life (Richenderfer and Sopper, 1979). Nitrogen can be particularly troublesome because, in the nitrate form, it is mobile and may readily move through soil to contaminate groundwater. Bohn et al. (1979) explain that anions are typically repelled from the electrical double layer surrounding soil colloidal particles (e.g. clay particles), and that nitrate and chloride move through most soils at about the same rate as water. Nitrate mobility may be reduced in very acid forest soils having a pH-dependent, net positive charge associated with organic matter or hydrous oxides (Bohn et al. 1979); phosphorus retention can also be very high in acid soils (DeVries 1979).

Because the behaviour of nitrogen is important, it is useful to briefly review the nitrogen cycle in soils. Organic nitrogen, and many other nutrients, initially contact the soil in the form of litter or plant root exudates. Once exposed to the soil, complex organic material is broken down into smaller and less complex units, mainly by soil animals and microorganisms. Soil invertebrates prepare the organic waste material for the fungi, bacteria and protozoans, which biologically re-mineralize much of this organic matter (Dindal et al. 1977). A large percentage of these newly mineralized nutrients are either immediately taken up by plants or are stored in the soil where they may be utilized later.

Nitrogen is mineralized by soil microorganisms to form ammonium, which may be volatilized as ammonia and escape into the atmosphere; or, be nitrified to nitrite or nitrate ions. Mineralization is enhanced by aerobic conditions, moderately high temperatures and carbon to nitrogen ratios below about 30 (Tisdale and Nelson, 1975). Volatilization of ammonium ions occurs most readily at or near the soil surface at high temperatures and high pH. Nitrification occurs most rapidly in soils having a pH range of 5.5 - 10.0, relatively high temperatures, moderate moisture levels, and which have aerobic conditions. Denitrification of nitrates to nitrogen gas or nitrogen oxides is favoured when soils become water logged. Under otherwise favourable conditions, denitrification may be limited by the availability of carbon (Miller et al. 1977).

Nitrate, unless taken up by plants or microorganisms or adsorbed by soil, may move rapidly into groundwater. Wastewater irrigation has been observed to promote nitrificaton, thus increasing leaching losses (Breuer et al. 1979). As a result of increased leaching losses of nitrate, feasibility of a land wastewater renovation program can be restricted (Miller et al. 1977). Nitrification also increases acidity in some soils, a condition which subsequently favours leaching of exchangeable cations. Harvesting of plant growth can greatly decrease nitrogen influx into groundwater (Quin and Forsythe, 1978), and thus could be an effective management tool.

There are other aspects of the nitrogen cycle that are of interest in wastewater application. For instance, ammonium is often retained by soils, in contrast to nitrate; and, at high pH ammonium can readily volatilize to ammonia. Also, nitrogen losses from volatilization are higher in poorly drained or saturated soils (Tisdale and Nelson, 1975). Therefore, effluent disposal on wetlands may be a favourable method for disposal of some wastewaters that are high in nitrate nitrogen (Williams, 1980).

An additional consideration regarding the reduction of nitrate leaching losses is the application of nitrification inhibitors. A number of commercially available substances are toxic to nitrifying bacteria and can delay the conversion of ammonium to nitrate for varying periods of time, depending on various soil characteristics and application rates (Tisdale and Nelson, 1975). Applicability of this technique to mine wastewater treatment may be worthy of attention for some special cases. However, high costs and potential environmental impacts may preclude widespread use of inhibitors.

4.2.2 Site selection considerations

Water storage capacity, nutrient retention properties, permeability, drainage, depth to water table and texture are some of the soil properties which require assessment (Brownlee, 1975). Slope and vegetation cover are two other very important site characteristics. As an example of the differences between soils, a coarse textured soil is permeable and can transmit large volumes of water without inducing overland flow, the opposite is true for a fine textured soil. However, since coarse soils also have low water and nutrient retention properties, their capacity to renovate wastewater is low. Clay soils, by contrast, have a high capacity for renovation but have low permeability, so they may be severely limited with respect to loading rates (Brownlee, 1975).

Because of their low infiltration rates, clayey or silty soils can readily experience overland flow which may carry nitrogen, phosphorus or other elements to surface waters. For example, overland flow would often be induced in certain locations in British Columbia if steady state infiltration were 5 mm/hr - a maximum rate for many clayey soils (Hillel, 1980). In coarse textured soils filtering of bacteria, viruses and nutrients may be inadequate and groundwater may become contaminated (Urie, 1979).

The rate and duration of effluent irrigation can largely determine the effectiveness of wastewater application. Hook and Kardos (1978), McAuliffe et al. (1979), Urie (1979), for example, found that the amount of nitrogen leached from the soil increased as the rate of application was increased. Quin and Forsythe (1978) and Hook and Kardos (1978), found that nitrogen removal from the effluent by various soils decreased over time.

A red pine plantation on a sandy soil in Michigan, sustained a 2.5 and 5.0 cm/wk irrigation with municipal sewage effluent for five years without increasing the subsoil nitrogen levels more than 10 and 15% respectively (Urie, 1979). Raising the application rate to 8.8 cm/wk increased the proportion of added nitrogen to about 30% (Urie, 1979). It appears that restriction of loading rates to a maximum of 5.0 cm/wk should result in good renovation in many medium textured soils. Well-drained, loamy soils would probably provide better renovation of wastewater than sandy or clayey soils. Mineral soils with high organic matter content should also improve renovation because of the high capacity of organic matter to absorb water and nutrients. Another desirable soil feature is a deep forest floor, or other protective surface layer.

Examples of suitable soils in British Columbia may include medium textured Gray Luvisols formed from glacial till; or, moderately well drained Humo-Ferric Podzols derived from morainal parent material; both of these soil subgroups are common in interior British Columbia. Valley floors, toe slopes and gentle subalpine slopes depict some of the more desirable site characteristics appropriate for wastewater irrigation. Greater success is probable in warm dry regions (e.g. Vernon, B.C.), where the evapotranspiration rate is high.

4.2.3 Health considerations and land use

The results from studies describing the fate of toxic organic chemicals, heavy metals and pathogens originating in municipal wastewater that has been and applied to the soil, vary (Sagik et al. 1979; Demirjian et al. 1983). In general, it seems that loading levels up to 5 cm/wk, applied over the short term (1 - 10 years) to medium textured soils, will significantly reduce the concentrations of wastewater contaminants. Potential health risks remain where the soil is used for agriculture; the hazard is due to plant uptake of heavy metals and organic toxins. Also, some pathogens can persist in the soil for several years; and, there is no clear indication of what constitutes a safe level.

Where land is used for forestry or mining, the above problems will not be as significant. The major concerns in these industries are the prevention of groundwater contamination by excessive loading rates (particularly of NO_2 and NO_3), care in site selection, and the potential for significant site alterations resulting from irrigation-induced changes in the microclimate and nutrient and microbial regimes.

4.2.4 Potential of irrigation for treating coal mine wastewater

The available literature (Sopper and Kerr, 1979; MOE, 1978; Norecol, 1985; Pommen, 1983) provides values for many of the physical and chemical parameters for comparison of both secondary municipal effluent and mine wastewater. For example, total and suspended solids, organic carbon and nitrogen frequently (but not always) occur in concentrations of the same order of magnitude; phosphorus levels in the mine wastewaters studied were at least an order of magnitude lower than those found for municipal wastewaters. Heavy metals, sulfur, and other constituents, on the other hand, may differ considerably (Norecol, 1985; Sopper and Kerr, 1979; Richenderfer and Sopper, 1979). The presence of grease and oil can be greater in mine wastewater then municipal wastewater because most of the oil and grease material has been removed in secondary municipal effluent. A further, important difference is the relative variability in mine wastewaters.

Mine wastewaters vary considerably depending on the type of ore, extraction methods, and source (i.e. tailings, waste rock dump, equipment storage, sewage treatment plant, etc.). Unlike municipal wastewaters, they are not routinely subjected to more or less uniform treatment processes. Treatment plants on the scale required to deal with the volumes of wastewater common in mines are cost prohibitive. Many mines employ settling ponds to remove settleable and suspended solids; and, some release untreated wastewater to natural drainages.

4.2.5 Conclusions

Information specific to mines is lacking in the literature and some uncertainty exists with respect to experience with municipal effluent compared to mine wastewater. However, many of the principles regarding municipal wastewater irrigation also apply to mine wastewater irrigation.

Most of the available data pertain to upland spray irrigation. However, wetland or marsh irrigation should also be considered. There is evidence to indicate that wastewater application to drained peat marshes can be less costly than upland irrigation (Williams, 1980). Wastewater irrigation of strip mine spoils overlying fine textured soils, for example, might transform unproductive areas into productive marshes.

Site and soil suitability determination should be very similar for mine or municipal wastewater applications. Where mine wastewater parameters such as suspended solids, pH, nitrogen, organic compounds and heavy metals are comparable to those commonly observed for municipal wastewaters, similar loading rates (e.g. 2-5 cm/wk) would be appropriate (if soil parameters are similar). Where levels of heavy metals, organic chemicals or sediment loads are higher, pretreatment, much lower application rates or different methods of application may be required.

Other variables may influence the transferability of wastewater irrigation technology. Because the relative concentrations of various constituents may vary considerably between municipal and mine wastewaters, the effect on soil may also vary. For example, the ratio of carbon to nitrogen to sulfur (C:N:S) has a great bearing on how these elements act in the soil. Sulfur may be immobilized by microbes if the C:S or N:S ratio is too wide. Nitrogen may be immobilized if the C:N ratio is too high or if the C:S or N:S ratios are too narrow. Very low ratios of C:N predispose ammonium to volatilization if pH is sufficiently high (Tisdale and Nelson, 1975). These relationships can have important implications with respect to the level of renovation (or conversely the level of leaching) that takes place and the effect that irrigation has on plant communities.

The pH has important correlations with the behaviour of many wastewater and soilwater constituents. Most heavy metals, other trace metals, and oxides of iron and aluminum tend to be more mobile under acid conditions - but there are exceptions such as molybdenum. Relative concentrations of sulfur, nitrogen, carbonate and other components have important effects on pH (Bohn, 1979; Tisdale and Nelson, 1978).

Soil colloids greatly affect the removal or stripping of wastewater contaminants. In general, higher levels of soil organic matter and clay minerals suggest high renovation capabilities (Sopper and Richenderfer, 1979; Brownlee, 1978).

Site factors such as slope, area, type of vegetation and climate are important. Gentle slopes, abundant vegetation and dry warm climates are generally favourable characteristics with respect to wastewater renovation.

Despite any difficulties in scientifically predicting the effectiveness of irrigation for treating mine wastewater, land application of mine wastewaters is, in principle and to varying degrees in practice, similar to municipal wastewater application. Mine wastewaters are more variable with respect to a number of physical and chemical properties (eg. suspended solids, pH, heavy metals, phosphorus, etc.) and they also tend to have much lower organic chemical contents. The properties of any particular mine wastewater will have to be identified so that, where significant differences exist, appropriate adjustments in treatment techniques can be made.

The principles of mine wastewater irrigation are fundamentally the same as for municipal wastewater. Based on their physical, chemical and site characteristics, many soils in British Columbia and elsewhere in Canada possess a high capacity to renovate wastewater. It is probable that spray irrigation can effectively be used to treat wastewater produced in some Canadian mines; and, such practices as cropping (which removes nutrients, metals and organics taken up by plants) might be employed to enhance the effectiveness of this treatment.

Upland irrigation has the potential for treatment of coal mine wastewaters. Additional work is needed to establish the criteria for its use along with field trials to consider application rates. Methods for reducing nitrogen mobility in the soil also require investigation.

5.0 EVALUATION OF NUTRIENT REMOVAL FROM COAL MINE WASTEWATER BY LABORATORY DUCKWEED CULTURES

5.1 Introduction

5.1.1 Overview

The use of aquatic plants in wastewater treatment has been gaining considerable attention in recent years, in Canada (Neil, 1974, Lakshman, 1979; Wile, 1980; Black, 1983; Reed et al. 1984) and in other countries (Ehrlich, 1966; Tourbier and Pierson, 1976; Oron et al. 1986). The ability of many species to contribute directly and indirectly to the removal of dissolved contaminants, including nutrients, metals and complex organic molecules, has been widely documented (N.A.S. 1976; E.P.A. 1979). While much of the European research has dealt with a wide variety of wastewaters, most of the North American studies have concentrated on municipal and, to a lesser extent, agricultural wastewaters. Research aimed at evaluating the potential of aquatic macrophytes for treating the wastewaters from the mining industry has apparently received limited attention to date (Salm and Arze, 1982; Brooks et al. 1985). This report presents the results of a preliminary evaluation of the potential usefulness of duckweeds (Lemna minor and Spirodela polyrhiza) for removal of nutrients from surface coal mine drainage water.

The use of explosives in surface mining has resulted in a measurable increase in nitrogen compounds in water bodies receiving the mine drainage. A study conducted in southeastern British Columbia has identified that nitrogen compounds - nitrate, ammonia and nitrite - discharged in mine drainage water, originated mostly from ammonium nitrate explosives (Pommen, 1983). Downstream monitoring of nitrogen levels in the receiving waters revealed nitrate concentrations as high as 11 mg NO_3 -N/L, compared to upstream levels of less than 0.3 mg NO_3 -N/L. Similarly, total ammonia and nitrite concentrations downstream were found to reach 5 mg NH_4 -N/L and 50 ug NO_2 -N/L, respectively, compared to upstream levels of <0.1 mg NH_4 -N/L and <10 ug NO_2 -N/L, respectively. The high nitrate levels at times exceeded the maximum acceptable concentration for drinking water, and the ammonia and nitrite concentrations, though not acutely toxic, were considered high enough to have sub-lethal effects on trout.

5.1.2 Information needs

Concern over the adverse environmental effects of nitrogen compounds from coal mining operations is prompting the search for means of reducing the discharge of nitrogen from surface mines. Conventional nitrogen treatment technologies are unlikely to be adopted because of their high cost. In this light, the investigation of non-conventional or innovative wastewater treatment technologies appears to warrant further emphasis. The information available on the use of aquatic plants in nutrient removal from wastewaters suggests that this concept may hold promise for use with surface mine drainage water.

A preliminary laboratory research program was designed with the aim of assessing the usefulness of macrophyte culture for reducing the nutrient concentrations in a typical mining wastewater. Two native duckweed species, Lemna minor and Spirodela polyrhiza, which occur naturally in mixed populations, were selected as the test plants. Their small size (<6 mm diameter), free- floating form, and rapid multiplication rate, were considered to render them well suited to a short term laboratory study. In addition, there was a considerable body of published information on these and other species of the Lemnaceae family (See Chapter 3.0).

5.1.3 Objectives

The specific objectives of the study were as follows:

- to determine if the quantities and relative concentrations of macro-nutrients (N and P) available in effluent from a coal mine settling pond can support duckweed growth;
- to quantify the removal of nitrogen and phosphorus from the water by the duckweed under conditions of static and intermittent wastewater flow;
- to ascertain whether regular cropping of the duckweed biomass could improve the nutrient removal capacity of an unmanaged duckweed population growing on coal mine wastewater.

5.2 Materials and Methods

5.2.1 Experimental design

Two concurrent experiments were carried out: static and flow-through trials. The treatments during both experiments consisted of exposing the wastewater to a duckweed population or mat which was either cropped weekly or not cropped at all. Duckweed-free tubs were covered with aluminum foil to prevent algal growth and served as the controls. The static treatments were not replicated, while all treatments in the flow-through trials were duplicated. Table 5-1 summarizes the experimental designs. The static experiments used a single initial batch of wastewater as the sole nutrient source for duckweed growth (aside from the nutrients already contained in the biomass). The flow-through trials received an intermittent loading of wastewater at a rate which allowed for a complete exchange of water every ten days. The 10-day residence time was selected as a representative summertime retention period experienced in mine drainage settling ponds in B.C. (Ferguson and Kelso, pers. com.).

5.2.2 Duckweed stocks

The duckweed stocks were clones of L. minor and S. polyrhiza originally collected in 1983 from an agricultural drainage ditch in the vicinity of Pitt Meadows, British Columbia (Alouette watershed). Species identification was based on the keys of Fassett (1957) and Muenscher (1972). In nature, the two species occur in combination, with Lemna being dominant.

TABLE 5-1

SUMMARY OF THE EXPERIMENTAL DESIGN

TREATMENTS	STATIC	FLOWING (10d HRT)
WITH DUCKWEED		
- CROPPED (5% area/week)	n[a] = 1	n = 2
- UNCROPPED	n = 1	n = 2
WITHOUT DUCKWEED (control)	n = 1	n = 2
DURATION (days)	78	69

a n = number of experimental units

5.2.3 Wastewater

Drainage water from a coal mine (Westar Ltd.) near Sparwood in southeastern British Columbia was used for the study. The water was collected in February, 1986, at the point of discharge into the Bodie Creek settling pond, and transported to Vancouver in polyethylene-lined barrels. (The pond was under winter ice at the time.) The water was stored at room temperature adjacent to the test apparatus.

Due to NO_3 -N concentrations in the original wastewater being lower than expected, the influent to the flow- through channels was conditioned with KNO_3. This occurred after the first 10-day flushing period, to obtain an NO_3-N concentration of approximately 12 mg/L. The addition of nitrogen occurred on the 18th day but the first subsequent sampling took place on the 24th day. No KNO_3 was added to the static experiment.

5.2.4 Apparatus

The plants were cultured in opaque plastic tubs, 33 cm long x 29 cm wide x 15 cm deep, containing a wastewater working volume of 7 L. Water depth was 7.5 cm and the surface area available for duckweed growth was approximately 940 cm^2. Each tub was initially stocked with 25 g, fresh weight, of duckweed. The static trials did not require modification of the tubs. For the static trials, evaporative water losses were measured and replaced weekly with tapwater.

An inlet, outlet and flow-control baffle were provided for the tubs used in the flow-through trials, as shown in Figure 5-1. A 10 day hydraulic retention time (HRT) was maintained by means of an electrically timed peristaltic pump. The overflow from each tub was collected in plastic bottles held in a refrigerated container.

The entire experimental setup was located in the heated greenhouse (temperature range of 18^0 - 30^0 C) of the University of British Columbia Plant Science Department. Natural lighting (late February through May) was supplemented with "cool white" fluorescent lamps placed 40 cm above the culture tubs. The 14/10 hour light/dark photoperiod was electrically timed to approximate summer illumination (05:00 to 19:00 hours).

5.2.5 Sampling and chemical analyses

A 50 ml aliquot of the common influent and of effluent from each tub was collected daily (5 to 6 days per week). A weekly composite sample was prepared from the pooled subsamples and stored at 4^0 C, without chemical preservatives. The water was analyzed for ammonia (NH_3 -N), nitrite plus nitrate (NO_2+NO_3 -N), total Kjeldahl nitrogen (TKN), ortho phosphate (ortho-P) and total phosphorus (TP), using a Technicon Auto Analyzer II. Preparation of TKN and TP extracts was carried out according to the method of Schumann (1973), using a 5.0 mL aliquot. Hydrogen ion concentration was measured periodically using a digital pH meter. (Triplicate water samples were also periodically collected by the Environmental Protection

Figure 5-1(a). Experimental setup for greenhouse duckweed culture consisting of six experimental units. Aluminum foil covers the two duckweed-free controls and the influent delivery tubes, to prevent algal growth. Multi-channel peristaltic pump is at lower right. Influent resevoir (white) and effluent storage container (blue) are just visible behind rows of tubs. One lamp housing has been removed to allow photography.

Figure 5-1(b). Detail of duckweed culture unit. Wooden brace supports transparent acrylic plastic baffle. Inlet is at upper right and outlet (not visible) is underwater at lower right.

Service for analysis at the Pacific Region laboratory; See Section 5.2.6.1.)

Weekly duckweed cropping involved removing all the plant biomass from 5% of the tub surface area. On each cropping occasion, a calibrated plastic frame was placed on the duckweed mat, and all the plants enclosed within the frame were harvested with a small dipnet. A different location within each tub was harvested each time. The wet duckweed was then dewatered by centrifuging and transferred to a preweighed paper bag. Fresh and dry weights, respectively, were measured before and after drying to constant weight at 70^0 C.

TKN and TP of the dry plant tissue was determined monthly for the cropped duckweed, and, for the uncropped mats, at the beginning, middle and end of the experiment. Tissue sample preparation involved digestion of 0.2 g of dried plant material in the same manner as for the water samples. The extracts were then diluted prior to colourimetric determination of TKN and TP using the auto-analyzer.

5.2.6 Data analysis and interpretation

5.2.6.1 Chemical analyses

Determinations of water quality were made at the Bio-Resource Engineering laboratory for all samples. Selected samplings and analyses were performed by the Environmental Protection Service (EPS). Discrepancies were found between the total-P measurements made by the two laboratories. These differences were attributed to variations in sampling location, different analytical methods and other variations.

The results reported here are based on the data obtained from the UBC Bio-Resource Engineering laboratory, because of the more complete data set.

5.2.6.2 Duckweed growth rates

Crop growth rates (CGR) were estimated by regression analysis of the cumulative areal dry biomass harvested over time. The slope of the line during any given time interval thus represented the dry matter production rate in grams per m^2 per day.

5.2.6.3 Nutrient removal efficiency

The analysis of nutrient removal was based on two different approaches, in order to account for the relatively brief duration of the experiment, the small container volumes and the high proportion of N and P in the duckweed stocks compared to the wastewater. Overall nutrient removal efficiencies were calculated as the difference between the total nutrient mass added to the culture unit (exclusive of the initial fill water in the flow-through treatment) and the total nutrient mass discharged via the effluent. The relationship is described below in Equation 1:

$$E_o = 100\% \, [(b+c) - e] \, / \, (b+c) \qquad (1)$$

where E_o is the overall removal efficiency; b is the nutrient mass loaded via the pumped influent; c is the nutrient mass stocked as duckweed; and e is the nutrient mass discharged via the effluent. Equation 1 was also used for evaluation of the static experiments.

The contribution of duckweed growth to the overall nutrient removal was calculated as the net nutrient mass uptake by the plants, expressed as a percentage of the total nutrient mass added to the culture unit (Equation 2):

$$E_d = 100\% \, [(f+h) - c] \, / \, (b+c) \qquad (2)$$

where E_d is the nutrient removal efficiency attributable to duckweed; b and c are as in Equation 1; f is the nutrient mass removed via duckweed cropping; and h is the nutrient mass remaining in the unharvested plant tissue at the end of the experiment.

5.3 Results and Discussion

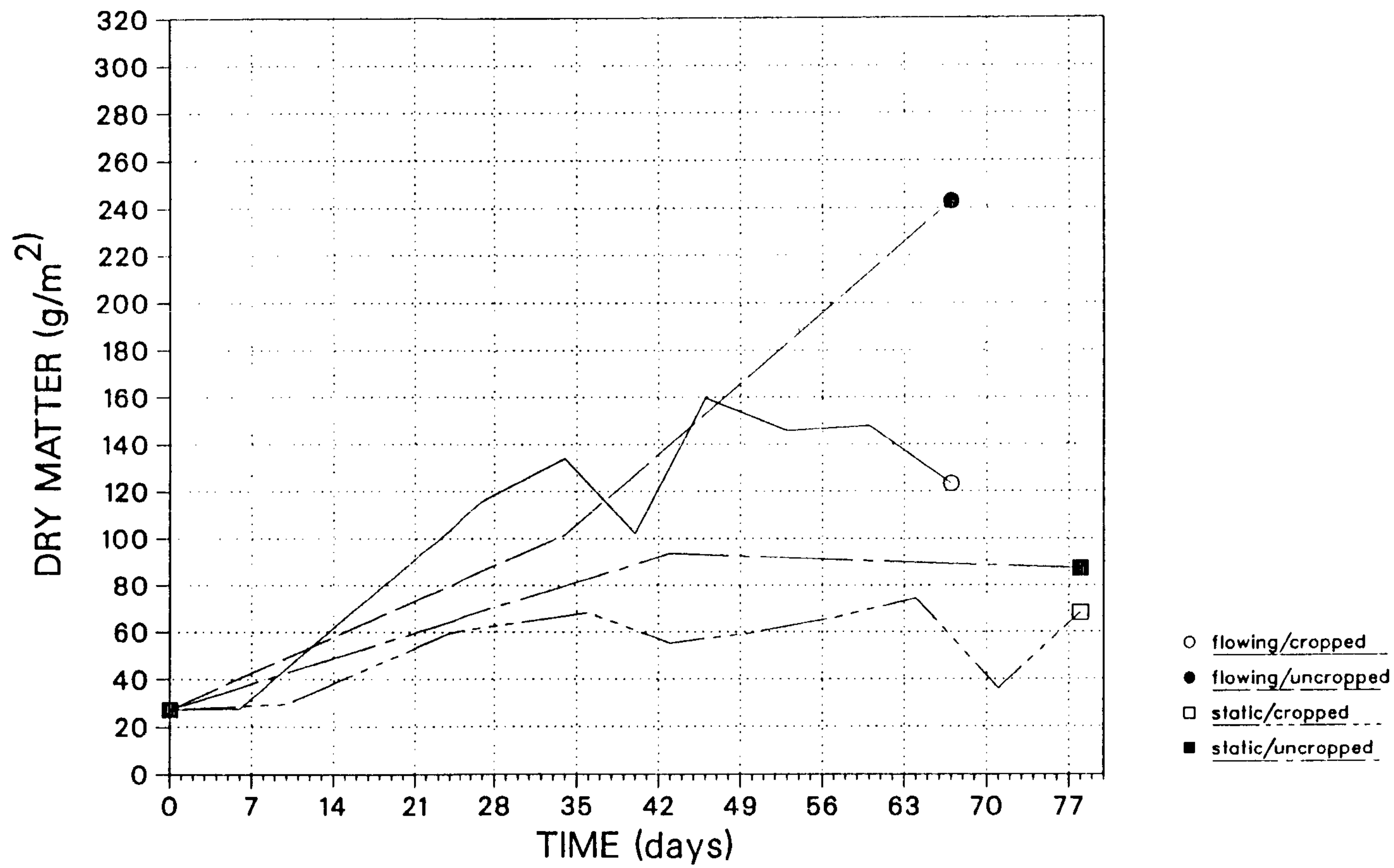
5.3.1 Static experiment

5.3.1.1 Duckweed growth response

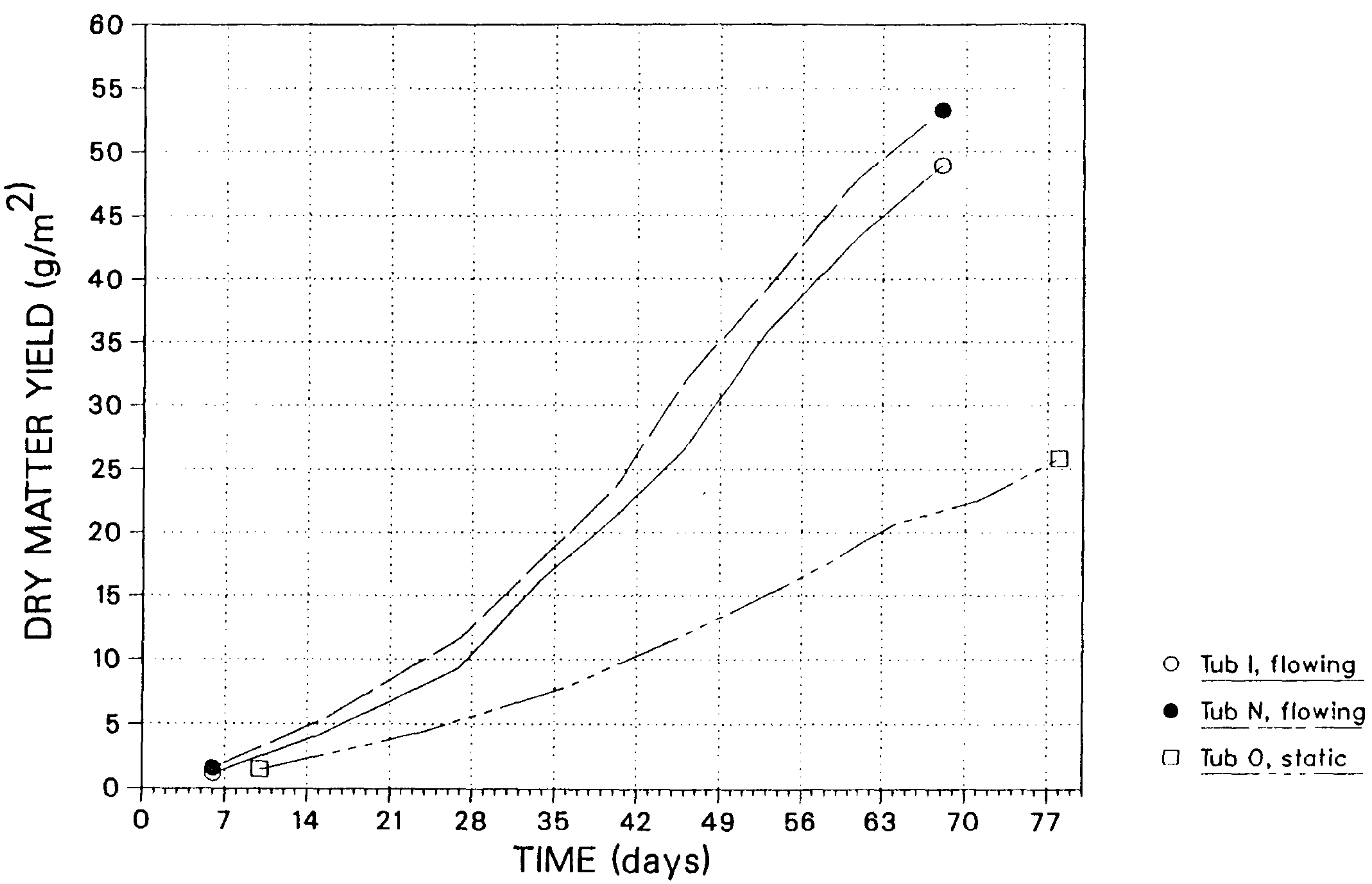
Duckweed growth was measured in both the cropped and uncropped static (batch) treatments. The plants, which were a vibrant green colour when initially stocked, began to turn chlorotic (indicating nitrogen limitation) within three weeks, and to show signs of P limitation (reddish colouration along the mid-portion of the fronds) by the fifth week. The corresponding measured concentration of dissolved (total inorganic) N and total P were 0.044 mg N/L and 0.200 - 0.375 mg P/L at the indicated times. The size of the "daughter" fronds decreased (indicating P limitation) with each succeeding generation, from an initial diameter of approximately 2 - 3 mm in Lemna and 4 - 6 mm in Spirodela, to <1 mm in Lemna and <2 mm in Spirodela by the tenth week. Also, root length increased (indicating low N and P concentrations) with time in both species, from <1 cm at the beginning to >4 cm by the end of the experiment. This led to considerable root entanglement (and subsequent plant submergence and mortality) from harvesting.

The cropped duckweed maintained a less dense cover than the uncropped population (Figure 5-2). From a dry matter starting density (standing crop) of 27.4 g/m^2, the cropped system reached a maximum density of 74.5 g/m^2 after eight weeks, decreasing to 68.1 g/m^2 by the end of the 11-week experimental period. The uncropped system was measured at the beginnning, middle and end of the period, and (from an equal starting density) showed a peak standing crop of 93.6 g/m^2 at week six, decreasing to 87.3 g/m^2 by the end of the experiment.

The rate of biomass production from the cropped static treatment averaged 0.372 $g/m^2/d$ over the whole experiment, with the production rate during the second half being slightly higher than during the first half (Figure 5-3). Reasons for the increase in production with time may be related to the release of nutrients from dying original stock, and possibly to the seasonal



DUCKWEED DENSITY IN THE STATIC AND FLOW-THROUGH EXPERIMENTS
Figure 5-2



CUMULATIVE DUCKWEED DRY WEIGHT PRODUCTION IN STATIC AND FLOW-THROUGH EXPERIMENTS
(CROPPED TREATMENTS ONLY)
Figure 5-3

increase in solar energy, (the vernal equinox occurred during week four of this experiment). Much higher yields have been reported for duckweed growing on fish ponds and nutrient rich wastewaters (see section 5.3.2.2).

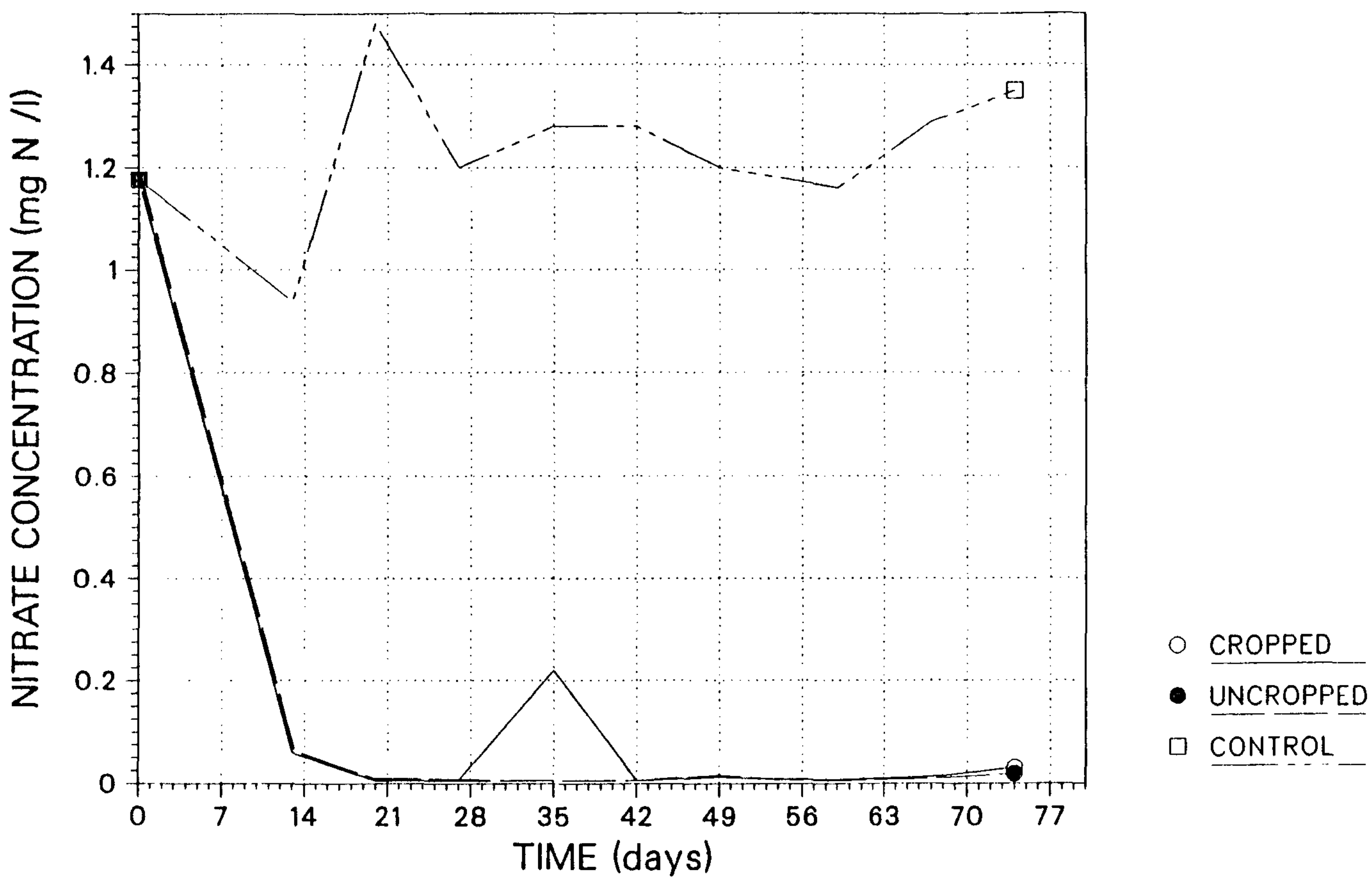
5.3.1.2 Water chemistry

Figures 5-4, 5-5, and 5-6 summarize the changes in the concentrations of nitrate, ammonia and total phosphorus measured in the cropped, uncropped and duckweed free treatments.

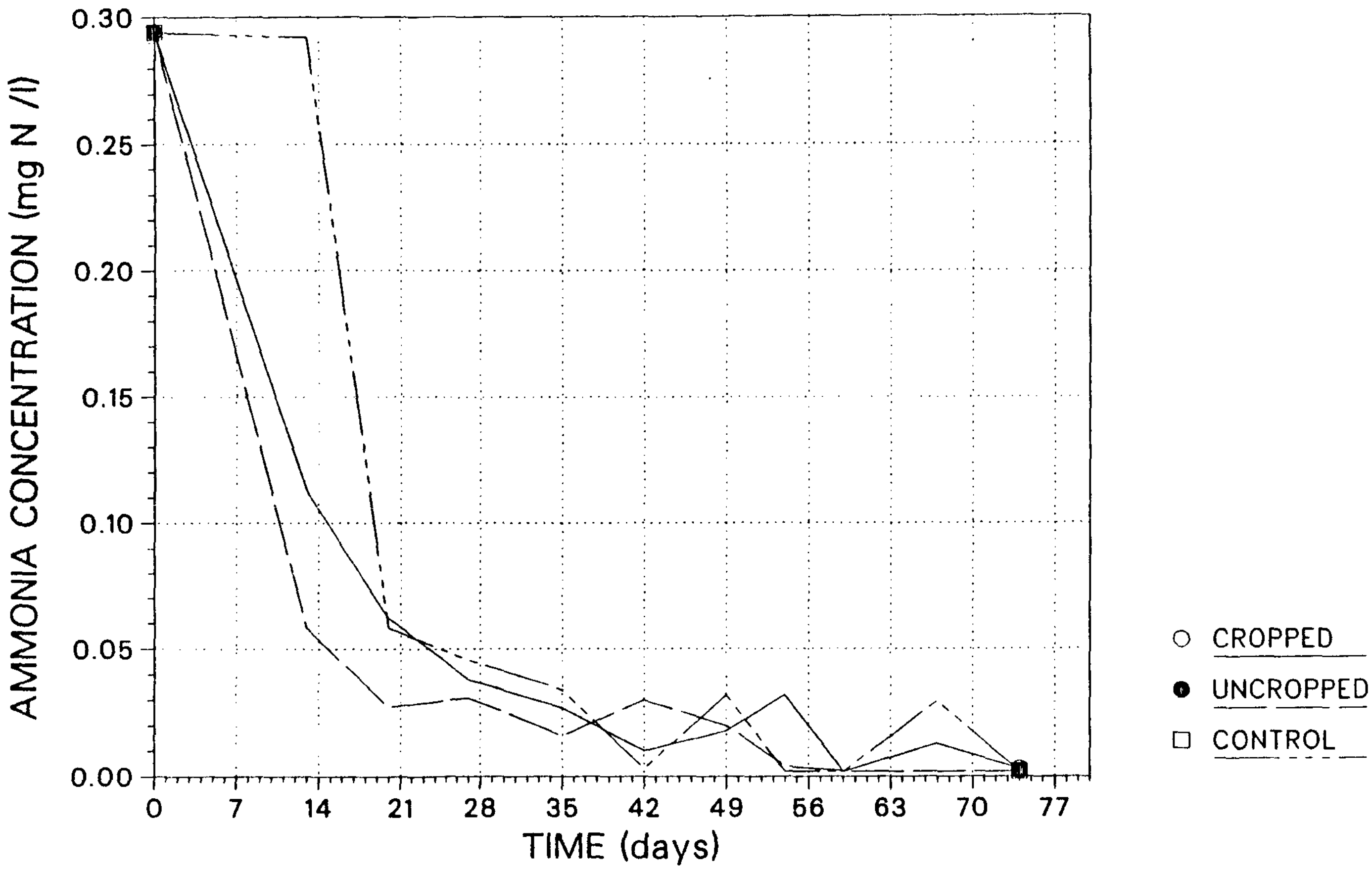
The nitrate curves indicate that the control underwent little change in nitrate concentration over time, compared to the duckweed treatments. There was little difference between the cropped and uncropped units, both reaching the nitrate detection limit (0.006 mg N/L) within the first week.

Ammonia concentrations decreased over time in all the static units, including the duckweed-free control. Given the absence of plants in the latter, the ammonia removal may have been due to nitrification as indicated by the concurrent increase in nitrate.

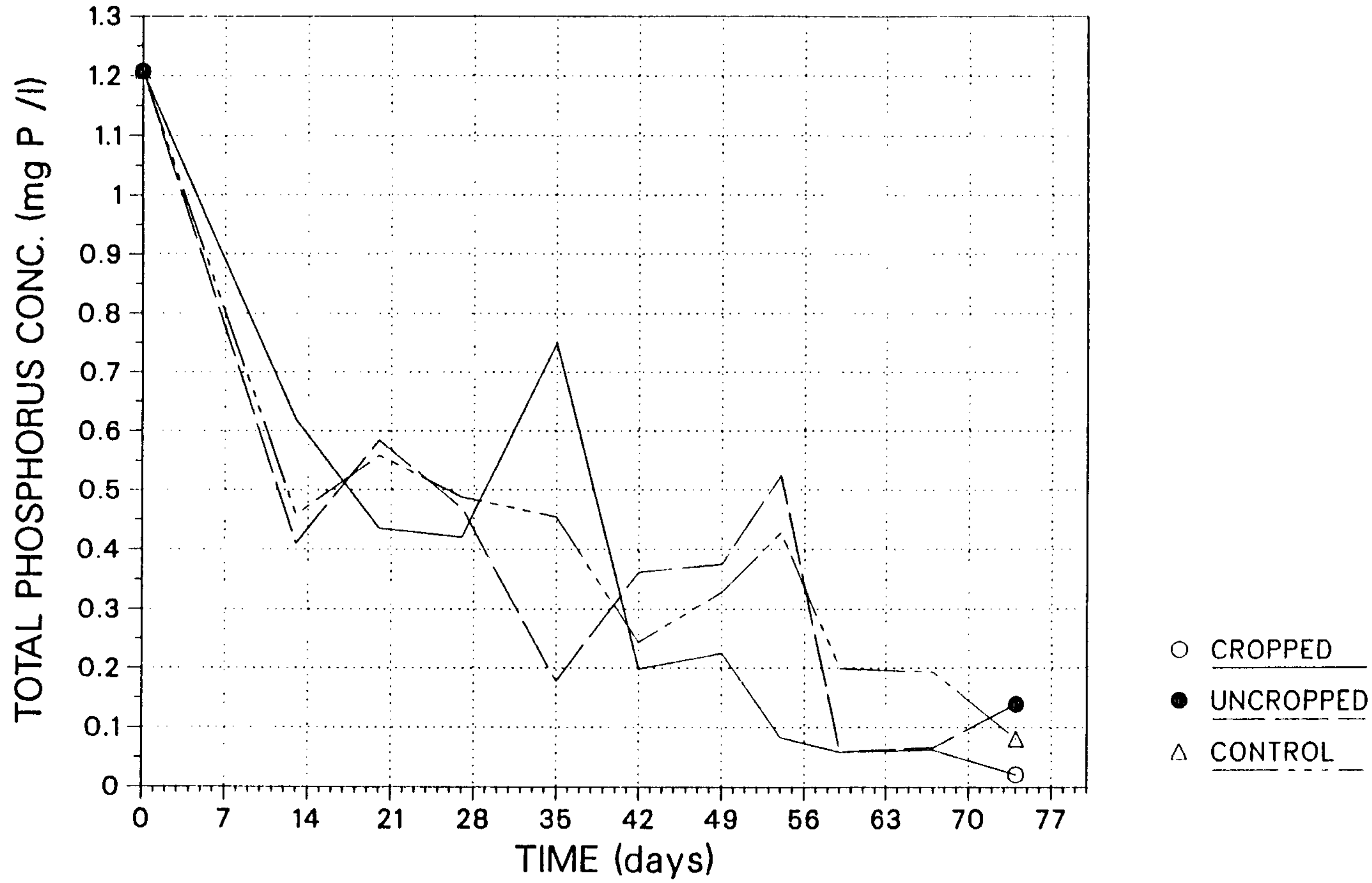
Phosphorus concentration in all treatments decreased with time, the rate of decrease being greatest in the cropped treatment followed by the uncropped treatment and finally the control. The measured final P concentrations were 20.8 ug/L, 139.4 ug/L and 80.7 ug/L, in the cropped, uncropped and duckweed-free containers, respectively. The relatively high final



NITRATE CONCENTRATIONS IN THE STATIC WATER DUCKWEED EXPERIMENTS
Figure 5-4



AMMONIA CONCENTRATIONS IN THE STATIC WATER DUCKWEED EXPERIMENTS
Figure 5-5



PHOSPHOROUS CONCENTRATIONS IN THE STATIC WATER DUCKWEED EXPERIMENTS
Figure 5-6

concentration in the uncropped tub may reflect the release of P from dead duckweed. Considerable variability was evident in the phosphorus data (Figure 5-6). Phosphorus losses in the control were most likely due to adsorption on the container walls, and possibly to some bacterial uptake.

5.3.1.3 Evapotranspiration

The rates of water loss from the duckweed covered units due to evaporation and transpiration are summarized in Table 5-2. As shown, the average water loss rate from the cropped system was slightly less than from the uncropped unit (approximately 179 vs 187 mL/d), though the difference is not statistically significant. The plant free unit was kept covered to prevent nuisance algal growth, and showed an evaporative water loss rate of only 22 mL/d. On an areal basis, the overall average rate of evaporative water loss from the static duckweed cultures was equivalent to 1949 mL/m^2/d. Water loss rates of 1705 - 3340 ml/m^2/d have been measured from open water in a greenhouse environment (Whitehead, unpublished).

5.3.1.4 Nutrient removal performance

Overall Nitrogen removal efficiency (E_o) was 97.8% in the cropped, 98.6% in the uncropped and 8.1% in the control treatment, over the 11-week period (Table 5-2). Nitrogen removal resulting from the presence of duckweed was substantial. Nitrogen removal (E_d) by duckweed from the initial fill water, in both the cropped and uncropped treatments, amounted to >800% of

TABLE 5-2

RATES OF EVAPORATIVE WATER LOSS
FROM CROPPED AND UNCROPPED DUCKWEED CULTURES
GROWING ON STATIC COAL MINE DRAINAGE WATER

TREATMENT	TIME INTERVAL (day no.s)	RATE OF WATER LOSS (ml $H_2O/m^2.d$)
CROPPED	5 - 20	2026
	20 - 76	1892
	5 - 76	1903
UNCROPPED	5 - 20	1880
	20 - 76	2023
	5 - 76	1995
CONTROL (covered)	5 - 20	289
	20 - 76	207
	5 - 76	230

TABLE 5-3

NITROGEN MASS BALANCE AND REMOVAL EFFICIENCIES FROM EXPERIMENTAL DUCKWEED CULTURES GROWN ON STATIC COAL MINE WASTEWATER

	TREATMENT		
	CROPPED	UNCROPPED	CONTROL
NITROGEN INPUT (mg)			
(a) Initial Water	10.297	10.297	10.297
(b) Initial Duckweed	78.813	78.813	0.0
(d) Initial N Input	89.110	89.110	10.297
NITROGEN OUTPUT (mg)			
(f) Duckweed Harvest	74.879	0.0	0.0
(g) Remaining Water	0.228	0.147	9.464
(h) Remaining Duckweed	209.198	339.737	0.0
(i) Total N Output	284.305	339.884	9.464
REMOVAL EFFICIENCIES (%)			
Overall:			
E_o = [(a-g)/a].100% =	97.8%	98.6%	8.1%
Duckweed Related:			
E_d = [(f+h)-b]/a.100% =	1993.4%	896.3%	-

the initial aqueous N (Table 5-3). This result suggests either gross analytical error or substantial unmeasured N additions to the duckweed covered experimental units, via microbial N fixation and faunal imports (see section 5.3.2.4, below).

Overall P removal efficiency was 94.5% in the cropped, 65.5% in the uncropped and 80.1% in the plant-free treatment (Table 5-4). Given that approximately 80% of the initial aqueous P could be removed by non-duckweed (physical- chemical) processes, the lower removal efficiency in the uncropped system appears to reflect the contribution of P to the water from the unharvested duckweed mat. It is evident from the data that cropping improved P removal performance. Duckweed removal of P from the initial fill water was 77.7% in the cropped and 40.4% in the uncropped system (Table 5-4). The beneficial effect of cropping on P removal in the static experiment is clearly evident.

5.3.2 Flow-through experiment

5.3.2.1 Water chemistry

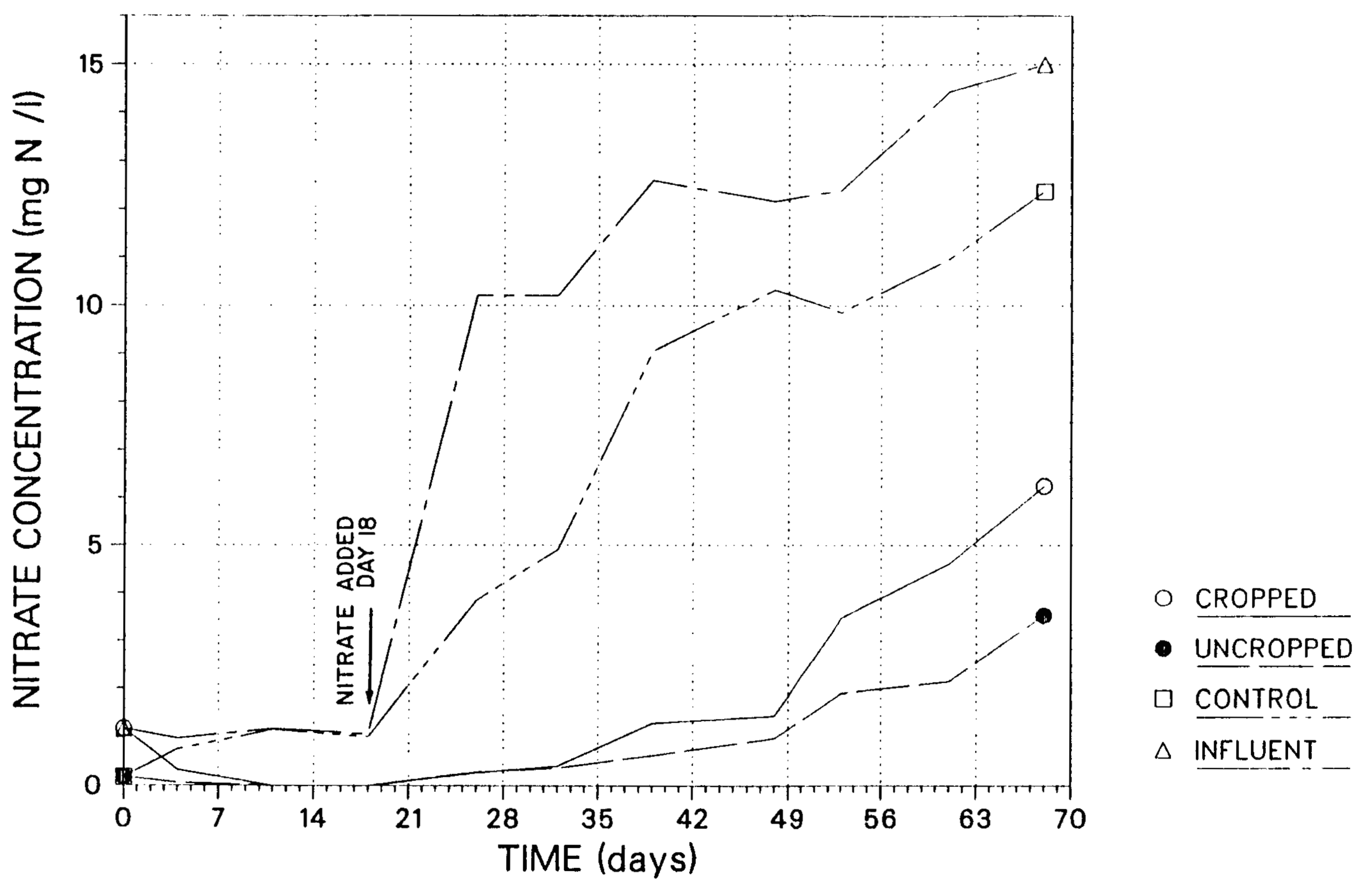
Figures 5-7, 5-8 and 5-9 summarize the concentrations of nitrate, ammonia and total phosphorus measured in the influent, as well as in the cropped, uncropped and duckweed free flow-through treatments.

Influent nitrate concentrations prior to NO_3 enrichment were approximately 1 mg N/L. Effluent nitrate content in the duckweed cultures was reduced to below detection limits (0.006 mg N/L) within 11 days

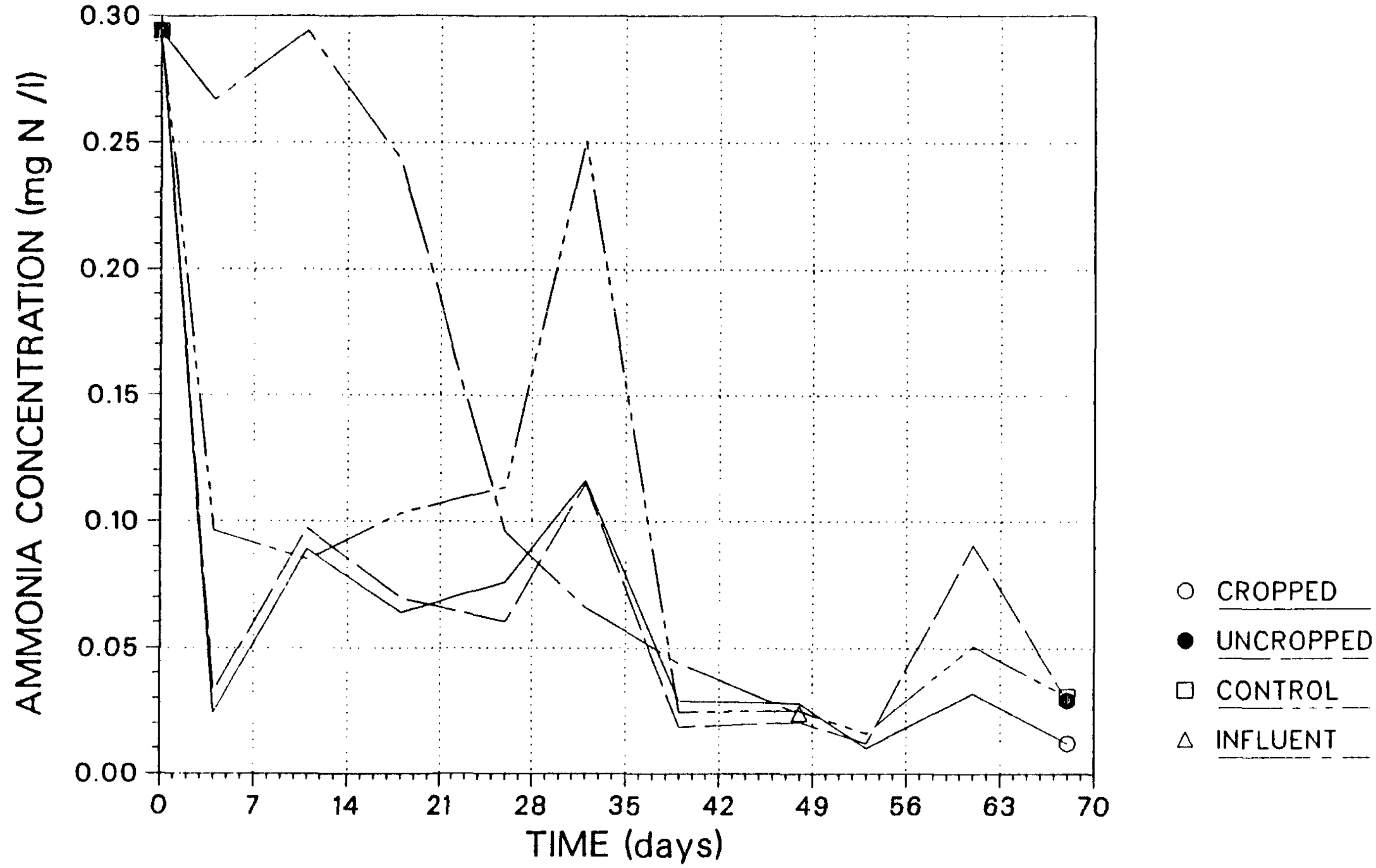
TABLE 5-4

PHOSPHORUS MASS BALANCE AND REMOVAL EFFICIENCIES
FROM EXPERIMENTAL DUCKWEED CULTURES
GROWN ON STATIC COAL MINE WASTEWATER

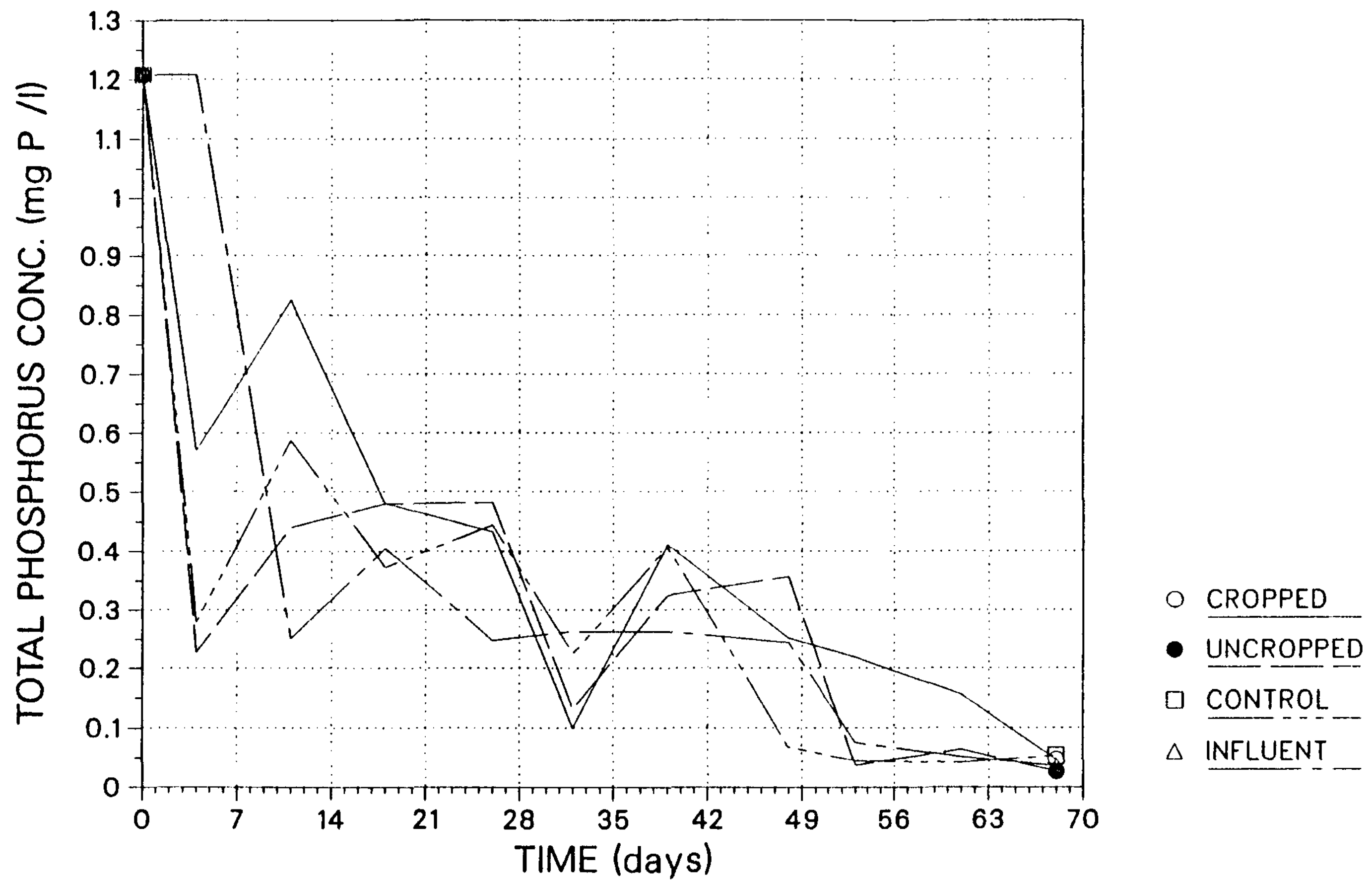
	TREATMENT		
	CROPPED	UNCROPPED	CONTROL
PHOSPHORUS INPUT (mg)			
(a) Initial Water	2.835	2.835	2.835
(b) Initial Duckweed	5.666	5.666	0.0
(d) Initial P Input	8.501	8.501	2.835
PHOSPHORUS OUTPUT (mg)			
(f) Duckweed Harvest	3.856	0.0	0.0
(g) Remaining Water	0.146	0.976	0.565
(h) Remaining Duckweed	4.014	6.811	0.0
(i) Total P Output	8.016	7.787	0.565
REMOVAL EFFICIENCIES (%)			
Overall:			
$E_o = [(a-g)/a].100\% =$	94.5%	65.5%	80.1%
Duckweed Related:			
$E_d = [(f+h)-b]/a.100\% =$	77.7%	40.4%	-



NITRATE CONCENTRATIONS IN THE FLOW-THROUGH DUCKWEED EXPERIMENTS
Figure 5-7



AMMONIA CONCENTRATIONS IN THE FLOW-THROUGH DUCKWEED EXPERIMENTS
Figure 5-8



PHOSPHORUS CONCENTRATIONS IN THE FLOW-THROUGH DUCKWEED EXPERIMENTS
Figure 5-9

(i.e. one retention period). Effluent nitrate concentrations during the "spiked" period steadily increased in all treatments but was more noticeable in the cropped than uncropped treatment effluent. The discharge nitrate concentration from the control approached the influent concentration approximately two weeks after the initiation of spiking, indicating that the actual retention time was, in fact, similar to the theoretical value of 10 days.

The marked increase in effluent nitrate concentrations from the duckweed treatments during the final three weeks is partly related to the rising influent nitrate levels. This does not however explain the increase in the duckweed treatment between days 48 and 52. These data would appear to reflect either a decrease in the nitrate uptake rate or nitrification of additional N, (more likely from decomposing plant tissue than from the water), or both. The possibilities of reduced growth rate due to P limitation and of N fixation within the duckweed mats are discussed further in subsequent sections.

Ammonia concentration in the influent was very low (<0.3 mg/L), and generally decreased with time, rising again during the final two weeks (Figure 5-8). Effluent ammonia concentrations from the duckweed and control treatments were consistently lower than influent levels. The cause of a peak in effluent NH_3-N concentrations on day 32 (Figure 5-8) is uncertain; this peak coincides with a marked drop in P (Figure 5-9). There was no marked difference between the control, cropped, and uncropped treatment effluent ammonia concentrations.

TABLE 5-5

ORGANIC NITROGEN CONTENT (mg/L)
IN THE INFLUENT AND EFFLUENTS,
MEASURED AT THE END OF THE EXPERIMENTS

	TREATMENT					
	Cropped		Uncropped		Control	
	static	flowing	static	flowing	static	flowing
Influent	--	0.160	--	0.160	--	0.160
Effluent	2.185	1.481	2.285	1.244	1.608	0.235

The above results suggest that similar ammonia removal mechanisms were operative in the planted and unplanted systems. Bacterial uptake (conversion to organic N) and volatilization are two possible mechanisms. The alkaline pH 8.3 of the wastewater would theoretically favour some ammonia volatilization. The higher final organic N content in the control treatment relative to the influent (Table 5-5) provides evidence of bacterial growth. The absence of higher final effluent ammonia concentrations in the duckweed systems than in the control indicates that similar processes are probably underway in both containers and that any nitrogen fixation taking place did not generate ammonia.

Influent total P concentrations generally decreased throughout the experiment, from an intial value of about 1.2 mg P/L to a final value of less than 0.4 mg P/L. A similar trend was evident in the effluents from all treatments. A marked decline in P concentration was evident in the influent and effluents after seven weeks. The effluent from the cropped treatment contained more P than that from the uncropped treatment during the final three weeks (Figure 5-9).

5.3.2.2 Duckweed growth response

Figures 5-2 and 5-3, respectively, show the changes in duckweed standing biomass and cumulative production over time. From an initial biomass density of 27.4 g/m^2, dry matter basis, the cropped population peaked at about 160 g/m^2 after seven weeks, declining slightly thereafter. The uncropped mats showed a continuous increase in density throughout the experiment, peaking at about 240 g/m^2 after ten weeks.

The average dry matter content of the duckweed was 13.4%. This value is higher than the 4-8% dry matter content commonly reported for duckweed grown on nutrient rich wastewaters, and is higher than the value (10.3%) obtained under static experiments.

The results show that the 5% weekly cropping rate was insufficient to maintain the highest plant productivity. At the time maximum plant density was reached, crowding slowed plant production and the standing biomass stabilized. It appears, therefore, that higher cropping rates may have been sustainable, provided that P limitation did not develop (see discussion of nutrient removal efficiency, below).

Figure 5-3 presents the cumulative dry matter production per square meter over time, for the cropped and uncropped treatments. As can be seen, the cropped units yielded, on average, approximately two times more biomass than the uncropped units, despite the higher plant density in the latter at the time of the final harvest.

Table 5-6 summarizes the productivity of the cropped duckweed during three intervals within the experimental period. The slowest crop growth rate of 0.316 $g/m^2/d$ in the flowing treatments was obtained before the addition of synthetic nitrate. The increased availability of nitrate after day 18 resulted in a growth rate of 1.032 $g/m^2/d$, over three times that of

TABLE 5-6

DUCKWEED CROP GROWTH RATES FROM HARVESTED POPULATIONS
GROWN ON STATIC OR FLOWING COAL MINE WASTEWATER

TREATMENT	TIME INTERVAL (day no.s)	CROP GROWTH RATE (g dry matter/m^2.d)	COEFFICIENT OF DETERMINATION
STATIC	0 - 24	0.188	0.991
	24 - 78	0.409	0.993
FLOWING (HRT = 10d)	0 - 15	0.316	0.959
	15 - 48*	0.772	0.960
	41 - 69*	1.032	0.969

* after NO_3 enrichment of the influent.

the "unspiked" condition. The slopes of the curves in Figure 5-3 indicate that the growth rate continued to increase during the nitrate enriched period.

Studies in Israel (Oron et al. 1986) using settled domestic sewage have measured yields of 3-15 g/m^2/d (dry weight). In Czechoslovakia, Rejmankova (in Culley et al. 1981) reported duckweed yields of 3.14 g (dry matter)/m^2/d from natural fish ponds. The duckweed yields that have been obtained in the present study are much lower than those reported above. It would appear that this may have been due to the dilute (nutrient poor) nature of the wastewater, and possibly to the lower availability of radiant energy.

5.3.2.3 Nutrient removal efficiencies

Overall nitrogen removal performance in the flow-through experiments (E_o) was highest in the uncropped duckweed system (91.2%), followed by the cropped treatment (83.6%) and the control (29.1%), as shown in Table 5-7. Interestingly, while approximately 30% of the influent N could be removed via non-duckweed processes, cropping appeared to have a slightly detrimental effect on overall N removal because of the lowered tissue N content with time. The reasons for this are not clear, but may be related to a phosphorus limitation, as discussed below.

The highest overall P removal (E_o) was obtained from the uncropped duckweed (31.3%), followed by the cropped treatment (14.5%) and the control (8.8%) (Table 5-8). Evidently, under the existing experimental conditions,

TABLE 5-7

NITROGEN MASS BALANCE AND REMOVAL EFFICIENCIES FROM EXPERIMENTAL DUCKWEED CULTURES GROWN ON FLOWING* COAL MINE WASTEWATER

	TREATMENT		
	CROPPED	UNCROPPED	CONTROL
NITROGEN INPUT (mg)			
(a) Initial Water	8.715	8.715	8.715
(b) Initial Duckweed	78.813	78.813	0.0
(c) Pumped Water	256.214	256.214	245.214
(d) Total N Input	343.742	343.742	264.929
NITROGEN OUTPUT (mg)			
(e) Discharged Water	53.003	29.562	181.593
(f) Duckweed Harvest	228.432	19.191	0.0
(g) Remaining Water	43.833	25.081	86.840
(h) Remaining Duckweed	525.073	1020.076	0.0
(i) Total N Output	850.341	1093.910	268.433
REMOVAL EFFICIENCIES (%)			
Overall:			
$E_o = [(b+c)-e]/(b+c).100\% =$	83.6%	91.2%	29.1%
Duckweed Related:			
$E_d = [(f+h)-b]/(b+c).100\% =$	201.4%	310.2%	-

* Hydraulic retention time = 10 days.

TABLE 5-8

PHOSPHORUS MASS BALANCE AND REMOVAL EFFICIENCIES
FROM EXPERIMENTAL DUCKWEED CULTURES
GROWN ON FLOWING* COAL MINE WASTEWATER

	TREATMENT		
	CROPPED	UNCROPPED	CONTROL
PHOSPHORUS INPUT (mg)			
(a) Initial Water	8.456	8.456	8.456
(b) Initial Duckweed	5.666	5.666	0.0
(c) Pumped Water	12.301	12.301	12.301
(d) Total P Input	26.423	26.423	20.757
PHOSPHORUS OUTPUT (mg)			
(e) Duckweed Water	15.385	12.347	11.224
(f) Duckweed Harvest	4.302	0.695	0.0
(g) Remaining Water	0.326	0.184	0.372
(h) Remaining Duckweed	5.778	14.649	0.0
(i) Total P Output	25.764	27.875	11.596
REMOVAL EFFICIENCIES (%)			
Overall:			
$E_o = [(b+c)-e]/(b+c).100\% =$	14.5%	31.3%	8.8%
Duckweed Related:			
$E_d = [(f+h)-b]/(b+c).100\% =$	24.6%	53.9%	-

* hydraulic retention time = 10 days.

approximately 15% less P was removed in the cropped than in the uncropped systems, suggesting that cropping was in some way disadvantageous.

One possible explanation for the higher overall performance in the absence of cropping is P and/or micronutrient limitation in the cropped population, due to P removal via harvesting. Examination of the total biomass production in the two duckweed treatments shows that there was a greater net production from the uncropped than the cropped systems, despite "crowding". Also, the content of P in the biomass decreased over time, with a lower final content being measured in the cropped than in the uncropped populations. This suggests that the available P and/or micronutrient reserves within the original plant stock became redistributed among the daughter plants and that there was less P or micronutrients available to the cropped population by the end of the experiment.

Nitrogen removal performance of the duckweed (E_d) was 201.4% and 310.2% of the loaded N, respectively, in the cropped and uncropped systems. These figures indicate that considerably more N exited the systems via duckweed production than was loaded, particularly in the uncropped treatment. Similar results were obtained from the static experiment. The implications of the data are discussed below in section 5.3.2.5.

The duckweed-related P removal (E_d) from the uncropped treatment was 53.9%, compared to 24.6% from the cropped system. Thus, over twice as much P was removed via the uncropped than via the cropped duckweed. The contribution of P to the water from the duckweed probably accounts for E_o being lower than E_d.

5.3.2.4 Nutrient mass flux

Table 5-9 summarizes the rates of nutrient input (via the influent) and output (via the removed plant matter and the effluent). Prior to nitrate "spiking" the nitrogen discharge in the effluent from the cropped treatment amounted to 6.7% of the loading rate, indicating a high degree of N removal. The corresponding fractions for the uncropped and plant-free treatment were 56.6% and 84.5%, respectively. During the nitrate-enriched or "spiked" period, the nitrogen discharge rates represented 22.9%, 12.3% and 76.2%, respectively, of the loading to the cropped, uncropped and plant free systems. The N removal rate via duckweed was 156.2% before, and 88.1% of the loading rate after "spiking". These results indicate that approximately 15% and 24% of the N removal rate was attributable to non-duckweed causes, before and after "spiking", respectively.

The P flux, "prespiking", via the effluents represented 138.3%, 88.0% and 96.1% of the P loading rate to the cropped, uncropped and control treatments, respectively (Table 5-10). The higher P outflow rate in the cropped duckweed treatment may represent the contribution of P to the water attributable to "leakage" from the plants (plus, as in all treatments, experimental error and contamination). The corresponding values for the "spiked" period were 123.5%, 111.8%, and 80.0%. The P

TABLE 5-9

NITROGEN FLUX RATES[a] IN EXPERIMENTAL DUCKWEED CULTURES GROWN ON COAL MINE WASTEWATER UNDER FLOWING[b] CONDITIONS

SYSTEM COMPARTMENT	TIME INTERVAL (day no's.)	TREATMENT		
		CROPPED (mg/d)	UNCROPPED (mg/d)	CONTROL (mg/d)
INFLUENT	0 - 18[c]	0.555	0.555	0.555
	26 - 66[d]	5.413	5.413	5.413
EFFLUENT	0 - 18	0.037	0.314	0.469
	26 - 66	1.240	0.666	4.123
DUCKWEED	0 - 15	0.867	-	-
HARVEST	16 - 69	4.771	-	-

a divide by $0.094 m^2$ to obtain rates on areal basis, i.e. $mg/m^2.d$.

b hydraulic retention time = 10 days.

c before enrichment with NO_3.

d after enrichment with NO_3.

TABLE 5-10

PHOSPHORUS FLUX RATES[a] IN EXPERIMENTAL DUCKWEED CULTURES GROWN ON COAL MINE WASTEWATER UNDER FLOWING[b] CONDITIONS

SYSTEM COMPARTMENT	TIME INTERVAL (day no's.)	TREATMENT CROPPED (mg/d)	UNCROPPED (mg/d)	CONTROL (mg/d)
INFLUENT	0 - 18[c]	0.332	0.332	0.332
	26 - 66[d]	0.110	0.110	0.110
EFFLUENT	0 - 18	0.459	0.292	0.319
	26 - 66	0.136	0.123	0.088
DUCKWEED HARVEST	0 - 15	0.077	-	-
	16 - 69	0.088	-	-

a divide by $0.094m^2$ to obtain rates on areal basis, i.e. $mg/m^2.d$.
b hydraulic retention time = 10 days.
c before NO_3 enrichment.
d after NO_3 enrichment.

removal rate via duckweed amounted to 23.2% before, and 80.0% of the loading rate after nitrate enrichment. On a rate basis, therefore, approximately 4% and 20% of the P removal rate was attributable to non-duckweed causes before and after "spiking", respectively.

5.3.2.5 Unmeasured nitrogen imports

Excess nitrogen output was measured in the duckweed covered systems, as mentioned above. Such results imply either an underestimation of the total N loading and unloading or chemical analytical error. It is most likely that nitrogen loading was underestimated because of nitrogen fixation occurring in the experimental containers, as experienced by Oron et al. (1984).

Underestimation of N loading is most likely, for two principal reasons. First, dissolved organic nitrogen was not quantified initially, in either the influent or effluents because, at the time, it was considered to be present in insignificant amounts. Analyses performed at the end of the experiment, however, revealed that the remaining unused influent contained 0.160 mg organic N per liter, and that organic N was also present in the effluents, as shown in Table 5-5.

Nevertheless, the additional N mass due to organic N loading amounts to only to 1.2% more than the total N loading based on inorganic-N alone. Since the effluent organic-N concentrations were higher in all treatments, it appears that unmeasured organic-N does not account significantly for the high N output via duckweed.

The second possible cause for actual N loading appearing to be higher than measured involves unmeasured N imports via macrofauna, algal or microbial nitrogen fixation within the duckweed mat. Small flies and spiders were observed in the plant-covered and plant-free tubs. The potential contribution from these sources to N loading was not quantified. Similarly, the potential contribution of non-duckweed biota to N removal was not quantified. The greater N removal in the uncropped than cropped treatment (Table 5-7) might reflect a larger population of attached microbiota in the uncropped (i.e. undisturbed) duckweed mat. Zuberer (1982) has reported that microbial populations associated with duckweed mats are capable of fixing atmospheric N (acetylene reduction). Reportedly, populations of diazotrophic cyanobacteria and heterotrophic bacteria are enhanced in dense duckweed mats, and have been measured to provide 15 - 20% of the N used for duckweed growth. Oron et al. (1984) have also reported evidence of N fixation in duckweed mats, where the N mass balance showed N output amounting to 160% of input. Nitrogen fixation in such systems may account for some of the increased nitrogen loading.

5.3.3 Implications for large scale applications

Phosphorus limitation and cropping frequency are two related factors which stand out from the present study as having important implications relative to the application of duckweed culture for mine drainage water treatment.

Nitrogen removal efficiency was higher in the uncropped than cropped treatments of both the static and flow-through systems. Under static conditions, cropping removed considerably more P than the uncropped treatment. Tissue N content in the cropped and uncropped treatments increased throughout the study in both hydraulic regimes, while P content decreased. Results suggest that nutrient removal via frequent plant harvests may have led to the development of a nutrient deficiency in the growing medium. The available reserves of phosphorus, and possibly other micronutrients, were depleted more rapidly in the cropped than in the uncropped treatments. If this interpretation of the data is correct, then it would appear that a single or few harvests during the growing season would yield greater N and P removal from nutrient-poor coal mine wastewaters than the frequent cropping that is advantageous with more nutrient-rich effluents.

The present study has not addressed other factors which will be of importance in larger scale systems. For example, water pH and micronutrient content, impoundment geometry, wind and possibly pests, will require site specific work. Harvesting methods and duckweed crop re-use alternatives will also have to be examined further.

5.4 Conclusions

5.4.1 Evaluation of the concept

The use of duckweed cultures for nitrogen removal from surface coal mine drainage water has been shown by this study to be technically feasible under laboratory conditions.

The experimental, continous-flow systems (10 day hydraulic retention time) were capable of reducing the nitrate concentration from a median influent value approximately 12 mg N/L to <1 mg N/L under uncropped conditions and to <2 mg N/L with 5% weekly cropping of the water surface. The static (non-flowing) systems were capable of reducing wastewater nitrate concentrations from approximately 1.5 mg N/L to <0.01 mg N/L within 20 days. Duckweed cropping, at the rate employed, did not have an effect on nitrate removal under static conditions.

5.4.2 Duckweed survival and production

The duckweed species *Lemna minor* and *Spirodela polyrhiza* were able to grow and reproduce on the coal mine wastewater provided, under conditions of static or continuous hydraulic loading.

Symptoms of phosphorus and nitrogen deficiency were evident, the former particularly in *Lemna* and the latter in both species. On the whole, *Lemna* appeared more "healthy" than *Spirodela*, though no special advantage was observed in one species over the other.

The standing crop measured in the cropped systems was always lower than in the uncropped systems. Under static conditions the maximum plant densities achieved were 160 and 240 g/m^2, dry matter, on the cropped and uncropped treatments, respectively. Under flow-through conditions the highest plant densities achieved were 74.5 and 93.6 g/m^2 in the cropped and uncropped treatments, respectively.

The increases in plant density despite 5% weekly cropping indicate the higher cropping rates, and therefore higher dry matter yields, may be sustainable.

Duckweed production was similar in the static and flowing systems prior to nitrate enrichment. Without nitrate enrichment, average production in the cropped static treatment was 0.37 $g/m^2/d$ of dry matter compared to 0.32 $g/m^2/d$ in the cropped flow-through treatment. The average production from the cropped flowing systems after nitrate enrichment was 0.90 $g/m^2/d$, almost three times the pre-spiking rate.

Evaporation rates from the duckweed covered wastewater averaged 1949 $mL/m^2/d$. This value is within the range of values recorded from duckweed-free water under similar experimental conditions.

5.4.3 Nutrient uptake and removal efficiency

Nitrogen uptake rates via duckweed harvests under flow-through conditions attained levels of 9.2 $mg/m^2/d$ on natural ("unspiked") wastewater and 50.8 $mg/m^2/d$ on nitrate enriched wastewater. Phosphorus uptake rates under the corresponding conditions were 0.8 and 0.9 mg/m^2/d, respectively.

Nutrient removal from the wastewater was greater in the presence than in the absence of duckweed (Table 5-11). The only exception was P removal under static conditions, where more P was removed in the control than in the uncropped treatment. Evidence of N and P release from the plants into the water was found. Though requiring confirmation, the data also suggested that considerable N fixation may have been taking place in association with the duckweed mats.

The highest overall nitrogen removal performance of 98.6% was attained in the uncropped batch treatment. Duckweed cropping appeared to have no significant effect on overall N removal under static conditions. Under flowing conditions, the maximum N removal efficiency of 91.2% was also obtained from the uncropped systems. Cropping appeared to slightly reduce N removal.

The maximum overall P removal efficiency of 94.5% was obtained from the cropped static treatment. Phosphorus removal was significantly lower (65.6%) in the uncropped static treatment. With continuous flow, the highest P removal performance of 31.3% was attained under uncropped conditions.

5.4.4 Effect of cropping

The negative effect of cropping on nutrient removal efficiency appears to be related to the creation of

TABLE 5-11

SUMMARY OF OVERALL NUTRIENT REMOVAL EFFICIENCIES

TREATMENT		NUTRIENT REMOVAL EFFICIENCY (%)	
		NITROGEN	PHOSPHORUS
WITH DUCKWEED:			
CROPPED	- Flowing	83.6	14.5
	- Static	97.8	94.5
UNCROPPED	- Flowing	91.2	31.3
	- Static	98.6	65.6
WITHOUT DUCKWEED:			
	- Flowing	29.1	8.8
	- Static	8.1	80.1

nutrient limiting conditions. The latter may result from a depletion of the P (and possibly micronutrient) reserves within the system.

6.0 CONCLUSIONS

Literature regarding aquatic plant treatment systems reflected the emphasis on the treatment of municipal wastewater. However, aquatic plant systems used in the treatment of wastewater from municipal and industrial sources were shown to be generally successful with few drawbacks. Aquatic plants were shown to remove nutrients, heavy metals and organic chemicals from wastewaters. Evidence indicates that aquatic plant systems can be useful in temperate climates. However, because the problem of nitrogen removal has not been addressed specifically the evidence verifying the success of this approach is not complete. Potential design considerations and management of aquatic plant treatment systems was also found in the literature and provides the basis for development of such systems as they may apply to coal mine wastewater treatment. The literature review led to the conclusion that the use of aquatic plants to treat coal mine wastewaters was valid but that further information is needed on candidate plant selection through investigations of plant physiology and ecology and more effort is needed in the design of applicable treatment systems.

Literature concerning upland irrigation also reflected the emphasis placed on treatment of municipal wastewaters. Irrigation was found to have merit in the potential treatment of coal mine wastewaters but the application of this technique would have to be evaluated for each situation because of the relatively high mobility of nitrogen in the soil. This technique would appear to be most useful during dry periods but must be managed to reduce migration of nitrogen, and perhaps other contaminants through the soil.

Experiments were conducted using duckweed (floating aquatic plants) in static and flow-through treatments. Cropped and uncropped populations were evaluated in both situations. Nitrogen removal was found to be greater in the uncropped experiments ranging up to 98.6% in the static tests and 91.2% in the flow-through tests. Phosphorus removal was more variable achieving 94.5% in the cropped static tests and 31.3% in the uncropped flow-through tests. Biomass production was greatest in the cropped tests but nutrient removal was lower because of decreasing tissue nutrient content with time. However, the duckweed did significantly reduce nitrogen concentrations and does indicate the feasibility of using aquatic plants to remove nitrogen from coal mine wastewaters.

7.0 RECOMMENDATIONS

Aquatic plants have been shown to have potential for treating coal mine wastewaters to remove nitrogen in temperate climates. However, further work is required on the selection of candidate plants and in development of more specific design criteria for treatment systems. The selection of candidate plants requires evaluation of plant physiology and ecology as well as test work on the efficiency and harvesting of the plants. Pilot scale testing should also be conducted to verify this treatment technique.

Upland irrigation also has the potential for treatment of coal mine wastewaters. Additional work is needed to establish the criteria for its use along with field trials to consider application rates and methods for reducing nitrogen mobility in the soil.

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APPENDIX 1

DUCKWEED EXPERIMENT DATA

DESCRIPTION OF EXPERIMENTAL UNIT LABELS

The experimental unit labels employed in the Appendix are defined as follows:

LABEL	DESCRIPTION
I	Flowing; duckweed present; cropped @ 5% area per week.
J	" ; no duckweed (control); covered w/ foil.
K	" ; duckweed present; uncropped.
L	" ; duckweed present; uncropped.
M	" ; no duckweed (control); covered w/ foil.
N	" ; duckweed present; cropped @ 5% area per week.
	Hydraulic retention time = 10 days.
O	Static; duckweed present; cropped @ 5% area per week.
P	Static; no duckweed (control); covered w/ foil.
Q	Static; duckweed present; uncropped.

EVAPORATIVE WATER LOSS FROM STATIC WATER

DATE	DAY	EXPERIMENTAL TUB O	P (ml water)	Q
86/03/03		1375	250	1250
86/03/07		750	250	500
86/03/11		625	0	625
86/03/18		1500	200	1500
86/03/24		1200	175	1250
86/04/01		1200	200	1250
86/04/08		1000	175	1090
86/04/15		1350	125	1375
86/04/22		1000	110	1250
86/04/28		1100	105	1125
86/05/06		1750	105	1875
86/05/13		1750	125	1750

NORECOL DUCKWEED PROJECT

NITRATE-N CONCENTRATIONS (ppm)

FLOWING WATER

DATE	DAY	Infl.	I	J	K
86/03/10	13	0.978	0.150	0.750	0.062
86/03/17	20	1.177	0.006	1.076	0.006
86/03/24	27	1.067	0.006	0.198	0.006
86/04/01	35	10.200	0.280	4.300	0.300
86/04/07	42	10.200	0.460	5.600	0.360
86/04/14	49	12.600	1.280	8.760	0.930
86/04/23	54	12.160	2.080	2.450	1.490
86/04/28	59	12.400	3.860	9.880	2.820
86/05/06	67	14.440	5.240	10.960	3.650
86/05/13	74	15.000	6.100	12.600	5.560

DATE	DAY	L	M	N
86/03/10	13	0.060	0.755	0.490
86/03/17	20	0.006	1.234	0.006
86/03/24	27	0.006	1.098	0.006
86/04/01	35	0.260	3.400	0.260
86/04/07	42	2.400	4.200	0.340
86/04/14	49	0.320	9.360	1.280
86/04/23	54	0.460	10.320	1.520
86/04/28	59	0.980	9.840	3.090
86/05/06	67	0.650	10.960	3.980
86/05/13	74	1.500	12.150	6.400

STATIC WATER

DATE	DAY	O	P	Q
86/03/10	13	0.060	0.936	0.060
86/03/17	20	0.006	1.480	0.006
86/03/24	27	0.006	1.200	0.006
86/04/01	35	0.220	1.280	0.006
86/04/07	42	0.006	1.280	0.006
86/04/14	49	0.012	1.200	0.016
86/04/23	54	0.009	1.180	0.008
86/04/28	59	0.008	1.160	0.006
86/05/06	67	0.013	1.290	0.010
86/05/13	74	0.030	1.350	0.019

AMMONIA-N CONCENTRATIONS (ppm)

FLOWING WATER

DATE	DAY	Infl.	I	J	K
86/03/10	13	0.267	0.290	0.170	0.039
86/03/17	20	0.294	0.698	0.061	0.050
86/03/24	27	0.243	0.530	0.118	0.060
86/04/01	35	0.096	0.820	0.075	0.098
86/04/07	42	0.066	0.162	0.402	0.156
86/04/14	49	0.044	0.027	0.022	0.021
86/04/23	54	0.024	0.027	0.035	0.025
86/04/28	59	0.020	0.010	0.016	0.014
86/05/06	67	0.080	0.012	0.026	0.074
86/05/13	74	0.149	0.136	0.020	0.005

DATE	DAY	L	M	N
86/03/10	13	0.026	0.023	0.020
86/03/17	20	0.145	0.109	0.190
86/03/24	27	0.078	0.880	0.075
86/04/01	35	0.230	0.152	0.070
86/04/07	42	0.074	0.100	0.070
86/04/14	49	0.016	0.027	0.031
86/04/23	54	0.016	0.015	0.028
86/04/28	59	0.010	0.002	0.010
86/05/06	67	0.108	0.128	0.052
86/05/13	74	0.053	0.042	0.010

STATIC WATER

DATE	DAY	O	P	Q
86/03/10	13	0.113	0.292	0.058
86/03/17	20	0.062	0.058	0.027
86/03/24	27	0.038	0.046	0.031
86/04/01	35	0.027	0.034	0.016
86/04/07	42	0.010	0.003	0.030
86/04/14	49	0.018	0.032	0.020
86/04/23	54	0.032	0.002	0.004
86/04/28	59	0.002	0.002	0.002
86/05/06	67	0.013	0.030	0.002
86/05/13	74	0.025	0.002	0.002

TOTAL KJELDAHL NITROGEN (ppm)

FLOWING WATER

DATE	DAY		Infl.	I	J	K
86/05/06	67	rep.1	0.065	1.005	0.035	1.045
		rep.2	0.065	1.100	0.705	1.080
		avg.	0.065	1.053	0.370	1.063
86/05/10	74	rep.1	0.160	1.600	0.235	1.100
		rep.2	NA	1.455	0.235	0.975
		avg.	NA	1.528	0.235	1.038

DATE	DAY		L	M	N
86/05/06	67	rep.1	1.115	0.025	1.115
		rep.2	1.115	0.025	1.115
		avg.	1.115	0.025	1.115
86/05/10	74	rep.1	1.450	NA	1.405
		rep.2	1.450		1.460
		avg.	1.450	NA	1.433

STATIC WATER

DATE	DAY		O	P	Q
86/05/06	67	rep.1	1.555	1.200	1.445
		rep.2	1.385	1.095	1.445
		avg.	1.470	1.148	1.445
86/05/10	74	rep.1	2.000	1.400	2.285
		rep.2	2.370	1.815	2.285
		avg.	2.185	1.608	2.285

TOTAL PHOSPHORUS CONCENTRATIONS (ppm)

FLOWING WATER

DATE	DAY		Infl.	I	J	K
86/03/10	13	rep.1	0.720	0.165	0.263	0.210
		rep.2	1.695	0.255	0.285	0.225
		avg.	1.208	0.210	0.274	0.218
86/03/17	20	rep.1	0.255	0.360	0.615	0.525
		rep.2	0.248	0.450	0.795	0.435
		avg.	0.251	0.405	0.705	0.480
86/03/24	27	rep.1	0.420	0.645	0.420	0.450
		rep.2	0.390	0.555	0.473	0.473
		avg.	0.405	0.600	0.446	0.461
86/04/01	35	rep.1	0.270	0.465	0.420	0.525
		rep.2	0.255	0.428	0.518	0.540
		avg.	0.263	0.446	0.469	0.533
86/04/07	42	rep.1	0.285	0.030	0.225	0.150
		rep.2	0.240	0.113	1.163	0.113
		avg.	0.263	0.071	0.694	0.131
86/04/14	49	rep.1	0.375	0.600	0.315	0.263
		rep.2	0.150	0.630	0.383	0.315
		avg.	0.263	0.615	0.349	0.289
86/04/23	54	rep.1	0.255	0.300	0.068	0.150
		rep.2	0.233	0.480	0.068	0.105
		avg.	0.244	0.390	0.068	0.128
86/04/28	59	rep.1	0.076	0.166	0.059	0.026
		rep.2	0.076	0.242	0.027	0.037
		avg.	0.076	0.204	0.043	0.031
86/05/06	67	rep.1	0.047	0.088	0.056	0.086
		rep.2	0.056	0.064	0.076	0.044
		avg.	0.051	0.076	0.066	0.065
86/05/13	74	rep.1	0.037	0.069	0.070	0.039
		rep.2	0.034	0.010	0.091	0.027
		avg.	0.036	0.039	0.080	0.033

TOTAL PHOSPHORUS CONCENTRATIONS (ppm)

FLOWING WATER

DATE	DAY		Infl.	L	M	N
86/03/10	13	rep.1	0.720	0.225	0.248	1.230
		rep.2	1.695	0.248	0.315	0.630
		avg.	1.208	0.236	0.281	0.930
86/03/17	20	rep.1	0.255	0.375	0.413	1.110
		rep.2	0.248	0.428	0.525	1.380
		avg.	0.251	0.401	0.469	1.245
86/03/24	27	rep.1	0.420	0.473	0.278	0.338
		rep.2	0.390	0.525	0.323	0.383
		avg.	0.405	0.499	0.300	0.360
86/04/01	35	rep.1	0.270	0.465	0.435	0.420
		rep.2	0.255	0.398	0.405	0.420
		avg.	0.263	0.431	0.420	0.420
86/04/07	42	rep.1	0.285	0.075	NA	0.173
		rep.2	0.240	0.188	0.030	0.113
		avg.	0.263	0.131	NA	0.143
86/04/14	49	rep.1	0.375	0.375	0.450	0.263
		rep.2	0.150	0.345	0.480	0.150
		avg.	0.263	0.360	0.465	0.206
86/04/23	54	rep.1	0.255	0.255	NA	0.075
		rep.2	0.233	0.915		0.150
		avg.	0.244	0.585	NA	0.113
86/04/28	59	rep.1	0.076	0.049	0.047	NA
		rep.2	0.076	0.037	0.046	
		avg.	0.076	0.043	0.046	NA
86/05/06	67	rep.1	0.047	0.056	NA	0.044
		rep.2	0.056	0.069	0.017	0.009
		avg.	0.051	0.062	NA	0.027
86/05/13	74	rep.1	0.037	NA	0.025	NA
		rep.2	0.034	0.020	0.028	0.025
		avg.	0.036	NA	0.026	NA

TOTAL PHOSPHORUS CONCENTRATIONS (ppm)

STATIC WATER

DATE	DAY		O	P	Q
86/03/10	13	rep.1	0.683	0.480	0.353
		rep.2	0.555	0.435	0.465
		avg.	0.619	0.458	0.409
86/03/17	20	rep.1	0.480	0.435	0.630
		rep.2	0.390	0.683	0.540
		avg.	0.435	0.559	0.585
86/03/24	27	rep.1	0.383	0.518	0.518
		rep.2	0.458	0.458	0.420
		avg.	0.420	0.488	0.469
86/04/01	35	rep.1	0.600	0.518	0.225
		rep.2	0.900	0.390	0.130
		avg.	0.750	0.454	0.178
86/04/07	42	rep.1	0.150	0.330	0.360
		rep.2	0.105	0.158	0.364
		avg.	0.128	0.244	0.362
86/04/14	49	rep.1	0.225	1.350	0.300
		rep.2	0.225	1.305	0.450
		avg.	0.225	1.328	0.375
86/04/23	54	rep.1	0.083	0.465	0.600
		rep.2	0.083	0.390	0.450
		avg.	0.083	0.428	0.525
86/04/28	59	rep.1	0.017	0.213	0.076
		rep.2	0.054	0.186	0.044
		avg.	0.035	0.199	0.060
86/05/06	67	rep.1	0.062	0.223	NA
		rep.2	0.062	0.164	0.066
		avg.	0.062	0.193	NA
86/05/13	74	rep.1	0.025	0.037	0.135
		rep.2	0.017	0.125	0.144
		avg.	0.021	0.081	0.139

DUCKWEED BIOMASS HARVESTS (grams)

FLOWING WATER

DATE	DAY	FRESH I (rep.1)	FRESH N (rep.2)	DRY I (rep.1)	DRY N (rep.2)
86/03/07	0				
86/03/12	6	1.42	1.95	0.11	0.15
86/03/21	15	1.77	2.13	0.28	0.34
86/04/02	27	4.15	5.14	0.49	0.60
86/04/09	34	4.85	3.92	0.66	0.60
86/04/17	40	3.55	4.37	0.45	0.51
86/04/23	46	4.94	5.73	0.70	0.80
86/04/30	53	4.23	4.79	0.68	0.69
86/05/07	60	3.84	4.87	0.64	0.75
86/05/14	67	4.38	4.28	0.59	0.57
	TOTAL:	33.13	37.18	4.6	5.01

STATIC WATER

DATE	DAY	MASS REMOVED fresh	MASS REMOVED dry wt.
86/03/09	10	1.63	0.14
86/03/21	24	2.03	0.28
86/04/02	36	3.33	0.32
86/04/09	43	2.46	0.26
86/04/17	50	2.40	0.28
86/04/23	57	3.00	0.31
86/04/30	64	3.03	0.35
86/05/07	71	2.77	0.17
86/05/14	78	3.14	0.32
	TOTAL:	23.79	2.43

Note: Harvested area = 47cm2.

DUCKWEED NITROGEN CONTENT (% of dry matter)

FLOWING WATER

DATE	DAYS		I	N	AVG.
03/06	0				3.06
03/12-21	06-15	rep.1	2.69	3.16	2.93
		rep.2	2.50	3.01	2.76
		avg.	2.60	3.09	2.84
04/02-09	27-24	rep.1	3.48	3.54	3.51
		rep.2	3.87	3.82	3.85
		avg.	3.68	3.68	3.68
04/17-23	42-48	rep.1	3.83	3.74	3.79
		rep.2	3.75	3.80	3.78
		avg.	3.79	3.77	3.78
4/30-5/07	55-62	rep.1	4.47		4.47
		rep.2	4.68	4.86	4.77
		avg.	4.58	4.86	4.72
05/14	68	rep.1	4.76	4.32	4.54
		rep.2	4.75	5.21	4.98
		avg.	4.76	4.77	4.76

DATE	DAYS		K	L	AVG.
03/06	0				3.06
03/12-21	06-15	rep.1			
		rep.2			
		avg.			
04/02-09	27-24	rep.1	4.5	3.53	4.015
		rep.2			
		avg.			
04/17-23	42-48	rep.1			
		rep.2			
		avg.			
4/30-5/07	55-62	rep.1			
		rep.2			
		avg.			
05/14	68	rep.1	2.61	5.18	3.90
		rep.2	6.05	4.99	5.52
		avg.	4.33	5.09	4.71

DUCKWEED NITROGEN CONTENT (% of dry matter)

STATIC WATER

DATE	DAYS		O	Q
03/06	0		3.06	3.06
03/12-21	06-15	rep.1	3.33	
		rep.2	3.08	
		avg.	3.21	
04/02-09	27-24	rep.1	2.83	
		rep.2	2.83	3.61
		avg.	2.83	3.61
04/17-23	42-48	rep.1	2.73	
		rep.2	2.88	
		avg.	2.81	
4/30-5/07	55-62	rep.1	3.26	
		rep.2	3.45	
		avg.	3.36	
05/14	68	rep.1		
		rep.2	3.44	4.14
		avg.	3.44	4.14

DUCKWEED PHOSPHORUS CONTENT (% of dry matter)

FLOWING WATER

DATE	DAYS		I	N	AVG.
03/06	0				0.220
03/12-21	06-15	rep.1	0.290	0.250	0.270
		rep.2	0.230	0.280	0.255
		avg.	0.260	0.265	0.263
04/02-09	27-24	rep.1	0.170	0.110	0.140
		rep.2	0.130	0.140	0.135
		avg.	0.150	0.125	0.138
04/17-23	42-48	rep.1	0.140	0.050	0.095
		rep.2	0.050	0.050	0.050
		avg.	0.095	0.050	0.073
4/30-5/07	55-62	rep.1	0.015	0.049	0.032
		rep.2	0.013		0.013
		avg.	0.014	0.049	0.032
05/14	68	rep.1	0.041	0.039	0.040
		rep.2	0.057	0.072	0.065
		avg.	0.049	0.056	0.052

DATE	DAYS		K	L	AVG.
03/06	0				0.220
03/12-21	06-15	rep.1			
		rep.2			
		avg.			
04/02-09	27-24	rep.1	0.170	0.120	0.145
		rep.2			
		avg.			
04/17-23	42-48	rep.1			
		rep.2			
		avg.			
4/30-5/07	55-62	rep.1			
		rep.2			
		avg.			
05/14	68	rep.1	0.066	0.063	0.065
		rep.2	0.064	0.074	0.069
		avg.	0.065	0.069	0.067

DUCKWEED PHOSPHORUS CONTENT (% of dry matter)

STATIC WATER

DATE	DAYS		O	Q
03/06	0		3.060	3.060
03/12-21	06-15	rep.1	3.330	
		rep.2	3.080	
		avg.	3.205	
04/02-09	27-24	rep.1	2.830	
		rep.2	2.830	3.610
		avg.	2.830	3.610
04/17-23	42-48	rep.1	2.730	
		rep.2	2.880	
		avg.	2.805	
4/30-5/07	55-62	rep.1	3.260	
		rep.2	3.450	
		avg.	3.355	
05/14	68	rep.1		
		rep.2	3.440	4.140
		avg.	3.440	4.140