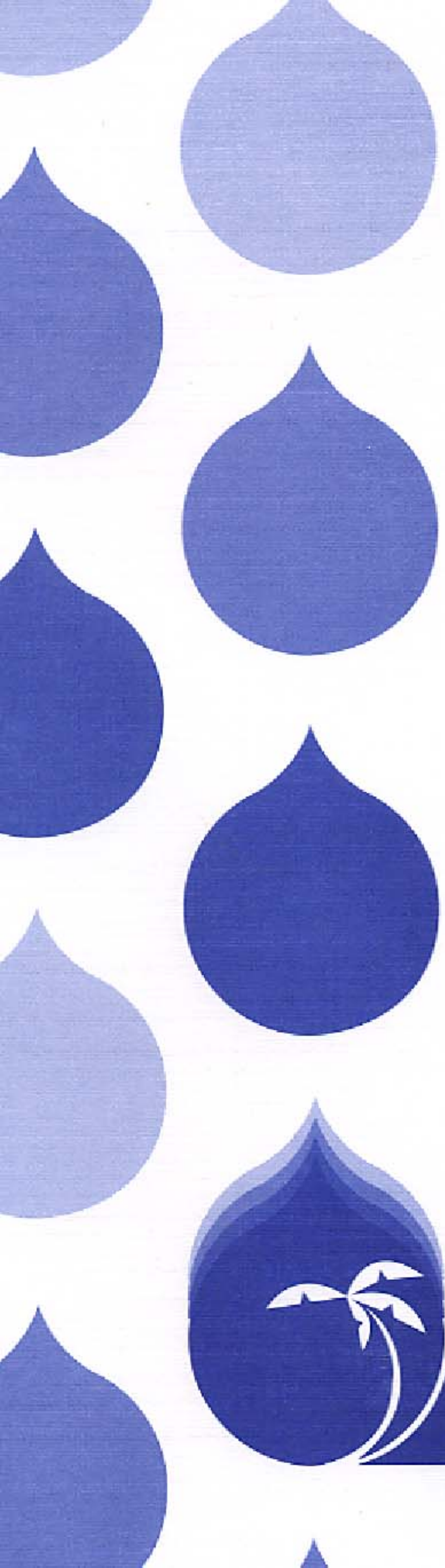

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ABSTRACT

Research is reported on the aqueous chemistry of soil pore waters and associated surface runoff from a small perennial riverine wetland astride the Taelayag River in southwestern Guam. The study took place over the onset and peak of the 1996 monsoon rainy season. The purpose of the project was to establish a first-order understanding of the chemical process that occur within wetlands, especially sediment pore waters, so that ultimately we can establish how this, and presumably many other wetlands of similar origin in southern Guam, influence or buffer downstream water quality.

The Taelayag River winds through badlands terrain that has developed in soft volcanic rock. Badlands slope process and erosion contribute the sediment that builds and maintains the wetlands along the floodplain of the Taelayag River. Groundwater seepage from the wetlands and overbank flooding maintain the sediment balance and waterlogged conditions.

Chemical weathering and erosion in the badlands fractionates particle grain sizes and chemistry, and influences the chemistry of waters moving into the wetlands below In the wetlands, REDOX- and pH-driven reactions coupled with bacterial activity alternately dissolve and precipitate metals. Filterable and total iron and manganese concentrations may reach three and two orders of magnitude respectively above ambient stream levels.

Low dissolved carbon, phosphate, and nitrogen levels are typical of Taelayag River wetland soil waters. The soils on local adjacent slopes that wash into the wetlands also are nutrient impoverished. It seems reasonable then to assume that the plant community is dependent for nourishment upon seasonal flooding and sedimentation from the main trunk stream.

Variation in chemistry across a given wetlands can be considerable. Proximity to hillside runoff and depth in the soil column are significant sources chemical variation. The major source of variation however is believed to be temporal, not spatial. Over the course of a given monsoon season, high ratios of water/soil promote elevated dissolved oxygen levels, increases in both Eh and pH, and dilution and short residence times of pore waters. Monsoonal flushing effects in these highly permeable wetlands override spatial variation that may develop over the rest of the year.

Downstream chemical effects on wetlands flushing were not investigated systematically. We noted however, a pervasive buildup at the mouth of the Taelayag River of precipitated manganese and iron oxides on reef debris. Coatings on coral fragments may reach 1.5 cm thickness. Oxide coatings are not common nor as thick on reef detritus lying at the mouths of the adjacent Sagua or Taleyfac Rivers.

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INTRODUCTION

Background

Perennial and seasonal wetlands are scattered throughout watersheds in southern Guam. Their location, configuration, day-to-day functions, and long term evolution result from several hundred years of diverse land-use activities superimposed on bedrock geology, weathering and erosional processes, plant coverage, and natural drainage development. Their ultimate value depends on biological, chemical and physical parameters that contribute to any given wetland functioning effectively as a wildlife habitat, chemical or physical buffer to water quality and stream flooding respectively, temporary sediment sink, recreational or research site, recharge basin, or even a small-scale shallow aquifer.

Water quality functions of a given wetland may be a major concern to water resource scientists and planners. Those functions relate to chemical inputs, reactions, and outputs that take place in pore and surface waters of the wetland. Although it is doubtful that any two wetlands are identical in this respect, we feel that certain physical and chemical conditions are common to many perennial riverine wetlands in south Guam, and that an analysis of these conditions can provide a better framework for understanding and managing water quality functions in local wetlands.

When a soil is flooded with freshwater, a whole series of chemical and electrochemical reaction are set into motion that profoundly affect the mobility and biological availability of plant nutrients and essential trace elements. This in turn, has a dramatic influence on the type of plants that can survive and successfully colonize the area. The primary electrochemical and chemical changes that occur in any recently flooded soil include a decrease in REDOX potential (Eh); changes in both soil and soil pore water pH; changes in electrical conductivity (EC); denitrification; accumulation of NH_4^+, reduction of Mn^{4+} and Fe^{3+}; changes in the biological availability of N, P, Fe, Mn, Cu and Zn, and reduction in the decomposition of organic matter.

This study focuses on the above parameters in surface and soil pore waters of a permanent wetland study site in southern Guam. Where possible the data are compared with findings from other areas and are discussed in terms of general requirements for optimum plant growth.

A reconnaissance study of a large perennial wetland complex in the Ugum River watershed of central Guam (Figure 1) recently completed for the Government of Guam, Bureau of Planning (Siegrist, et al, 1995), indicated that the assumed function of wetlands to buffer or modify water quality was poorly demonstrated and perhaps as poorly understood. This absence of understanding arises chiefly because levels and ranges of natural chemicals in Guam's wetlands, and their possible and probable reactions were essentially unknown. The previous study dealt with a large wetland flowing into a relatively major river where chemical anomalies would be quickly dissipated by dilution. The present study in the Taelayag River watershed (Figure 1) targeted a smaller, better delineated perennial wetland where a controlled program of chemical and physical measurements can be more efficiently designed and carried out, but more importantly, where results can be better translated into a functional model applicable to many of southern Guam's wetlands.

Perennial Non-Tidal Wetlands in Southern Guam

Geomorphology

Perennial wetlands in south Guam are frequently located on riverine floodplains where short, high gradient, and intermittently flowing tributaries debouch from wadis (gulches) that collect and drain runoff from nearby badlands. Most wetlands are less than a hectare in area and are floored with weathered volcanic rock alluvium deposited from wadis during heavy prolonged rains associated with the monsoon season.

These wetlands are generally flat surfaced and slope gently ($<1^{\circ}$) upward toward the end of the each source tributary, a geomorphology reflective of an alluvial fan. As illustrated in Figure 2, typical riverine wetlands are overprinted with slow and intermittently flowing distributary channels or ditches, the main branch of which can be traced upstream toward the mouths of individual wadis. These conduits also serve as collection-redistribution of the wetlands "plumbing, the groundwater seepage moving through pores, most notably in monsoon seasons.

Sediment thickness for this type wetlands has not been reported elsewhere on Guam, but two holes hand-augered in 1992 at the Taelayag River site reached weathered volcanic bedrock substrate at 92 and 210 cm. The former was recorded approximately sixteen meters from the nearest bedrock outcrop at the southern or upstream end of the wetland; the latter near Sampling Station P4 (Figure 3) in the center of the lower end of the wetland. Deposits at both locations lie directly on eroded surfaces of weathered volcanic bedrock which, if similar to surrounding volcanic terrain, are highly variable.

Plants

Guam's flora have been comprehensively described by Fosberg (1960). Wetland plants in the Ugum River valley were later detailed by Raulerson et al (1978) and Rinehart (1996) in connection with environmental impact and resource assessments respectively. Plant communities along the Taelayag River have not been specifically described nor was it our intention to do so in this study.

Wetlands in the middle reaches of the Taelayag River basin are essentially marshes, populated largely with grasses, sedges, reeds, and ferns (Plate) to the general exclusion of woody plants. A notable exception to the latter is the ubiquitous *Pandanus* , a monocot tree that seems to concentrate in stands along narrow and low relief natural levees that border several of the larger wetland distributary ditches (Plate 3 and 4). Patchy clusters of the reed *Phragmites karka* abound throughout most wetlands on the Taelayag River. Seedlings of woody plants are scattered about on wetlands but mature individuals are apparently excluded because of the high water table and/or an inhospitable soil chemistry.

Marsh plants baffle and stabilize wetlands sediment, provide food and habitats for animals, and provide nutrients to the soils through decay. Their role also includes influencing the solution chemistry of soil pore waters. There, chemical fields and gradients around root systems alter valance states of dissolved polyvalent elements (e.g. iron, chromium, manganese, etc.), and in doing so, changing their ionic strength and solubility in the pore waters, and ultimately their mobility, concentration, and bioavailability within the wetlands. The cumulative effects of root-induced chemical changes on the water quality of a small stream, taken over all wetlands on that stream, is unknown

Chemistry

Chemical reactions form the basis of any water quality function of a wetland, and they occur within interconnected and isolated soil pores and in flowing and ponded surface waters. Moreover, those reactions are constrained by an enormous number of environmental variables that can be broadly grouped into:

Geologic Variables: Mineralogy, particle size, degree of chemical weathering, volume and rate of incoming sediment.

Chemical Variables: Ion concentration and activity, pH, Eh, water temperature, reaction kinetics and thermodynamics, and buffering reactions.

Meteorological & Hydrogeologic Variables: Rainfall amount and intensity, stream discharge, depth to water table, piezometric surface, porosity, and hydraulic conductivity

Biological & Biochemical Variables: Plant type and coverage, root penetration, selective chemical uptake of plants, plant decay rates, and bacterial activity.

Objectives and Goals

The objective of this research was to develop a working understanding of the water quality function of a typical wetland in south Guam. The goals were to a) establish concentration levels and ranges of key environmental chemicals and chemical variables (metals, nutrients, pH, dissolved oxygen, etc.) in the wetland system, and b) develop a model of the dominant aqueous chemical processes occurring in the wetland and contributing to the quality of exported waters. A collateral goal was to develop a conceptual model of the origin and evolution of this wetland, especially as related to chemical inputs and outputs. Achieving these goals required a series of field and laboratory tasks detailed below under Methods.

Scope

The study concentrated on water chemistry as it evolved in a single wetland in the Talaeyag River basin from the onset to the peak of one monsoon season. Comparative analyses of waters from other wetlands are included in this report. Our emphasis is on the concentration levels and chemical reactions taking place during a time interval when pore waters should undergo major chemical changes as presumed stagnant pore waters of the dry season are gradually then rapidly flushed and replaced with new, and presumably more oxygenated waters of the wet season.

Table 5: Physical and Electrochemical Characteristics of Soils from Wetland Study Stations

			% Air Dry Weight of Soil				
Stations	%Water[1]	Sand	Silt	Clay	Organic Matter[2]	Volatile Solids[3]	pH[4]
P1	52.01	15.2	31.2	53.6	3.32	16.1	6.14
P2	49.51	23.2	29.2	47.6	1.50	13.9	6.25
P3	49.36	21.6	31.2	47/2	2.32	14.6	6.38
P4	47.34	17.6	30.8	51.6	2.54	13.7	6.15
P5	51.98	22.4	40.4	37.2	3.18	14.9	6.31
P6	48.09	20.4	30.0	49.6	2.76	15/0	6.22
P7	43.06	32.0	28.0	40.0	2.30	12.1	6.13
P8	50.75	26.0	29.6	44.4	3.32	13.4	6.26

[1] determined upon drying wet soil at 100 °C

[2] estimated from Total Organic Carbon (TOC) data assuming that 1.724 g of organic matter has a TOC of 1 g

[3] determined upon heating dry soil (100°C) to 550 °C.

[4] determined in a 1:1 slurry of wet soil and MillQ water

Table 6: Nitrogen, Phosphorus and Organic Carbon Concentrations in Sub-surface Soils from the Wetland Study Stations (concentrations* expressed on dry weight basis)

Station	NH_4^+-N (μg/g)	NO_3^--N (μg/g)	Total N (%)	PO_4^{3-}-P (μg/g)	Total P (%)	Total Org. C (%)
P1	2.00 (8.66)	0.46 (6.80)	0.049	0.84 (0.65)	<16	1.66
P2	1.85 (8.40)	0.35 (6.99)	<0.015	0.88 (0.68)	<16	0.75
P3	2.00 (7.80)	0.01 (7.15)	0.022	1.03 (0.74)	<16	1.16
P4	1.14 (5.88)	0.02 (6.73)	0.034	1.04 (0.60)	<16	1.27
P5	1.77 (3.94)	0.18 (6.68)	0.019	0.94 (0.63)	<16	1.59
P6	3.95 (3.61)	0.14 (6.78)	0.026	0.79 (0.59)	<16	1.38
P7	1.94 (3.31)	0.19 (7.26)	0.015	1.20 (0.61)	<16	1.15
P8	2.78 (12.6)	0.03 (7.36)	0.032	1.04 (0.60)	<16	1.66

* values in parenthesis are confirmatory analyses performed by the Soil Research and Testing Laboratory, CALS, UOG

METHODS

Field Methods

Wetland Site and Lysimeter Sampling Stations

The research wetlands are located astride the upper middle reaches of the Taelayag River where it trends approximately north-south through the volcanic terrain on the western edge of the Agat 7.5 Minute Quadrangle (U.S. Geol. Survey, 1964) (Figure 1). The watershed in this section of the basin is about evenly divided between badlands terrain and savanna or ravine forests. The Taelayag River originates at around +325 foot elevation near the Agat-Umatac Road about 500 meters south of the GovGuam Department of Public Works Transfer Station, and empties onto the reef platform at Taelayag Beach about 300 meters south of the mouth of the Talayfac River (Spanish Bridge).

The site is accessed as shown in Figure 1 and Plate 2, by hiking about 550 meters almost due west from the Government of Guam, Garbage Transfer Station on the Agat-Umatac Road, or by driving to the end of the San Isidro Chapel Road (via a bridge over the lower Taelayag River), then proceeding on the only possible extension of that road by 4-wheel vehicle south southeast about 750 meters across savanna to the site.

Wetland groundwater sampling stations P1 - P6 were laid out on a pyramidal grid pattern as indicated by the ■ symbol, and highlighted on the field photo by white plastic garbage bags as seen in the lower left portion of Plate 4 in Appendix B. Locations were chosen to collect a representative high volume of pore waters flowing along a presumed hydrologic

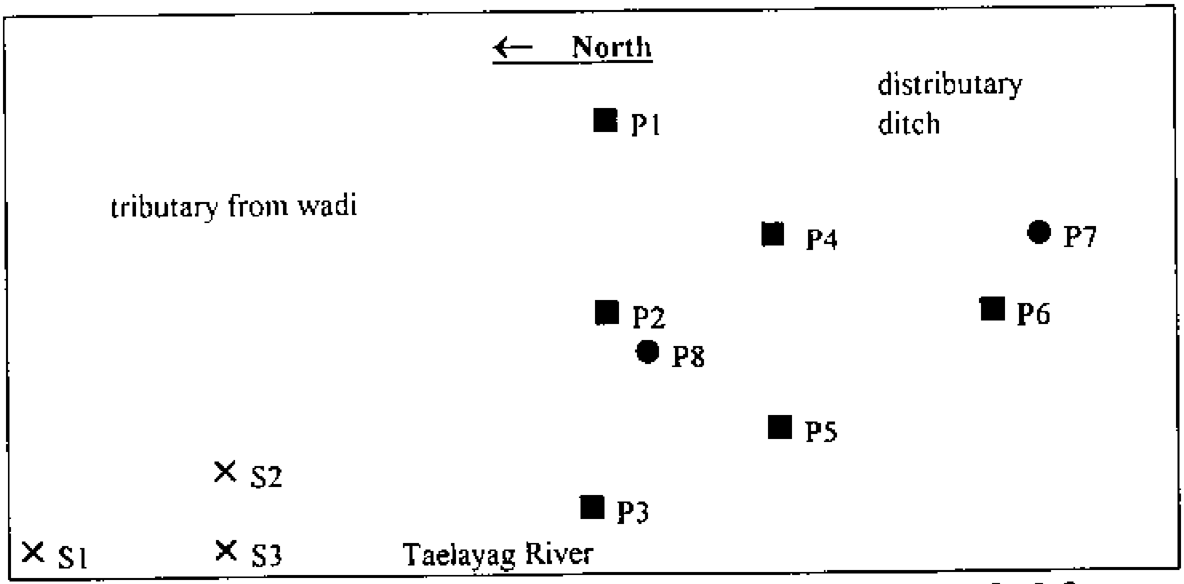

Figure 3: Sketch Map of Lysimeter Stations (expanded from lower left side of Plate 4, Appendix B) Scale About 1:300.

gradient toward the Taelayag River. Groundwater sampling stations P7 and P8, adjacent to distributary drainage ditches, are indicated by the ● symbol. Station P8 does not appear on the airphoto owing to its location under a large *Pandanus* tree. Surface water sample stations S1-S3 are indicated by the symbol x.

Water Sampling

All samples were collected during mid-wet season conditions when the study area was almost always completely water-logged and all adjacent streams were flowing freely. Surface waters and pore-waters were collected from the study site for analysis on four occasions between August 29 and October 15, 1996. All water samples were collected in either glass or polypropylene syringes fitted with in-line prefilter assemblies to remove coarse particulate material (>5 µm). Where possible, estimates of organic carbon, Fe, Mn, NO_2^-, NO_3^-, NH_4^+ and PO_4^{3-} were made on both filtered (0.45 µm) and unfiltered water samples in an attempt to distinguish between the particulate and "dissolved" contributions for each component. All electrochemical measurement (pH, Eh, and electrical conductivity) were made in the field using portable instruments.

Soil and plant samples were taken once, on October 10, 1996. The soils were analyzed for the above parameters in addition to total N, total P, and the trace elements Zn, Cu, Cr and Ni. The plant samples were analyzed for trace elements only. All sample containers and collection devices used during the study were cleaned by soaking in dilute HCl for at least 24 hours prior to use, followed by liberal rinsing in MilliQ water. Specific details on the collection procedures are outlined below.

Surface Waters

Surface water samples were collected from two sites on the Taelayag River, that bordered the study area, and from one station on an adjoining tributary that skirted along the western (seaward) edge of the wetland. The sampling stations were strategically positioned at distances of ~3 m upstream and downstream from their point of confluence (Plate 3).

Samples for organic carbon analysis were collected using a 10 ml glass syringe held just below the water surface. Two, 2 ml aliquots (filtered/unfiltered) were immediately dispensed into separate 10 ml screw cap glass vials. Water samples for all other chemical analyses were collected in 50 ml polypropylene syringes that had been filled with nitrogen gas for several days prior to use. Two syringes were filled with water at each site. Immediately after collection, a 2 ml subsample was taken from each syringe for Fe and Mn analysis (filtered/unfiltered). Each sample was placed in a separate 2.5 ml polypropylene vial containing 20 µl of concentrated HNO_3 as a preservative. The syringes were then quickly capped, placed on ice and transported to the laboratory within 1-4 hours of collection. Great care was taken to exclude air bubbles from the syringe interior at all stages of operation.

Pore Waters

Soil pore-water samples were collected in 4.5 cm (i.d.) PVC lysimeters, from eight sites within the study area (Figure 3, Plates 5-8, Appendix B). Each lysimeter was fitted with a porous ceramic cup (porosity ~5 microns) at its leading end and a rubber bung at the other end. The internal collection tube, installed through the rubber bung, was composed of polypropylene. Two lysimeters were installed at depths of 30 and 60 cm at sites 1-6. However, at sites 3 and 6, the depth of the deeper lysimeter was increased to 90 cm in an attempt to overcome the low volumes collected at the intermediate depth. Single lysimeters were installed at sites 7 and 8 at depths of 60 cm and 90 cm respectively.

The lysimeters were filled with nitrogen gas at the time of deployment and left to equilibrate to field conditions for several weeks prior to pore-water removal. Upon collection, samples were withdrawn by syringe and treated in accordance with the protocols described above for surface water sample collection. The process was assisted by pressurizing the interior of each lysimeter with oxygen-free nitrogen from a small, portable gas cylinder. The gas was blown into each lysimeter via a syringe needle pushed through the rubber bung at the exposed end of the unit. Where possible, two syringes were filled with water from each lysimeter for chemical analysis. A third syringe was used to withdraw samples for pH and Eh determinations. Any additional water remaining was blown into a 500 ml measuring cylinder for volume and specific conductance determinations. Each lysimeter was then purged with nitrogen and resealed.

All pH and Eh measurements performed on anaerobic soil pore-waters were undertaken using a "closed system" arrangement to avoid sample oxidation. This was achieved using the modified barrel of a 10 ml polypropylene syringe in which the instrument probe was contained. A rubber seal placed behind the probe's electrode ensured a snug fit and prevented oxygen from migrating towards the sample head-space at the front of the syringe barrel. While holding the device in an upright position, the water sample to be measured was injected through an inlet port at the bottom of the head-space and escaped through a small hole at the top. By maintaining a steady stream of sample through the head-space, both pH and Eh readings could be satisfactorily accomplished with a minimum volume of around 5 ml.

Soil Sampling

Anaerobic wetland soil samples were obtained from freshly augured holes within 1 m of the lysimeters at each site. They were hand collected from a depth of ~60 cm, immediately sealed in polyethylene bags to minimize oxidation processes, and transported on ice to the laboratory within 2 hours of collection. In the laboratory they were held at 4 °C until required for analysis.

Plant Sampling

The sedge, *Fimbrystylis tristachya*, and the delicate climbing fern, *Lygodium microphyllum*, were very abundant throughout the study area. Leaf and stem samples of each of these facultative wetland plants were clipped from representatives found growing in the immediate vicinity of each lysimeter site. Composite samples from at least 10 plants were sealed in Whirl-Pak bags and transported to the laboratory for analysis within 2 hours of collection.

Chemical Analysis

All chemical analyses were undertaken at the WERI Water Quality Laboratory with the exception of those performed on soil samples for total nitrogen, total phosphorus and total carbon. These were carried out by the Soil Research and Testing Laboratory with the College of Agriculture and Life Sciences at UOG. The latter laboratory also carried out the soil particle size analysis and ran confirmatory soil tests for NH_4^+, NO_2^-, and reactive (ortho) PO_4^{3+}.

Water

All aqueous samples were maintained in the laboratory at 4 °C until required for analysis. During analysis, every effort was made to keep the samples as cool as possible and minimize standing times at room temperature. The analytical procedures adopted for the inorganic forms of nitrogen and phosphorus, and organic carbon, were essentially the same as those described in the American Public Health *Standard Methods* (APHA 1989). They were as follows:

NH_4^+	Determined on 5 ml of sample by the indophenol colorimetric method (Method 4500-NH_3 D.)
NO_2^-	Determined on 2.5 ml of sample by the azo dye colorimetric method (Method 4500-NO_2^- B.)
NO_3^-	Determined on 2.5 ml of sample using a modification of the cadmium reduction colorimetric method (Method 4500-NO_3^- E.) as described by Jones (1984).
PO_4^{3-}	As orthophosphate, determined on 5 ml of sample by the ascorbic acid *colorimetric method (Method 4500-P E.)*
COD	For estimating organic carbon, determined on 2.5 ml of sample using the closed reflux (potassium dichromate reduction), colorimetric method (Method 5220 D.)

The above colorimetric tests for inorganic N and P were run in duplicate and completed within 6 hours of returning to the laboratory. The chemical oxygen demand (COD) determinations were done the following day to ensure that all reduced components present in the water samples were fully oxidized and, thus, did not interfere with the analysis.

It should be noted here that the chemical oxygen demand (COD) provides a measure of the oxygen equivalent of the organic matter of the sample and while most organic compounds are completely oxidized by this method, pyridine and related compounds are not (APHA 1989). Based on the carbon content of the potassium hydrogen phthalate standard, used in this method, an approximation of organic carbon equivalent was obtained by multiplying the COD by a factor of 0.45. The organic carbon value can, in turn, by numerically adjusted to provide an approximation of the organic matter by multiplying it by a factor of two (Nelson and Sommers 1982).

The analysis of Fe and Mn was performed on all water samples by conventional flame atomic absorption spectroscopy (AAS) employing a hydrogen continuum lamp to correct for non atomic absorption effects. Samples were aspirated into the flame directly or following dilution in 10% HNO_3 if necessary. All absorbencies were calibrated against serial dilutions of the appropriate standards after making any adjustments for blank values.

Soils

Soils analyzed for NH_4^+, NO_3^- and PO_4^{3-} by the WERI Water Quality Laboratory were prepared for chemical extraction within 4 hours of collection. Every effort was made to keep the samples cool and minimize aerial contact in order to preserve their anaerobic integrity during the subsampling process. The inorganic nitrogen and phosphorus components were determined by the methods cited above for water, following extraction (3 h) of 3-5 g of wet soil (~50% water) in 20 ml of 20% KCl and 20% HCl respectively. The 20% HCl soil extracts were also analyzed for labile Fe, Mn, Cu, Zn, Ni and Cr by flame AAS. Levels of these metals were also examined in oven dried (65 °C) soil samples following a more rigorous digestion with hot concentrated HNO_3.

The soil samples examined by the CALS Soil Research and Testing Laboratory were held at 4 °C for a little over 2 weeks prior to analysis. The following analytical procedures were adopted:

NH_4^+	Determined on wet soil samples following extraction in 2 M KCl (QuikChem Method 12-107-06-2A: Lachat Instruments 1993)
NO_3^-	Determined on wet soil samples following extraction in 2M KCl (QuikChem Method 12-107-04-1B: Lachat Instruments 1992)
PO_4^{3-}	Determined as orthophosphate on wet soil samples following extraction with 0.05N HCl and 0.025N H_2SO_4 (Nelson and Sommers 1982)
TKN	Determined as total nitrogen following Kjeldahl digestion of air dried soil samples (QuikChem Method 13-107-06-2D: Lachat Instruments 1992)
TKP	Determined as total phosphorus following Kjeldahl digestion of air dried soil samples (QuikChem Method 13-115-01-1-B: Lachat Instruments 1992)

TOC	Determined as total organic carbon in air dried soil samples using the modified *Mebius* (potassium dichromate reduction) procedure (Nelson and Sommers 1982).

Particle size analysis was conducted using a simplification of hydrometer method following Stokes Law (Bouyoucos 1962) as described by Lee and Bander (1986). Volatile (combustible) solids, pH, and % water determinations were carried out at WERI. Definitions of particle sizes follow those used by the U.S. Department of Agriculture.

Plants

All plant tissues for trace element analysis were oven dried to constant weight at 65 °C prior to wet oxidation with concentrated HNO_3 on a hot plate at 135 °C. Digested samples were evaporated to dryness, redissolved in 10% HNO3 and analyzed for Fe, Mn, Cu, Zn, Ni and Cr by flame AAS as described above.

RESULTS AND DISCUSSION

Water Elecrochemistry

All results are listed in Tables 1-8 appearing in Appendix B. Electrochemical data for stream surface waters and swamp pore waters are presented in Tables 1 and 2 respectively.

Redox Potential

The REDOX potential (Eh) scale ranges from +700 mV in a well oxidized environment down to -300 mV in a highly reducing environment. It can therefore be used as an indicator of the intensity of the oxidation or reduction processes occurring in soil, sediments and water. In the present study, the adjusted REDOX potentials (Eh_7) in stream surface waters ranged from +49 mV to -85 mV (Table 1) and in pore waters from +73 mV to -88 mV (Table 2). Values between +100 mV and -100 mV are indicative of depleted oxygen levels and are characteristic of a reducing or anaerobic environment.

Under such conditions the stable forms of nitrogen, manganese and iron are very different from those occurring under aerobic conditions. For example, after oxygen is depleted by aerobic respiration, denitrification begins when the REDOX falls to +421 mV (Cho 1982). When nitrate is depleted, reduction of Mn^{4+} begins below a REDOX of +396 mV (Schlesinger 1991) followed by reduction of Fe^{3+} at Eh +200 mV (Patrick 1964). These reactions are catalyzed by various bacteria that use fermentation to obtain energy, with Mn^{4+} and Fe^{3+} acting indirectly as electron acceptors (Lovley and Phillips 1989). Apparently there is very little overlap between the zones of reduction for each of these components. The reduction of Fe^{3+}, for example, will not start until Mn^{4+} is depleted as a result of an enzymatic preferences by soil bacteria for the latter element (Lovely and Phillips 1988).

The low Eh_7 values recorded in the surface water samples were surprising considering the substantial flow observed in each stream and the fact that dissolved oxygen levels were around 7.0 mg/l. This suggests that a major component of the stream flow was derived from subterranean drainage waters leaving the wetland study area, and that equilibrium between the various oxidation states of the inorganic species present, had not established in the short time it took to flow past the sampling points on each stream.

pH

Similar pH readings were determined in both surface and pore water samples (Tables 1-2) and rarely deviated by more that 0.5 pH units about the point of neutrality (pH 7). This is to be expected largely because of the buffering action of substances produced by the reduction reactions, e.g. Fe^{2+} and Mn^{2+} hydroxides and carbonates (Patrick and Mikkelsen 1971), and to organic acids derived from the microbial decomposition of organic matter (Motomura 1962). In the latter context, it is worth noting that anaerobic bacteria function best at a pH of around 7 (Ponnamperuma 1984).

Specific Conductivity

The specific conductance of surface waters (Table 1) samples was consistently lower than that of the pore waters examined (Table 2). Values in the former samples ranged from 319-716 µs/cm and presumably reflect varying increases in the ionized forms of Fe and Mn together with increased mobilization of Ca^{2+} and Mg^{2+} from the surrounding water-logged soil by cation exchange reactions (Ponnamperuma 1965).

The data shows obvious differences in the chemical signatures of pore waters collected at the different depths within sites, and at the same depth between sites. We attribute this to variations in residence times of discrete water masses moving through the system. The implication is, therefore, that the vertical and horizontal drainage capacity of the wetland soil is markedly heterogeneous over the study area.

Difference in the chemical signatures of the surface waters analyzed were also apparent, with the main Taelayag River having a significantly higher specific conductance than it's smaller adjoining tributary Once again this points towards a significant subterranean seepage of anoxic waters from the wetland area into the main body of the former stream.

Surface waters, particularly in soft water areas, have specific conductances of around 30-50 µs/cm, whereas in hard water areas, readings in the order of 100-300 µS/cm can be expected. The specific conductance of groundwaters is generally higher still and may approach 1000 µS/cm in limestone areas.

Water Chemistry

The chemical parameters measured in the surface waters and pore waters from the wetland study area are shown Tables 3 and 4a-d respectively.

Carbon

The COD determined for the surface water samples over the study period (Table 3) translated to a total organic carbon concentration (TOC) that ranged from <2-81 mg/l. This, in turn, is equivalent to an organic matter range of <4-162 mg/l. The average TOC in rivers on a global basis has been estimated at 9.5 mg/l (Schlesinger 1991) which is within the range reported here. Thus, the wetland study area exports appreciable amounts of organic carbon to adjacent streams, at least, during wet season conditions.

In the majority of samples analyzed, dissolved organic carbon concentrations exceeded particulate organic carbon concentrations. This trend seems to apply to almost all water bodies (Furnas 1992).

A comparison between the two streams examined indicated that the small tributary leading into the main Taelayag River occasionally had relatively high loadings of dissolved organic matter (DOM). In contrast, levels in the Taelayag River above the point of confluence were consistently low and less variable with detectable DOM concentrations that ranged between 5.0 and 11.7 mg/l. In point of interest, freshwater inland lakes normally contain DOM concentrations in the range of 0.1-50 mg/l with the highest levels occurring in dystrophic lakes (Birge and Juday 1934).

Much of the DOM in natural waters consists of complex, brown or yellow acidic polymers formed, at least in part, during the decomposition of lignin and simpler organic molecules and collectively known as humic matter (Christman and Ghassemi 1966, Jackson 1975). Dissolved humic matter has traditionally been divided into fulvic acid and humic acid fractions. Most of the humic acid in flowing waters is fulvic acid (Reuter and Perdue 1972, Jackson 1975). Humic matter plays an important role in the complexing and recycling of several divalent and trivalent metals (Shapiro 1964, Ghassemi and Christman 1968, Reuter and Perdue 1972). Humic-metal-phosphorus associations have also been proposed for surface waters (Jackson 1975).

It is well known that the decomposition of organic matter is slower in submerged soils compared with well-drained soils (DeLaune *et al.* 1976). This is because resident facultative anaerobes in water-logged soils operate at a much lower energy level than their aerobic counterparts and, hence, they decompose organic matter at a much slower rate. The accumulation of DOM in the pore waters sampled during the present study amply illustrates this point (Tables 4a-d). This was especially evident during the earlier part of the study when concentrations ranging from 35-573 mg/l were determined in pore-waters from the lysimeter sites (Table 4b). As time progressed, however, levels of DOM generally declined and ranged from 5-92 mg/l on the last date of sampling (Table 4c). We strongly suspect that this trend represents the subterranean flushing of soluble decomposition products from the wetland during flood conditions.

Nitrogen

The bacterial decomposition of wetland plants is probably the most important source of nitrogen within the study area. Much of this nitrogen in this environment (>95%) will undoubtedly be in the organic form, as amino and amide groups associated with plant material. While estimates of organic nitrogen were not performed directly on any of the water samples collected, indirect first order approximations were obtained from the TOC estimates given above, by adopting the overall C:N ratio of 11.1 reported by Schlesinger (1991) for global river flow. Thus, total nitrogen in our surface waters ranged from around <0.2-7 mg/l whereas, in pore waters, higher levels ranging from 0.5 mg/l to upwards of 41 mg/l, were estimated. From Schlesinger's data, the global average value for total nitrogen in rivers is in the order of 0.9 mg/l.

Nitrogen, as nongaseous, inorganic nitrogen, exists in soil and water as ammonium, nitrate and nitrite ions. Shifts in the oxidation states of inorganic nitrogen are commonly mediated by microorganisms and reflect the reduction oxidation potential and oxygen availability of the immediate environment (Paul and Clark 1989). Ammonium (NH_4^+) and nitrate (NO_3^-) are the principal sources nitrogen to aquatic plants. Nitrite (NO_2^-), urea [$CO(NH_2)_2$], and free amino acids ($R-NH_2$) can also be directly taken up but concentrations of these nitrogen species are usually considerably lower than those of ammonium and/or nitrate in most unpolluted aquatic systems (Furnas 1992).

In the present study, ammonium-N and nitrate-N levels were all extremely low (<10 µg/l) in the Taelayag River above the point of confluence (site 3). However, relatively high concentrations of nitrate-N, ranging from 153-378 µg/l, were consistently found in the small adjoining tributary stream (site 2) and may possibly reflect anthropogenic activities occurring, on a small scale, within the watershed area (Table 3). Nitrate-N levels have previously been reported for surface waters from the Ugum and Bubulao Rivers on Guam (Siegrist *et al.* 1995) and ranged from 1.2-16.8 µg/l (mean 5.7 µg/l). These concentrations are similar to those reported here for station S3 on the Taelayag River and are probably fairly representative of unimpacted wetlands on the island.

Levels of ammonium-N in the pore waters examined were generally much higher than found in the surface water samples (Tables 4a-d). Over the entire study, ammonium-N concentrations ranged from 6->2000 µg/l and averaged around 50 µg/l (upper extreme omitted). These relatively high levels were not surprising in view of the fact that the mineralization of organically bound nitrogen in anaerobic soils stops at the NH_4^+ stage because of lack of oxygen for nitrification to NO_3^- (Ponnamperuma 1972). Also, in the absence of oxygen, certain microbial species obtain their energy from decomposing organic matter by indirectly using NO_3^- as an electron acceptor to produce NH_4^+ which accumulates in the soil pore water (Paul and Clark 1989). Such denitrification process have been shown experimental to begin when the REDOX falls to +421 mV (Cho 1982) which is well above the Eh values recorded in all aqueous samples collected during the present study.

As a general rule, pore water concentrations of ammonium-N were higher than the corresponding nitrate-N levels although ratios between the two varied from approximately 1:1 to >100:1 depending upon the depth and position of the lysimeter and time of sampling. Pore water nitrate-N levels, in turn, were generally higher than found in the Taelayag River above the point of confluence (site 3) and ranged from <1-85 µg/l with a mean value of 14 µg/l. It is also interesting to note that ammonium-N and nitrate-N, like organic carbon, were predominantly present in the dissolved form in both surface and pore water samples taken from within the study area.

Nitrite-N was consistently below the limits of analytical detection (<1 µg/l) in the Taelayag River and all pore water samples. Very low concentrations (2-3 µg/l) were however detected in the small tributary stream (site 2), presumably a consequence of the relatively high levels of nitrate-N found there.

While there were very definite difference in ammonium-N and nitrate-N levels in pore waters within and between lysimeter sites, consistent temporal trends were not apparent. This was due in part to the relatively narrow window of time over which samples were collected and possibly to rapid changes in drainage rates (and concurrent changes in soil chemistry) in response to intermittent rainfalls of varying duration and intensity.

Phosphorous

In natural conditions, phosphorus is derived almost exclusively from the weathering of igneous rocks and inputs to wetland ecosystems are relatively small (Ahl 1988). Unlike inorganic nitrogen, there is little phosphorus in the atmosphere or precipitation. Moreover, phosphorus is normally retained much more efficiently in surrounding terrestrial watersheds by vegetation and by chemical interactions with soil minerals (Schlesinger 1991). Consequently, phosphorus is frequently the most limiting nutrient in aquatic environments, especially in the form of ionic, reactive orthophosphate (PO_4^{3-}), which is immediately useful for autotrophic plants.

Average PO_4^{3-}-P levels for rivers and inland lakes are in the range of 10-30 µg/l (Hutchinson 1957, Meybeck 1982, Freedman 1989). In the present study, unfiltered surface water PO_4^{3-}-P concentrations were at the lower end of this range, averaging 10 µg/l with a spread of 2-16 µg/l. Yarbro (1980), working on a surface waters draining a floodplain swamp on North Carolina over a four year period, reported annual mean levels of PO_4^{3-}-P of 2-20 µg/l, for unimpacted streams, with an overall average of 8 µg/l. She concluded that her data was similar or lower than concentrations measured in other coastal plain streams of the eastern United States.

Based on the above comparisons, we conclude that inorganic PO_4^{3-}-P levels of in the Taelayac River and its adjoining tributary are most likely below world average values for surface waters. From the literature, it would also appear that most of the total phosphorus leaving the wetland study area, in surface streams, will be bound to organic matter or inorganic particles (Yarbro 1980).

Pore water concentrations of PO_4^{3-}-P varied considerably between and within lysimeters over the study period. Levels recorded in unfiltered water collected from the shallower (30 cm) of the two lysimeters, at station P5, were consistently high and ranged from 25-84 µg/l with a mean value of 52 µg/l. It should be noted here that water from this lysimeter had a foul-smelling odor consistent with the anaerobic degradation (putrification) of protein-rich substances. Initially, water from this lysimeter also contained extremely high levels of ammonia (Table 4a), and had a relatively high COD. We concluded that the lysimeter was situated directly over a small, enriched pocket of decaying organic material, possibly of animal origin.

Elsewhere in the study area, unfiltered pore waters yielded PO_4^{3-}-P levels of <1-48 µg/l, with an overall average of 5 µg/l. These values are surprisingly low considering the anoxic nature of the wetland soil and the well established tendency for inorganic phosphorus to be released from its association with iron under such reducing conditions (Shapiro 1958a & b, Ponnamperuma 1985).

The concentration of PO_4^{3-}-P in soil interstitial water is normally in the order of 100-1000 µg /l (Paul and Clark 1989). It is this water soluble pool that is directly available to plants and thus effects crop yields. Optimum levels recorded for a variety of agricultural crops range from 10-300 µg /l (Fox 1981).

The data presented here is therefore suggestive of impoverished reserves of phosphorous in the wetland soil, coupled with rapid uptake and recycling of the limited amounts of dissolved PO_4^{3-} by soil microbes and higher plants within the area. The relatively high levels of organic matter in the pore waters may, in fact, facilitate this process by supporting large populations of decomposer organisms that readily assimilate mineralized phosphorus produced from the organic matter (Shapiro 1958b).

A comparison of the filterable and none filterable data indicates that reactive phosphate was not always in the dissolved form. Yarbro (1980), points out that a significant amount of both reactive and unreactive phosphorus in surface waters is colloidal rather than dissolved. It is possible, therefore, that some of these colloids were retained by the filter during the filtration process.

Iron and Manganese

The elements Fe and Mn were very abundant throughout the wetland study area reflecting high levels in the soil and parent rock materials. Fe in unfiltered surface waters draining the wetland study area ranged from 0.20-0.57 mg/l in the main Taelayag River with little difference at stations above (S3) and below (S1) the point of confluence (Table 3). On the other hand, levels determined in samples from the adjoining tributary (S2), were consistently lower and did not exceed 0.2 mg/l over the entire study period.

Mn levels at both stations on the Taelayag River were marginally higher than for iron and ranged from 0.43-0.59 and 0.49-0.71 mg/l at S1 and S3 respectively. Once again, surface waters from S2, on the small adjoining tributary stream, contained the lowest metal concentrations with Mn ranging from 0.11-0.23 mg/l. A comparison of unfiltered and filtered data indicated that the dissolved contributions of each metal almost always exceeded amounts associated with suspended particulate material.

In oxygenated surface water, at normal pH, Fe and Mn form insoluble oxides and oxyhydroxides and consequently levels are generally low. A broad concentration range that encompasses much of the published data for Fe and Mn in river waters throughout the world, is in the order of 0.003-0.3 mg/l and 0.001-0.1 mg/l for each metal respectively (Wilson 1980).

Dissolved Fe and Mn levels in soil interstitial waters are generally higher than found in most surface waters although, in well drained soil, they rarely exceed 0.5 mg/l (Kabata-Pendias 1972). However, under the anoxic conditions that prevail in most wetland soils, insoluble Fe^{3+}, Mn^{4+} and Mn^{3+} are reduced to their respective soluble forms, namely Fe^{2+} and Mn^{2+}, and may reach very high concentrations in the surrounding pore waters. For example, Fe^{2+} and Mn^{2+}, levels of up to 450 mg/l and 90 mg/l respectively were reported for acid soils, high in organic matter within two to three weeks of submergence (Cho and Ponnamperuma 1971). Apparently, increased organic matter increases the build up of both elements in flooded soils by serving as an energy source for the microorganisms that facilitate the reductive process (Redman and Patrick 1965).

In the present study, Fe concentrations in pore waters were predominantly in the dissolved form and extended over three orders of magnitude from 0.11-140 mg/l (Tables 4a-d). Levels varied considerably between and within lysimeters over the study period, presumably in response to localized difference in the composition of the underlying soil and to discrete spatial and temporal variations in the flushing and drainage rates of water through the system. Interestingly, the highest levels were once again confined to the 30 cm lysimeter at station P5.

Levels of Mn determined in pore waters were considerably less variable than for Fe and ranged from 0.82-30.1 mg/l. Levels within lysimeters rarely varied by more than a factor of two over time, and unlike iron, were consistently higher than elsewhere in the 30 cm lysimeter at station P4. Rarely was there any difference between filtered and unfiltered samples suggesting that this element was almost exclusively confined to the dissolved form.

Wetland Soil Chemistry

The acidic clay soils found in the savanna and upland slopes adjacent to the wetland study area belong to the Akina soil series. These soils are deeply weathered saprolites, with a high kaolinitic clay content (Young 1987). They are particularly subject to slumping and erosion, a process which, no doubt, has played a key role in the formation and maintenance of wetlands in this part of the island.

The physical parameters measured in this soil taken from within the wetland study area are shown Table 5. The data for nutrient and elemental analysis are summarized, on a dry weight basis, in Tables 6 and 7 respectively.

Particle Size Analysis

Demeterio *et al.* (1986a) found that Akina Soil, taken from a depth of 5-30 cm, was composed of 69% clay, 23 % silt and 8 % sand. Siegrist (1986, unpublished) measured clay and silt content of two soils (B-horizon) samples of the Akina Soil near the present the Transfer Station Site at 87.1 and 93.8% percent. Wetland soil, taken from the eight lysimeter stations (P1-P8) at a depth of approximately 60 cm during the present study, differed significantly from these findings, yielding a mean clay, silt and sand content of 46.4%, 31.3% and 22.3% respectively (Table 5).

We suspect that the lowered clay content of the wetland soil is a physical effect of the erosion process caused by the differential settling of the heavier sand and silt particles in the swamp during storm water run-off, and the export out of the area of proportionately high amounts of clay in surface streams and rivers. The relatively large amounts of silt and sand deposited in the wetlands by this fractionation process, improves water permeability of the soil and gives credence to the idea that there is a substantial subterranean output of water from the wetland area into the Taelayag River, and adjacent streams, during wet season conditions.

pH

Typical pH values for Akina Soils rang between 4.5 and 5.5 (Demeterio *et al.* 1986a & b, Young 1987). In the present study we observed pH values in our wetland soil samples ranging from 6.13-6.38 (Table 5) with an overall mean pH of 6.23. The relatively high pH values recorded by us seem reasonable in view of the fact that the overall effect of flooding acid soils is to raise the pH and buffer it around the point of neutrality (Ponnamperuma 1972, 1984). This is explained by the action of substances produced as a result of reduction reactions, and in Akina soil, as with most other mineral soils, predominantly involves the anaerobic reduction Fe^{3+} to Fe^{2+}.

Volatile Solids

Volatile solids, reported here, ranged from 12.1-16.1% with an overall mean value of 14.2%. They represent contributions to dry wetland soil made by carbonate materials, organic matter and structural water locked up in the clay plates. Considering that the carbonate content was insignificant and that the organic matter only accounted for 2-3% of the total weight, most of the weight loss was attributable to the removal of structural water -- equivalent to 10-13% of the total dry weight of soil

Carbon

The organic matter content of the wetland soil samples collected at a depth of ~60 cm ranged from 1.5-3.3% (Table 5). The overall mean value of 2.7% was somewhat lower than the value of 4.6% previously reported by Demeterio *et al.* (1986a) for Akina soil, although the fact that their sample was taken from a shallower depth (5-30 cm) may partly explain this difference. In support of this, reference is made to the work of Young (1987) who found that the organic matter content of Akina soil dropped dramatically with depth from a high 5% in the top 10 cm, to 0.2 % at depths between 110-150 cm. He also reported an organic carbon content of 0.6 % for soil taken from 40-60 cm which is approximately half the mean value found at an equivalent depth during the present investigation.

Absolute amounts of organic carbon vary considerably from one soil to another, being as low as 1%, or less, in sandy soils to as much as 3.5% in prairie grassland soils (Stevenson 1986). On Guam, relatively high amounts of organic matter (3-8%) have been recorded in the upper 20 cm of various soils from around the island (Young 1988). As a general rule, levels tend to be even higher than this in any poorly drained and waterlogged soil, as discussed earlier, and may range between 10% and 40% depending on depth and location (Smock 1980, Stevenson 1986). The general conclusion from such comparisons therefore leads us to believe that organic carbon levels determined in our wetland soil samples are generally on the low side by world standards.

Nitrogen

From Table 6, it can be seen that inorganic nitrogen accounted for <2% of the total nitrogen present in the wetland soil samples indicating that >98% of it is bound up in the organic form. In fresh, anoxic wetland soils, NH_4^+-N was the dominant inorganic form of nitrogen (Table 6) with levels ranging from 1.14-3.95 µg/g. NO_3^--N concentrations were much lower in comparison and ranged between 0.01-0.46 µg/g.

Soils held in the laboratory at 4 °C for two weeks prior to analysis yielded significantly higher levels of both inorganic forms as a result of the continued microbial decomposition of organic matter present. Although these soil samples were wrapped in plastic bags, the proportionately higher levels of NO_3^--N found, indicates that the anaerobic integrity of the sample had deteriorated during storage. As a consequence some of the NH_4^+ present had been oxidized to NO_3^-- by resident nitrifying microorganisms.

It is well known that NH_4^+ is present in soil in the exchangeable and fixed form (Bremner 1965). Fixation results from the substitution of NH_4^+ for interlayer cations (Ca^{2+}, Mg^{2+}, Na^+) within the expandable lattice of clay minerals (Stevenson 1986). This process may significantly reduce the availability of NH_4^+ to microorganisms and higher plants (Stevenson 1986). The extent to which it occurs depends on the abundance and type of clay mineral present. For example, vermiculite and illite, followed by montmorillonite, are the main minerals that retain NH_4^+ in a nonexchangeable form (Alison *et al.* 1953). In contrast, virtually no fixation occurs in those soils where kaolinite is the predominant clay mineral (Stevenson 1986). Thus, most of the NH_4^+ present in our wetland soil is in the exchangeable form and presumably is freely available for assimilation by the indigenous plants within the area.

Total nitrogen levels in 106 soils from all over the world have been reported by Snowden *et al.* (1977; cited in Stevenson 1986) and extend from 0.02-1.61 % on a dry weight basis. It is significant to note that the levels recorded during the present study are at the lower end of this range and point towards a relatively impoverished environment from a nutritional standpoint and/or rapid nutrient uptake and recycling by soil microorganisms and indigenous plants.

C/N Ratio

As carbon and nitrogen are derived from soil organic matter, they are roughly correlated with one another. For most soils, the C/N ratio is usually around 10-12 (Stevenson 1986). Ratios of less than 20:1 indicates that there is a net gain in mineralized forms of the nutrients within the soil, whereas ratios of 30:1, or over, indicates nutrient immobilization by the soil biomass, i.e., that the decay of organic residues in soil is accompanied by conversion of mineralized carbon and nitrogen into microbial tissue (Stevenson 1986). It is therefore pertinent to note that the C:N ratios found during the present study ranged from 34-84 (mean 56) indicating that the labile nutrients are, indeed, quickly assimilated and recycled back into the biota.

Phosphorus

Naturally occurring phosphorus in soil is predominantly derived from apatite, a complex tricalcium phosphate compound that can exist in the fluoro-, chloro-, hydroxy-, and carbonate form. During weathering and soil development, phosphorus is liberated from this mineral matrix and assimilated by the soil microbes and plants, that eventually die, decay, and ultimately return it to the soil. Here it exists in the insoluble or sparingly soluble mineral form, adsorbed to soil components and occluded in hydrated oxides (Stevenson 1986). In acidic soils, particularly those rich in Fe like the lateritic Oxisols of the tropical and subtropical regions of the world, phosphorus is readily precipitated as Fe or Al-phosphate which is relatively unavailable to plants (Stevenson 1986).

According to Cope *et al.* (1985), the lower soil optimum PO_4^{3+}-P requirement for agricultural purposes is 15 μg/g dry wt. Levels ranging from 4-7 μg/g dry wt. are considered low while those between 0.5-1.2 are classed as extremely low (Demeterio *et al.* 1986a & b). The PO_4^{3+}-P concentrations recorded here ranged from 0.79-1.2 μg/g dry wt. and obviously fall into the latter category. They also substantiate the conclusion of Demeterio *et al.* (1986b), namely that acid soils from southern Guam generally have PO_4^{3+}-P levels lower than 2 μg/g dry wt.

Total phosphorus concentrations in common soil forming rocks vary from as little as 100 μg/g in sandstone to 2000 μg/g or more in high phosphate limestones (Stevenson 1986). Acid volcanic rocks are generally low in phosphorus as are the soils formed from them. In the latter context, it is noteworthy that background concentrations of phosphorus determined in the volcanic parent rock of Akina soil on Guam (Facpi formation), ranged from 175-269 μg/g (Reagan and Meijer 1983).

Total phosphorus concentrations in soil generally mirror levels in the parent rock, although substantial losses can occur during the weathering and soil forming process particularly in high rainfall areas (Stevenson 1986). This notwithstanding, average levels are in the order of 600 μg/g dry wt. In the US, the highest phosphorus levels are found in soils from the Northwest which typically contain 1000-1300 μg/g. In contrast, the sandy soils the Atlantic and Gulf coast plains are extremely low in phosphorus with levels generally <100 μg/g dry wt. (Stevenson 1986).

Total phosphorus concentrations in the wetland soils examined here showed greatly diminished phosphorus reserves that were below an analytical detection limit of 16 μg/g. Such low levels strongly suggests that extensive leaching has occurred over time. The importance of this process as a major mechanism of phosphorus loss from the wetland area is therefore highlighted.

A first order approximation of the total phosphorus levels present in the wetland soil was obtained by assuming that the phosphorus available in soil solution accounted for only 0.35% of the total phosphorus pool in the soil (Chapin *et al.* 1978). Using the overall mean values for PO_4^{3+}-P levels determined in pore waters, a total phosphorus concentration of around 3 μg/g dry wt. was conservatively estimated.

Iron and Manganese
Fe is the fourth most abundant element in the lithosphere behind oxygen Al and Si. Levels in soil vary from about 200 μg/g dry wt. in coarse textured soils, to well over 10% (100,000 μg/g dry wt.) in ferruginous latisols from tropical regions of the world

(Stevenson 1986). The acidic Akina soils of Guam fall into the latter category and are known to be high in iron oxide (Demeterio *et al.* 1986a, Young 1987). This was confirmed during the present study with levels in the wetland soil samples ranging from 8.2-10.5% on a dry weight basis (Table 7).

A comparison of data for both extraction procedures indicates that around 50% of the Fe load in the anoxic wetland soil was in a labile or relatively mobile form. However, net losses of soluble Fe leached from the wetland in times of flood are probably small and balanced, for the most part, by amounts brought into the area from outside in surface runoff and subterranean seepage. Also, much of the Fe will be recycled within the wetland itself as changing gradients in redox potential are encountered. Nevertheless, some net loss of particulate and sediment bound Fe is expected to occur during periods of high runoff. This is considered to be the principal mechanism controlling transport of Fe out of the area

Mn in soils covers a particularly wide range from as little 20 µg/g to well over 6,000 µg/g on a dry weight basis (Stevenson 1986). Such variation seems to apply equally to Guam. For example, Demeterio *et al.* (1986) analyzed 13 local soil types and reported total Mn concentrations of 344-5,486 µg/g dry wt. Extractable concentrations (in 0.2N HCl) ranged from 4-750 µg/g. More recently, Motavalli *et al.* (1996) presented data for extractable Mn in 499 soil samples collected from 19 districts on the island, as a mean and standard deviation of 167 and 208 µg/g dry wt. respectively. Approximately 30% of these soil samples contained concentrations of 50 µg/g or less, and, from an agricultural perspective, were considered to be Mn deficient.

In the present study, relatively high concentrations of Mn were found in the wetland soils although the degree of variation between stations was much higher than for Fe. This more than likely reflects a difference in the distribution pattern of each element, with Fe being much more diffuse and widespread throughout the area. Concentrations ranged from 599-3,036 µg/g dry wt. for the weak acid soil extractions and from 895-1,798 µg/g dry wt. after digestion in concentrated nitric acid. The higher range observed for the weak acid extractions reflects sample heterogeneity. Even so, a comparison between the two data set does suggest that a high proportion of Mn in the anoxic wetland soil is in the labile form.

Sediment transport in surface water runoff is more than likely the major transportation mechanism controlling losses of Mn from the wetland area. Thus, Mn, like Fe, is exported primarily in the solid phase. However, since the oxidation of Mn^{2+} is substantially slower than the oxidation of Fe^{2+} at neutral pH (Hem 1977), proportionally greater amounts of it will be flushed from the wetland in dissolved form compared with Fe. In support of this, we have previously identified relatively high levels of filterable Mn (60-150 µg/l) at the mouth of the Taelayac River, some 500 m (??) downstream of the wetland area. Dissolved Fe levels, on the other hand, were consistently below a detection limit of 30 µg/l (unpublished data).

Trace Elements

A trace element is a natural element in natural materials such as the lithosphere that is present at a level of <0.1% (Adriano 1986). In the present study, we considered the trace elements Cu, Zn, Cr and Ni, all of which are known to be associated with Fe and Mn oxides in soil and rock materials (Kabata-Pendias and Pendias 1992).

Levels of each element determined in the wetland soil samples following treatment with weak and strong acids, are listed in Table 7. It can be seen that there was little difference between treatments suggesting these elements were relatively mobile in the anoxic swamp environment and are not precipitated as the insoluble metal sulfides. Presumably, the high background levels of Fe and Mn maintain the ambient soil redox potential at Eh values that do not favor the reduction of sulfate (SO_4^{2-}) to hydrogen sulfide (H_2S) and hence do not facilitates this process.

Copper and Zinc: Total levels of Cu and Zn in an Akina soil from Guam were previously published by Demeterio *et al.* (1986a) They reported 242 µg Cu and 76 µg Zn/g dry wt. of soil which is reasonably close to levels found here for each element. Moreover, the data they presented for 12 other local soil types were of the same order and ranged from 111-250 (mean 171) µg/g for Cu, and 50-212 (mean 130) µg /g for zinc. However, extractable levels for both elements were one to two orders of magnitude lower, a finding that led Motavalli *et al.* (1996) to conclude that Cu and Zn deficiency could be a problem facing crop growers on Guam. Clearly, in terms of metal abundance, such deficiencies do not apply to the wetland soil examined here.

On a broader scale, normal levels of Cu and Zn in soil can range from a low of 1-2 µg/g dry wt. to a high of 200-300 µg/g dry wt. (Stevenson 1986, Kabata-Pendias and Pendias 1992). The iron-rich Ferralsols are typically high in both elements (Kabata-Pendias and Pendias 1992).

Nickel and Chromium: The distribution of Ni in the earth's crust parallels that of Fe. Consequently, levels in ultramafic rocks may be has high as 2000 µg/g (Kabata-Pendias and Pendias 1992). Ni levels found in local volcanic rocks, from which the Akina soil series are derived, ranged from 9-342 µg/g with a mean of 173 µg/g (Reagan and Meijer 1983). As the Ni status in soils is highly dependent on the Ni content of the parent rock, the levels found the wetland soil samples are, therefore, not surprising. What is noteworthy, however, is the fact that the levels found during the present study exceed levels reported for unenriched and uncontaminated US soils (<5-150 µg/g dry wt.) and generally rank among the higher concentrations (0.2-450 µg/g dry wt.) found elsewhere in the world (Kabata-Pendias and Pendias 1992).

Natural levels of Cr found in soils range from <10 to >1000 µg/g on a dry weight basis, although mean values for most lie between 10 and 200 µg/g (Kabata-Pendias and Pendias 1992). Relatively high concentrations are found in soils derived from mafic and volcanic rocks similar to the Facpi formation on Guam. The data gathered from the wetland soil samples during the present study supports this statement (Table 7).

Cr can exist in several valency states, the most common of which are Cr^{3+} and Cr^{6+}. Most soil Cr occurs as non toxic Cr^{3+} and is within the mineral structures or forms of mixed Cr^{3+} and Fe^{3+} oxides (Kabata-Pendias and Pendias 1992). Trivalent Cr compounds are of limited water solubility and are relatively immobile in the soil profile, whereas the reverse is true for hexavalent Cr compounds which are also highly toxic. The ready conversion of unstable Cr^{6+} to stable Cr^{3+} under normal soil conditions is, therefore, of great importance because it greatly reduces the toxicological impact of this element on the indigenous biota. However, the fact that Cr^{3+} can be oxidized to Cr^{6+} by oxides of Mn^{3+} and Mn^{4+} in soil (Bartlett and James 1979, Fendorf and Zasoski 1992) is noteworthy in the context of the present study, particularly in view of the great abundance of Mn throughout the study area.

Trace Element Chemistry of Wetland Plants

Data obtained from the elemental analysis of the sedge, *Fimbrystylis tristachya*, and the fern, *Lygodium microphyllum*, are summarized in Table 8. We point out here, that this was a preliminary study prompted by the discovery of high levels of transition elements, particularly Fe and Mn, in the wetland soil. . As yet, we have no comparative data for these plants from other locations. Nonetheless, some important points emerge when weighed against previously reported data for other plant species from elsewhere in the world.

Iron and Manganese

Fe and Mn play essential and interrelated roles in plant metabolism and, while the uptake of both elements is metabolically controlled, passive absorption occurs at elevated soil concentrations (Kabata-Pendias and Pendias 1992). Both elements are readily available to plants in the reduced, divalent form that predominate in anoxic waterlogged soils (Ponnamperuma *et al.* 1969).

Surprisingly, Fe levels in both plants examined here were considerably below that expected of vegetation growing in a soluble Fe enriched area. Grass growing on serpentine, for example, have been shown to accumulate Fe to concentrations in excess of 3,000 μg/g dry wt. (Johnston and Proctor 1977) and usually contain between 100-200 μg/g dry wt. under normal pasture conditions (Underwood 1962). Mn concentrations, on the other hand, were generally at the higher end of the range normally encountered in other plant species (Loneragan 1975, Stevenson 1986).

Because of their chemical similarity and related biochemical functions in plants, Fe and Mn are mutual antagonists and can influence the uptake one another in plants depending upon their respective concentrations in the growth medium. Such antagonism is well known and usually occurs in acid soils that contain large amounts of Mn (McKenzie 1977, 1980). However, it is conceivable that similar antagonistic effects can also occur in waterlogged soils enriched with both elements in a biologically available form. This would certainly explain the somewhat depressed Fe levels in both wetland plants examined here.

Evidence to indicate that Mn is suppressing the uptake of Fe in the two local plants analyzed can be seen by examining the Fe/Mn ratios of each species. Under normal conditions the Fe/Mn ratio in plants generally lies between 1.5 and 2.5. Below this range, symptoms of Mn toxicity and/or Fe deficiency may occur, while above 2.5 the reverse is true (Kabata-Pendias and Pendias 1992). In the present study Fe/Mn ratios ranged from 0.15-0.45 for *F. tristachya* and from 0.22-1.61 in *L. microphyllum*. According to Kabata-Pendias and Pendias (1992), the phytotoxic threshold for Mn, in most plants, is around 500 µg/g on a dry weight basis. This value is very close to levels determined in several plant samples analyzed during the present study. Thus, Fe deficiency and Mn toxicity may well be two problems facing the indigenous plants growing in the wetland study area.

Of additional importance here is the fact that Fe and P are also antagonists and high soil concentrations of one will suppress uptake of the other (Olsen 1972). Thus, the issue of diminished P reserves in the wetland soil may be exacerbated to the point where local plants are also P deficient as a result of the high background concentrations of Fe in the surrounding soil.

Copper and Zinc

Cu and Zn are essential for plant growth and concentrations found in plant tissues are generally a function of levels in the surrounding soil (Kabata-Pendias and Pendias 1992). However, high concentrations of soluble Fe can reduce Cu and Zn absorption from soil solutions, and vice versa, and this may be important here. Apparently species growing in waterlogged soils frequently show signs of Cu and Zn deficiency (Neue and Mamaril 1985, Ponnamperuma 1985).

Cu levels in grasses normally range from 3-14 µg/g dry wt. (Underwood 1962) and in other plants may be as high as 40 µg/g dry wt. (Stevenson 1986). Concentrations below 2-3 µg/g dry wt are usually indicative of Cu deficiency (Stevenson 1986).

Background levels of Zn in grasses and other pasture plants generally range between 10-100 µg/g dry wt. with mean values in the vicinity of 25-50 µg/g dry wt (Underwood 1962). Symptoms of deficiency usually manifest themselves at tissue concentrations of 10-20 µg/g dry wt. depending upon individual requirements for this element (Kabata-Pendias and Pendias 1992).

On the strength of the above information, it would appear that the sedge, *F. tristachya*, growing in the wetland study area contains sub-optimal concentrations of Cu and Zn in their tissues, whereas the fern, *L. microphyllum*, does not (Table 8). One possible reason for this may be the ferns rather shallow rooting system which is not exposed to the anoxic wetland soils in the same way as the deeper roots of the sedge.

Nickel and Chromium

There is no evidence that Ni and Cr play an essential role in plant metabolism, although the reported beneficial effects of both elements on plant growth have stimulated speculation that they may have some beneficial function (Mishra and Kar 1974, Metz 1969). Natural levels of both elements in plants are generally low and range from around 0.1-4.0 µg/g dry wt (Underwood 1962, Kabata-Pendias and Pendias 1992). Interactions of both elements with Fe have been reported (Cotaldo *et al.* 1977, Turner and Rust 1971) and may possibly explain the lower concentrations determined in sedge during the present study

CONCLUSIONS

Floodplains in the middle Taelayag River valley afford excellent opportunities to study, in considerable detail, natural evolutionary processes occurring in perennial wetlands. These processes combine to alter waters from several sources moving into and through the pore system of the wetland, and as such affect the chemistry of downstream waters The following conclusions may be drawn from this study.

- Wetlands develop because high volumes of sediment funnel episodically from adjacent badlands through wadis and spread onto the floodplain of the Taelayag River. The Taelayag River is not capable of transporting away this accumulation until the next typhoon season brings large storms, and the deposit stabilizes to a marsh by the acquisition of a plant cover

- The accumulated sediment is porous and permeable, and perfectly situated to intersect all upstream runoff and groundwater seepage thus becoming waterlogged, in effect, a shallow aquifer. In the ensuing dry season waters within this aquifer become acidic and anoxic as plant decay proceeds to take up free oxygen. By the peak of the next wet season, however, the high volume of oxygenated waters displace the anoxic waters.

- Chemistry of Taelayag River wetland waters is ultimately dictated by the chemistry of volcanic rocks and seasonal variation in rainfall and pore water volumes. However, the detailed chemistry is controlled by subtle fractionation driven by REDOX and pH dependent processes. Partitioning begins in the surface and shallow subsurface weathering of volcanic rock where transition metals (Fe, Mn, Zn, etc.) split from alkai and alkaline earth metals (e.g. Na, K, Ca, Mg) and both groups split from silica and alumina.

- Wetland soil pore waters can buildup dissolved iron concentrations that reach three orders of magnitude above surrounding stream waters and may have manganese concentrations of two orders of magnitude beyond those same streams. . Both metal levels exceed Safe Drinking Water Standards. Metal concentrations appear to vary inversely with rainfall. The high concentration of iron and manganese in Taelayag River wetlands is probably instrumental in the pervasive precipitation of iron and manganese oxides around reef carbonate detritus near the mouth of this river.

- Wetland soil pore waters are impoverished in nutrients. Dissolved carbon, total phosphate, and nitrogen levels indicate rapid, almost instantaneous take-up by the wetland plant community The lack of nutrients in the local, adjacent badlands soils indicate that nutrients are probably supplied by the main Taelayag River during infrequent flood sedimentation.

Further Recommended Research

The limited scope of this project was consistent with time and budget constraints. We are aware that the research frames only a snapshot of one wetland in one year. As many questions were raised as were answered; they include questions related to kinetics (rates and completion) or certain reactions, seasonal cyclically in wetland chemistry and water quality, and cause-effect relationship between wetlands and downstream and adjacent marine water resources.

We consider the site in the Taelayag River Basin to be a scientific treasure and are making here a plea that the work described in this report be extended to include a) undertaking a five-year water chemistry monitoring program and b) developing the site for educational field trips. Already, the writers, at their own expense, have led students from high school and college science classes into these badlands/wetlands for the purpose of hands-on instruction on environmental issues. An extended monitoring program could lead to a major breakthrough relating environmental chemistry of upstream wetlands to downstream water chemistry and quality on a river that has already been targeted and approved for hotel and resort development near its mouth. The monitoring program could be effectively run at a maintenance budget of about $5,000 per annum.

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APPENDIX A

DATA TABLES

Table 1: Electrochemical Parameters Measured in Surface Waters from the Wetland Study Stations

Date	Station	Sp. Cond. (µS/cm)	pH	Eh (mv)
29 Aug. '96	S1	247.3	6.93	+5
	S2	152.7	7.43	+20
	S3	253.5	7.28	+14
11 Sept. '96	S1	nd	nd	nd
	S2	nd	nd	nd
	S3	nd	nd	nd
01 Oct. '96	S1	227.9	7.44	+23
	S2	165.1	6.09	- 32
	S3	238.6	7.40	+21
15 Oct. '96	S1	230.6	6.58	- 17
	S2	144.2	6.12	- 32
	S3	239.8	6.83	- 7

nd = no electrochemical data taken

Table 2: Electrochemical Parameters Measured in Pore Waters from the Wetland Study Stations

Date	Station	Depth (cm)	Sp. Cond. (μS/cm)	pH	Eh (mv)
29 Aug. '96	P1	30*	nd	nd	nd
		60	nd	7.34	+17
	P2	30	nd	6.81	- 12
		60	nd	6.71	- 14
	P3	30*	nd	nd	nd
		60	nd	6.85	- 10
	P4	30	nd	7.55	+26
		60	nd	6.76	- 12
	P5	30	nd	6.95	- 3
		60	nd	6.78	- 11
	P6	30	nd	7.48	+24
		60	nd	nd	nd
	P7	60*	nd	nd	nd
	P8	90	nd	7.68	+33
01 Oct. '96	1	30	nd	nd	nd
		60	nd	6.23	- 33
	2	30	396	7.32	+15
		60	273	6.93	- 4
	3	30	540	7.31	+18
		90	278	6.82	- 11
	4	30	nd	7.68	+34
		60*	nd	nd	nd
	5	30	769	7.16	+6
		60	404	7.38	+20
	6	30	nd	nd	nd
		90	319	7.49	+24
	7	60	493	7.14	+7
	8	90	nd	7.24	+20
15 Oct. '96	1	30	nd	nd	nd
		60	326	6.31	- 35
	2	30	385	6.64	- 20
		60	321	6.34	- 33
	3	30	nd	nd	nd

	90	nd	6.23	- 43
4	30	436	7.06	+3
	60*	nd	nd	nd
5	30	nd	6.70	- 16
	60	400	6.75	- 12
6	30	nd	nd	nd
	90	nd	6.59	- 22
7	60	499	6.76	- 13
8	90	332	6.65	- 18

nd insufficient sample available for electrochemical analysis

* lysimeter dry inside

Table 3: Chemical Parameters Measured in Unfiltered Surface Waters from the Wetland Study Stations

(values[1] in parenthesis represent data for filterable or "dissolved" fraction expressed as a % of the "total" concentration determined in the corresponding unfiltered sample)

Date	Station	ug/l				mg/l		
		NH_3-N	NO_2^--N	NO_3^--N	PO_4^{3-}-P (ortho)	COD[2]	Fe	Mn
9 Aug. '96[3]	S1	3	<1	16	6	<5	0.39	0.54
	S2	2	3	210	5	9	<0.05	0.20
	S3	3	<1	9	2	<5	0.42	0.56
11 Sept. '96	S1	6 (88)	<1	12 (100)	<1	13 (100)	0.45 (87)	0.43 (97)
	S2	8 (66)	3 (?)	153 (100)	2 (100)	13 (63)	<0.05	0.11 (100)
	S3	6 (64)	<1	3 (68)	2 (100)	37 (15)	0.45 (100)	0.49 (100)
01 Oct. '96	S1	4 (100)	<1	42 (100)	3 (?)	180 (100)	0.40 (73)	0.51 (100)
	S2	7 (53)	2 (?)	378 (73)	2 (?)	100 (100)	0.10 (100)	0.19 (100)
	S3	7 (81)	<1	4 (60)	5 (?)	13 (100)	0.57 (89)	0.71 (100)
15 Oct. '96	S1	3 (?)	<1	21 (86)	<1	9 (78)	0.33 (61)	0.59 (100)
	S2	<1	<1	225 (100)	2 (?)	30 (10)	<0.05	0.23 (100)
	S3	1 (100	<1	<1	<1	30 (32)	0.20 (100)	0.63 (100)

---

[1] (?) indicates that a value was not calculable as the level in the corresponding filtered sample was below the limits of analytical detection

[2] the chemical oxygen demand (COD), multiplied by a factor of 0.45, provides an approximation of the organic carbon content of the sample

[3] unfiltered samples only were analyzed on 29 August, 1996.

Table 4a: Chemical Parameters Measured in Unfiltered Pore Waters from the Wetland Study Stations

Date	Station	Depth (cm)	ug/l				mg/l		
			NH_3-N	NO_2^--N	NO_3^--N	PO_4^{3-}-P (ortho)	COD	Fe	Mn
29 Aug. '96	P1	30*	nd	nd	nd	nd	nd	nd	nd
		60	nd	nd	nd	nd	312	3.42	6.12
	P2	30	15	<1	14	13	121	8.89	9.53
		60	28	<1	35	11	52	8.18	6.03
	P3	30*	nd	nd	nd	nd	nd	nd	nd
		60	39	nd	nd	nd	250	27.4	4.61
	P4	30	9	<1	9	2	85	<0.1	28.1
		60	19	<1	16	6	106	5.34	66.5
	P5	30	>2000	<1	27	16	505	95.7	11.8
		60	23	<1	34	6	100	10.3	2.99
	P6	30	16	<1	13	14	129	4.09	5.09
		60	nd	nd	nd	nd	>1000	10.3	1.59
	P7	60*	nd	nd	nd	nd	nd	nd	nd
	P8	90	nd	nd	nd	nd	578	3.13	11.5

nd insufficient sample available for chemical analysis

* lysimeter dry inside;

† the chemical oxygen demand (COD), multiplied by a factor of 0.45, provides an approximation of the organic carbon content of the sample

Table 4b: Chemical Parameters Measured in Unfiltered Pore Waters from the Wetland Study Stations

(values[1] in parenthesis represent data for filterable or "dissolved" fraction expressed
as % of "total" concentration determined in corresponding unfiltered sample)

Date	Station	Depth (cm)	ug/l				mg/l		
			NH_3-N	NO_2^--N	NO_3^--N	PO_4^{3-}-P (ortho)	COD[2]	Fe	Mn
11 Sept '96	1	30	92 (100)	<1	10 (100)	3 (100)	56 (100)	9.26 (97)	7.41 (94)
		60	28 (?)	<1	8 (?)	7 (?)	132 (96)	11.6 (100)	6.82 (93)
	2	30	52 (97)	<1	22 (74)	4 (100)	114 (100)	24.9 (95)	9.29 (99)
		60	43 (100)	<1	20 (100)	7 (87)	352 (41)	32.3 (94)	7.18 (98)
	3	30	11 (?)	<1	27 (?)	3 (?)	57 (83)	25.2 (62)	1.90 (100)
		90	22 (100)	<1	23 (100)	7 (100)	102 (100)	33.3 (97)	6.47 (100)
	4	30	29 (90)	<1	<1	3 (100)	68 (81)	1.39 (95)	26.7 (100)
		60*	nd	nd	nd	nd	nd	nd	nd
	5	30	103 (80)	<1	<1	25 (45)	405 (78)	131 (98)	8.59 (96)
		60	6 (95)	<1	<1	3 (100)	223 (57)	<0.05	2.82 (100)
	6	30	14 (100)	<1	12 (100)	10 (100)	43 (90)	15.1 (100)	6.24 (100)
		90	64 (57)	<1	22 (12)	4 (100)	256 (42)	24.9 (62)	1.07 (100)
	7	60	89 (52)	<1	9 (43)	11 (46)	230 (86)	66.1 (73)	4.59 (100)
	8	90	57 (73)	<1	9 (92)	11 (69)	896 (71)	32.4 (45)	5.18 (100)

nd insufficient sample available for chemical analysis

* lysimeter dry inside

[1] (?) indicates that a value was not calculable as the level in the corresponding filtered sample was below the limits of analytical detection

[2] the chemical oxygen demand (COD), multiplied by a factor of 0.45, provides an approximation of the organic carbon content of the sample

Table 4c: Chemical Parameters Measured in Unfiltered Pore Waters from the Wetland Study Stations

(values[1] in parenthesis represent data for filterable or "dissolved" fraction expressed as % of "total" concentration determined in corresponding unfiltered sample)

Date	Station	Depth (cm)	ug/l				mg/l		
			NH_3-N	NO_2^--N	NO_3^--N	PO_4^{3-}-P (ortho)	COD[2]	Fe	Mn
1 Oct. '96	P1	30	174 (100)	<1	3 (83)	3 (?)	44 (100)	0.17 (100)	5.00 (100)
		60	22 (100)	<1	17 (79)	3 (53)	129 (100)	19.8 (98)	5.38 (100)
	P2	30	38 (100)	<1	1 (?)	48 (100)	46 (45)	0.20 (55)	8.38 (97)
		60	45 (96)	<1	2 (?)	<1	61 (100)	0.20 (85)	6.38 (100)
	P3	30	45 (96)	<1	2 (100)	2 (?)	37 (100)	0.34 (41)	1.34 (100)
		90	13 (100)	<1	2 (100)	<1	59 (100)	0.40 (100)	6.25 (98)
	P4	30	18 (77)	<1	2 (84)	<1	50 (100)	0.11 (?)	30.1 (100)
		60*	nd	nd	nd	nd	nd	nd	nd
	P5	30	23 (?)	<1	8 (?)	70 (?)	225 (83)	140 (96)	4.88 (100)
		60	23 (92)	<1	17 (100)	2 (100)	22 (100)	27.0 (92)	3.63 (100)
	P6	30	10 (92)	<1	15 (38)	8 (33)	27 (100)	17.0 (100)	5.63 (93)
		90	97 (91)	<1	13 (100)	7 (50)	131 (100)	28.3 (100)	0.88 (94)
	P7	60	164 (90)	<1	9 (76)	14 (12)	491 (77)	69.1 (98)	3.5 (100)
	P8	90	110 (100)	<1	14 (89)	3 (75)	89 (46)	20.1 (97)	3.5 (100)

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nd insufficient sample available for chemical analysis

* lysimeter dry inside

[1] (?) indicates that a value was not calculable as the level in the corresponding filtered sample was below the limits of analytical detection

[2] the chemical oxygen demand (COD), multiplied by a factor of 0.45, provides an approximation of the organic carbon content of the sample

Table 4d: Chemical Parameters Measured in Unfiltered Pore Waters from the Wetland Study Stations

(values[1] in parenthesis represent data for filterable or "dissolved" fraction expressed as % of "total" concentration determined in corresponding unfiltered sample)

Date	Station	Depth (cm)	ug/l				mg/l		
			NH_3-N	NO_2-N	NO_3-N	PO_4^{3-}-P (ortho)	COD[2]	Fe	Mn
15 Oct. '96	P1	30	241 (91)	<1	3 (100)	<1	24 (100)	<0,04	5.12 (100)
		60	35 (58)	<1	19 (95)	3 (50)	51 (71)	22,6 (98)	5.79 (100)
	P2	30	35 (100)	<1	4 (100)	<1	12 (100)	2.54 (100)	8.33 (100)
		60	49 (100)	<1	13 (100)	<1	38 (63)	21.5 (93)	6.78 (100)
	P3	30	52 (?)	nd	19 (?)	nd	66 (8)	26.5 (97)	2.80 (65)
		90	22 (98)	<1	85 (22)	<1	202 (40)	26.9 (100)	7.00 (81)
	P4	30	10 (100)	<1	2 (?)	2 (?)	34 (100)	0.07 (?)	28.3 (100)
		60*	nd	nd	nd	nd	nd	nd	nd
	P5	30	9 (21)	<1	10 (93)	84 (2)	95 (100)	84.4 (100)	4.13 (100)
		60	11 (76)	<1	14 (100)	2 (?)	41 (100)	30.4 (96)	4.13 (100)
	P6	30	6 (?)	<1	11 (100)	<1	16 (100)	10.8 (90)	5.34 (100)
		90	100 (?)	<1	16	<1	24 (?)	20.1 (100)	0.82 (99)
	P7	60	44 (86)	<1	7 (100)	9 (17)	118 (87)	58.3 (88)	3.69 (100)
	P8	90	139 (95)	<1	17 (100)	3 (?)	93 (44)	26,9 (89)	4.02 (100)

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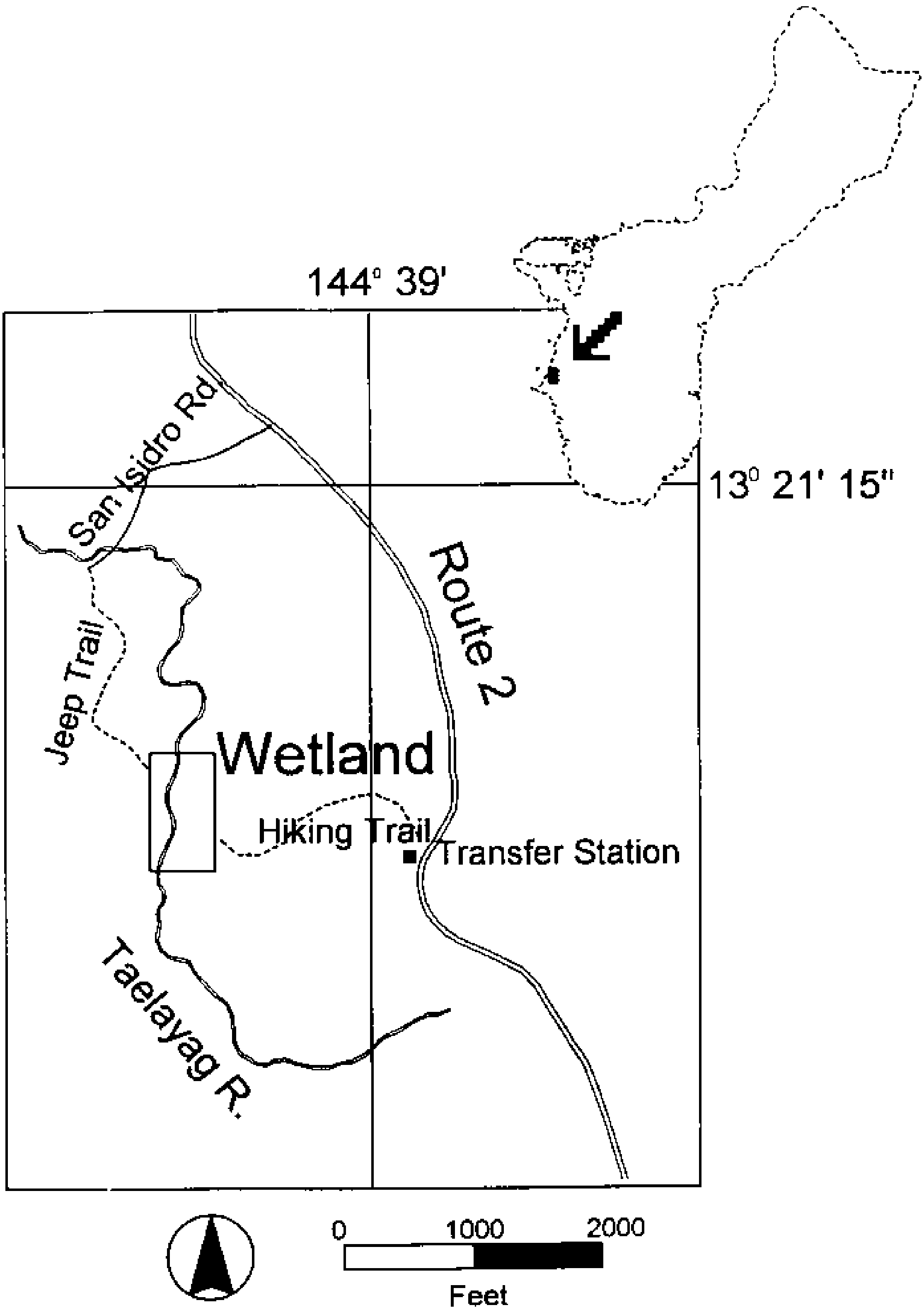
nd insufficient sample available for chemical analysis

* lysimeter dry inside

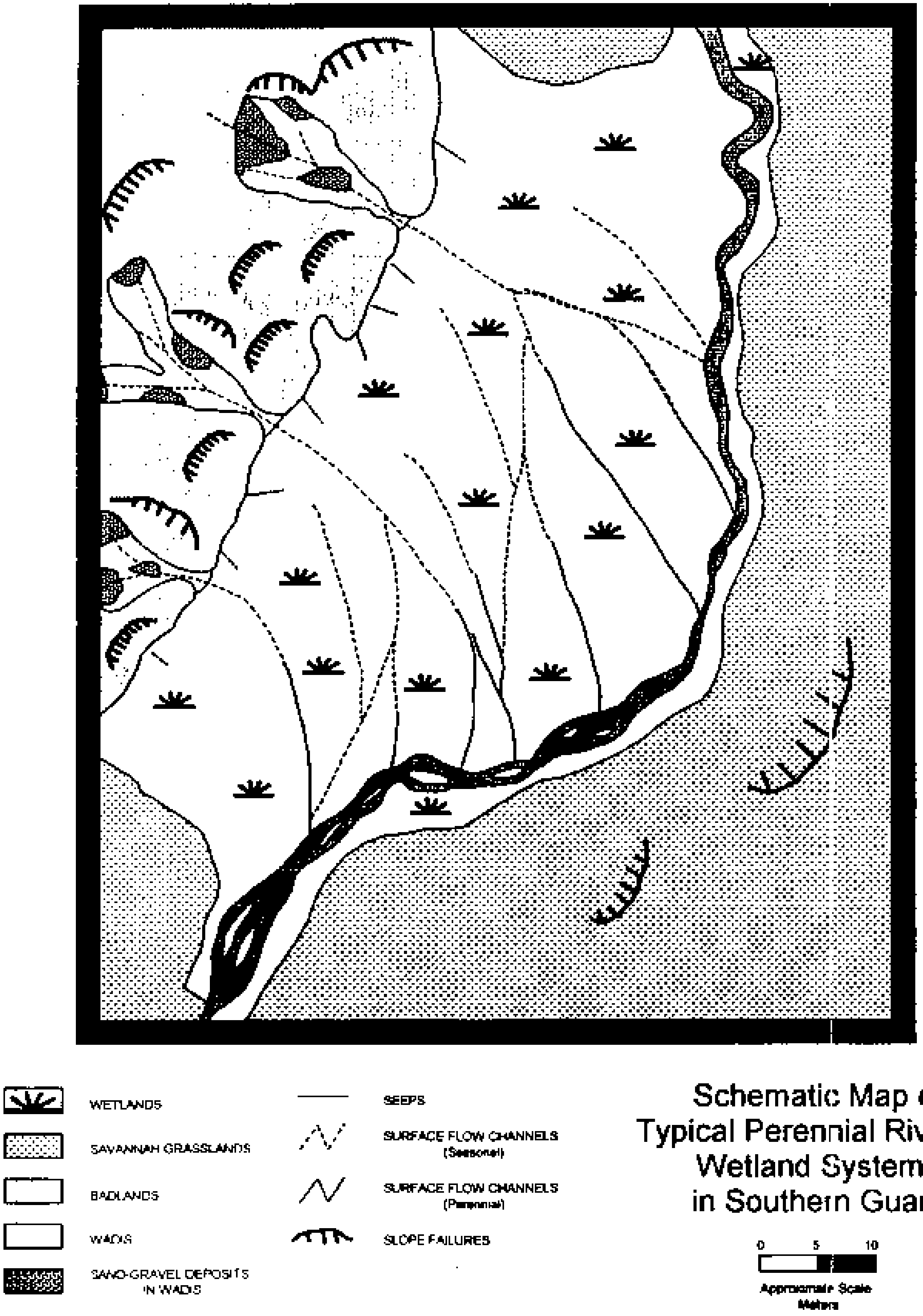
[1] (?) indicates that a value was not calculable as the level in the corresponding filtered sample was below the limits of analytical detection

[2] the chemical oxygen demand (COD), multiplied by a factor of 0.45, provides an approximation of the organic carbon content of the sample



Wetland Location

FIGURE 1



Schematic Map of
Typical Perennial Riverine
Wetland Systems
in Southern Guam

FIGURE 2

Table 7: Trace Metal Concentrations (µg/g dry wt.) in Sub-surface Soils from the Wetland Study Stations

Station	Fe	Mn	Cu	Zn	Ni	Cr
2N HCl Extracts:						
P1	53,432	1,220	164	96.8	242	219
P2	55,532	1,029	151	151	269	242
P3	45,796	3,036	168	130	231	262
P4	40,042	1,064	158	123	247	283
P5	60,735	1,384	181	135	287	295
P6	41,873	1,741	157	126	265	272
P7	50,772	599	135	114	216	220
P8	34,371	839	183	118	178	269
HNO3 Digests:						
P1	96,169	829	177	107	298	351
P2	92,880	1,115	153	143	318	326
P3	96,444	1,798	177	118	274	365
P4	99,659	1,100	176	131	320	390
P5	90,561	1,032	188	126	295	385
P6	95,857	1,553	178	132	301	372
P7	105,148	895	147	124	382	317
P8	82,229	902	172	126	283	335

Table 8: Trace Metal Concentrations (μg/g dry wt.) in Plant Tissues from the Wetland Study Stations

Station	Fe	Mn	Cu	Zn	Ni	Cr
Sedge (*Fimbrystylis tristachya*):						
P1	71.1	485	1.3	8.3	<0.5	<0.3
P2	63.7	468	1.4	14.8	<0.5	<0.3
P3	78.6	540	1.7	7.0	<0.5	<0.3
P4	67.4	268	1.5	7.0	<0.5	<0.3
P5	57.8	246	3.3	8.9	<0.5	<0.3
P6	121	268	2.1	13.1	<0.5	<0.3
P7	99.3	509	1.7	10.0	<0.5	<0.3
P8	58.0	180	1.8	11.8	<0.5	<0.3
Fern (*Lygodium microphyllum*):						
P1	234	483	8.7	70.1	1.8	0.7
P2	210	458	7.2	118	3.8	0.7
P3	74.9	344	10.3	128	9.0	0.7
P4	97.6	169	11.0	54.0	4.4	1.2
P5	67.5	134	10.6	62.3	4.4	1.6
P6	241	149	7.5	65.9	6.3	4.2
P7	112	92.6	10.1	45.8	4.3	1.1
P8	78.2	224	9.0	78.7	2.0	3.7

APPENDIX B

PLATES

Plate 1. Regional Geomorphology of Southern Guam: Photo shows prolific badlands development of uplands, slopes, and divides in Inarajan Fintasa River watershed. Sediment delivered during wet seasons sustains wetlands along that river.

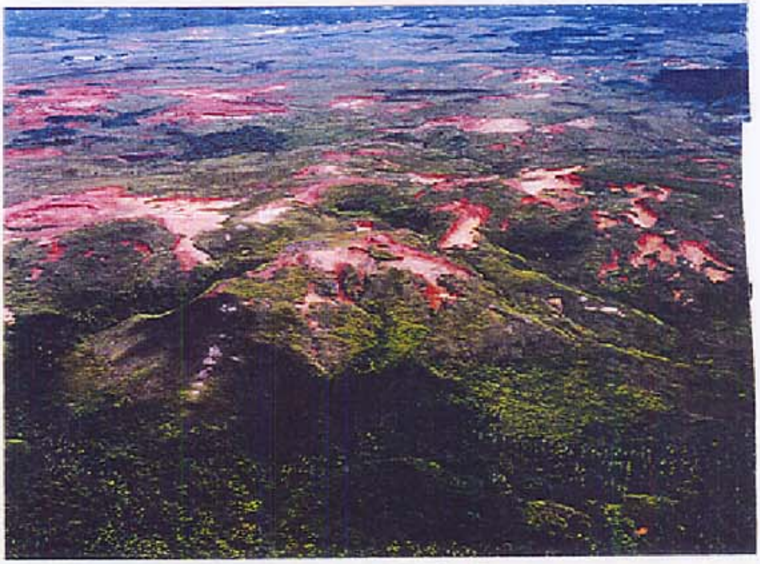


Plate 2. Study Area Overview: Agat-Umatac Rd (Rte 2) in background. Taelayag River in central photo flowing right (S) to left (N). Wetlands approximated by small overprinted rectangle. Boonie rd from San Isidro Chapel Rd in left foreground.

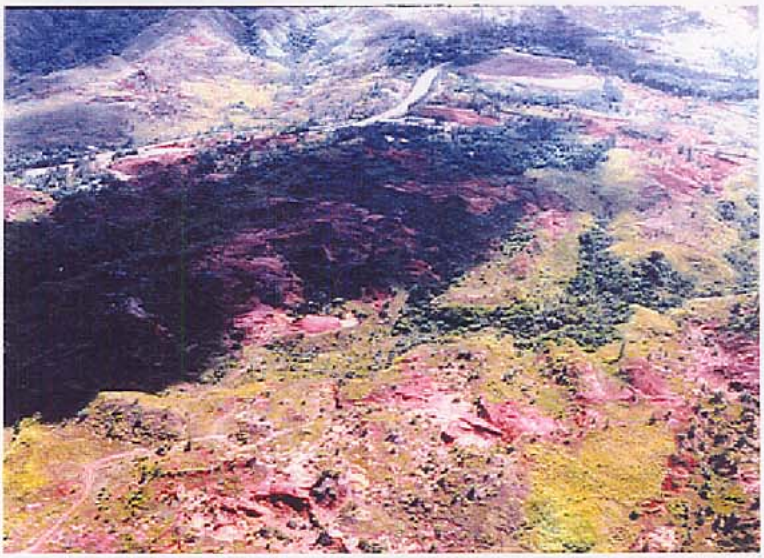


Plate 3; Study Wetlands, Vertical Airphoto: Taelayag R. flowing right (south) to left (north) across central photo (alignment of *Pandanus).* Main water sampling area outlined with overprinted rectangle. Distance from right to left side of photo approximately 350 meters.

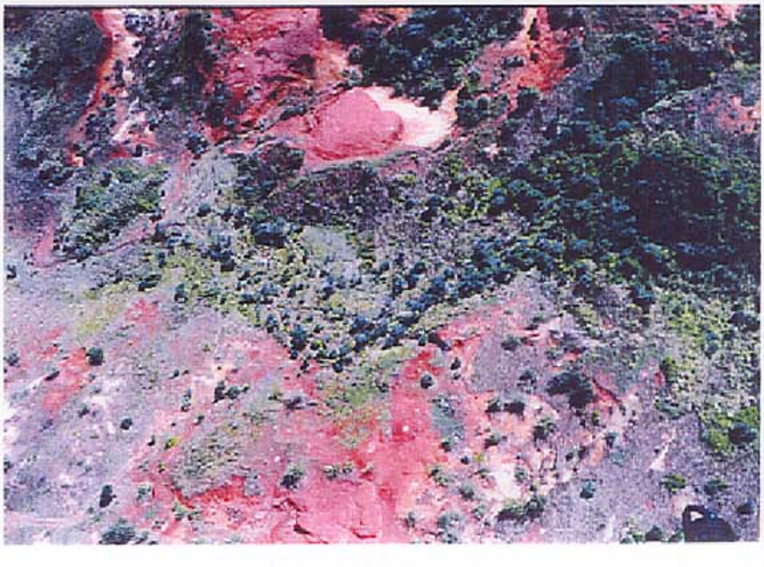


Plate 4: Sampling Stations in Wetlands: Lower left are white bags that mark pore water sampling stations. Water flowing from right to left through wetland. Area overprinted at top of photo is approximate outline of alluvial fan that dams and diverts wadi waters and provides seepage into wetlands

Plate 5: Water Sampling Photo 1: Lysimeter, with marking ribbon and clear tubing that introduces N_2 (tank behind legs). Graduated cylinder used to measure volume of pore waters in lysimeter.

Plate 6: Water Sampling Photo 2: Drawing off water sample from black tube extending into N_2 pressurized lysimeter

Plate 7: Water Sampling
Photo 3: Introducing collected soil-pore water into bottles.

Plate 8: Water Sampling
Photo 4: Cooling down and securing water sampling in ice-filled container.