# NUTRIENT CYCLING IN TROPICAL GRASSES IRRIGATED WITH DAIRY EFFLUENT IN A TROPICAL ISLAND ENVIRONMENT

A DISSERTATION SUBMITTED TO THE GRADUATE DIVISION OF THE
UNIVERSITY OF HAWAI'I IN PARTIAL FULFILLMENT OF THE
REQUIREMENTS FOR THE DEGREE OF

DOCTOR OF PHILOSOPHY

IN

TROPICAL PLANT AND SOIL SCIENCES

MAY 2007

By
Rowena B. Valencia-Gica

Dissertation Committee:

Russell S. Yost, Chairperson
Carl I. Evensen
Nguyen V. Hue
Robert E. Paull
Chin N. Lee

We certify that we have read this dissertation and that, in our opinion, it is satisfactory in scope and quality as a dissertation for the degree of Doctor of Philosophy in Tropical Plant and Soil Sciences.

DISSERTATION COMMITTEE

Chairperson

**To**

**GOD**

**and to His Precious Gifts to me,**

**Edison**

**&**

**Gabriel Elindil**

## ACKNOWLEDGMENTS

Thanks, praises, glory and honor to the ALMIGHTY GOD who makes all things possible and leads us to triumph (II Corinthians 2:14).

I am deeply indebted to my professors who served as my committee members. My committee chair, Dr. Yost, has been a great mentor who unselfishly shared his brilliant ideas and expertise. He has been like a father to me. He never looked at my limitations; rather he patiently honed the skills and potentials that he sees in me. The invaluable guidance and advice provided by Dr. Evensen, Dr. Hue, Dr. Paull and Dr. Lee are greatly appreciated. Thank you very much for helping me to overcome the challenges and to reach significant milestones.

The completion of my studies would not have been possible without the project funding from USDA-CSREES T-STAR program at the University of Hawaii at Manoa and partial financial support from USDA-CSREES 406 Water Quality Initiative Competitive Grants Program. The cooperation of the Pacific Dairy, Inc. management and staff is also highly appreciated.

My very special thanks to the best man of my life, Edison, who has always loved, supported, encouraged and cheered me. His indomitable character has inspired me to also be strong. Thanks for being such a good husband to me and a great loving father to the fruit of our love, Gabriel Elindil, who also inspires me to keep persevering.

I also express my heartfelt gratitude to my fellow graduate students and friends in Sherman Lab. (Tony, Guy, Rosy, Hamidou, Gaoussou, Aminata, Amy and Kyle) and the student helps (Andrea, Billy, Kekoa, Joe, Ted, Sierra, and Doris) who unselfishly shared their time and strengths during the various activities of the project such as harvesting, soil sampling and maintenance of the irrigation system. Thanks a lot to Dr. Uehara for letting us use his lab and equipment, and Jonathan for lending me his soil science book. Thanks also to Dr. Yost, Dr. Evensen, Dr. Uehara and Dr. Deenik for partially supporting my trip to Utah, to Dr. Yost for supporting my trips to Washington & Indiana. Thanks also to Laura, Richard, Tiger, Zhijun and Xiufu for their encouragement. Thanks to Merry Cris and Huang for e-mailing me the ADSC reports; Dr. DeFrank for lending us the sickle mower; Glen and Milton for providing the grass planting materials. Thanks to the secretaries of TPSS, SM-CRSP and NREM, especially Shirley, Susan T., Agnes, Lynne, Gayle and Susan Y., for administrative helps.

I also thank my family in the Philippines—Tatay (Rogeng), Inay (Vicky), Mama (Fely), Papa (Tony), brothers, sisters, nephews, nieces, brothers-in-law, and sisters-in-law for their moral and spiritual support. Thanks also to my brethren from the International Baptist Fellowship for their prayers and encouragement. Thanks also to those people who challenged me leading me to aspire for career development.

To GOD in whom I trust I give all the glory!

## ABSTRACT

In Hawaii and other island environments, dairy producers accumulate large quantities of effluent in lagoons. These lagoons can potentially overflow causing the nutrients and other contaminants to pollute the land and associated water bodies. Alternative uses of effluent are urgently needed for a sustainable and environment-friendly dairy production. This study assessed the effects of effluent irrigation on plant and soil (Cumulic Haplustoll) properties. Five tropical grasses—bana (*Pennisetum purpureum* S.), California (*Brachiaria mutica* S.), signal (*Brachiaria decumbens* S.), star (*Cynodon nlemfuensis* V.), and suerte (*Paspalum atratum* S.)—received subsurface drip irrigation of dairy effluent at two rates based upon the potential evapotranspiration ($ET_p$) at the site (Waianae, Hawaii)—2.0 $ET_p$ (7 to 44 mm $d^{-1}$) and 0.5 $ET_p$ (2 to 11 mm $d^{-1}$).

No excessive increases in extractable soil P (81 to 176 mg $kg^{-1}$) and soil solution total P (3 to 9 mg $L^{-1}$) was observed after two years of effluent irrigation. Soil pH and soil solution pH fluctuated over time due to the high soil buffering capacity. Salinity and sodicity were not observed in this effluent-irrigated soil. Soil electrical conductivity ($EC_{spc}$) declined from 18.0 dS $m^{-1}$ in July 2003 to 2.7 dS $m^{-1}$ in Aug 2006—lower than the U.S. Salinity Laboratory's critical level for classifying soils as saline (4.0 dS $m^{-1}$). Soil exchangeable sodium percentage (6.4 to 10.2%) remained below 15%—critical value critical value for classifying soils as sodic. *Brachiaria mutica* and *P. purpureum* yielded the highest dry matter of 57 and 53 Mg $ha^{-1}$ $y^{-1}$, respectively. Average nutrient removal of grasses was 30 to 187%, 13 to 86% and 2 to 14% of applied effluent N, P and K, respectively. Forage quality was within acceptable levels for feeding to dairy cattle. Modeling results showed that total applied phosphorus determines how many animals

may be raised and how much area may be utilized to produce the forage. Results indicated that irrigating high yielding tropical grasses with effluent at 2.0 $ET_p$ was acceptable for recycling of nutrients from the effluent. Additional monitoring is needed to determine the longer-term impacts of effluent application on soil and plant properties.

## TABLE OF CONTENTS

## LIST OF TABLES

# LIST OF FIGURES

# LIST OF PLATES

# LIST OF APPENDICES

## LIST OF ABBREVIATIONS & SYMBOL

ADF – Acid Detergent Fiber

ADSS – Acid Decision Support System

ANOVA – Analysis of Variance

AWR – Actual Water Requirement

CAFO –Confined Animal Feeding Operations

CAST –Council for Agricultural Science and Technology

CIAT – Center for International Agriculture and Technology

CP – Crude Protein

CTAHR – College of Tropical Agriculture and Human Resources

DOC – Dissolved Organic Carbon

DCPD – Dicalcium Phosphate Dihydrate

DRP – Dissolved Reactive Phosphorus

DSSAT – Decision Support System for Agrotechnology Transfer

EC – Electrical Conductivity

EPA – Environmental Protection Agency

EPP – Exchangeable Potassium Percentage

ESP – Exchangeable Sodium Percentage

$ET_p$ – Potential Evapotranspiration

HA – Hydroxyapatite

ICP-AES – Inductively Coupled Plasma-Atomic Emission Spectroscopy

LF – Leaching Fraction

LSD – Least Significant Difference

MCP – Monocalcium Phosphate

NDF – Neutral Detergent Fiber

NDSS – Nitrogen Decision Support System

NMR – Nuclear Magnetic Resonance

NPDES – National Pollutant Discharge Elimination System

NRCS – Natural Resources and Conservation Service

NSF – National Science Foundation

NuMaSS – Nutrient Management Support System

OCP – Octacalcium Phosphate

PDSS – Phosphorus Decision Support System

PSC – Phosphorus Sorption Capacity

SAR – Sodium Absorption Ratio

TCP – Tricalcium Phosphate

# CHAPTER 1. INTRODUCTION

## Challenges Confronting Dairy Producers in Island Environments

Livestock and dairy production in Hawai‘i and possibly, other island environments is a concentrated, animal-intensive operation. They are also mostly dependent on imported feeds and other materials to sustain their operation (Hao, 2006), resulting in a tremendous influx and accumulation of nutrients and salts. These nutrients and salts can cause pollution of land and water bodies receiving the effluent. If the animal wastes are not re-utilized or exported, an open nutrient cycle is created in the island milk production system.

Producers are challenged to maintain a profitable business while dealing with environmental regulations imposed by the U.S.-Environmental Protection Agency (U.S.-EPA) and mounting pressures from urbanization. Although most animal feeding enterprises in Hawai‘i and other island environments operate on a relatively small-scale, most of them generate large amounts of wastes, especially liquid wastes, for which they have limited land area for re-utilization. Manures/liquid wastes from animal industries are among the most important contributors of nutrients, sediments, toxins and pathogens (Department of Health-Hawaii, 1996; U.S.-EPA, 1998). This is especially true in cases where organic sources are applied to the land more for the purpose of disposal rather than for the utilization of the nutrients they can supply.

Environmental regulations require collection, control and use of manures and wastewater in a manner that prevents discharge into water bodies or into environments outside the boundaries of the generating enterprise. Under the U.S.-EPA’s effluent limitation guidelines, confined animal feeding operations (CAFOs) are required to contain all the manures and wastewater as well as the runoff from a 25-y recurrence interval, 24-h duration rainfall event (Sweeten et al., 2003; U.S.-EPA, 1995). Complying with this regulation is very critical for animal producers in Hawai‘i and island environments due to the very limited land available for disposal of animal wastes, proximity of their operations to the ocean, and the sensitivity and vulnerability of the receiving coastal waters to pollution.

### Approaches to Animal Waste Management at the Farm Level

Efforts to comply with these regulations have encouraged animal feeding operators to implement numerous strategies for animal waste management. These practices include minimizing waste generation, modifying animal diet, recycling wastes, adding amendments to reduce P in the wastes, storing liquid wastes in lagoons, waste pasture irrigation, and solid separation from slurries, among others.

On-site storage of wastewater in lagoons is still the most common option for animal producers and is intended to increase the residence time to allow the liquid to evaporate and the solids to settle, which in turn, are used for composting (Plate 1.1). Dairy operators usually set up multiple lagoons to increase the surface area for evaporative removal of water. However, this practice results in more concentrated solutes and dissolved elements in the remaining effluent that cannot just be applied on land due to the high salinity, sodium and nutrient contents. Productive re-use of the nutrients and organic materials in the effluent is, therefore, limited. Many lagoons have limited capacity and can potentially overflow during the rainy period, allowing the nutrients, soluble solids and microbiological contaminants to be carried by floodwaters that eventually drain into the streams and lagoons. Concerns are also raised about odor pollution, especially when effluent is being applied on the land using sprinklers.

**Plate 1.1. Dairy effluent lagoon near a coast, Oahu, Hawai'i.**

The inability of many livestock and dairy producers to address waste management problems, in addition to the soaring costs of imported feeds, have driven them out of business, leaving Hawai'i and other island environments more dependent on external supplies of meat, eggs and milk. In 1999, there were about 67 livestock operations in Hawai'i. By 2004, only about two-thirds of the livestock producers in Hawai'i remained in business (The Honolulu Advertiser, 2004). Those who are still in operation have to invest in technology to comply with environmental regulations on handling wastes and lagoon effluents. Thus, livestock and dairy producers urgently need information on all possible options for handling animal wastes effectively to ensure their economic existence while also protecting the environment.

### Alternative Solutions to Dairy Waste Management Problems

Collecting the effluent in the evaporation lagoons is usually not sufficient to dispose of the wastewater and its nutrients. Given the quantities of effluent generated by dairy producers, it makes sense to utilize the effluent for the production of forage, which in turn, can be fed to dairy cattle. This is a sound approach to addressing the problems on animal waste disposal in Hawai'i and island environments.

Under the grazing system (Plate 1.2), cattle typically recycle 60 to 90% of the nutrients they consume or remove from pastures in manure and urine (Haynes and Williams, 1993; Bussink, 1994). This does not always happen in dairy operations because the liquid animal wastes are normally collected in storage lagoons. Thus, the only way nutrients from these wastewaters can be recycled is to apply them back to soils planted to forage grasses. The valuable water and nutrients contained in dairy effluent may be effectively and sustainably utilized to maintain or improve soil fertility and enhance grass production and quality. This strategy recycles the nutrients and water, instead of allowing them to pollute the surface and coastal waters. This recycling closes the open nutrient cycle in the milk production system. Applying effluent to the soil also helps to reduce the nutrient levels in wastewater because the soil can act as a filter. The soil physically filters the particles (and the nutrients) from the effluent, breaks the particles down, and incorporates them into the soil structure. The soil also absorbs the nutrients and serves as a medium for conversion into a plant-available form.

**Plate 1.2. Dairy cattle grazing operation, O‘okala, Hawai‘i.**

Forage production areas are generally very desirable recipients of animal wastes and effluents, both for economic and environmental reasons. Animal feeds in Hawai‘i are currently imported, thus, increased productivity of forage grasses would allow dairy/animal producers to save on feed costs. Forage grasses normally need fertilization for improved production especially in intensive grazing/cutting systems and this could account for part of the expenses in animal/dairy production. Re-applying animal effluent for forage production enhances the efficiency in dairy production and minimizes environmental pollution. Animal solid wastes and effluent are excellent sources of plant nutrients (Sharpley and Sisak, 1997) since they contain all the elements necessary for plant growth (CAST, 1996). This organic resource can supply the nutrients and water required by the forage grasses, thereby eliminating or reducing the need for inorganic fertilization of the grasses, or reducing freshwater irrigation. These types of amendments also supply a lot of organic matter that improves the soil structure and water holding capacity. The presence of organic matter in the topsoil contributes to the soils net negative charge, thereby increasing the cation exchange capacity of the soil (Uehara and Gillman, 1981). With high biomass productivity of the grasses, this strategy could minimize feed importation, thereby reducing the dairy production costs and influx of imported nutrients to the island(s). Environmental benefits include reduced risk of erosion and runoff because the grasses provide excellent ground cover and act as filter strips. For example, erosion in grassland was only 1/84 to 1/668 that of erosion in areas planted to wheat or corn (Mugaas et al., 1997). Runoff was only 1/84 to 1/120 in turfgrass than in a tobacco field (Gross et al., 1990).

## The Concerns

Numerous benefits are obtained from land application of manures and lagoon effluents, however, improperly managed application can contribute to the degradation of soil and water bodies. For one, application could elevate the concentrations of nutrients and salts, and pathogens in the soil and vegetation (Mathers and Stewart, 1974; Dantzman et al., 1983; Aldrich et al., 1997). Land application of effluent could also lead to accumulation of P in the soil to excessive levels beyond the capacity of the plant to absorb or be in forms that the plant cannot absorb (Zhang et al., 2002). For example, a survey in Hawaiʻi (Yost et al., 1999, unpublished) revealed that many soils used for animal effluent application have high P levels, ranging from 800 to 3000 mg $kg^{-1}$ Modified Truog P. Nutrients and salts that accumulated in the soil could either be transported by runoff to surface waters or leach into the groundwater. Concerns with excess loading rates become greater when rainfall is very low or irrigation water is insufficient to ensure good growth and nutrient uptake of the crops (Aldrich et al., 1997). A high concentration of soluble salts, especially Na, relative to other ions such as Ca and Mg in soils that are high in clay and silt could result in clay dispersion, which then reduces the soil's permeability to air and water (Clark et al., 2000). Application of animal effluents with high sodium absorption ratio (SAR) and low electrical conductivity (EC) could also reduce the soil hydraulic conductivity through dispersion and destruction of soil aggregates (Patterson, 1999).

When crops take up nutrients from effluents in excessive amounts, nutrient imbalance in the forage may result, that can cause health problems to grazing animals (NSF, 1996). For example, high levels of K in the soil can suppress grass uptake of Mg, causing a disease in cattle known as grass tetany (Robinson and Eilers, 1996).

Another important consequence of repeated effluent application is the leaching of the water, which carries nutrients that could potentially contaminate the groundwater. In Hawai'i, many soils are characterized by very good aggregation (Uehara and Gillman, 1981) and the downward movement of nutrients can be a problem.

Given the potential negative impacts of animal wastes and effluents, managing nutrients from these resources is an urgent priority especially in light of the increasing pressures on dairy producers from the society and environmental regulators. Developing effluent management strategies related to nutrient accumulation and leaching are urgently needed. These strategies need to capture the benefits of utilizing animal wastes as inputs to livestock forage production systems as well as to minimize environmental pollution associated with this kind of enterprise. New information on the performance of different forage grasses in soils receiving animal effluents must be developed so that animal effluent application for forage production can be actively promoted as a mode of using this important resource.

Taking advantage of the P sorption capacity of the soil can also be an important means by which effluent application can be maximized without creating environmental problems. Phosphorus sorption studies are useful in assessing the suitability of a site for animal waste disposal. Such studies provide an estimate of whether the crops/grasses will have sufficient P for uptake and normal growth. Also provided is information on how much P can potentially be sorbed by the soil, thus, not leached from the system. Another effluent management alternative is to develop ways by which nutrients, particularly P, can be incorporated in the deeper layers of the soil profile, especially at the zone where plant roots are concentrated. Such alternative offers several benefits that include preventing erosional losses of P, maximizing the total area for effluent application, supplying P nearest the zone of plant root absorption, and maximizing the use of effluent water.

# CHAPTER 2. REVIEW OF LITERATURE

## Impacts of Effluent Application on Soil and the Environment

### Positive impacts

The application of animal effluents to the soil can increase the amount of nutrients such as N and K in the soil (Aldrich et al., 1997; Roach et al., 2001). Silage corn yields are increased by the application of dairy waste compost or liquid dairy waste, and repeated application of dairy waste compost results in higher soil organic C, total N and available P and K (Habteselassie et al., 2006). Effluent application can also supply significant amount of P, but this is more often viewed as a negative impact rather than positive due to the likelihood of P accumulation and consequent pollution of associated water bodies.

### Negative impacts

Excessive and long-term applications of lagoon effluents can have numerous negative agronomic and environmental effects. It can elevate the nutrient concentrations and human pathogens in the soil, which could either be transported by runoff to surface waters or leach into the groundwater. Effluent application could also change the chemical balance in the soil especially C-N-S-P (100:10:1:1) (Patterson, 1999). Excessive application of lagoon effluents may also lead to high salinity/sodicity (Evans et al., 1977). Consequently, a high concentration of soluble salts especially Na in soils high in clay and silt could also result in clay swelling and dispersion (Clark et al., 2000), with consequent reduction in soil hydraulic conductivity and infiltration rate (Patterson, 1999), or development of a manure seal that clog the soil pores (Maule et al., 2000).

Repeated applications of animal manures and lagoon effluents can elevate the levels of N, P, K and other nutrients in the soil and vegetation (Mathers and Stewart, 1974; Dantzman et al., 1983; Aldrich et al., 1997). Long-term application of farm effluent can lead to accumulation of P and heavy metals in the soil (Wang et al., 2004). Concerns of excess loading increase in areas with very low rainfall or where irrigation water is insufficient to ensure good growth and nutrient uptake by the crops (Aldrich et al., 1997). Thus, improperly managed land application of manures and lagoon effluent can contribute to the degradation of soil and water bodies.

High amounts of P are not harmful to the crop or the soil. However, the accumulation of P on the surface can increase the potential of P being carried in surface runoff (Gerritse, 1977; Ebeling et al., 2002; Tabbara, 2003; Hart et al., 2004) or leached in the soil profile (Djodjic et al., 2004; Simard et al., 2000; Sharpley et al., 1994; Simard et al., 1995; Nair et al., 1995). Pollution impacts of excessive P application have been widely reported (e.g., Pierzynski et al., 1994; Williams, 1995; US-EPA, 1998). Hountin et al. (1997) working on Le Bras silt loam soil showed that the leaching or translocation of P compounds is responsible for the distribution of P in the various soil layers. Accordingly, annual applications of various rates of liquid swine manure into the soil increased the labile soil P pool at the surface or the profile. Manure and effluent application can also result in soil pH changes (Patterson, 1999; Nair et al., 2003) with important consequences for P sorption and leaching (Nair et al., 2003).

## Effects of Animal Waste Application on Soil Phosphorus Sorption, Accumulation and Leaching

Long-term and high application rate of P, whether from inorganic (fertilizer) or organic (animal wastes, composts, etc.) can eventually lead to P accumulation in the soil profile (Breeuwsma et al., 1995; Sharpley et al., 1993, 2004; Nair et al., 1998; Sims et al., 1998; Kingery et al., 1994; Kleinman et al., 1999). Plants remove a limited amount of P due either to the low availability of P in the soil (Schachtman et al., 1998), or low plant P requirement, between 1 to 5 μM of soil solution P for most crops (Tiessen, 2005; Barber, 1995), thus, any excess applied P can be expected to accumulate in the soil (Pautler and Sims, 2000). Phosphorus accumulation is more likely in soils that have high P sorbing capacity such as soils derived from volcanic ash (e.g., Andisols) and rich in Fe- and Al-oxides (Uehara and Gillman, 1980). Soils that are rich in Ca can also accumulate P via precipitation (Busman et al., 1998).

Aside from soil characteristics, the quality of effluent in terms of ion composition also affects P accumulation in the soil. Dairy farm effluents contain high concentrations of N, P, and K, as well as various trace contaminants (e.g., heavy metals, organic compounds, and endocrine-disrupting chemicals) (Wang and Bolan, 2004). Wastewater/effluents also contain a high proportion of monovalent cations such as $Na^+$ and $K^+$ (Aldrich et al., 1997), which could cause greater P desorption from the soil (Barrow and Shaw, 1979; Patterson, 2001). Calcium and Mg in both the soil and effluent can affect P sorption/precipitation (Iyamuremye et al., 1996; Celi et al., 1999; Rietra et al., 2001; Josan et al., 2005). Dissolved organic C can also inhibit the crystallization of stable phosphate forms (Harris et al., 1994; Huang and Violante, 1986; Borggaard et al.,

1990) or alter the P sorption capacity of the soil (Breeuwsma and Silva, 1992; Holford et al., 1997; Mozaffari and Sims, 1996; Siddique and Robinson, 2003; Hutchison and Hesterberg, 2004). All of these have important implications on P accumulation and leaching in the soil.

Application of animal wastes has varied effects on the P sorption capacity of the soil. Many factors affect the P sorption behavior of a certain soil receiving such type of inputs. Reduced P sorption capacity of the soil after animal waste addition has been reported (e.g., Reddy et al., 1980; Sharpley et al., 1984; Holford et al., 1997; Phillips, 2002). Reddy et al. (1980) applied different types of animal wastes at relatively high and variable P rates of up to 345 mg P $kg^{-1}$ soil. Decreases in P sorption capacity and increases in several measurements of P solubility and availability per unit of P (100 mg $kg^{-1}$ soil) applied were observed. The type of waste was among the controlling factors on the extent of these changes per unit of P and followed the order: beef = poultry < swine. Similarly, Sharpley, et al. (1984) reported that cattle feedlot manure application resulted in reduced P sorption capacity of the soil and increased levels of total, inorganic, organic and available P in the profile to a depth of 30 cm. A study on the long-term effects of animal waste application on soil properties showed that piggery effluent-irrigated samples generally sorbed less P than non-irrigated soils (Phillips, 2002; Holford et al., 1997).

Various explanations and mechanisms have been offered regarding the capacity of animal waste addition to the soil to reduce P sorption, with implications for P leaching. Lower P sorption with manure application has been contributing to the saturation of P sorption sites (Reddy et al. 1980); competition for (Reddy et al., 1980) or blocking of

(Holford et al., 1997) P adsorption sites from organic acids that are released during the decomposition of organic matter in the wastes; lowering of the P sorption strength by organic anion interactions (Holford et al., 1997); and loss of high-affinity P sorption sites due to previous wastewater additions (Phillips, 2002; Holford et al., 1997).

Contrary to the above studies, many researchers noted an opposite effect of the addition of animal wastes. The soil P sorption capacity can increase. Robinson and Sharpley (1996) observed that poultry litter leachate-treated soils sorbed P more strongly due to the formation of Ca-P compounds. More P as poultry litter is needed to raise soil test P to a given level and that fertilization with poultry litter results in more residual P than inorganic fertilizers. Laboski and Lamb (2004) reported a higher P sorption capacity in one of the soil series they studied (Nicolett series--fine-loamy, mixed, mesic Aquic Hapludoll) although the others had reduced or unchanged P sorption capacity after manure application. The sorption strength of several Australian soils was reduced by the continuous addition of dairy, pig or sewage effluents (Holford et al., 1997) except in a soil with the shortest period of effluent application (three years) and had the highest increases in organic C and Fe.

The mechanisms suggested for the increased P sorption capacity vary. Eghball et al. (1996) attributed it to the precipitation by Ca contained in the soil and in the effluent, particularly in soils with very high Ca content. Also, effluent addition increased pH beyond 8.3 such that it reached the point where P can be precipitated by the Ca present in the soil and effluent. The displacement of Si in soil was also reported to increase the number of potential sites for P adsorption (Celi et al., 1999). The ionic strength (I) of the soil can also be altered by waste/effluent application by affecting the charge density of the diffuse double layer that counters the charge on P adsorbing surfaces (Bar-Yosef et al., 1988). Increased ionic strength reduced the diffuse double layer and consequently, reduced the masking of P sorption sites on clay edges by the negative electric field of clay faces. The diffuse double layer is also thinner when Ca ions, which have two positive charges, balance the negative charge in the soil surfaces, rather than ions that has single positive charge such as Na. Increased P sorption has also been attributed to the effects of organic matter or dissolved organic C (see Section C.4).

The change in P sorption capacity of the soil has important consequences in relation to the leaching of P in the soil profile. In general, soils have a finite capacity to hold P (Muneer and Lawrence, 2004). When the limit of P sorption is reached, as is often the case in soils used in waste/wastewater studies, P begins to move in the soil profile beyond the zone of application (Latterell et al., 1982; Sharpley et al., 1994; Simard et al., 1995; Nair et al., 1995). For example, the low sorption capacity and strength of the waste-treated soils indicated that the new sites adsorb P very weakly and perhaps allowed leaching to occur (Holford et al., 1997). On the other hand, P leaching was low despite high P applications due to the high P sorption capacity of the soil (Djodjic et al., 2004).

A method to determine the soil P sorption capacity was developed by Fox and Kamprath (1970). This method involves equilibration of soil with graded concentrations of P (soil to solution ratio of 1:10 or 1:20) for 4-6 days and subtracting the concentration of P in the solution after equilibration from the concentration of P initially added. The difference between P added and final P concentration in the equilibrated solution is assumed to have been sorbed by the soil. The data from this P sorption determination are usually fitted into different equations such as the Langmuir, Freundlich and Tempkin equations.

### Phosphorus Movement in the Soil Profile

Phosphorus is generally considered very immobile in the soil and low in availability for plant uptake because of the various reactions it undergoes once in the soil such as adsorption, precipitation, or conversion to the organic form (Beck et al., 2004; Holford, 1997). If P moves, it does so for only a short distance (Eghball et al., 1995; Sims et al., 1998) or moves very slowly laterally with interflow (Akhtar et al., 2003). In the past, leaching as a mechanism for P loss was considered less important, especially because P is held very tightly by the soil. Data are now being obtained on the movement of P in the soil profile from soils fertilized with mineral fertilizers (Eghball et al., 1996; Heckrath et al., 1995; Borling, 2003) and animal wastes/effluents (Eghball et al., 1996; Toor et al., 2004; Holford, 1997; Hawke and Summers, 2003). The movement and accumulation of P in the soil profile has been related to the soil's P sorption capacity (Reddy et al., 1980; Gerritse, 1981; Holford et al., 1997; Weaver and Ritchie, 1994), organic matter content of the soil and effluent (Singh and Tabatabai, 1977; Sanyal et al., 1993; Dodor and Oya, 2000), ions present in the effluent (Aldrich et al. 1997; Patterson 2001; Nair et al., 1995; Iyamuremye et al. 1996; Celi et al. 1999; Rietra et al. 2001), soil type/structure (Singh and Tabatabai, 1977; Sanyal et al., 1993; Dodor and Oya, 2000), nutrient uptake of the plants (Pautler and Sims, 2000), forms of P (Phillips 2002) and subsoil properties such as the presence of fissures/cracks, wormholes and holes created by decayed roots (Toor et al., 2004).

In a 2-year study on dairy farm effluent application to grasslands, over half of P loss was observed to occur immediately following effluent application (Toor et al., 2004). These authors also observed that orthophosphate monoesters and diesters comprised only 13% of P in the dairy effluent, but 88% in the leachates. This result suggests that organic P forms are mobile in the soil profile, and are selectively transported through the soil. The results also indicated that P fixation sites in the soil might become occupied by the applied P inputs such as mineral fertilizer and dairy effluent.

## Factors Affecting Phosphorus Sorption and Movement in Soils Receiving Animal Wastes

### Mineralogy, clay content and surface area of the soil

The P sorption capacity of the soil is determined by a combination of the soil chemical and mineralogical properties (Burt et al., 2002; Zhang et al., 2005a). These include Fe and Al oxide (Sanyal et al., 1993; Dodor and Oya, 2000; Börling et al., 2001), clay type and content (Singh and Tabatabai, 1977; Sanyal et al., 1993; Dodor and Oya, 2000), organic matter and C (Singh and Tabatabai, 1977; Sanyal et al., 1993; Dodor and Oya, 2000), pH (Brennan et al., 1994; Dodor and Oya, 2000) and calcium carbonate ($CaCO_3$) contents (Burt et al., 2002). The mineralogy is one of the most important determinants of the P sorption capacity of the soil, with volcanic ash soils having the highest P sorption capacity (Uehara and Gillman, 1980). It is also well-established that fine textured/high clay soils will adsorb much more P compared with coarse textured/low clay soils. Among the fine textured soils, those dominated by 1:1 clays (kaolinite) will adsorb more P than those with 2:1 clays (montmorillonite).

**Adsorption reaction time**

The adsorption reaction time also influences the P sorption capacity of the soil. Initial P sorption is very rapid then slows with time (Barrow, 1983; Agbenin and Tiessen, 1985). Phosphorus sorption estimates are, therefore, affected by the length of incubation time chosen. For example, Aktar and Alam (2001) observed that increasing incubation time resulted in a higher percentage of the P sorbed in an alkaline calcareous soil that received either dicalcium phosphate (contains 17.5% citrate soluble P and 1.3 water soluble P) or single superphosphate (with 7.6% citrate soluble P and water soluble P).

**Ions (Na, Ca, Mg) present in the effluent**

*Effects of Na*

Animal wastes and effluents also contain ions that alter the P sorption capacity of the soil. Application of solutions or liquid wastewater/effluents containing a high proportion of monovalent cations can cause greater P desorption from the soil (Barrow and Shaw, 1979; Patterson, 2001). Animal effluents normally contain high amounts of $Na^+$ and $K^+$ (Aldrich et al., 1997), thus, application to the soil may result in lower P sorption.

*Effects of Ca and Mg*

High Ca and/or Mg concentrations are typical of dairy manure–impacted soils (Nair et al., 1995). For example, almost 80% of the P in the surface horizons of highly manure-impacted New Zealand soils was identified as labile P (readily soluble P) and was associated with Ca- and Mg-P forms.

Calcium in both the soil and effluent can markedly affect the P sorption capacity of the soil (Iyamuremye et al., 1996; Celi et al., 1999; etc). Rietra et al., (2001) studied the interaction of phosphate and Ca adsorption on goethite and observed that more P is sorbed in the presence of Ca and more Ca is sorbed in the presence of P. They suggested that P adsorption reduces the positive surface charge, and therefore, less repulsion for the positive Ca ions. Increased P sorption with increasing Ca concentration could be due to specific adsorption of Ca ions on hydrous oxides and an increase in the positive charge on colloidal surfaces (Barrow, 1972; Kinniburgh et al., 1975). Iyamuremye et al. (1996) explained that the addition of Ca ions to the soil increases the positive charges thereby reducing the repulsion of negatively charged phosphate ions away from soil surfaces. Also, the addition of soluble low molecular weight (<50,000) organic compounds may form complexes with Ca, Fe, and Al, thereby increasing the area for adsorption of the soil material (Gerritse, 1981; Frossard et al., 1995).

Calcium released from some organic wastes may have a significant influence on the sorption and availability of P in soils (Robinson and Sharpley, 1996; Siddique et al., 2000). The addition of Ca in the soil through organic waste application can lead to the formation of Ca-P precipitates in the soil. Thus, the high Ca contents of some manure may be responsible for modifying the P sorption characteristics of manured soils (Robinson and Sharpley, 1996). Increased P sorption in several Oklahoma and Texas soils amended with poultry, beef or swine manure for up to 35 years has been attributed to Ca-P formation (Sharpley and Smith, 1995). Similarly, Robinson and Sharpley (1996) suggested that the increased P sorption after addition of poultry leachate to the soil was due to the formation of stable Ca-P compounds at soil surfaces, which is a strong P sorption mechanism, and/or a larger number of sorption sites in the soil treated with leachate (soil pH 5.4-6.1 in 1:2.5 soil:water ratio). The formation of relatively stable calcium phosphates (e.g., apatite, tricalcium phosphate) in soils amended with animal waste is favored by the high Ca concentrations and pH (approximately 8), as observed by Wang et al. (1995) in the leachates of Florida sandy soils amended with dairy manure. Formation of various Ca phosphate compounds was also reported by Sharpley et al. (2004) who studied the amounts, forms and solubility of P in manure-amended soils in eastern states of the U.S.

The lower P availability in poultry litter and sewage sludge was related to the high Ca content (13.5 and 42 g $kg^{-1}$, respectively) of these wastes (Siddique and Robinson, 2004). They found a positive relationship between Ca and P sorption. Accordingly, Ca in the form of sparingly soluble precipitates influences P chemistry by contributing to the P sorption at the colloid surfaces (Siddique and Robinson, 2004). Davis and Burgoa (1995) observed reduced runoff P losses in soils where dairy lagoon effluent was applied. They suggested that soil $Ca^{2+}$ ion formed the ion pair $CaHPO_4$ with phosphate from dairy lagoon effluent, which could have been readsorbed to the soil colloids.

Qian and Schoenau (2000) investigated the forms and distribution of inorganic and organic P at 2-wk and 16-wk after liquid swine manure addition through sequential extraction. They noted that the initial fate of P from the manure was primarily to enter the moderately labile and stable fractions such as Ca phosphate and organic P forms. Calcium phosphates can form by precipitation following an initial P adsorption onto calcite (Cole et al., 1953; Freeman and Rowell, 1981; Syers and Curtin, 1989). The surface catalyzed precipitation of Ca phosphate even below saturation or the formation of ternary surface complexes can also be expected (Rietra et al., 2001).

Numerous forms of Ca phosphates occur in the soil, ranging from the very soluble monocalcium phosphate (MCP) to the very insoluble fluorapatite (Jones and Jacobsen, 2002). Dicalcium phosphate will keep soluble P levels at approximately 1.5 mg/L or ppm (at pH 7.5), whereas tricalcium phosphate will maintain soluble P levels at about 0.03 mg/L (Lindsay, 1979).

Aside from Ca, Mg in the soil and effluent can also affect P sorption. Josan et al. (2005) tested the hypothesis that in abandoned and active dairy manure–impacted soils, solution P is controlled by a sparingly soluble Mg-P phase that requires many years for depletion. Their results showed that the P released from high intensity areas in dairy-impacted soils during repeated water extractions was more closely associated with Mg than Ca release. They explained that if a sparingly soluble Mg-P phase is responsible for the continued release of P from the manure-impacted soils, additions of amendments such as water treatment residuals is the only possible solution to stabilize the P because no stable Mg-P phase is expected to form. Water treatment residuals are by-products of drinking water purification that contain appreciable amounts of reactive hydrous oxides (Novak et al., 2004). The crystallization of stable Ca phosphate forms can be inhibited by components such as Mg or dissolved organic carbon (DOC) (Harris et al., 1994). Similarly, Arvin (1983) found that Mg may stabilize the amorphous Ca phosphates, such as tricalcium phosphate, and inhibit further crystallization of the more stable forms such as hydroxyapatite.

Nair et al. (1995), however, speculated that Ca-P and Mg-P associations were loosely bound, probably by some weak adsorption mechanism or in the form of poorly crystalline solids, and were available for sustained leaching under suitable conditions.

Recently, Cooperband and Good (2002) presented evidence that sparingly soluble Ca- and Mg-P minerals that are more soluble than apatite, controlled the solution P concentrations in soils amended with poultry manure. Their forms, however, have not been directly identified (Josan et al., 2005).

The reactions of Ca and Mg with the soil and resulting effect on pH are the basis for the use of various types of amendments to increase P sorption. For example, Boruvka and Rechcigl (2003) applied $CaCO_3$, dolomite, gypsum, and $CaCl_2$ in an Ap horizon of a Spodosol in Florida to reduce P leaching by enhancing P retention of this sandy soil. Except for $CaCl_2$, all the other amendments raised the soil pH and improved the soil P sorption capacity.

**Organic carbon**

Effluents contain both organic and inorganic fractions that could interact with soil P components. Erich et al. (2002) showed that sorption of P was accompanied by release of C, suggesting that soluble C may influence P sorption. Reports on the interaction of organic matter and P sorption have been inconsistent. Some researchers claimed that organic matter additions will increase P sorption (e.g., Robinson and Sharpley, 1996; Sharpley and Sisak, 1997), while others reported the opposite (e.g., Swenson et al., 1949; Nagarajah et al., 1970; Singh and Jones, 1976; Hue 1991; Sharpley et al., 1993; Mozaffari and Sims, 1996; Eghball et al., 1996; Sharpley, 1996; Ohno and Crannell, 1996; Ohno and Erich, 1997; Holford et al., 1997; Daly et al., 2001). The inconsistency in observations could be attributed to differences in soil properties and location, types of organic additions, soil pH, and soil mineralogy. For example, highly weathered soils retained more P than slightly weathered ones regardless of P source (Sharpley and Sisak, 1997). Ohno and Crannell (1996) reported that dissolved organic matter derived from green manure decreased P sorption, but those from animal manure did not affect it. Soils with oxidic mineralogy usually have high P sorption capacity, but when such soils contain high concentration of coarse minerals such as goethite, lowering their surface area, P sorption decreases (Jackman et al., 1997). An earlier study by Anderson et al. (1974) indicated that at lower pH, P sorption involved complexation with sesquioxides, but at higher pH, organic P sorption was dependent upon organic constituents rather than on Al and Fe oxides. The duration of incubation may also be a factor for the differences in the findings on the effect of organic additions on P sorption. Bumaya and Naylor (1988) found that longer incubation resulted in increased P sorption.

Several explanations have been offered regarding the capacity of organic amendments to reduce P retentions in soils. Organic matter reportedly reduced P sorption by competing with P for sorption sites (Hutchison and Hesterberg, 2004; Hue, 1991; Holford et al., 1997; Iyamuremye et al., 1996, etc.) or by some other mechanisms. Most investigators agreed that the effect of organic additions on reducing P sorption results from the adsorption of organic matter decomposition products such as organic acids onto sesquioxide surfaces or to formation of organo-metallic complexes with Al and Fe, thereby blocking P retention by soils (e.g., Nagarajah et al., 1970; Singh and Jones, 1976; Traina et al., 1986; Ohno and Crannell, 1996; Iyamuremye et al., 1996a and b; Ohno and Erich, 1997). Organic anions probably block exposed hydroxyl groups on sesquioxide surfaces (Li et al., 1990), complexed surfaces of P sorption sites (Habib et al., 1994; Iyamuremye et al., 1996a and b; Bahl and Toor, 2002), or form five- or six-membered rings with the metal ions (Nagarajah et al., 1970). Others proposed that adsoption of organic matter decreased the bonding energy of adsorbed P in neutral (Weir and Soper, 1963) and calcareous soil (Holford and Mattingly, 1975).

Organic species have been identified as key potential inhibitors of Ca phosphate precipitation (van der Houwen and Vaisami-Jones, 2001; Inskeep and Silvertooth, 1988). Using two small molecular weight organic ligands, acetate and citrate, at $10^{-3}$ *M*, the supersaturation for Ca-P precipitation was not affected by acetate, but was increased by citrate from 10.93 to 11.73 (van der Houwen and Vaisami-Jones, 2001). The increase in supersaturation required for Ca-P precipitation caused by citrate was attributed to binding of citrate on the active growth sites of newly formed crystals, thereby inhibiting precipitation. Meanwhile, Inskeep and Silvertooth (1988) found that organic C reduced the rate of Ca phosphate precipitation in solution. They also observed that humic and fulvic acids inhibit hydroxyapatite formation via organic C adsorption onto seed crystals.

Hue (1991) found reduced P sorption and more available P for lettuce when organic anions, such as protocatechuic acids, were added to selected Hawai‘i soils (Andisol, Oxisol, Ultisol and Vertisol). Accordingly, aliphatic acids were more easily consumed by soil microorganisms than protocatechuic acids and so, the former stays at a much shorter time in the soil and at lower concentration, thereby reducing their effectiveness to minimize P sorption. His study also supported the idea that organic matter competes with P for sorption sites and concluded that greater fertilizer P efficiency was achieved with the addition of organic acids or materials with the same effect such as green and animal manures. Sorption curves of compost:soil mixture shifted slightly to the right relative to that of the soil, indicating that compost additions slightly minimized P sorption (Hue et al. 1994).

Others have found that simple organic acids, fulvic and humic acids had no effect on P adsorption by allophanic soils, which preferentially adsorbed P (Appelt et al., 1975). Singh and Jones (1976) observed an increase in P sorption after the addition of some organic residues in highly P sorbing soils, with an increasing P content in the residue. But not all types of residues with high P content reduce the P sorption capacity of the soil. Haynes and Mokolobate (2001) reported that crop residues (e.g. lucerne, barley and grain opposed to sawdust, wheat straw and maize) with greater P content was more effective in reducing the P sorption capacity of soils.

The presence of organic matter or C can enhance P sorption. Organic C can also inhibit oxide crystallization resulting in higher proportion of poorly-ordered Fe and Al oxides, thereby increasing P sorption. Organic matter sorbs P primarily through complexation with P-reactive cations (Holford et al., 1997; Mozaffari and Sims, 1996; among others). Sample et al. (1980) explained that humic acids could react with Al from soil minerals to form these complexes, giving rise to more surfaces for P adsorption. They concluded that higher levels of organic matter in the soil would, thus, increase P adsorption rather than decrease it. This was supported by Holford et al.'s (1997) and Mozaffari and Sims (1996) studies which indicated higher P sorption capacities of animal effluent-treated soils. Holford et al. (1997) attributed it to the enhancement of soil total C, which forms complexes with P-reactive cations. Soluble organic compounds of low molecular weight form complexes with Al, Fe, and Ca, which then increases soil P sorption capacity and reduces the water extractability of soil P (Siddique and Robinson, 2003). Phosphate can be sorbed through these complexes of humic compounds with Al, Fe and Ca (Bloom 1981; Gerke and Hermann, 1992) possibly by the formation of ternary humic acid-metal-phosphate compounds (Cegarra et al., 1978). An increase in P sorption can also occur when the addition of organic compounds to amorphous oxides in soils prevents their crystallization and increases their specific surface, thus, increasing P sorption (Huang and Violante, 1986; Borggaard et al., 1990). Dissolved organic acids such as citrate, tartrate or formate also can extract metallic ions from mineral surfaces creating new P sorption sites (Traina et al., 1986a). It is also possible that P displaces soluble organic ligands from adsorption sites. Erich et al. (2002) found that amended (compost or manure) and unamended fine, loamy frigid Typic Haplorthod soils both released C as they sorbed increasing amounts of P. The amended soils always had greater amounts of C in solution. Previously, Beck et al. (1999) noted the link between P sorption and C desorption in allophanic soils.

Humic and other organic acids form phosphohumate complexes either by coadsorption or direct complexation reactions (Tan, 1998). Accordingly, phosphates will react with the exposed hydroxyl ($OH^-$) groups present in the functional groups of humic matter, the same way they react with the hydroxyl groups of clay minerals.

Phosphorus sorption was reportedly correlated not with organic matter directly, but with organic C content--due to Al and Fe in the organic colloids (Tisdale et al., 1993) mentioned that. Another indirect effect of organic matter on P sorption can result from the anaerobic decomposition of organic matter. This causes Fe compounds to be dissolved and re-precipitated as amorphous minerals with strong P sorption properties (Sah and Mikkelsen, 1989; Sah et al., 1989a). Although Holford et al. (1997) agreed with other researchers' claims that P and organic anions in effluent block P adsorption sites, yet, they explained that P sorption can still increase because of the creation of new sites for P adsorption on enlarged organic matter-Fe surface complexes.

The addition of organic matter to the soil has also been reported to modify the transformation of phosphate compounds already present. For example, organic acids adsorbed on dicalcium phosphate dihydrate (DCPD) crystal surfaces can act as new nuclei for DCPD crystals preventing any further transformation to octacalcium phosphate (OCP) or hydroxyapatite (HA) (Grossl and Inskeep, 1991; Frossard et al., 1995). A recent study showed that humic materials present in soil can inhibit Ca phosphate transformation and modify the P availability in soils by altering the crystallization behavior in solution (Alvarez et al., 2004).

The dissolved organic C in effluent has been reported to alter soil properties and behavior such as the soil redox potential (Patrick and Jugsujinda, 1992), and can interact with and enhance the transport of other nutrients such as P and other potential contaminants (Mingelgrin and Biggar, 1986; Gooddy et al., 1995; Han and Thompson, 1999; Fine et al. 2002)

**pH and ionic strength**

The amount of P sorbed in effluent-irrigated soils also depends on soil pH. The pH for maximum soil phosphorous availability is between 5.5 and 6.5 (Havlin and Beaton, 1998). At pH below 5.5, P is held by Fe- and Al-oxides, while at pH greater than 6.5, P is precipitated by Ca, thereby making it less plant-available or immobile in the soil (Lindsay, 1979).

With the addition of animal effluent, there is usually a change in the pH of the soil, normally with a tendency towards becoming alkaline. For example, in a laboratory study, Dendooven et al. (1998) reported that immediately after the addition of swine slurry, the soil pH increased from 6.2 to 7.1, increased to 8.3 after 3 d, and then declined to 7.4 after 28 d. Studying the effect of poultry lagoon effluent on soil, vegetation and surface runoff quality, Aldrich et al. (1997) found that effluent application did not change the soil pH of the calcareous, clayey, Burco series soils planted to bermudagrass and Bermuda-ryegrass. But the soil pH at 0-15.2 cm increased in the fine loamy, silicious, Bowie series in bermudagrass irrigated at twice the amount of effluent N (1076 kg $ha^{-1}$ $y^{-1}$) and all bermuda-ryegrass plots. In soils that had been receiving effluent, the increased pH of the soil reduced P adsorption by Ca- and K-montmorillonite and kaolinite (Bar-Yosef et al. 1988). Several workers have also reported increased soil pH with manure application (Eghball, 2002; Iyamuremye et al., 1996; Kingery et al., 1994). However, others reported a decrease in pH with swine effluent application (King et al. 1985). Accordingly, the decline in pH with increasing application rate is consistent with the loss of $Ca^{2+}$ and $Mg^{2+}$ from the surface layer due to effluent additions of $Na^{+}$, $K^{+}$ and $NH_4^{+}$.

The increased pH during microbial reduction in soils where animal effluent is applied may have little direct effect on phosphate dissolution (Hutchison and Hesterberg, 2004). However, the increase in dissolved organic C concentration with rising pH brought about by the deprotonation of acidic functional groups increases (Swift, 1996) may increase the phosphate dissolution (Hutchison and Hesterberg, 2004). Effluent pH and ionic composition, especially monovalent salts (e.g., Na salts), can enhance the release of soil organic C (Reemtsma et al., 1999).

## Forms of phosphorus

The form of P found in animal waste can provide information on the possible fate of animal waste-derived P. Unfortunately, the proportion of a certain form of P found in animal manures and effluents is not always the same. Some studies report that most of the P in animal manure is in the inorganic form (27 to 92% of total P) (Rasnake, 1996; Sharpley and Moyer, 2000; Hansen et al., 2004; He et al., 2004), with smaller amounts in organic and residual forms (Peperzak et al., 1959; CAST, 1995; Nahm, 2003). Mostafa (1999) reported that most P in swine lagoon effluent exists as ortho-phosphate, which is chemically similar to inorganic P from fertilizer. Australian piggery effluents contain as much as 34% organic P (Redding, 2001). Toor et al. (2003) reported that organic P comprises only a small fraction of the total P in dairy effluent. Also, they found that organic P was the dominant form of P in leachates collected from dairy effluent irrigated soils, suggesting that soluble phosphate in effluent is not necessarily mobile in the soil. It appears that the fraction of organic and inorganic P in animal waste varies with animal species, animal diet, storage method, and other factors.

Soil organic P is divided into three main groups (inositol phosphates, nucleic acids and phospholipids), with other organic P compounds existing in diverse unknown forms (MacDonald, 2000). The largest characterized organic P fraction in soil comprises derivatives of inositol hexaphosphate (Dalal, 1977). He and Honeycutt (2001) used ortho-phosphate releasing enzymes (wheat phytase, alkaline phosphatase, nuclease P1, nucleotide pyrophosphatase, or their combinations) to characterize the various organic P fractions in swine and cattle manure. They found that part of the organic P in the different P fractions (obtained by sequential extractions) could be identified by the enzymatic treatments as phytate, simple phosphomonoesters, nucleotides-such as phosphodiesters and nucleotide phyrophosphate. The most dominant form of organic P in NaOH-soluble organic P in pig manure was the simple phosphomonoesters (43%) followed by phytate (39% in water soluble organic P). The two forms also dominate in cattle manure (15% and 17%, respectively). Using $^{31}$P nuclear magnetic resonance (P-NMR), Hansen et al. (2004) found that solid dairy manure had 26% organic P, while liquid lagoon dairy manure had 18 to 73%. The differences in the organic P content of liquid manure were attributed to differences in ponds, seasons, and effluent source. The presence of greater amounts of suspended materials in the samples also accounted for the higher organic P. Phytic acid dominated in solid dairy manure whereas more deoxyribonucleic acid (DNA) was in the liquid dairy manure.

To understand the behavior of animal waste-derived P, some researchers attempted to describe the forms of P in the profile of soils that have received or been treated with animal waste/effluent. For example, Chang et al. (1991) found that most of the P from sludge-amended soils transported downward in the soil profile moved as an organic form. Toor et al. (2003) also found that monoesters and diesters are the major components of organic P in leachate collected from Lismore silt loam soil (Udic Haplustept) fertilized with different combinations of P fertilizer, farm dairy effluent, N fertilizer and cow urine. But more recently, Muneer and Lawrence (2004) reported that most (90%) of the P that had accumulated in the sewage treated soil was in inorganic form reflecting the high dissolved inorganic P content in wastewater.

Phillips (2002) compared the forms of P that leached in undisturbed columns of two soils of varying P sorption capacity wherein piggery wastewater was surface-applied.

In a Vertisol, unreactive P (includes dissolved organic P, soluble organic P, particulate P and non-reactive P) was the more dominant form that leached in the soil cores since molybdate reactive P (MRP) was sorbed by the soil colloids. On the other hand, a higher proportion of leached P in another soil with low P sorption capacity was in the MRP form due to its limited capacity to sorb molybdate reactive P.

**Types of animal waste**

Researchers' findings differ with regard to the form of P and its sorption behavior in the soil, especially in comparison with inorganic P from fertilizers. Eghball and Frank (1995) claimed that the P from manure and inorganic fertilizer P sources do not differ in the way they react with the soil because manure contains 40 to 75% inorganic P out of the total P. Toor et al. (2004) employed $^{31}P$ nuclear magnetic resonance (NMR), which revealed that over 86% of P in farm dairy effluent is in the inorganic orthophosphate form. Some researchers found that manure P was less available than $KH_2PO_4$-P (Elias-Azar et al., 1980), indicating that manure P is more strongly sorbed. Similarly, more inorganic P from pig slurry was sorbed than inorganic P from $KH_2PO_4$ (Bhat and O'Callaghan, 1980). On the other hand, some researchers found the opposite. For example, in a clay loam soil, only a portion of inorganic P in feedlot waste was readily sorbed while 100% of inorganic P from fertilizer was sorbed (Sharpley et al., 1984). Although Toor et al.'s (2004) NMR study showed that organic forms (orthophosphate monoesters and diesters) was only 13% of total P in farm dairy effluent, it dominated in the leachate (88%) collected from stony silt loam soil receiving farm dairy effluent, suggesting the mobility and selective transport of organic P forms in the soil profile.

Between types of manure, there are differences in the amount and forms of P. For example, poultry litters can have as much as 10 times more P than semi-solid dairy manures on a per wet weight basis, but the latter can have more soluble P (17 to 66%) than the former (14 to 22%) (Cooperband and Good, 2002). In swine manure, there are conflicting reports regarding the amount of soluble P depending on the addition of phytase or addition of low phytate feeds in the diet. For example, Xavier et al. (2004) found that 2.0% of total P in manures of pig fed with corn-soybean meal control diet was soluble, whereas higher soluble P (4.7 to 13.1%) were found in the pig manure when phytase or low phytate corn-soybean meal were included in the diet.

The differences in the P composition and behavior between solid and liquid animal wastes possibly explain the differences in fate of P from these sources when applied to the soil. For example, Hansen and Strawn (2003) found that concentrations of P found in the subsurface of the two fields receiving solid dairy manures vs. lagoon effluents were significantly different. Phosphorus concentrations in the subsurface soils of the field receiving effluents were 2-3 times greater than those found in the subsurface of soils receiving solid manures. These results suggest that the liquid lagoon effluent has a higher potential to move through the soil profile below the root zone and potentially reach the groundwater. Their study also showed that organic P did not desorb into the soil solution. Ruling out the possibility of mineralization of organic P, they suggested that inositol phosphate (the major form of organic P) possibly displaced inorganic ortho-phosphate into the solution by competing for the same binding sites. Hansen et al.'s (2004) study on Kecko loamy fine sand (Xeric Haploalcids) also showed that the total P concentration of the subsurface soils was also higher in lagoon-fertilized than in manure-fertilized soils despite the higher P concentration of solid dairy manure than lagoon manure. But the P forms in solid and liquid dairy manures were the same, with about 30% organic P mostly in orthophosphate monoester form. Inorganic P dominated the leachates from lagoon-fertilized soils, suggesting the greater mobility of this P form in the soil studied.

**Waterlogging/Reduced Condition**

Under waterlogged conditions, P leaching is expectedly greater due to the conversion of Fe(III) to Fe(II) leading to release of bound P (Holford and Patrick, 1981; Willlet, 1989; Jensen et al., 1998; Phillips, 1998; Maguire et al., 2000) and due to mineralization of organic P (Goto and Patrick, 1974). Inorganic phosphorus is more mobile through subsoils (Sims et al., 1998; Frossard et al., 1989; Hannapel et al., 1964a,b)

When effluents are added to soil, an anaerobic condition can be created. Under such condition, organic matter oxidation is reduced thereby causing the production of water-soluble metabolites and less carbon dioxide production (Hutchison and Hesterberg, 2004). In effect, dissolved organic C concentration increases, which then enhances phosphate dissolution and its vulnerability to leaching. In fact, the results of their study showed that increased dissolved organic C due to microbial reduction exerts a greater effect on P dissolution than other mechanisms/factors such as pH or reductive dissolution of ferric oxide minerals.

To recapitulate, animal waste application on soils has important consequences on the changes in soil properties, especially the P sorption capacity. Being rich in organic constituents, effluents can influence the P sorption behavior of soils. However, this effect is still not well studied and largely not understood. The conflicting reports on the effects of organic C on soil P sorption can be attributed to the differences in soil properties used by different authors, the duration of waste/effluent application to these soils, the types of organic C source studied, and methods of P sorption determination, among others. It appears that among these factors, soil properties and effluent characteristics have the most important impact on P sorption.

The general effect of organic matter on soil P sorption is towards decreasing it, either by releasing organic acids that block P sorption sites or reducing the bonding energy of sorbed P. Also, when the dominant form of organic P in the organic amendment is inositol hexaphosphate or phytic acid, the P sorption may be lower because these compounds are more preferentially sorbed by clayey and sesquioxide-rich soils (Stewart and Tiessen, 1987) such as those in southeastern U.S. (Leytem et al., 2002), rendering the inorganic P free to remain in the soil solution. Also, the adsorption sites could already be pre-occupied by inorganic P released from mineralization of organic P fractions. The addition of organic materials also induces pH changes in the soil resulting in the changes in surface charge on soil colloids. These latter two explanations though may hold true in strongly acidic soils, but may not in clay or organic matter-rich, alkaline soils, which has usually high pH buffering capacity. However, when potential P adsorbents/precipitants such as Fe, Al and Ca are present in the soil and/or amendments (residues, wastes, effluent), the P sorption may increase or P precipitation may occur.

In soil systems where both Ca and organic C are present in substantial amounts and animal wastes are applied, the combined effects of Ca and organic C on soil P sorption or precipitation have not yet been explored. But it can be hypothesized that Ca present in the soil and animal waste could increase P sorption (possibly by increasing the positive charge in the soil), or precipitate P and eventually the Ca-P compound can be re-adsorbed to soil colloids, leading to P accumulation in the soil. Since the action of organic C is more indirect by reacting with Fe and Al (hydr)oxides, then, it can be expected that the addition of organic C-rich material such as effluent in this soil system will result in competition for sorption sites and consequently, reduced P sorption. When both Ca and organic C are present, there are two possibilities: P sorption or precipitation may increase because both are known sorbents of P and probably, Ca will precipitate P particularly if the soil pH is very high (>8.0). On the other hand, it is also possible that P sorption will decrease because the organic P components of the amendment may compete for P for sorption sites and also, Ca-P precipitation is much slower compared with dissolved organic C-facilitated transport.

The type of organic amendment and its composition (principal form of P and C) as well as soil properties are important factors in predicting the fate of the added P in the soil. Even if the total P concentrations in different types of organic amendments are the same, yet the forms of P in them may vary. These different P forms may also react differently with different types of soils. Some soils receiving amendments with inositol hexaphosphate as the dominant fraction may retain more organic P and allow inorganic P to move in the soil profile by competing for binding sites (Condron et al., 2005). On the other hand, other soils may retain more inorganic P and allow organic P to move in the soil profile. Some forms of organic P (e.g., single phosphate orthophosphate monoesters) are more mobile in the soil especially in soils where animal manures have been applied (Chardon et al., 1997; Condron et al., 2005).

The "saturation" or maximization of P sorption capacity of the soil can be estimated in many ways. However, there is no single method that is universally used due to site-specificity or inherent variability in the characteristics of soils at different places. Also, the use of each method depends on the objective of the study, i.e., whether to re-derive the old equations to consider the new parameters (e.g., linear Langmuir vs. two-surface Langmuir), to present a new approach that is more feasible/easier to use (e.g., degree of P saturation that uses oxalate extraction vs. one that uses soil test P), to simplify the methodology (e.g., traditional P sorption isotherm vs. single point isotherm), or to arrive at an indicator for risk assessment (e.g., degree of P saturation, change point, sorption maxima, etc.), among others. It appears though that those concepts such as the change point and soil P sorption capacity have the widest application since these have been applied to study both P sorption and leaching.

Methods that are easy to implement and interpret are very useful for risk assessments and development of decision-aid tools. Some P sorption methods such as the Fox and Kamprath (1970) are tedious and costly to implement particularly for routine determination. However, this method of P sorption determination is still the most useful for comparison of the P retention capacities of a wider range of soil. The other methods are limited in their application. For example, some indices use the Mehlich or Olsen tests and these may not be applicable to some soils such as the Andisols of Hawai'i. Others are based on soil tests involving the determination of Fe and Al contents, which may not be present in some soils such as the calcareous or alkaline soils. Also, various equations can be fitted to data from P sorption isotherms to provide other means for assessing the extent or strength of P sorption.

## Practical Implications of Phosphorus Sorption

The effect of effluent on P sorption has important implications for the movement of P in the soil. If effluent application causes an increase in the P sorption capacity, a reduced land area can be utilized for the application of this resource. However, there will also be a concern for P accumulation at the surface, especially in soils that have high P sorption capacity such as Andisols and Oxisols. Phosphorus accumulation may also be a concern in soils that are high in pH and Ca content because these conditions can result in Ca-P precipitation.

Application methods that allow for incorporation of effluent to subsurface layers may be an important strategy to maximize effluent utilization and minimize P accumulation on the surface. From an agronomic standpoint, subsurface effluent application will be beneficial since nutrients are applied directly to the root zone where plants can readily access them. This is especially so in alkaline soils where plant root growth can be greatly inhibited thereby limiting the access to macro- and micronutrients. Since the P is redistributed to the subsurface soil layers, subsurface application will also eliminate the concern with surface P being carried by erosion and runoff to water bodies. If effluent application does not increase the P sorption capacity of the soil, then, the downward movement of this nutrient, particularly in soils that sorb low amounts of P, may be the concern instead of P accumulation. From a groundwater quality perspective, this is a threat to consider.

## Impacts of Effluent Application on Nutrient Supply to the Grasses

### Positive impacts

The response of crops and grasses to animal effluent application frequently leads to higher biomass production and nutrient uptake because of the nutrients and water provided to the plants. Roach et al. (2001) reported improved growth and an increase in dry matter yields of pasture grasses equivalent to that from using fertilizer N. Dairy lagoon effluent increases N and P uptake of corn (Kurunc et al., 2004). Anaerobically treated poultry manure effluent, urea and non-anaerobically digested poultry manure influent applied on corn at the same rate based on N resulted in approximately the same grain yield (Field et al., 1986).

### Negative impacts

Effluent application can lead to nutrient toxicity, commonly chloride toxicity (Evans et al. 1977) and reduced crop yield (Brown, 1995). When crops take up nutrients from effluents in excessive amounts, nutrient imbalance in the forage may result that can cause health problems to the grazing animals (NSF, 1996). For example, high levels of K in the soil can suppress grass uptake of Mg, causing a disease in cattle known as grass tetany (Robinson and Eilers, 1996; Wang et al., 2004).

Effluent application can also change the availability of micronutrients such as Fe, Zn, Mn, Mo, Cu, B, Cl and Ni to plants. With effluent application, the level of P in the soil is increased, which in turn, could cause an imbalance in the supply of micronutrients especially Fe and/or Zn (Bennet, 1993). The high pH of effluent and soil are also important factors that could lead to micronutrient deficiency (Moraghan and Mascagni, 1991).

The critical tissue concentration of micronutrients range from 60 to 80 mg $kg^{-1}$ for HCl-extractable Fe and 400 to 1000 for total Fe; 10 to 15 mg $kg^{-1}$ for Mn; and 15 to 30 mg $kg^{-1}$ for Cu and Zn (Moraghan and Mascagni, 1991). Many tolerant crops, however, can have critical leaf concentration of 100 to 5000 mg $kg^{-1}$ for Mn (Hannam and Ohki, 1988) and 200 to 8000 mg $kg^{-1}$ for Zn (Moraghan and Mascagni, 1991). Mills and Jones (1988) suggested that the adequate tissue concentrations for bahia grass (*Paspalum notatum*) are 2.80% N, 0.40% P, 1.80% K, 0.52% Ca, 0.32% Mg, 0.40% S, 100 mg $kg^{-1}$ Fe, 105 mg $kg^{-1}$ Mn, 9 mg $kg^{-1}$ B, 11 mg $kg^{-1}$ Cu, 31 mg $kg^{-1}$ Zn and 0.8 mg $kg^{-1}$ Mo.

Phosphorus and Zn can have either antagonistic or synergistic interactions, with important consequences on Zn uptake, concentrations, translocation and crop response. These interactions are believed to be more dominant in the soil system, where Zn availability and diffusion rates are controlled by P supply (Marschner, 1995). The cause of the antagonistic interaction was suspected to be the formation of an insoluble Zn phosphate, which reduced the concentration of Zn in the soil solution to deficiency levels. The antagonistic interaction is explained by the fact that plants take up Zn as a cation ($Zn^{2+}$) (or also as $Zn(OH)_2$ at high pH) and P as a species of the phosphate ($PO_4^{3-}$) anion (Hopkins and Ellswort, 2003; Alloway, 2004). Positively and negatively charged ions have an electrical attraction to one another. This facilitates the formation of a chemical bond that can form either in the soil or the plant tissue. The P-Zn bond is strong and does not readily break without causing changes in the physical or chemical environment (Alloway, 2004). If excess P binds a large amount of the Zn normally used by the plant, a P-induced Zn deficiency will occur. But P-induced Zn deficiency is most likely to occur only if soil test Zn levels are low (<1.0 ppm) (Alloway, 2004).

There are three major factors responsible for increased plant need for Zn when P is applied to low P soils or reduced Zn plant availability in soil when P is applied such as (1) dilution effect due to growth enhancement, (2) inhibition of Zn uptake or depressed translocation of Zn from roots to shoots by cations, and (3) P-enhanced Zn adsorption in soil (Loneragan et al., 1979; Loneragan and Webb, 1993). In wetland rice soils, the conditions that are likely to induce Zn deficiency include high pH (>7.0), low available Zn, high organic matter content, continuous wetting or waterlogging, a Mg to Ca ratio greater than one, high levels of available phosphate and silica and irrigation with alkaline water (Dobermann and Fairhurst, 2000). The high pH conditions and excessive Ca concentrations in most saline soils are responsible for the low availability of Zn and the occurrence of deficiencies in crops on these soils. In sodic soils, Na ions dominate the exchange sites causing Zn ions to be lost by leaching especially under irrigation with water having a high Na content (Alloway, 2004).The dissolved organic carbon in wastewater (e.g., sewage sludge) can also lead to downward movement of metals such as Zn (Antoniadis and Alloway, 2002).

Similar to Zn, Fe deficiency is common in soils with high pH (Moraghan and Mascagni, 1991). Increasing the organic matter content can also increase Fe availability (organic Fe chelates) in soils (Bar-Ness and Chen, 1991; Moraghan and Mascagni, 1991). Excessive P applications can also cause Fe deficiency by reducing the rate of Fe dissolution (Marschner, 1995) or reduced proteoid root formation (Handreck, 1991). Although reduced conditions converts Fe into more available forms (Lindsay, 1991), this condition may also induce Fe chlorosis in some cases due to reduced supply of oxygen to plant roots (Moraghan and Mascagni, 1991; Lindsay, 1984) and ethylene accumulation

(Smith and Restall, 1971). In some cases, irrigation can also result in Fe chlorosis if the water used for irrigation contains high amounts of $HCO_3$ (Harley and Lindner, 1945).

Other micronutrients such as Mn and Cu also become deficient at a high soil pH (Moraghan and Mascagni, 1991). Formation of relatively insoluble Mn oxide (Sparrow and Uren, 1987) and increased Cu adsorption (Cavallaro and McBride, 1984) occur under high soil pH and high organic matter (Pavanasasivam, 1973; Stevenson and Fitch, 1981) conditions, although high organic matter can also enhance Mn (Hue, 1988) and Cu (Stevenson and Fitch, 1981) availability.

## Nutrient Recycling Potential of Tropical Pasture Grasses

The capacity of pastures, especially in a cut and carry system, to recycle nutrients can be enhanced if the grasses have high dry matter yield and nutrient uptake (Pederson et al., 2002). In this section, the characteristics and performance of the different tropical grasses under various growing conditions will be reviewed. The review will focus only on the forage species which will be considered in this study, namely, *Brachiaria decumbens, Brachiaria mutica, Cynodon nlemfuensis, Paspalum atratum* and *Pennisetum purpureum.*

### General characteristics of tropical forage grasses

Signal grass *(Brachiaria decumbens)* is a vigorous, trailing perennial grass, with morphology that are very similar to paragrass (*Brachiaria mutica*) (Hernandez-Daumas, 2000). Roots and shoots developed at each node of the stolons, eventually forming a dense cover. Signal grass is adapted to humid, tropical areas with a minimum rainfall of 1000 mm per year and a dry season of up to 5 months (Fisher and Kerridge, 1996; Thomas and Grof, 1986). It grows rapidly, forming an aggressive, high-yielding sward

(Hernandez-Daumas, 2000). *Brachiaria decumbens* is classified as a rapid regenerating grass (Chin 1995), which tolerates infertile soils and heavy grazing and trampling (Hernandez-Daumas, 2000). It is the major grass species planted for grazing in Malaysia because it is aggressive and hardy, highly responsive to N fertilization, able to withstand heavy and close grazing, resistant to drought and responsive to cultural management (Chin, 1989; Chin, 1994a).

A relative of signal grass, paragrass or California grass produces very long trailing stems that can reach up to 2.5 m. It can quickly grow to a height of 1.8 m with roots to about the same depth. It tolerates annual precipitation of 870 to 4100 mm and annual temperature of 18.7 to 27.4°C (Duke, 1983). It readily invades disturbed low areas such as canals, and also displaces native vegetation along river and lake shorelines and in marshes and swamps (Holm et al. 1977). This grass persists at areas with rainfall as low as 900 mm $y^{-1}$ (Binh, 1998).

Known as star grass, *Cynodon nlemfuensis* is taller and larger than Bermuda grass (*C. dactylon)* (Harlan, 1976). Star grass spreads by seed and stolons but lacks rhizomes. It is softer, more palatable and has a higher digestibility than common bermuda grass. Star grass usually contains hydrocyanic acid or prussic acid glucosides, but reports of livestock poisoning are rare (Duke, 1983). Star grass grows from sea level to more than 935 m elevation in areas having rainfall of a minimum 381 mm to more than 2032 mm (Fukumoto and Lee, 2003). The most predominant variety in Hawai'i is 'Florico', more commonly referred to as Puerto Rican stargrass. Star grass will not tolerate long periods of flooding and requires a soil pH of 5.5 to 6.0 (Mislevy and Brown, 1991).

Suerte (*Paspalum atratum*) is an erect rhizomatous perennial with a dense fibrous root system, and with good seed production (Partridge, 2003). It can produce good weight gains in cattle when fertilized with N. Together with paragrass, it is a recommended pasture species in the lowland areas of Thailand (Phoatong and Phaikaew, 2001). Suerte is a perennial bunch grass that grows well on wet, flatwoods pastures, and is not recommended for dry sites (Kalmbacher and Kretschmer, 1994).

Bana grass (*Pennisetum purpureum* cv. Bana), also called elephant grass, forms thick clumps by extensive tillering or colonies from basal offshoots or short rhizomes (Langeland and Burks, 1998). It thrives well in areas with annual precipitation of 200-4000 mm and annual temperature of 13.6-27.3°C (Duke, 1983). It grows best in rich, well-drained soil (Duke, 1983), but this grass species is generally able to persist in changing conditions owing to its extensive, deep, fibrous root system (Holm et al., 1977). Now widespread in the United States, it can be found in many areas such as roadsides, canal banks, and fields, but also in scrub, pine rockland, hammock, sink, lake shore, swamp, and prairie habitats (Hall, 1978). It has been well-studied in Hawai‘i for bioenergy production. It was among the few species identified as the most promising crops for ethanol production, having been demonstrated to attain very high biomass yields and strong commercial potential as an energy crop (Gieskes and Hackett, 2003). It requires similar cultural practices, agronomic conditions, and production infrastructure as sugarcane. The projected yield of bana grass for ethanol production, based on several trials conducted in Hawai‘i, is about 37 Mg $ha^{-1}$ $y^{-1}$ for irrigated fields, and about 30 Mg $ha^{-1}$ $y^{-1}$ for unirrigated fields (Gieskes and Hackett, 2003).

**Nutrient uptake/removal potential of tropical pasture grasses**

Nutrient uptake and dry matter production of forage grasses differ among species (McLaughlin et al., 2004a and b, Pederson et al., 2002; Brink et al., 2003; Brink et al., 2004; Zemenchik and Albrecht, 2002; Pierzynski and Logan, 1993), varieties (Brink et al., 2003; Muir, 2001), climate, soil, and management practices (Rowe and Fairbrother, 2003; Macoon et al., 2002). Also, the rate of growth of pasture grasses influences their nutrient uptake and nutritive value (Humphrey, 1999). Nutrient uptake also depends on whether the grass is an annual or perennial. Annual forage grasses normally have lower dry matter production and are less responsive to fertilization, but the nutrient concentrations in their tissues may be higher compared with perennials (Robinson, 1996). Nutrient uptake from soils by different forages is also highly variable and is directly related to the soil nutrient content and physical properties, and forage biomass harvesting (Pierzynski and Logan, 1993).

Only a few studies involve the use of tropical grasses for nutrient removal, especially from animal effluent or other types of liquid wastes being applied to the soil. One study on wastewater reuse was conducted in Hawai‘i to remove N from secondarily treated domestic sewage effluent (Handley and Ekern, 1981). The authors took advantage of *B. mutica*'s rapid, dense, and monocultural growth. With effluent N application rates ranging from 475 to 2600 kg ha$^{-1}$ y$^{-1}$, an average of 69% was harvested in the grass.

With its abundant and extensive fine root system, bana grass/elephant grass has a high nutrient uptake. In Tobago, West Indies, a crop of elephant grass reportedly removed 463 kg N, 96 kg P and 594 kg K ha$^{-1}$ y$^{-1}$ (Walmsley et al., 1978). A recent study on the nutrient uptake of crop-forage systems in the Andean hillsides showed that *P.*

*purpureum* acquired greater amounts of N, P, and K from the soil, although shoot Ca and Mg uptake was similar with imperial grass (*Axonopus scoparius* cv. Imperial) (Zhiping et al., 2004).

Some cultivars of *B. decumbens,* such as the Basilisk, responds to low fertility by increasing the root to shoot ratio (Rao et al., 1996; Hernandez-Daumas, 2000), thereby improving its nutrient uptake. Another characteristic that makes it adapt well to low-fertility acid soils is its capacity to acquire P through an extensive root systems; acquire and use both nitrate and ammonium forms of N and Ca through an extensively branched root system with large numbers of root tips (Rao et al., 1996; CIAT, 1984; Miranda and Boddey, 1987; Hernandez-Daumas, 2000).

Similar to other highly productive forage species, star grass has high nutrient requirements, especially when produced through a cut and carry system (Vicente-Chandler et al., 1974; Mislevy, 2002; Pant et al., 2004). In Vicente-Chandler et al.'s (1974) study, it was found that star grass yielded 28 Mg $ha^{-1}$ annually, which removed 390, 65, 470, 150, and 55 kg $ha^{-1}$ $yr^{-1}$ of N, P, K, Ca, and Mg, respectively. In another study, an average of 140 kg K $ha^{-1}$ was removed annually by tropical star grass when no N was applied, compared with 500 kg K $ha^{-1}$ when 450 kg N $ha^{-1}$ $yr^{-1}$ was applied (Vicente-Chandler and Pearson, 1960).

**Forage productivity**

The productivity of the various tropical grasses varies among locations, season, species/varieties, and cultural management. For example, the different cultivars of *B. decumbens* have been reported to produce 9.5 Mg dry matter $ha^{-1}$ $y^{-1}$ in Costa Rica (Bustamante et al., 1998) and 11.4 Mg dry matter $ha^{-1}$ $y^{-1}$ in Brazil. About one-fourth of the total biomass is produced during the dry season, and the leaf to stem ratio varied from 1.07 to 1.51 (Valle et al., 1993). Well-managed and N- fertilized *B. decumbens* pastures produce high quality, palatable forage enabling good animal performance (Valle et al., 1993). In a cutting experiment in Australia, the annual dry matter production of *B. decumbens* (33,000 kg ha) significantly exceeded that of para grass and guinea grass (*Panicum maximum*) (Register of Australian Herbage Plant Cultivars, 1990). In Africa, tropical America and Australia, dry matter yields ranged from 8 to 36 Mg $ha^{-1}$ $y^{-1}$ (Bogdan, 1977; Harding and Grof, 1978; CIAT, 1978 and 1979). *Brachiaria decumbens* had an average dry matter yields of 24 Mg $ha^{-1}$ $y^{-1}$ in the Philippines (Furoc and Javier, 1976); 83 to 91 Mg $ha^{-1}$ $y^{-1}$ with 5.5 to 15% crude protein (CP) in dry matter in Fiji (Roberts, 1970); and 30 Mg $ha^{-1}$ in North Queensland (Skerman and Riveros, 1990).

Dry matter yields of *B. mutica* ranged from 6 to 12 Mg $ha^{-1}$ $y^{-1}$ in Australia (Miller, 1976), 4 to 26 Mg $ha^{-1}$ $y^{-1}$ in Cuba; 16 Mg $ha^{-1}$ $y^{-1}$ in Surinam and 38 to 40 Mg $ha^{-1}$ $y^{-1}$ in Puerto Rico (Duke, 1983). Novoa and Rodriguez-Carrasquel (1972) obtained 8-week yields of 1 to 6 Mg dry matter $ha^{-1}$ $y^{-1}$ from fertilized and irrigated *B. mutica* during the dry season in Venezuela. Sotomayor-Rios et al. (1973) reported CP yield >2.5 Mg $ha^{-1}$ $y^{-1}$ in Puerto Rico. Dry matter yields of *B. mutica* in lowland paddy pastures in Thailand ranged from 9 to 15 Mg $ha^{-1}$ $y^{-1}$ with CP of 6 to 10% (Phoatong and Phaikaew, 2001).

*Pennisetum purpureum* is one of the highest yielding tropical forage grasses, but its stemmy nature limits its usefulness for grazing purposes (Duke, 1983). In Brazil, some cultivars such as the 'Mineiro' yielded 21 and 30 Mg dry matter $ha^{-1}$ $y^{-1}$ in two rainy seasons and 3.3 and 6.0 Mg $ha^{-1}$ $y^{-1}$ in two dry seasons (Duke, 1983). One Indian cultivar, Pusa Giant Napier, was reported to yield as high as 279 Mg fresh fodder ha$^{-1}$ y$^{-1}$ (Duke, 1983). Average yield of unfertilized grass during the wet season ranges from 3.2 to 5.3 Mg ha$^{-1}$, and for dry season, from 2.4 to 4.4 Mg ha$^{-1}$, with cutting interval of 42 to 63 days (Duke, 1983). Bogdan (1977) concluded that in practice, dry matter yields of *P. purpureum* are more likely to be only 2 to 10 Mg ha$^{-1}$ y$^{-1}$ for low or no fertilizers, and 6 to 30 Mg ha$^{-1}$ y$^{-1}$ for well-fertilized farms. This is far less than a report of 85 Mg dry matter ha$^{-1}$ y$^{-1}$ when fertilized with about 900 kg N ha$^{-1}$ year$^{-1}$ and cut every 90 days under natural rainfall of about 2000 mm per year (Vicente-Chandler et al., 1959).

In Hawai‘i, bana grass/elephant grass can produce as much as 336 Mg of green forage ha$^{-1}$ y$^{-1}$ (Takahashi et al., 1966). It yielded 11 Mg dry matter ha$^{-1}$ in spring and 4 Mg dry matter ha$^{-1}$ in summer, carrying 2.5 and 2.4 animals per hectare, respectively (FAO Grassland Index, no date). In Vietnam, dry matter yields of 25 to 40 Mg ha$^{-1}$ y$^{-1}$ for *P. purpureum* and about 18 Mg ha$^{-1}$ y$^{-1}$ for *B. mutica* (Nguyen et al., 2003) were reported. In Malaysia, average dry matter yields of *P. purpureum* was 25 Mg ha$^{-1}$ y$^{-1}$, with CP of 5 to 10% (Idris and Najib, 2003). In regions with over 125 cm annual rainfall, dry matter forage yields of *P. purpureum* ranged from 27.3 to 37.1 Mg ha$^{-1}$ (Duke, 1983).

In lowland paddy pastures in Thailand, *P. atratum*’s yield was reportedly higher than that of *B. mutica* at 18 to 25 Mg dry matter ha$^{-1}$ y$^{-1}$, although its CP (6 to 7%) was lower (Phoatong and Phaikaew, 2001). Another study in northeast Thailand showed that *B. mutica* and *P. atratum* produced an average of 20 Mg ha$^{-1}$ dry matter in the first year (Hare et al., 1999a). In the first wet season, no significant differences in production were found between the two species and no production differences between 45-d and 60-d cutting intervals. In the second wet season following establishment, *P. atratum* produced close to 30 Mg dry matter $ha^{-1}$, which was approximately 10 Mg $ha^{-1}$ more than *B. mutica.*

A comparative study in Florida of dwarf elephant grass *(Pennisetum purpureum cv.* 'Mott'*)* and *P. atratum* where both grasses received 0, 100, 200, or 300 kg $ha^{-1}$ $y^{-1}$ of N and cut (15 cm stubble) every 21, 42, or 63 days, *P. purpureum* yielded more forage (10.7 Mg $ha^{-1}$) than *P. atratum* (9.4 Mg $ha^{-1}$) during the 261-d growing season (Kalmbacher et al., 1997). But these grasses responded differently in terms of the time the forage was produced. *Pennisetum purpureum* has very good drought tolerance and produced much of its forage during the period when it was dry and relatively cool (March-June). On the other hand, *P. atratum* produced most of its forage during the time of the year when it was wet and warm (July and September). In terms of the nutritive value, *P. purpureum* appeared to be superior to *P. atratum* at all harvest frequencies and N fertilizer rates. The average CP in *P. atratum* was less than 6.0% when harvested at a more mature stage (>42-d interval). There was very little improvement in CP at higher rates of N fertilization.

**Response to drought/waterlogging**

Tropical pasture grasses vary in their tolerance to drought and waterlogging. For example, *P. purpureum* is a drought-resistant grass species (Duke, 1978; Holm et al., 1977) and the FAO's Grassland Index (no date) indicated that it does not tolerate flooding. For example, 'Capricorn' cultivar is relatively tolerant of drought and will tolerate dry spells, responding quickly when adequate moisture is provided again (Register of Australian Herbage Plant Cultivars, 1972). However, this cultivar also makes little growth during dry periods. However, Hernandez-Doumas (2000) reported that it is a water-loving grass species that can be grown on bunds, swamp areas, riverbanks, and upland areas.

Similar to *P. purpureum*, *B. mutica* is semi-aquatic and can persist under prolonged flooding or waterlogging, or in standing and running water (Cook et al., 2005). Halim and Hamid (1989) demonstrated that para grass was the species most adapted to flooded conditions, whereas guinea and signal grasses were the least tolerant. Although para grass flourishes in wet conditions, it can also withstand dry weather and drought (Holm et al., 1977; Duke, 1983) although growth may be significantly reduced (Nitis, 1999). In an experiment in Thailand, *B. mutica* and *B. ruzisiensis* grasses showed poor persistence under extended dry conditions, while *P. maximum* and *P. purpureum* exhibited greater tolerance to drought (Phaikaew et al., 2001). An earlier study showed that *P. atratum* had better performance under both wet and dry conditions than *B. mutica* or *B. ruzisiensis* (Hare et al., 1999).

*Brachiaria decumbens* is another drought-tolerant grass, which cannot withstand waterlogging for more than a three days (Partridge, 2003). It grows well on quick-drying, shallow, hillside soils (Cook et al., 1005). Its deep rooting system (>1.5m) prevents the crop from suffering drought stress during dry spells and acts as a safety net against nutrient leaching (Hernandez-Daumas, 2000).

Several studies have shown that *P. atratum* is well suited to waterlogged acid soils, such as those in northeast Thailand, which also become seasonally dry (Hare et al., 1999a; 1999b; 2001a) and in similar soils in Florida (Kalmbacher et al., 1997a,b).

**Response to irrigation and fertilization**

In Western Australia, Robert and Carbon (1969) showed that irrigated *P. purpureum* yielded about 42 Mg dry matter ha$^{-1}$; while irrigated *B. mutica* yielded over 20 Mg dry matter ha$^{-1}$. In the Burdekin area of north Queensland, Allen and Cowdry (1961) showed that the highest yields of irrigated grass-legume mixtures of *B. mutica-Centrosema, P. maximum-Stylosanthes* and *P. maximum-Centrosema* were 2.5, 1.9, and 1.4 Mg dry matter ha$^{-1}$, respectively.

Some improved grasses such as *B. decumbens* and *P. purpureum* respond well to N fertilizer application when grown in pure stands (Skerman and Riveros, 1990). In Puerto Rico, *P. purpureum* fertilized with 897 kg N ha$^{-1}$ produced dry matter of about 85 Mg dry matter ha$^{-1}$ y$^{-1}$ (Vicente-Chandler et al. 1959). Cameron et al. (1996) reported a 40% increase in plant production in pasture where dairy pond sludge was applied. For example, dry matter yield increased from about 6 Mg ha$^{-1}$ y$^{-1}$ to nearly 10 Mg ha$^{-1}$ y$^{-1}$ in both surface and injected applications. When applied at the right concentration, organic fertilizers applied at a rate of 15 Mg ha$^{-1}$ can increase dry matter yield of *P. purpureum* 121% more than the N-P-K fertilizer applied at 300 kg N, 80 kg P and 50 kg K ha$^{-1}$ (Halim, 1993).

The P requirement of most tropical grasses such as star grass (*Cynodon nlemfuensis* Vanderyst), guineagrass (*Panicum maximum* Jacq.), and bana grass (*Pennisetum purpureum*) was at least 73 kg P ha$^{-1}$ y$^{-1}$ when grown in some Puerto Rican soils and harvested at 40- to 60-d intervals (Vicente-Chandler et al., 1974). Their study also suggested that these grasses did not consume a lot of P.

Star grass is responsive to N fertilizer, with yields increasing from 12.2 to 21.7 Mg $ha^{-1}$ $yr^{-1}$ (30-d cutting interval) and from 20.2 to 30.0 Mg $ha^{-1}$ $yr^{-1}$ (45-d cutting interval) in Puerto Rico when N application was increased from 0 to 900 kg $ha^{-1}$ $yr^{-1}$ (Caro-Costas et al., 1972). Star grass herbage accumulation in Puerto Rico was 25.1, 31.3, and 40.6 Mg $ha^{-1}$ $yr^{-1}$ for N rates of 225, 450, and 900 kg $ha^{-1}$ $yr^{-1}$, respectively (Velez-Santiago and Arroyo-Aguilu, 1983). In Tabasco, Mexico, star grass yields increased from 17.2 to 24.8 Mg $ha^{-1}$ as N rate increased from 0 to 400 kg $ha^{-1}$ $yr^{-1}$ (Meléndez et al., 1980). No further increments in herbage production were observed with higher N rates. In another study, crude protein concentrations of *C. nlemfuensis* range from 10% to 20% when fertilized with 80 kg N $ha^{-1}$ every 5 weeks (Andrews, 1976).

Recently, Pant et al. (2004) conducted a study on P and K fertilization of star grass. Results showed that star grass receiving 10 and 93 kg $ha^{-1}$ $yr^{-1}$ of P and K, respectively, removed the maximum P (161% of the applied P) from the soil and maintained forage nutritive value and quantity. They suggested that star grass can utilize P from subsurface soils if there is an adequate supply of K. The authors also indicated that star grass may be useful for phytoremediation on sites where P build-up is a concern.

Nitrogen fertilizer requirements of *B. decumbens* may be low (Hernandez-Daumas, 2000), although it was reported to respond dramatically to N amendments (Chin, 1994a). This is because of its capacity to obtain significant proportions of plant N from associative $N_2$ fixation under natural conditions, (estimated as up to 40 kg N $ha^{-1}$ $y^{-1}$, Boddey and Dobereiner, 1988), which suggests that production is only partially contingent upon N amendments. Such estimates coincide with field observations that pastures of *B. decumbens* can remain productive for many years in the absence of N fixing legumes or N fertilizer (Rao et al., 1996). In fact, *B. decumbens* performed better than other *Brachiaria* species in unfertilized experimental conditions (Alvim et al., 1990). With applications of 75 kg N, *B. decumbens* produced 10.3 Mg dry matter $ha^{-1}$ and the N use efficiency was as high as 195 g biomass (shoots and roots) per gram of N taken up. However, no increment in biomass production was obtained when the N application rate was doubled. The CP content, however, increased linearly with incremental additions of N, from 7.2 to 10.6 to 13.4% for 0, 75 and 150 kg N $ha^{-1}$, respectively. A study conducted in Venezuela (Alvarado et al., 1990) concluded that N application of 100 to 150 kg $ha^{-1}$ $y^{-1}$ and harvesting of pastures at 42 to 56 d of re-growth allowed getting pastures of high nutritive value and adequate levels of dry matter production. *Brachiaria* species also have much lower requirements of other nutrients, especially of P and Ca, than other grasses such as *P. maximum*, although differences within species may exist (Rao et al., 1996).

The experience in Hawai'i also shows the low nutrient requirement of *B. decumbens*. Fukumoto and Lee (2002) established field plots (10 x 100 ft) of *B. decumbens* at Mealani, Waiakea, and Waimanalo and the grasses were harvested at regrowth intervals of 4, 8, and 12 weeks. The plots received a minimum level of fertilization, consisting of 440 kg of urea and 410 kg of muriate of potash $ha^{-1}$ $y^{-1}$. Yields obtained were 790, 2400 and 3600 kg ha (Mealani); 680, 4600, 5700 kg $ha^{-1}$ (Waiakea) and 1600, 3500 and 4900 kg ha (Waimanalo) for the harvest intervals of 4, 8 and 12 weeks, respectively. Yield differences among locations may be related to the differences in temperatures and rainfall at each location. Waiakea receives the highest rainfall while Waimanalo the least among the three (Fukumoto and Lee, 2002). Minimum/maximum temperatures were lower in Mealani (10-21°C) and higher in Waiakea and Waimanalo (16-27°C).

*Brachiaria decumbens*, however, is sensitive to flooding. Dias-Filho and Carvalho (2000) compared three species of *Brachiaria*, which included *B. decumbens*. The plants were grown in pots under flooded and well-drained conditions for 14 d. They found that flooding reduced the relative growth rate as well as the specific leaf area and biomass allocation to roots. Also, a high degree of premature leaf senescence was observed in flooded *B. decumbens* plants.

*Brachiaria mutica* responds readily to N (Roberts, 1970) and irrigation, being a water-loving grass (Duke, 1983). Nitrogen applied near the end of summer was reported to result in better winter growth (Currie, 1975). In 4-year irrigated swards in Australia, it gave yields of 6 to 12 Mg dry matter $ha^{-1}$ $y^{-1}$ (Miller, 1976). In India, 100 kg N $ha^{-1}$ increased yield by 49.2%, total digestible N (TDN) by 79.2% and CP by 23.3%. Planted in an area in Vietnam with an average rainfall of 1680 mm $y^{-1}$ and temperature of 23.4°C, and fertilized with 160N:80P:800K kg $ha^{-1}$ and manure of 10 Mg $ha^{-1}$, *B. mutica* had dry matter yield of about 14 Mg $ha^{-1}$ with CP of 1.17% (Binh, 1998). This species had the highest yield compared with other four other *Brachiaria* species tested (*ruziziensis, brizantha, humidicola, decumbens*). In terms of P fertilization, particularly on P-deficient soils, a dressing of 500 kg P $ha^{-1}$ was recommended prior to planting with subsequent top-dressings of 120 to 250 kg P $ha^{-1}$ $y^{-1}$ for a few years (Duke, 1983).

For bana grass, N application of 100 to 150 kg $ha^{-1}$ after each harvest gave the best uniform production, with a total of 900 kg ha of N for six harvests (Duke, 1983). Also, the potential carrying capacity of this grass is very high with application of 50 kg ha$^{-1}$ of N after each harvest, thus maintaining about 27 heads ha$^{-1}$. A study in Pretoria, South Africa showed that irrigated bana grass produced 24 to 26 Mg ha$^{-1}$ y$^{-1}$ of dry matter (Koster et al., 1992).

In a six-month wet season experiment in Thailand, *P. atratum* produced over 30 Mg dry matter ha$^{-1}$ y$^{-1}$ when fertilized every 30 to 40 d (Hare et al., 1999a) and when 20 kg ha N were applied dry matter yields was more than doubled (Hare et al., 1999d). In comparison to other improved tropical grasses, *P. atratum* has relatively low CP content but frequent cutting (Hare et al., 2001a; Kalmbacher et al., 1997a) and frequent N applications (Hare et al., 1999b; Kalmbacher and Martin, 1999) can maximize forage quality and palatability. In a separate study, N significantly increased dry matter yields of two cultivars of *P. atratum* (cv. Hi-Gane and cv. Ubon) on infertile, low lying, seasonally wet soils in northeast Thailand (Hare et al., 1999b). Nitrogen at 20 kg ha applied six times every 30 d in the wet season increased dry matter yields by almost 90% in one trial and over 250% in a second trial. Applying higher rates of 40 and 80 kg ha N every 30 days further increased dry matter yields, although the increase in dry matter per unit of N was less. The yield response (kg dry matter per kg N) from applying N as urea in the wet season ranged from 18 (480 kg N ha$^{-1}$) up to nearly 70 (120 kg N ha$^{-1}$). Nitrogen rate of 80 kg N ha$^{-1}$ every 30 days was required to consistently increase CP levels above 7%, provided the fields were not waterlogged. Cow manure alone did not increase dry matter yields of *P. atratum*. When cow manure (3 to 6 Mg ha$^{-1}$) and compound fertilizer (15% N: 15% P: 15% K) at 312 kg ha$^{-1}$ were applied together, dry matter yields increased by 35 to 40% above the yields produced by similar rates of compound fertilizer without cow manure.

**Response to alkalinity and salinity/sodicity**

Forage grasses also differ in their tolerance to alkalinity and salinity. *Brachiaria mutica* was reported to tolerate pH of 4.3 to 7.7 (Duke, 1983) and is moderately salt tolerant (Skerman and Riveros, 1990), within the 3.0 to 6.0 dS $m^{-1}$ range (Ayers and Westcot, 1989). It is also tolerant of brackish water (Holm *et al,*. 1977; Sainty and Jacobs, 1981). In southeast Queensland, *B. mutica* grows on deep loamy soils overlying saline clays and merges with saline grasses on marine flood plains (FAO, no date). The high salt (Na) tolerance of this grass species (with 48% yield at 58% ESP and survived even at 78% ESP) was attributed to its low root cation exchange capacity (Bajwa and Bhumbla, 1971). *Brachiaria decumbens* was found to have low salinity tolerance (Deifel et al., 2006).

*Pennisetum purpureum* tolerates high salinity (FAO, no date), thrives in coastal areas (Reynolds, 1995) and adapts well to both high and low pH (4.5 to 8.2) (Cook et al., 2005). It was also found to respond well to sewage sludge application (Duke, 1983). Wang et al. (2002) found 50% reduction in yield of *P. purpureum* when the EC of the irrigation water was increased from 5 dS $m^{-1}$ to 25 dS $m^{-1}$.

As mentioned earlier, *P. atratum* was found to be well suited in waterlogged acid, humic gley podzolic soils of northeast Thailand (Hare et al., 1999a). *Paspalum* species were reported to tolerate salinity levels of 2.6 dS $m^{-1}$ without reduction in yield and 3.3 dS $m^{-1}$ with 10% yield reduction (Evans, 2006). Many species of *Paspalum* and *Brachiaria* tolerate acid soils (Bogdan, 1977).

*Cynodon nlemfuensis* prefers soils with pH between 5.5 and 7.0 and is not as salt-tolerant as *C. dactylon* (Cook et al., 2005). In one study, *C. nlemfuensis* was initially grouped under salinity category of 2 dS $m^{-1}$ although the study found that it could tolerate salinity up to 4 dS $m^{-1}$ (Toth et al., 1997).

**Response to harvesting frequency**

The decision of a livestock producer regarding the frequency at which the forage grasses is harvested usually depends on targeted nutritive value. Frequent harvest intervals produce young, leafy forage of high nutritive value but lower yield, whereas infrequent cutting results in more stemmy forage with inferior quality leaf material but higher yield (Humphrey, 1999). For example, under ordinary conditions, *B. mutica* may be cut at six to eight week intervals but CP was highest (14.19%) when about 30 days old (Duke, 1983). For highest nutritive values, the recommendation for cutting this grass every 30 days, although this may vary with soil and other environmental conditions. In one trial in a low fertility soil in Thailand, *P. atratum* cv. Ubon produced dry matter yields that are generally significantly different only between 20- and 60-d cutting intervals (Hare et al., 2001b). Cutting every 20 d over a 240-d period produced less (21.6 Mg $ha^{-1}$) dry matter yield compared with cutting every 60 days (28.9 Mg $ha^{-1}$) but CP concentration was nearly twice as high (10.0% vs. 5.3%, respectively).

In an experiment carried out on savanna Tropudalf soil in Venezuela, dry matter production of *B. decumbens* reached only 2.2 to 4.8 Mg $ha^{-1}$ $y^{-1}$ with N application of 100-150 kg $ha^{-1}$ $y^{-1}$ and 42 to 56 d of re-growth of *B. decumbens* (Alvarado et al., 1990). Nonetheless, they observed that yields increased as the grass matured, with the accumulation of dry matter reaching its maximum near the 70 d of re-growth, when a balance between the production of leaves, stems and dead materials is usually attained. Beyond this point, the rate of production of dry matter was reduced. The dry matter yield of *B. decumbens* harvested at 6 weekly intervals in various locations in Malaysia ranged from 16.5 to 26.3 Mg $ha^{-1}$ $y^{-1}$ (Chee, 1986). In other locations, the yield of this grass species is very low, possibly due to less favorable environmental conditions under which it is grown.

Well-fertilized cultivars of *P. purpureum* and *Panicum maximum* harvested at an interval of 28 to 35 and 21 to 28 days, respectively, could achieve CP levels over 10% (Chin, 1995). *Panicum maximum* varieties harvested at 21 to 24 days interval could attain CP levels above 13%. *B. decumbens*, when adequately managed, produces forage of over 14% CP (Vallejos, 1988).

Vicente-Chandler et al. (1953) suggested that the highest yields of *P. purpureum* could be expected from cutting at 12-week intervals and applying N after every cut. The average yields reported for this grass were 4.85 and 7.27 Mg $ha^{-1}$ at 45- and 60-d re-growth intervals, respectively (Duke, 1983). Yields reached Mg dry matter and 3.4 Mg CP $ha^{-1}$ $y^{-1}$ when cut every 56 d at CIAT, Colombia, and 40 to 50 Mg green matter $ha^{-1}$ when cut each 35- to 40-d at the Tulio Ospina Station, Colombia (FAO, no date). At Kitale, Kenya, Odhiambo (1974) showed no drop in nutritive value in bana grass tissue analyses taken at seven to 12 weeks.

Chaparro and Sollenberger (1997) studied the response of *P. purpureum* cv. Mott to four cutting intervals (3, 6, 9, and 12 wk) and four cutting heights (10-, 22-, 34-, and 46-cm stubble). Harvested dry matter ranged from 8.3 to 16.4 Mg $ha^{-1}$ in 1989 and from 4.7 to 11.9 Mg $ha^{-1}$ in 1990. Lower maximum values in 1990 were attributed to drought conditions. Greatest values for herbage dry matter, CP, and digestible organic matter harvested occurred at the longest cutting interval and shortest cutting height. This grass persisted well and was very productive when harvested every 9 wk or longer at a height of 10 cm. Using this management scheme, CP concentration was 85 g $kg^{-1}$ dry matter or greater.

In a two-year study conducted on a dry, infertile site and under the subtropical conditions near Gainesville, Florida, the response of *P. purpureum* to three harvest frequency regimes was assessed (Woodard, 1990). Mean dry biomass yields for the four tall elephant grasses during 1986-87 were 27, 24, and 18 Mg $ha^{-1}$ $y^{-1}$ for one, two, and three harvests per year, respectively. Crude protein contents were 4.0, 5.8, and 7.9% (dry basis), while the ash-free neutral detergent fiber (NDF) contents were 81, 76, and 74% (dry basis), respectively. For dwarf elephant grass, two-year mean dry biomass yields were 13, 12, and 11 Mg $ha^{-1}$ $y^{-1}$ for one, two, and three harvests per year, respectively. Mean CP contents were 5.3, 7.1, and 9.6%, respectively, while NDF levels were 77, 73, and 70%, respectively.

Sotomayor-Rios et al. (1976) tested nine *Brachiarias*, nine *Digitarias* and 'Florico' star grass for yield, CP and percentage dry matter under three harvest intervals in a study involving high fertilization of the grasses (359-124-242 kg $ha^{-1}$ $y^{-1}$ N-$P_2O_5$-$K_2O$). Results indicated that 'Florico' star grass was among the highest dry matter and CP producers at 7.3 Mg $ha^{-1}$ $y^{-1}$ and 14.0%, respectively, when harvested at the 30-d interval. Its percentage dry matter (29.5%) was also the highest. Delaying harvest frequency to a 45-day interval resulted in highest dry matter yield (10.9 Mg $ha^{-1}$ $y^{-1}$) for 'Florico' compared with all 19 grasses, with a CP and percentage dry matter of 10.3 and 32.5, respectively. A further delay in harvest frequency to 60 days resulted in an additional dry matter yield increase for 'Florico' of 13% above the 45-d interval, with a CP content of

9.2%. A 1978 trial in Ona, Florida showed an increase in dry matter yield of 'Florico' stargrass (6-15 Mg $ha^{-1}$) when grazing frequency was reduced from 2- to 7-week intervals. However, when allowed to grow for seven or more weeks before harvest or grazing, stargrass produces low quality, mature forage, with low palatability (Mislevy et al., 1989).

Mislevy and Brown (1991) recommended that the stubble height of most star grasses should be maintained at 6 to 10 in (15 to 25 cm) for best performance, but plants should be harvested when the height above the stubble is 6 to18 in (15 to 46 cm). Stubble height is important on root development of star grass and studies have shown that root dry matter yield was reduced by 97% compared with the unharvested control when plants were repeatedly harvested back to a 2 in (5 cm) stubble after attaining 6 in (15 cm) of top growth above the stubble (Mislevy and Brown, 1991). Forage dry matter yield in Florida soils harvested at a 5-wk interval averaged about 11 Mg $ha^{-1}$ $y^{-1}$ (Mislevy et al., 1989).

Hare et al. (1999a) compared the performance of *B. mutica* and *P. atratum* as affected by cutting intervals in seasonally wet soils in Thailand. Under both 30-d and 60-d cutting intervals in the second wet season, *P. atratum* produced significantly more dry matter than *B. mutica*. Cutting *P. atratum* every 30 d produced higher quality grass compared to grass cut every 60 d and did not significantly reduce dry matter. Meanwhile, *B. mutica* produced nearly 40% less dry matter when cut at a 30-d interval compared to yields from a 60-d interval. Throughout the study, however, *B. mutica* had 2-3 % higher CP than *P. atratum*.

Another study conducted by Hare et al. (2001b) determined the effect of varying cutting height and interval on growth and forage quality of *P. atratum* cv. Ubon pastures grown on low fertility soils in Thailand. In one trial, they observed that increasing the cutting height from 0 cm to 20 cm above ground level resulted in higher total dry matter yield at 20-d, no effect at 30-d and lower yields at 60-d cutting intervals. Also, increasing the interval between harvests reduced concentrations of CP, K and P, but increased the concentrations of NDF and Acid Detergent Fiber (ADF). They observed also that longer cutting intervals and heights increased the stubble and root dry matter per plant.

### Basis for Using Plants to Remove Nutrients from Effluent

One of the unique uses of plants that are gaining increased research focus from the scientific community and widespread public acceptance is phytoremediation. This technology uses plants to remove, contain or render harmless the pollutants from the soil or water (Salt et al., 1998). Although the basic idea for this technology is very old, it has developed into a very effective, less expensive and environment-friendly approach to cleaning polluted sites.

At present, there are many plant species that have been identified to accumulate ions in their tissues, especially the leaves. These plants are called "hyperaccumulators" or sometimes "bioaccumulators". Examples are orchardgrass and tall fescue, which can accumulate significant amounts of P (Sleugh et al., 2002).

The major basis for using plants for clean-up is that they need to acquire macronutrients (N, P, K, S, Ca, and Mg), and essential micronutrients such as Fe, Zn, Mn, Ni, Cu, and Mo in order to grow and complete their life cycle. Plants can thus clean up pollutants as deep as their roots can grow and store these in their leaves, stems and roots. This prevents the pollutants in one site from moving or being transferred to other areas. Plants have evolved highly specific mechanisms to take up, translocate, and store these nutrients. For example, proteins with transport functions (also called transporters) mediate the movement of metal across biological membranes (Taiz and Zieger, 2002). Also, sensitive mechanisms maintain intracellular concentrations of metal ions within the physiological requirements of the plant. The uptake mechanism is generally selective, and there seems to be a linkage or synchronization between the plant's selectivity and the transporters. Some plants preferentially acquire some ions over others, depending upon the structure and properties of the transporters, allowing the latter to recognize, bind, and mediate the transmembrane transport of specific ions (Lasat, 2000). Thus, some transporters mediate the transport of divalent cations but do not recognize mono- or trivalent ions.

Effluent clean-up using chemicals or some other remediation technologies may not be feasible or affordable to the livestock producers. Because of the high cost, there is a need for less expensive clean-up technologies. A good alternative is to take advantage of the natural plant processes to do the work. Utilizing forage or pasture grasses for effluent clean-up requires the lowest capital cost and management input. In addition, because this strategy remediates both the effluent and soil *in situ*, the technique of utilizing the grasses for clean-up avoids dramatic landscape disruption. One drawback is the treatment time—several years may be needed in some cases for soil clean-up. Plant uptake of ions also depends on the season, usually greater nutrient removals during the wet season when sufficient water is available. Many plant species can also have a very low growth rate, making it necessary to select new varieties capable of accumulating nutrients with high biomass production.

## Knowledge Gaps

While much research has been conducted regarding the effects of animal waste application to the soil, most of these studies focused on identifying the forms of P in the soil and assessing the risk of P being carried in runoff. The few studies that dealt on the response of grasses to animal manure or effluent application have been conducted mostly on the mainland U.S. and involve temperate grass species. Studies on the response of tropical grasses to inorganic fertilization and cutting intervals were mostly conducted elsewhere (Florida, Malaysia, Puerto Rico, Australia, Venezuela, Thailand). The results generated by these studies vary greatly depending on location/growing conditions, cutting height, fertilization rates, varieties used, among others. Given the diversity in the results of the studies on grasses, it is difficult to use this information in developing practical recommendations to dairy operators in Hawai‘i. There is, therefore, a need to conduct a separate study that can generate information on the adaptability of these grasses specifically their performance in soils receiving effluent irrigation. This information could then be used to develop decision aids or prediction tools that cater to Hawai‘i dairy producers.

Very few studies on tropical grasses have been done in Hawai‘i (e.g., Whitney and Green, 1969) and none of these studies looked into the effects of animal waste application on the growth, productivity and forage quality of tropical grasses. Ways by which effluents can be re-used for enhanced pasture productivity have also not yet been fully explored. There is still a dearth of information on the response of the various tropical grasses to animal effluent application, especially at varying rates of application.

Also, sampling of both the plants and the soil at the surface and root zones are necessary to properly evaluate the effects of sustained manure application on soil properties. Seasonal differences in the response of the tropical grasses have also not yet been studied.

Many management questions need to be considered when applying effluent for irrigation such as what criteria should be used to determine the application rate (crop's nutrient requirement or crop water requirement). Also important are the effects on soil properties (salinity and sodicity, macro- and micronutrient balance, changes in soil pH, etc.). One of the issues is how to predict when there is a problem with effluent application and how to provide alternative solutions to such problems. For example, there could be a problem with unbalanced and/or excess N and P applied with effluent irrigation. There are also limited opportunities to irrigate with wastewater during the rainy season, and irrigating during the winter could result in wastewater and its contaminants being carried by runoff to water bodies. Often, rates of application of organic materials or animal wastes are based on crop's N needs. As reviewed earlier, this approach results in overapplication of P, especially if the soil sorbs a lot of this nutrient, which leads to P accumulation. Meanwhile, if application will be based on P, the crop may not receive as much N as it requires. There is, therefore, a need to create a balance between N and P application rates. Effluent application can also easily result in excess N and P in the soil, if the rate of application will be based not on crop nutrient needs but on evapotranspiration rates or crop water requirements. It is, thus, important to determine which factor should be used as the basis for effluent irrigation, with due consideration of the soil properties and other environmental factors.

Another important problem that is often overlooked is the micronutrient deficiencies induced by the increase in pH and excessive P resulting from the application of animal wastes and effluents. Forage plants need micronutrients to ensure optimal forage production and nutrient utilization. None of the studies, to our knowledge, has considered the possible micronutrient imbalances resulting from effluent application.

Many studies on the impacts of animal waste applications had already been conducted, but few considered at what point the downward movement of P begins. In other words, there is a need to better understand at what point or interval animal waste-derived P begins to move in the soil profile. Since the added animal effluent has relatively low P content, it is assumed that it may take a lot of effluent application before P sorption sites of the topsoil layer becomes fully occupied and P begins to move downward. However, in soils with very low P sorption capacity, P may potentially move down despite low application rates, especially if the application is continuous and long-term.

Despite the abundance of literature on soil P sorption, there is still a dearth of information on the effect of animal effluent-derived P on the dynamics of P sorption of soils. Many of the studies conducted on the characterization of P forms in soils receiving animal wastes did not indicate the P sorption capacity of the soil they used, thus, it is difficult to evaluate whether the forms of P in the leachate are related to the P sorption capacity of the soil. A need exists to study the important factors that affect P sorption and are relevant to effluent management. Elucidating these factors could help improve our knowledge of managing animal effluents and allow for identification of management strategies for land application and crop utilization of this resource. We can capitalize on these factors to find ways to increase the nutrient holding capacity of the soil and therefore, maximize effluent application per unit area.

Decision aid tools such as an effluent calculator has not yet been developed for Hawai‘i dairy operators use, especially one that takes into account the wide diversity in soil and environmental conditions in Hawai‘i. For example, one of the dairies on the Big Island is operating on a site receiving high rainfall all year round, with soil that is of volcanic ash origin (Andisol) and has very high P sorption capacity.  On the other hand, some dairies in Oahu are located on the dry side of the island, with soil that has high clay content but very low P sorption capacity.

# CHAPTER 3. RETENTION, ACCUMULATION AND LEACHING OF PHOSPHORUS IN DAIRY EFFLUENT-IRRIGATED SOIL IN A TROPICAL ENVIRONMENT

## Abstract

Dairy operations generate large quantities of effluent, which are stored in lagoons to allow solids to settle and water to evaporate. Lagoons, however, have finite storage capacity and may overflow causing pollution of land and associated water bodies. Dairy producers, thus, need alternative uses of effluent for an environment-friendly dairy production. This study assessed the effects of effluent irrigation on the retention, accumulation and leaching of nutrients, especially phosphorus (P), on a soil (Cumulic Haplustoll). Five tropical grasses—bana (*Pennisetum purpureum* S.), California (*Brachiaria mutica* S.), signal (*Brachiaria decumbens* S.), star (*Cynodon nlemfuensis* V.), and suerte (*Paspalum atratum* S.)—were planted in an augmented completely randomized design. Two rates of dairy effluent were supplied daily through a subsurface (20 to 25 cm depth) drip irrigation system based on the potential evapotranspiration ($ET_p$)—2.0 $ET_p$ (7 to 44 mm $d^{-1}$) and 0.5 $ET_p$ (2 to 11 mm $d^{-1}$)—at the site (Waianae, Hawai'i). Soil samples collected between Jul 2003 and Aug 2006 were used in the P sorption experiment and selected samples for extraction with the recommended extractant (Olsen) and acidic extractants (Modified Truog and HCl). No excessive increases in Olsen extractable soil P and soil solution total P were observed. Fluctuations in extractable soil P and soil solution P was explained by the variation in the amounts of effluent P applied due to varying $ET_p$ and rainfall. A relatively low flux of P was observed at the 2.0 $ET_p$ irrigation rate (95 to 131 kg $ha^{-1}$ $y^{-1}$). Calcium phosphate may be

forming due to the high pH of the soil (7.4 to 8.9) and effluent (8 to 9) and the large amounts of applied Ca (800 to 2900 kg $ha^{-1}$ $y^{-1}$). Higher acid extractable soil P than Olsen soil P suggested the dissolution of a difficultly soluble Ca-P compound, possibly hydroxyapatite. The extractable soil Ca also declined over time, further supporting the hypothesis of Ca-P formation. These findings suggested that this type of soil can be irrigated with dairy effluent at 2.0 $ET_p$ – a rate that is substantially higher than that designed for most irrigation objectives. Additional monitoring is needed to determine the longer-term impacts of dairy effluent application on plant and soil properties.

## Introduction

Dairy producers in Hawai‘i and other island environments are highly dependent on imported feeds to sustain dairy cattle operation. In addition, they generate large quantities of wastewater, which are commonly stored in open lagoons (Plate 3.1). A result is a tremendous influx and accumulation of nutrients and salts. Apart from nutrients, effluents normally contain considerable amount of organic matter, microbial contaminants and suspended solids, all of which can reduce soil and water quality when carried by runoff and through leaching. Also, effluent lagoons have finite capacity to contain the effluent and may overflow during the rainy season. As such, many dairy producers have to invest on technology or find alternative uses of effluent to comply with environmental regulations .

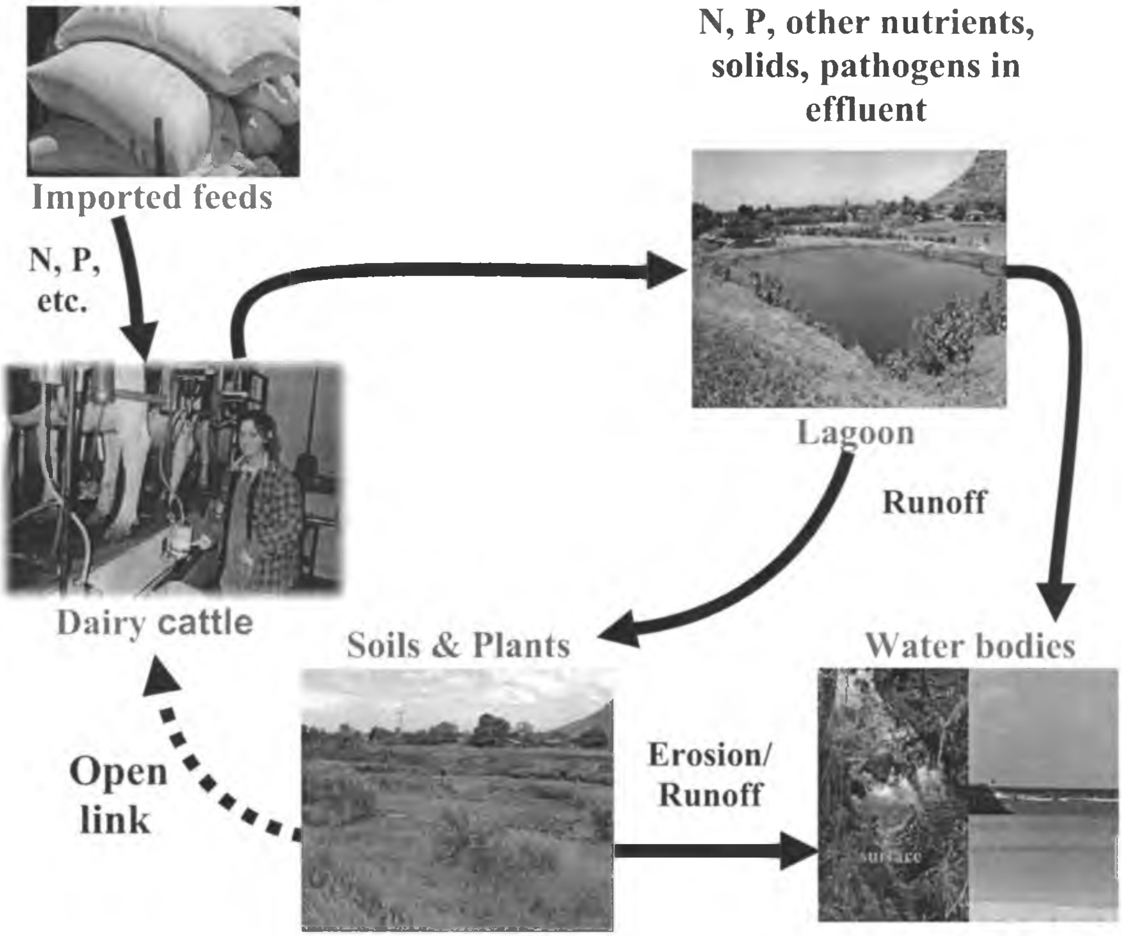



**Plate 3.1. A schematic of the flow of nutrients, salts and water in Hawaii's dairy production system that creates an open nutrient cycle.**

Various crops and tropical grasses have been used for phytoremediation of N and P from wastewater. For example, stargrass (*Cynodon nlemfuensis* V.), bermudagrass (*Cynodon dactylon* L.) and johnsongrass (*Sorghum halepense* L.) have been used for nutrient removal from swine effluent (e.g., Adeli and Varco, 2001; McLaughlin et al., 2004). Switchgrass (*Panicum virgatum* L.) was planted as filter strips to remove nutrients from dairy manure (Sanderson et al., 2001). McLaughlin et al., (2005) irrigated *C. dactylon* with swine effluent to supply sufficient amount of N to the forage. Bahiagrass

(*Paspalum notatum* F.) was successfully grown with domestic wastewater (Adjei and Rechcigl, 2002). Corn (*Zea mays L.*) is an example of a crop that effectively utilized N from anaerobically digested swine lagoon effluent (Evans et al., 1977). Nutrient removal from swine effluent was also tested on some winter forage crops such as *C. dactylon* fall seeded with 'Kenland' red clover (*Trifolium pratense* L.), 'Bigbee' berseem clover (*T. alexandrinum* L.), or 'Marshall' annual ryegrass (*Lolium multiflorum* Lam.) in temperate environments (Rowe and Fairbrother, 2003). Various forage systems had also been irrigated with dairy effluent or slurry (e.g., Woodard et al., 2002; Johnson et al., 1995).

Effluent irrigation to produce forage, which can then be fed to dairy cattle, is an attractive alternative that recycles the nutrients and water from the effluent. This strategy "closes" the nutrient cycle of the milk production system and allows producers to save on feed costs. In addition, this approach minimizes the entry of imported nutrients to Hawai'i while reducing environmental pollution associated with effluent application– thus, creating a win-win option. However, there is a concern about the accumulation and movement of nutrients, especially P, when applying effluent to the soil. Phosphorus accumulation has been a major issue in areas with concentrated and intensive animal feeding operations and where animal wastes are repeatedly applied (Sharpley et al., 1984). For example, repeated manure applications to the same fields in Coastal Plain soils resulted in excessive soil test P concentrations (Sharpley and Withers, 1994; Sharpley et al., 1984). In Hawai'i, P accumulation was reported for some soils currently used for pasture (Yost et al., 1999, unpublished). For example, soil samples at 0 to 20 cm in a dairy pasture where effluent had been applied exhibited elevated levels of P (up to 1055 mg $kg^{-1}$ of Modified Truog P). The soil is an Oxisol with moderate P sorption capacity (Fox and Searle, 1978). A study in Texas involving dairy effluent fertilization to 'Alamo' switchgrass (*Panicum virgatum* L.) planted as filter strip showed that soil P increased with increasing rates of effluent application (Sanderson et al., 2001). Phosphorus accumulated in systems that received both short-term (Johnson et al., 2004) and long-term (Redding, 2001) animal effluent application.

Phosphorus has been generally considered immobile being readily sorbed by the soil, yet, its downward movement had been reported in undisturbed soil core experiments (Phillips, 2002), grasslands utilized for effluent application (Toor et al., 2004; Barton et al., 2005), horticultural crop land fertilized with swine effluent (Redding, 2001), and in fields receiving high levels of animal manure application or extensive animal feeding operations (Hansen et al., 2002; Mozaffari and Sims, 1994). Holford et al. (1997) examined the effects of dairy, swine, or sewage effluent and other materials containing P on the P sorption characteristics of two Australian soil groups. They found that P leaching and downward movement might commence even before all the P sorption sites are occupied.

Whereas studies on effluent application have usually been based on N and P loading or requirement of the plants, this study implemented a weather-based effluent management scheme. In this approach, the rates of effluent applied are adjusted based on the changes in rainfall and $ET_p$ rates. This approach differs, in other words, by targeting the irrigation requirement and then assessing the resulting N and P status in the soil, rather than basing applications on arithmetic estimates of N and P requirements of the crops. This contrast may be important in implementing management strategies that dairy farmers in Hawai'i and elsewhere can consider in order to pursue economic and ecological sustainability. This strategy, however, may either lead to excessive application or insufficient application of N and P. Also, most of the studies on forage utilization for recycling of nutrients from effluent have been conducted on the U.S. mainland and involved temperate species, and these results have limited application to the tropical setting. Other studies used summer forage species but the soils they used were entirely different from those being utilized for animal and dairy production in Hawai‘i. This study was conducted to assess the effects of dairy effluent irrigation on the retention, accumulation and downward movement of nutrients in the soil planted to highly productive tropical forage species. The focus of this paper is on the accumulation and leaching of P in the soil irrigated with dairy effluent.

## Methodology

### Soil and Climate

The experimental site for dairy effluent irrigation was in Waianae (21° 26’ 56” N/158° 10’ 37” W), located on the leeward west coast of the Island of O’ahu, Hawai‘i (Plate 3.2). The area has a mean elevation of 2.4 meters (8 ft) above sea level--low-lying and relatively flat. From Aug 2004 to Aug 2006, the monthly rainfall varied from 0 to 320 mm and the average $ET_p$ ranged from 5 to 21 mm $d^{-1}$ (Fig. 3.1). The rainfall is very low and the $ET_p$ is, thus, high compared with other locations in the island.

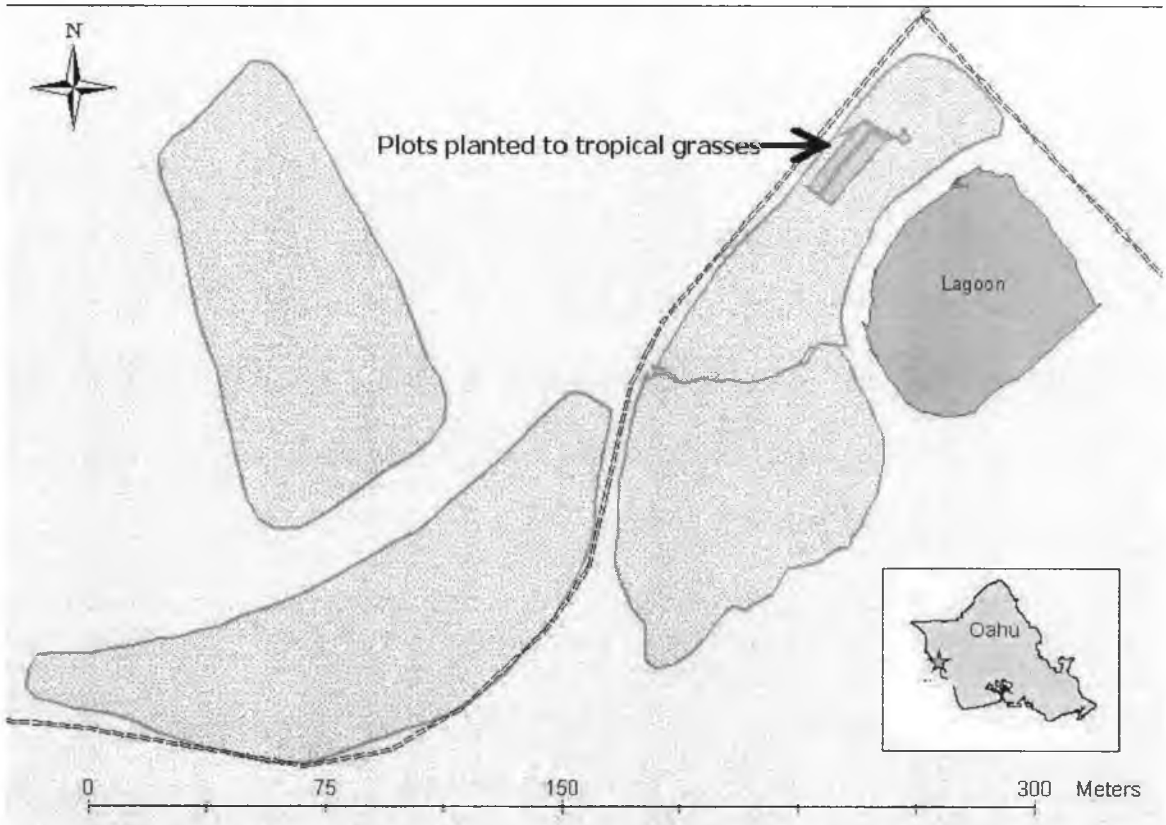



**Plate 3.2. Location of the dairy effluent irrigation experiment, Waianae, O'ahu, Hawai'i.**

The soil is a Mollisol with Vertic properties. It belongs to the Pulehu series, which is a fine-loamy, mixed, semiactive, isohyperthermic Cumulic Haplustoll (Soil Survey Staff, 1972). Particle size analysis showed that the soil has 63% clay, 19% silt and 18% sand. However, there was disturbance of the original soil profile during the installation of the irrigation system, with intensive mixing of soils at the 0 to 35 cm depth and removal of most of the boulders and cobbles. The properties of the soil and dairy effluent prior to effluent irrigation study were presented (Table 3.1).

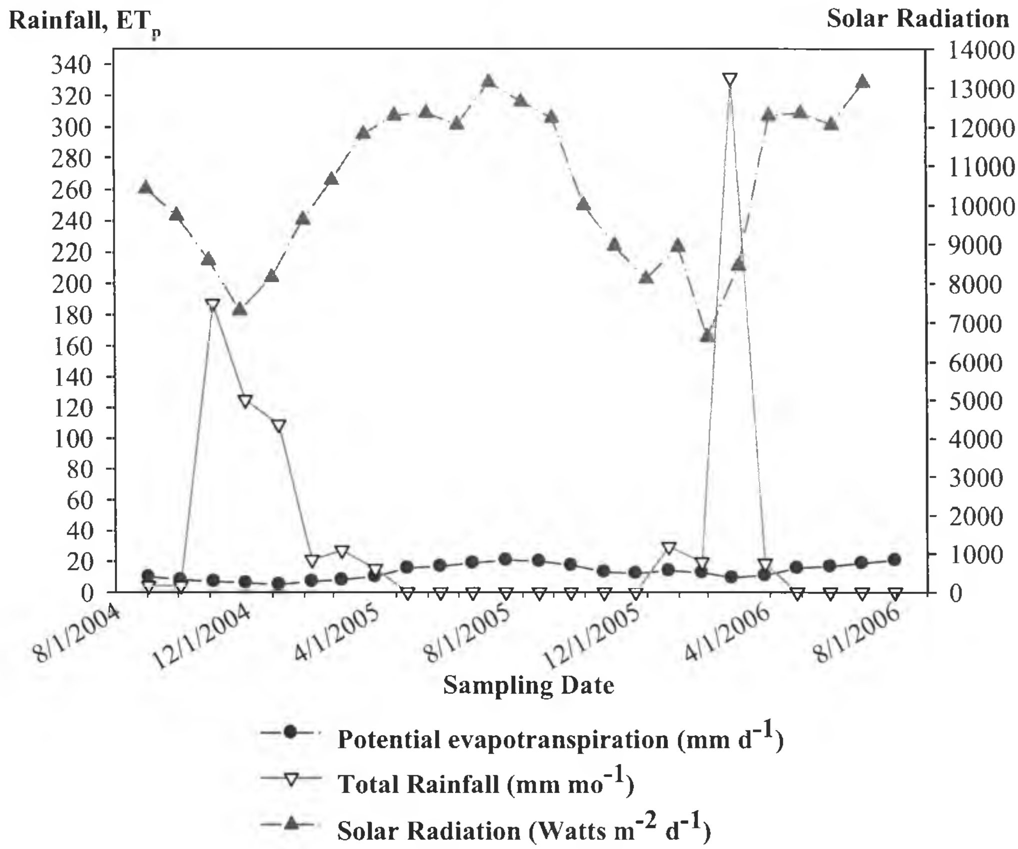



**Figure 3.1. Potential daily evapotranspiration, total monthly rainfall and daily solar radiation in Waianae, O'ahu, Hawai'i.**

**Table 3.1. Soil and effluent properties prior to effluent irrigation study, Waianae, Oahu, Hawaii.**

**Soil properties (Mollisol)**

| Sampling Depth | $pH_{sp}$[‡] | $EC_{1:1}$ | OC | N | P | K | Ca | Mg | Na |
|---|---|---|---|---|---|---|---|---|---|
| | | dS/m | -------%------ | | --------------------mg $kg^{-1}$ | | | | -------------------- |
| 0-15 cm[a] | 7.5 | 7.1 | 0.94 | 0.08 | 125 | 2263 | 6426 | 3440 | 2472 |
| 0-15 cm[b] | 7.5 | 8.3 | 1.25 | 0.10 | 143 | 2204 | 5841 | 3050 | 2521 |
| 15-30 cm | 7.6 | 6.1 | 0.73 | 0.07 | 91 | 2754 | 5702 | 3225 | 2195 |

[a]refers to the average data before dripline irrigation installation (Apr 2003)
[b]refers to average data after dripline irrigation installation (Jul 2003)

**Dairy effluent properties**

| Sampling Date | pH | EC | SAR | N | P | K | Ca | Mg | Na |
|---|---|---|---|---|---|---|---|---|---|
| | | dS/m | % | -------------------------mg $L^{-1}$ | | | | | ------------------------ |
| Dec 2003 | 8.3 | 3.4 | 5 | 165 | 13 | 856 | 70 | 113 | 284 |
| Aug 2004 | 8.2 | 3.2 | 4 | 59 | 15 | 648 | 63 | 111 | 238 |

[‡]Saturated paste extract

**Grass Selection and Management**

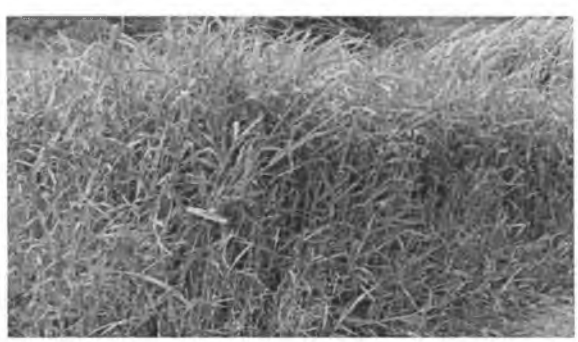
Grass selection for this study was based, in part, on the results of a fertilization study at Mealani Experiment Station demonstration grass plots on the Island of Hawai'i as well as from preliminary selection trials in Waianae, O'ahu. Of the three grass species initially tested in this high salt Waianae soil ($EC_{spe}$ = 11 to 26 dS $m^{-1}$), dwarf napier (P. purpureum S. cv. 'Mott') had the highest nutrient concentration, but it was unable to tolerate the salinity of the soil at the site. Therefore, only those tropical grasses that have demonstrated tolerance to saline soil condition—bana (*P. purpureum* cv. HA 5690), signal (*B. decumbens)*, California (*B. mutica*), star (*C. nlemfuensis)*, and suerte (*P. atratum*)—were included in the effluent irrigation study (Plate 3.3). Grasses were planted in December 2003 and allowed to establish to near 100% groundcover with freshwater irrigation (30 min, twice a day) until Jul 2004.

bana grass (*P. purpureum*)

signal grass (*B. decumbens*)

suerte grass (*P. atratum*)

star grass (*C. nlemfuensis*)



California grass (*B. mutica*)

**Plate 3.3. Five tropical grasses irrigated with dairy effluent, Waianae, O'ahu, Hawai'i.**

## Irrigation System

A subsurface (20 to 25 cm depth) drip effluent irrigation system was established near a dairy effluent lagoon (Fig. 3.2; Plates 3.4 to 3.5). The experimental site was surrounded by a berm that was constructed to avoid entry of the dairy effluent in the event that the lagoon overflows during the rainy season. The use of drip irrigation, wherein water is applied very frequent to keep the soil water potential high and meet the crop's water requirements, has been successfully used to grow plants using high salt waters (Rhoades et al., 1992). From Aug 2004 to Aug 2006, the plots received two rates of irrigation based on the $ET_p$ at the site: 2.0 $ET_p$ (7 to 44 mm $d^{-1}$) and 0.5 $ET_p$ (2 to 11 mm $d^{-1}$). The nutrient application rate varied every three to four months due to variation in the $ET_p$ rate and changes in nutrient concentration in the effluent.

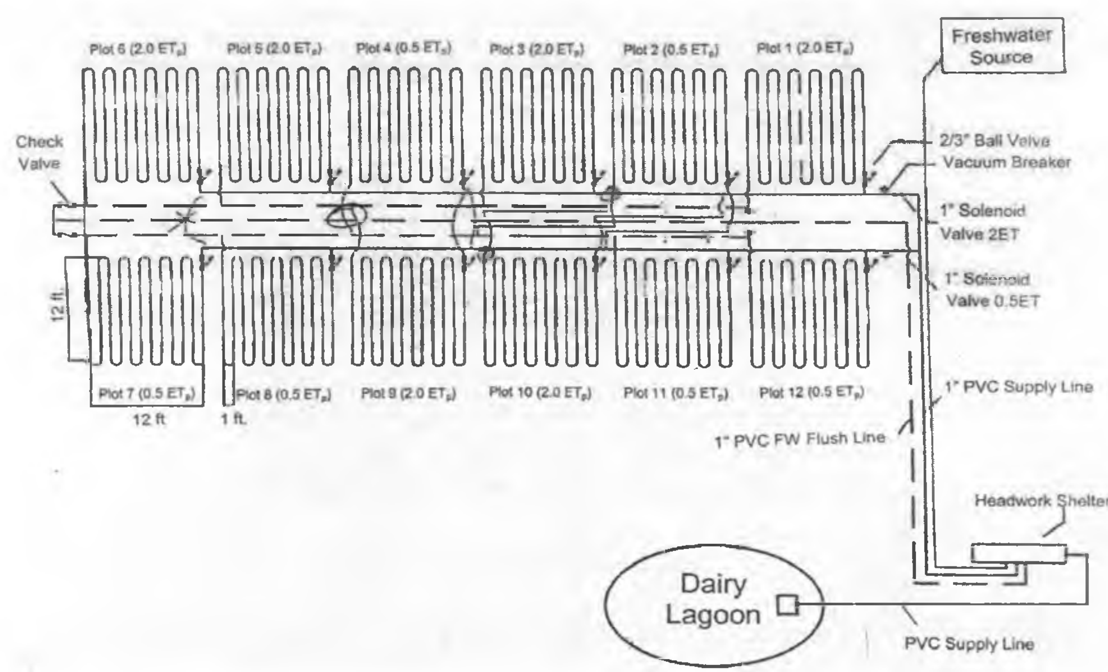



**Figure 3.2. Layout of the drip irrigation system.**

**Plate 3.4. The source of irrigation was a dairy effluent lagoon, O'ahu, Hawai'i.**

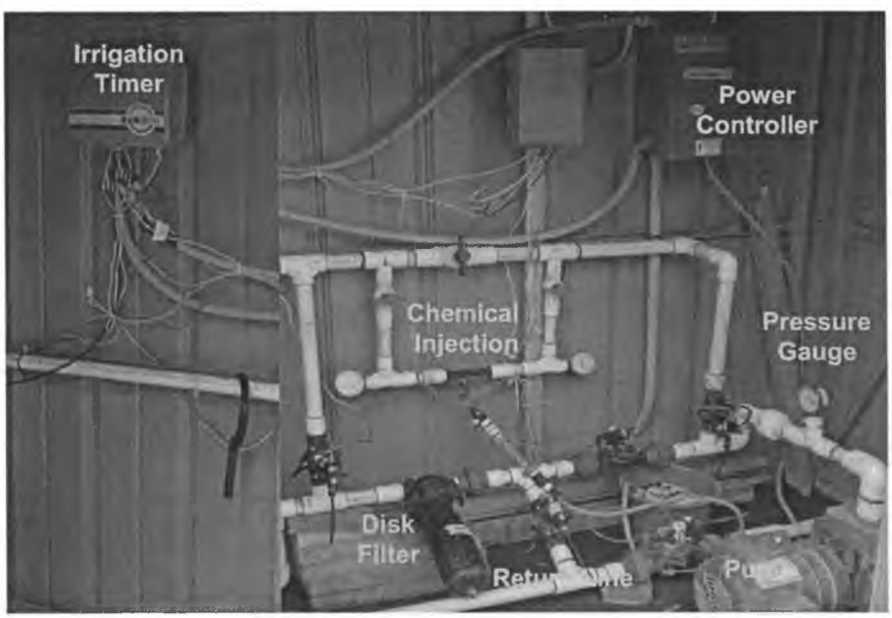



**Plate 3.5. Controls in the drip irrigation system located inside the headwork shelter.**

Adjustments of rates were made depending on the rainfall and calculated $ET_p$ data that were collected using the Hobo[†] weather station installed at the site (Plate 3.6). The $ET_p$ was calculated using the Reference Evapotranspiration Calculation and Software (Ref-ET v. 2.0) (Allen, 2001), which uses the Penman equation (Allen et al., 2001) (Eq. 1).

$$ET_o = \left( \frac{\Delta(R_n - G) + K_{time}\rho_a c_p \frac{(e_s - e_a)}{r_a}}{\Delta + \gamma\left(1 + \frac{r_s}{r_a}\right)} \right) / \lambda \qquad \text{(Eq. 1)}$$

where $ET_o$ is the reference evapotranspiration (mm $d^{-1}$ or mm $h^{-1}$), $R_n$ is the net radiation (MJ $m^{-2}$ $d^{-1}$ or MJ $m^{-2}$ $h^{-1}$), $G$ is the soil heat flux (MJ $m^{-2}$ $d^{-1}$ or MJ $m^{-2}$ $h^{-1}$), $(e_s - e_a)$ represents the vapor pressure deficit of the air (kPa), $e_s$ is the saturation vapor pressure of the air (kPa), $e_a$ is the actual vapor pressure of the air (kPa), $\rho_a$ is the mean air density at constant pressure *(kg* $m^{-3}$*)*, $r_a$ is the mean air density at constant pressure, $c_p$ is the specific heat of the air (MJ $kg^{-1}$ $°C^{-1}$), $\Delta$ represents the slope of the saturation vapor pressure temperature relationship (kPa $°C^{-1}$), $\gamma$ is the psychrometric constant (kPa $°C^{-1}$), $r_s$ and $r_a$ are the (bulk) surface and aerodynamic resistances (s $m^{-1}$) and $\lambda$ is the latent heat of vaporization (MJ $kg^{-1}$). $K_{time}$ is a unit conversion equal to 86400 s $d^{-1}$ for ET in mm $d^{-1}$ and equal to 3600 s $h^{-1}$ for ET in mm $h^{-1}$. The monthly $ET_p$ is calculated as the average of the daily $ET_p$ for a given month, while the rainfall was the total of the daily rainfall for a given month.

---

[†]Reference herein to any specific product, by trade name, trademark, manufacturer, or distributor does not necessarily constitute or imply its endorsement or recommendation by the authors and publishers.

**Plate 3.6. A portable weather station was installed at the site to monitor various climatic data.**

At the 2.0 $ET_p$ rate, cumulative nutrient loads were estimated at 1200 kg N $ha^{-1}$ $y^{-1}$, 620 kg P $ha^{-1}$ $y^{-1}$, 42000 kg K $ha^{-1}$ $y^{-1}$, 2900 kg Ca $ha^{-1}$ $y^{-1}$ and 5900 kg Mg $ha^{-1}$ $y^{-1}$. At the 0.5 $ET_p$ rate, cumulative nutrient loads were estimated at 370 kg N $ha^{-1}$ $y^{-1}$, 190 kg P $ha^{-1}$ $y^{-1}$, 12000 kg K $ha^{-1}$ $y^{-1}$, 850 kg Ca $ha^{-1}$ $y^{-1}$ and 1700 kg Mg $ha^{-1}$ $y^{-1}$.

## Experimental Design

Four tropical grass species were established in the plots, each measuring 13.4 sq m (12 ft by 12 ft)—bana (*P. purpureum*), signal (*B. decumbens),* star (*C. nlemfuensis)* and suerte (*P. atratum*). Treatments were arranged in an augmented completely randomized design (Federer, 1956), with *P. atratum* and *B. decumbens* having two replicates for each rate of effluent irrigation (Plate 3.7). By the fifth month, *B. decumbens* succumbed to effluent irrigation. In January 2005, two of the *B. decumbens* plots that received the 2.0 $ET_p$ and 0.5 $ET_p$ effluent irrigation rates, respectively, were replanted to

California grass (*Brachiaria mutica*), while the other two *B. decumbens* plots were replanted to *P. purpureum*. Due to unsatisfactory performance of the irrigation system in one of the plots planted to *P. purpureum* at the 2.0 $ET_p$ irrigation rate, this plot was not considered in the data analysis and discussion presented here.

**Grass Harvesting, Soil and Soil Solution Sampling**

The grasses were well-established for a first measured harvest in August 2004. Biomass harvesting was done using a sickle mower (Plate 3.8a), with a 0.30 m area on each border side of the plot excluded in the yield calculations resulting in an effective harvest area of 9.29 $m^2$ per plot (Plate 3.8b). Thereafter, harvesting was done at an interval of 28 to 42 d during the first year (Aug 2004 to Jul 2005) and 32 to 52 d during the second year (Aug 2005 to Aug 2006) to determine biomass and tissue nutrient concentrations. Dry matter production was based on the effective harvest area and nutrient uptake was estimated by multiplying dry matter with nutrient concentration.

Prior to the application of the treatments, soil samples were collected at the 0 to 15 cm depth from 12 plots and at the 15 to 30 cm depth from 6 of the plots. From the first harvest onwards, soil samples were taken at monthly intervals from two depths (0 to 15 cm and 15 to 30 cm) within a week after each harvesting. Assessment of the impact on soil properties involved analysis of soil samples for pH, EC, N, organic C, P, K, Ca, Mg and Na. The changes in soil properties over time at both depths were, thus, monitored to determine whether an accumulation of nutrients or salts might occur and, if so, at what depth.

**Plate 3.7. Treatments were arranged in an augmented completely randomized design. Dairy effluent was supplied by subsurface drip irrigation.**

(a)

(b)

**Plate 3.8. (a) Sickle mower used in harvesting the grasses. (b) A 0.30 m border was excluded from yield calculations.**

Tension lysimeters were installed in eight of the 12 plots representing the following treatments: *P. atratum* (2.0 $ET_p$ and 0.5 $ET_p$), *B. decumbens* (2.0 $ET_p$ and 0.5 $ET_p$), *B. mutica* (2.0 $ET_p$ and 0.5 $ET_p$), *P. purpureum* (2.0 $ET_p$) and *C. nlemfuensis* (2.0 $ET_p$) (Plate 3.9). Soil solution samples were collected (at depths of 15 cm, 35 cm, 70 cm and 100 cm) also within a week after harvesting to help assess the downward movement of nutrients from the application zone (20 to 30 cm). Soil solutions were analyzed for pH, EC, P, K, Ca, Mg, Na, $NO_3$-N, total N, and $NH_4$-N. The flux of the solute was calculated as water flux (mm mo$^{-1}$) multiplied by the concentration of solute (mg L$^{-1}$) in the soil solution. The water flux for each irrigation treatments was calculated using the water balance method as follows:

$$WaterFlux\,(mm\ mo^{-1}) = (Irrigation,\ mm\ mo^{-1} + Rain,\ mm\ mo^{-1}) - ET_p,\ mm\ mo^{-1} \quad \text{(Eq. 2)}$$

The changes in effluent properties during the period of the experiment were monitored (Fig. 3.10). The effluent Sodium Absorption Ratio (SAR), calculated from

$$SAR = \frac{Na^{+}\ (meq\ L^{-1})}{\sqrt{\dfrac{Ca^{2+}\ (meq\ L^{-1}) + Mg^{2+}\ (meq\ L^{-1})}{2}}} \quad \text{(Eq. 3)}$$

and EC have both remained low (3.1 to 5.3 and 2.7 to 4.4 dS m$^{-1}$, respectively) throughout the period of the study, indicating the effluent's suitability for irrigation.

**Plate 3.9. Tension lysimeters were installed in eight of the experimental plots to measure changes in the soil solution properties.**

**Laboratory Analyses**

Total soil N was determined following the micro-Kjeldahl method (Bremner and Mulvaney, 1982). Extractable soil P was analyzed following the Olsen (Olsen et al., 1954; Olsen and Sommers, 1982) method and select soil samples for Modified Truog (0.02*N* or 0.01 *M* $H_2SO_4$ buffered at pH 3 with 0.3% $(NH_4)_2SO_4$) (Hue et al., 1997) and 1*M* HCl (Guo et al., 2000) methods. Extractable soil cations such as Ca, K, Mg, and Na were measured by inductively coupled plasma-atomic emission spectroscopy (ICP-AES).

Phosphorus sorption experiments were conducted following the modified Fox and Kamprath (1970) method (Linquist et al., 1997). A 0.02*M* KCl solution was used instead of 0.01*M* $CaCl_2$ to reduce the effect of Ca from the background electrolyte and more importantly, reduce the confounding effects of the background electrolyte on P sorption. Duplicate samples of soils (3-g oven dry weight basis) collected from the field experiment at six-month intervals beginning in Jul 2003, were equilibrated for six days after the addition of five different P concentrations (0, 50, 100, 250, 500 mg $kg^{-1}$). The pH (using a pH meter), EC (using an EC meter) and Ca content (using the inductively ICP-AES) of the equilibrium P solutions were also obtained.

Dairy effluent and soil solution samples were analyzed for P, K, Ca, Mg, Na, Fe, Mn, Zn, Cu and B using the ICP-AES method (Hue et al., 1997). Some studies (e.g., Phillips, 2002) usually measured the dissolved reactive P (DRP), which is a measure of orthophosphate--the soluble, inorganic (filterable in 0.45 μm membrane) fraction of P and the form taken up by the plants. But the ICP-AES method was chosen in this study because it represents all the species of P that are present in the solutions, dissolved or particulate, that are found in the sample and was, therefore, of greater interest when studying wastewater such as dairy effluent due to its important effects on soil and plant properties. Dairy effluent and soil solution samples (Sept 2005) were also analyzed for inorganic P using the Murphy and Riley (1962) colorimetric method and for total Kjeldahl N following the modified procedure of Technicon's Industrial Method No. 376-75W of August 21, 1975 (Technicon, 1977). The EC of the soil solution and dairy effluent were measured using an EC meter. Nitrate-N in the soil solution and effluent was analyzed using an auto-analyzer following the method of Henriksen and Selmer-Olsen (1970). Analysis of $NH_4$-N in soil solution and effluent involved auto-analysis using an auto-analyzer with a detection range of 0.05 to 5 mg $L^{-1}$ of $NH_4$-N, following the method of Gentry and Willis (1988).

The phosphate potentials for the different Ca-P compounds were calculated using the total Ca concentration in the soil solution and solubility products of these compounds (Table 3.2). However, the solubility diagrams were developed based on the assumption that the system is in equilibrium and all of the Ca and P are in free ionic forms (Lindsay, 1979), whereas the system in this study was not a steady-state system. Therefore, a sensitivity analysis was done assuming various percentages of ionic Ca (15% to 25%) and phosphates (45%, 70% and 100%) out of the total concentrations measured in the soil solution. About 15% to 25% ionic Ca out of the total Ca can be found in the soil (N. V. Hue, pers. comm.). Forty-five percent and 70% were the average inorganic P measured from soil solutions collected in Sept 2005 from plots irrigated at 0.5 $ET_p$ and 2.0 $ET_p$ rates, respectively. The dairy effluent had 66% inorganic P. Assuming 15% free Mg of the total Mg in the soil solution, the phosphate potential was also calculated using the solubility product of struvite (log K° (25°C) = 6.40) to determine the possibility of the precipitation of this compound.

**Table 3.2. Solubility products of the different calcium phosphate compounds.**

| Name of the Compound | Chemical Formula | Log K° (25°C) |
|---|---|---|
| Monocalcium phosphate (MCP) | $Ca(H_2PO_4)_2.H_2O$ | -1.15 |
| Dicalcium phosphate dihydrate (DCPD) | $CaHPO_4.2H_2O$ | 0.63 |
| Tricalcium phosphate (TCP) | $Ca_3(PO_4)_2$ | 10.18 |
| Octacalcium phosphate (OCP) | $Ca_8H_2(PO_4)_6 \cdot 5H_2O$ | 11.76 |
| Hydroxyapatite (HA) | $Ca_{10}(PO_4)_6(OH)_2$ | 14.46 |

Source: Lindsay (1979)

**Data Analysis**

Best subset procedures were conducted using the MINITAB 14.13 (Minitab Inc., 2004) to determine which variables are the best predictors of the dependent variable. The model with the lowest Mallows' Cp value was chosen, and if any two or more models had the same Mallows' Cp value, the next criterion was the highest R-squared value. The main and interaction effects of the treatments as well as the dynamics of change in soil and soil solution P were analyzed with Repeated Measures ANOVA using PROC MIXED of SAS 9.1 (SAS Institute, 2004) using the reduced model.

When PROC MIXED was used, the parameters of the matrix have been estimated using the restricted/residual maximum likelihood method (REML) and various covariance structures (compound symmetry or CS, unstructured or UN, first order autoregressive or AR(1), ante-dependence or ANTE(1), and Toeplitz or TOEP). The covariance structures were first tested for convergence and compared using the fit statistics (Akaike's Information Criterion or AIC, Akaike's Information Criterion-Corrected or AICC, and Schwarz' Bayesian Information Criterion or SBC/BIC). The covariance structure chosen based on the convergence and fit statistics criteria were AR(1) with random intercept for tissue monthly data, and compound symmetry for soil and soil solution data. For factors and interactions with significant effects on soil and solution properties, least square (LS) means (which adjusts for any missing data) were reported. Data were plotted using SigmaPlot 9.0 (SYSTAT Software Inc., 2004). To facilitate data analysis and discussion, the "time" variable was used to refer to sampling dates which in turn, correspond to varying amount of effluent irrigation over time at each of the two irrigation rates.

## Results and Discussion

### Soil and Soil Solution Phosphorus

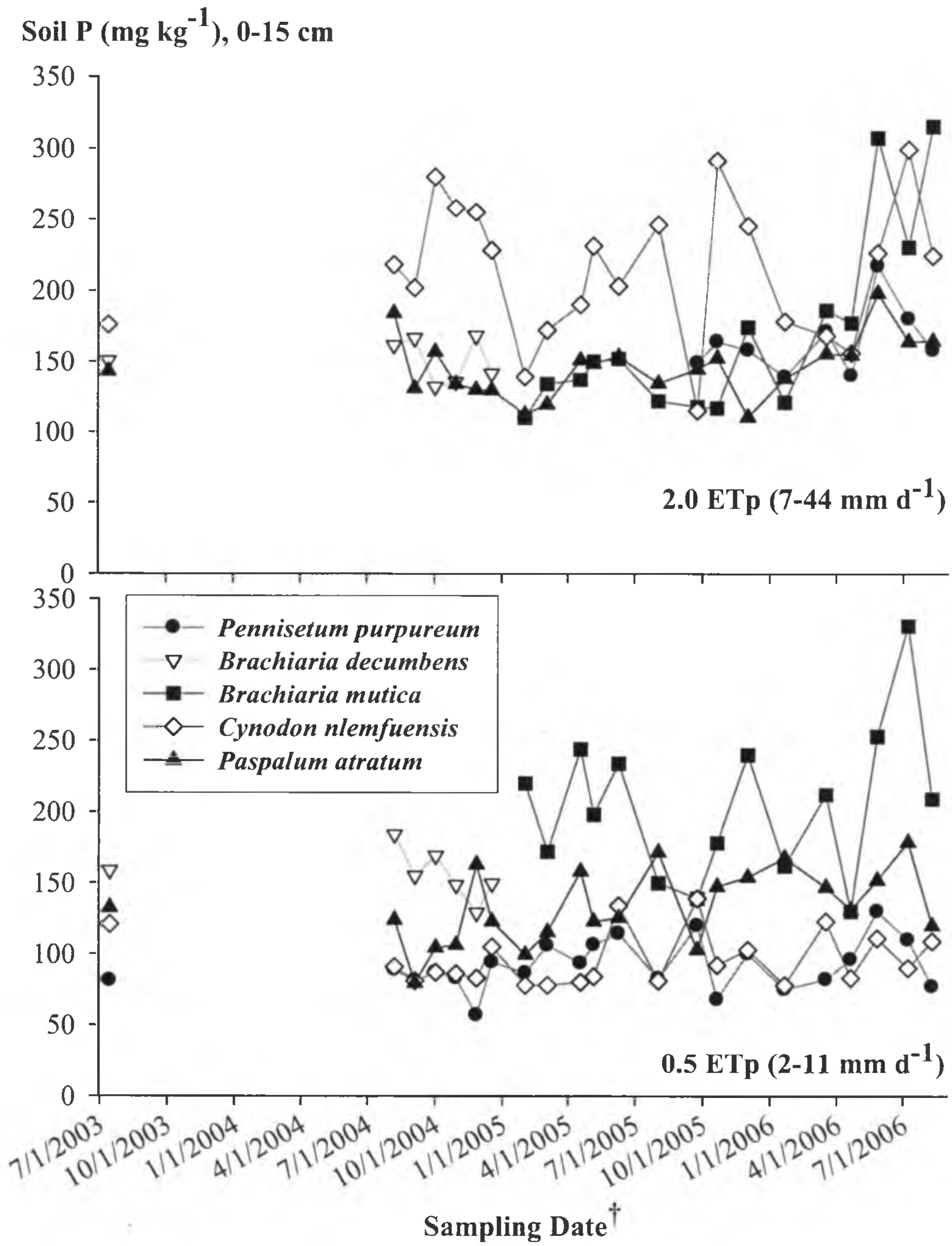
The extractable soil P prior to the installation of the irrigation system (Apr 2003) and after the installation (Jul 2003) of the irrigation system was already high ranging from 93 to 178 mg kg$^{-1}$ and 81 to 176 mg kg$^{-1}$, respectively. Between the beginning of the experiment (Jul 2003) and one year after the effluent irrigation installation (Aug 2004) (i.e., grass establishment phase), extractable soil P did not considerably increase ($P$>0.10) (Figs. 3.3 and 3.4). Statistical analysis of Olsen extractable soil P from Aug 2004 to Aug 2006 revealed that the main effects of time, irrigation rate and sampling depth were significant, while the main effect of grass species was not significant (Table 3.3). Olsen extractable soil P fluctuated over time, with slight declines occurring during the rainy season that may be ascribed to reduced amount of effluent applied during this period (Figs. 3.3 and 3.4). Also, relatively higher extractable soil P was obtained in Mar to Aug 2006 when higher amounts of effluent were applied due to the higher $ET_p$ during this period. Plots irrigated at the 2.0 $ET_p$ rate had generally higher mean extractable soil P (156.3 mg $kg^{-1}$) than those irrigated at 0.5 $ET_p$ (129.4 mg $kg^{-1}$). Higher mean extractable soil P was obtained at the 0 to 15 cm depth (145.8 mg $kg^{-1}$) than at the 15 to 30 cm depth (137.8 mg $kg^{-1}$). This was possibly due to the capillary movement of water that carried with it the nutrients from the application zone (20 to 25 cm depth) to the surface considering the very high $ET_p$ at the site.

**Table 3.3. Proc Mixed ANOVA[‡] statistics showing the Type III test of fixed effects on Olsen-extractable soil P of dairy effluent-irrigated soil, O'ahu, Hawai'i.**

| Effect | Numerator DF | Denominator DF | F Value | *P* |
|---|---|---|---|---|
| Time | 21 | 333 | 3.98 | <0.001 |
| Grass | 4 | 333 | 1.48 | 0.2091 |
| Irrigation rate | 1 | 333 | 3.13 | 0.0776 |
| Sampling depth | 1 | 333 | 7.90 | 0.0052 |
| Grass x Irrigation rate | 4 | 333 | 1.42 | 0.2259 |
| Grass x Time | 62 | 333 | 1.07 | 0.3532 |
| Irrigation rate x Sampling depth | 1 | 333 | 10.64 | 0.0012 |
| Irrigation rate x Time | 21 | 333 | 1.87 | 0.0121 |

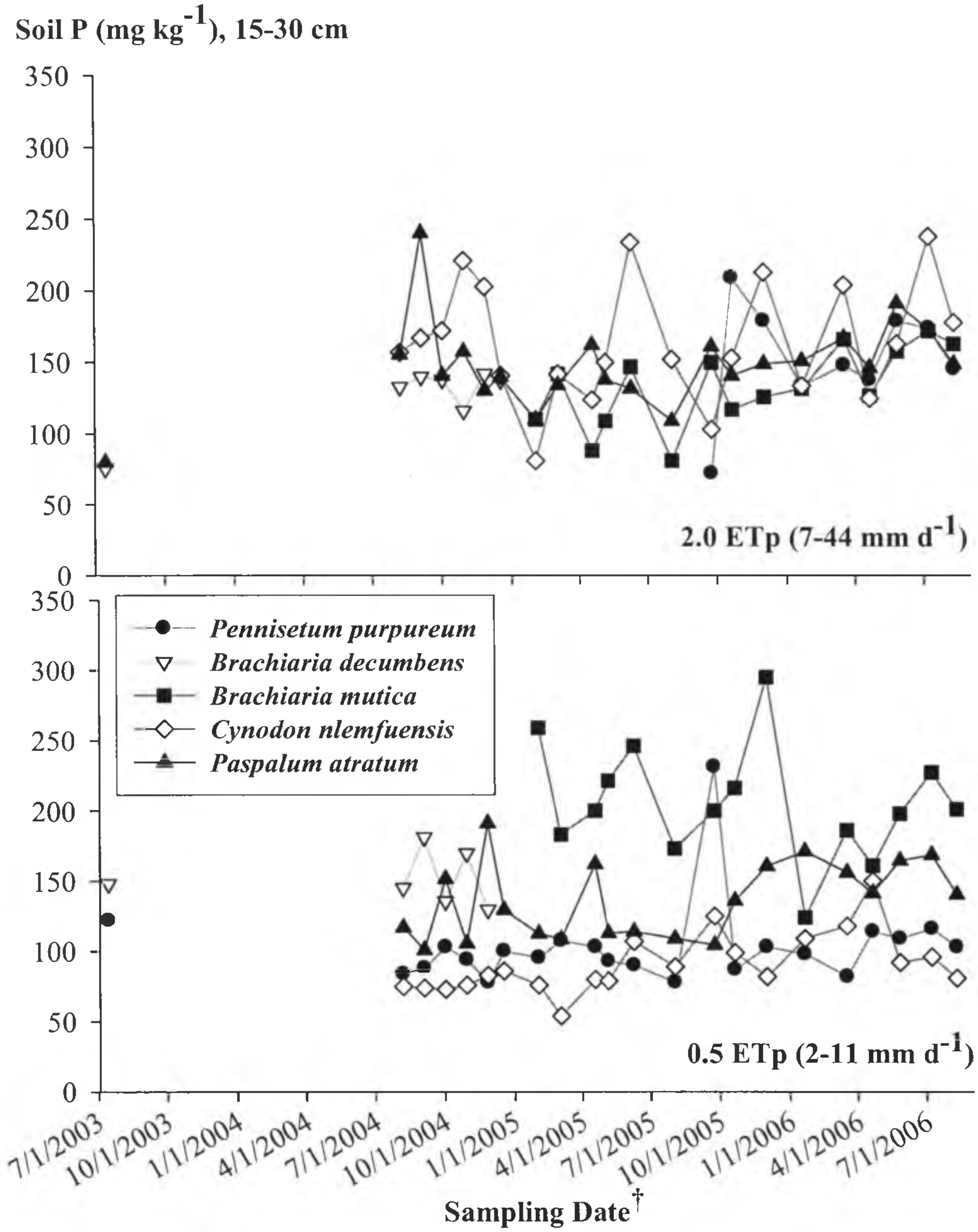
‡Based on best subset regression results.

The interaction between irrigation rate and sampling depth ($P<0.01$) and irrigation rate and time ($P<0.05$) was significant. At the 0 to 15 cm depth, extractable soil P was higher in plots irrigated at 2.0 $ET_p$ (165.9 mg $kg^{-1}$) than in plots that received the 0.5 $ET_p$ (128.9 mg $kg^{-1}$) rate. At the 15 to 30 cm depth, the extractable soil P was similar at the 2.0 $ET_p$ and 0.5 $ET_p$ rates (146.7 and 130.0 mg $kg^{-1}$, respectively). These results relate again to the high $ET_p$ at the surface. While Olsen soil P did not increase much over time at the two irrigation rates, the plots irrigated at 2.0 $ET_p$ showed greater fluctuations in soil P over time (Fig. 3.5).




†Soil samples were collected before the irrigation was installed in July 2003; grass establishment period was from December 2003 to July 2004

**Figure 3.3. Extractable (Olsen) phosphorus at the 0 to 15 cm depth of soils from dairy effluent-irrigated plots at 0.5 $ET_p$ and 2.0 $ET_p$ rates, Waianae, O'ahu, Hawai'i.**




†Soil samples were collected before the irrigation was installed in July 2003; grass establishment period was from December 2003 to July 2004

**Figure 3.4. Extractable (Olsen) phosphorus at the 15 to 30 cm depth of soils from dairy effluent-irrigated plots at the 0.5 $ET_p$ and 2.0 $ET_p$ rates, Waianae, O'ahu, Hawai'i.**

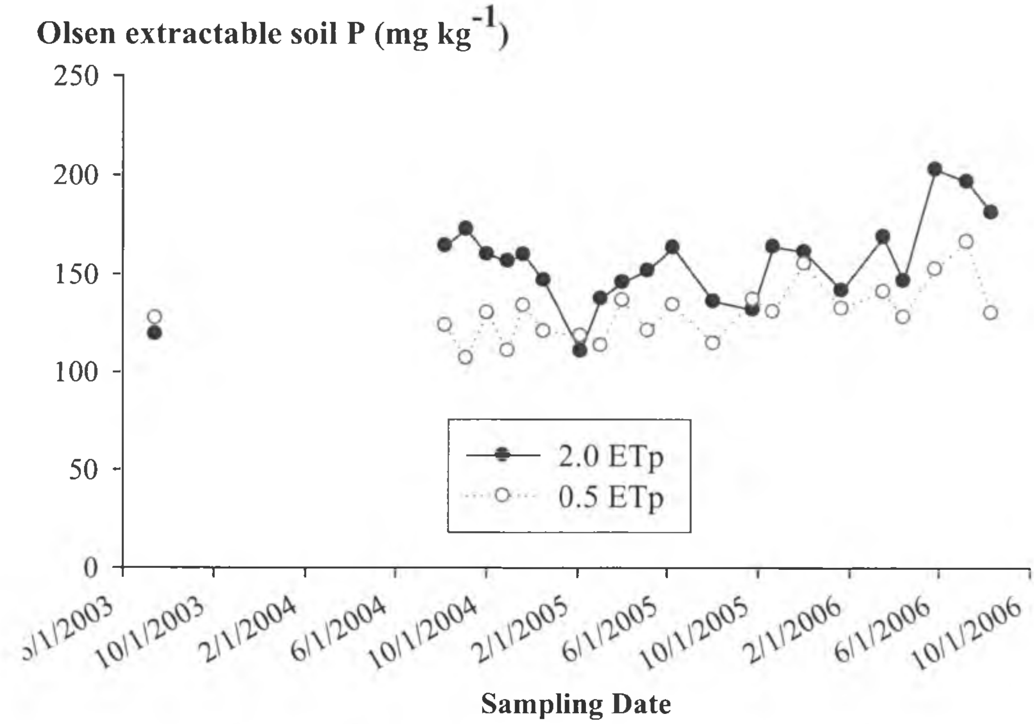



**Figure 3.5. Changes in Olsen extractable soil P over time at the 0.5 $ET_p$ and 2.0 $ET_p$ irrigation rates, Waianae, O'ahu, Hawai'i.**

Soil solution P also fluctuated over time, usually with lower concentrations (<5 mg $L^{-1}$) observed during the rainy period (Table 3.4 and Figs. 3.6 to 3.9). The relatively higher concentrations of soil solution P in Mar to Aug 2006 period also may be ascribed to the higher amounts of effluent were applied due to the higher $ET_p$ during this period. The interaction effect on soil solution total P of grass species and sampling depth, irrigation rate and sampling depth as well as irrigation rate and time were significant. The change in soil solution total P over time followed a similar trend for all the grass species ($P$<0.10). Soil solution total P was slightly higher at the 15 cm and 35 cm depths for *B. mutica* and *P. atratum* than at the 70 and 100 cm depths, whereas an opposite trend was observed for *C. nlemfuensis* and *B. decumbens* (Figs. 3.6 to 3.9). In fact, *C. nlemfuensis* plot had a mean soil solution total P of 5.5 mg $L^{-1}$ at the 100 cm depth, compared to 4.0 mg $L^{-1}$ at 15 cm, and 3.3 mg $L^{-1}$ at 35 cm and 70 cm depths. It is possible that some leaching of P occurred in this plot as a result of the high irrigation rate. Except at the 70 cm depth, the average soil solution total P at all depths was higher in plots irrigated at 2.0 $ET_p$, which can be explained by the higher amount of applied P to these plots. During the 2-y period of irrigation, the excess applied P ranged from 120 to 580 kg $ha^{-1}$ $y^{-1}$ in plots that received the 2.0 $ET_p$ rate and 12 to 104 kg $ha^{-1}$ $y^{-1}$ for plots that received the 0.5 $ET_p$ rate. Also, the levels of soil solution total P was relatively high from the beginning of the study which may be due to the effluent, dissolved manure and sediments carried by runoff during the rainy season.

**Table 3.4. Proc Mixed ANOVA[‡] statistics showing the Type III test of fixed effects on soil solution total P of dairy effluent-irrigated soil, O'ahu, Hawai'i.**

| Effect | Numerator DF | Denominator DF | F Value | *P* |
|---|---|---|---|---|
| Grass species | 3 | 3 | 0.11 | 0.946 |
| Time | 19 | 78 | 14.5 | <.0001 |
| Grass species x Irrigation rate | 2 | 3 | 0.04 | 0.9587 |
| Grass species x Sampling depth | 9 | 18 | 10.33 | <.0001 |
| Irrigation rate x Sampling depth | 3 | 24 | 12.54 | <.0001 |
| Irrigation rate x Time | 19 | 78 | 2.7 | 0.0011 |

‡Based on best subset regression results.

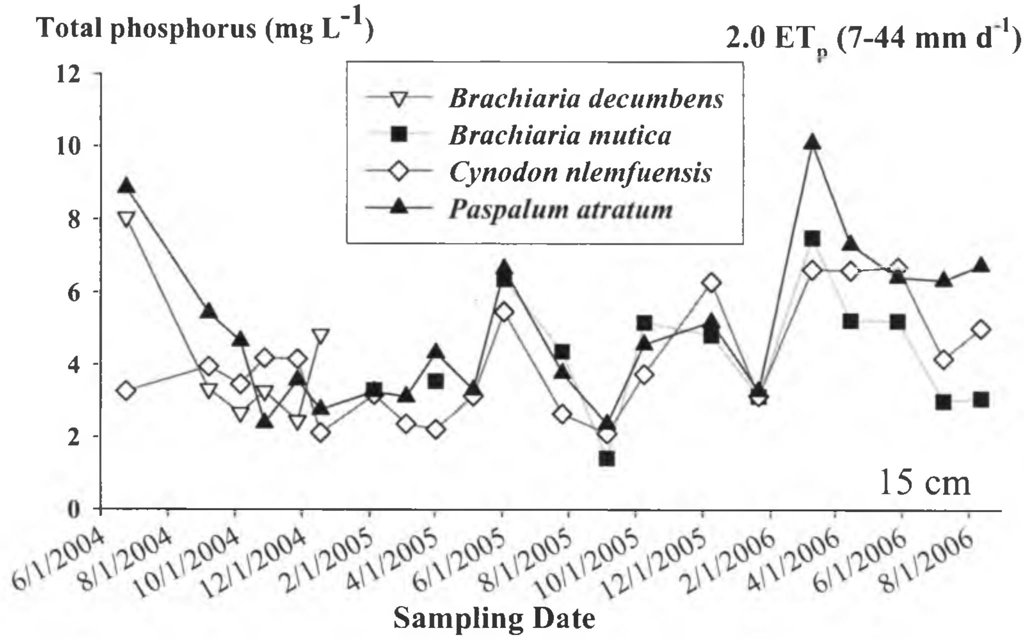



**Figure 3.6. Soil solution total phosphorus collected at the 15 cm depth from plots irrigated at the 2.0 $ET_p$ rate, Waianae, O'ahu, Hawai'i.**

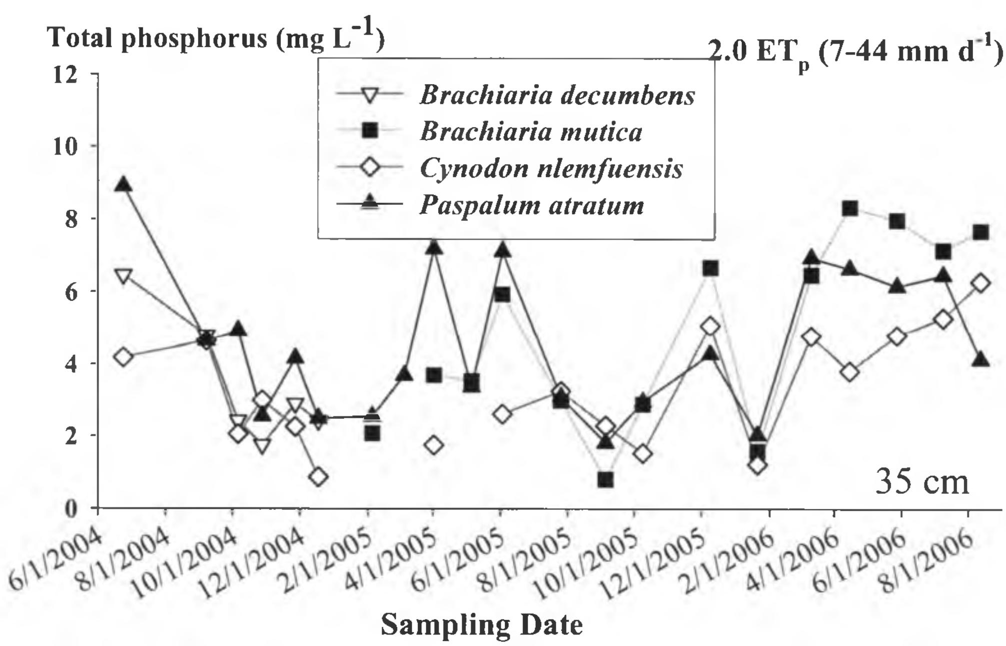



**Figure 3.7. Soil solution total phosphorus collected at the 35 cm depth from plots irrigated at the 2.0 $ET_p$ rate, Waianae, O'ahu, Hawai'i.**

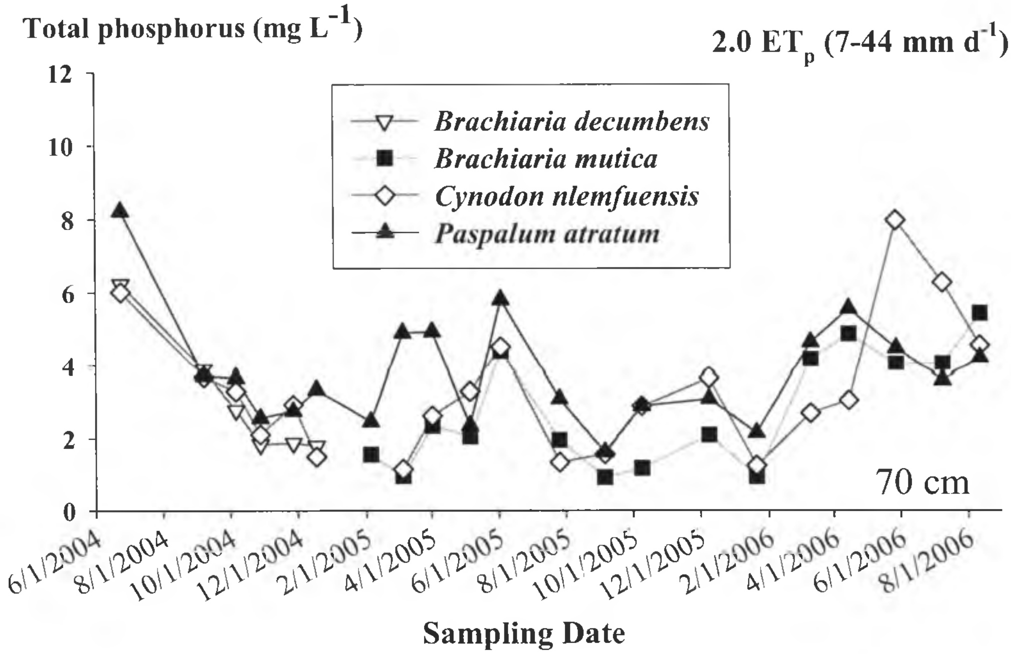



**Figure 3.8. Soil solution total phosphorus collected at the 70 cm depth from plots irrigated at the 2.0 $ET_p$ rate, Waianae, O'ahu, Hawai'i.**

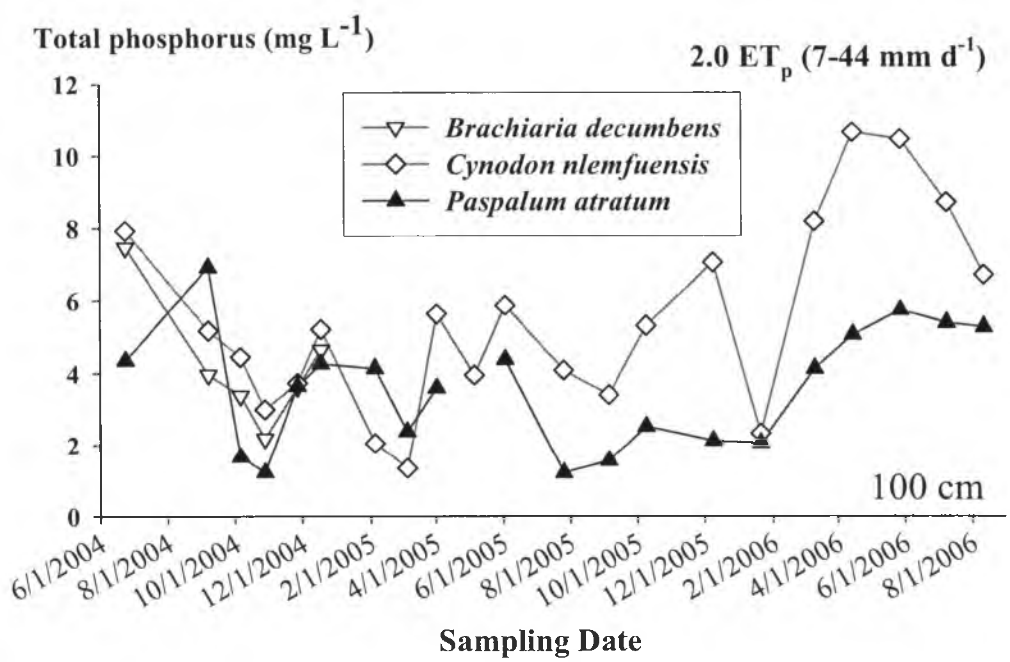



**Figure 3.9. Soil solution total phosphorus collected at the 100 cm depth from plots irrigated at the 2.0 $ET_p$ rate, Waianae, O'ahu, Hawai'i.**

The water flux (Table 3.5) of about 50 mm $mo^{-1}$ with a mean total P concentration of 3.2 mg $L^{-1}$ in Aug 2005 lead to a downward flux of 2.0 kg P $ha^{-1}$ or an annual equivalent of approximately 24.0 kg P $ha^{-1}$ $y^{-1}$ resulting from the irrigation. In Aug 2006, the water flux was 17 mm $mo^{-1}$ with a mean total P concentration of 6.0 $L^{-1}$ lead to a downward flux of 0.7 kg P $ha^{-1}$ or an annual equivalent of 8.4 kg P $ha^{-1}$ $y^{-1}$. Using the average measured moisture content of the soil (30% at the 15 to 30 cm depth from the plots irrigated at the high rate) and a dry bulk density of 1.3 g $cm^{-3}$, the 3.2 mg P $L^{-1}$ was equivalent to about 1.5 kg P $ha^{-1}$ (or 17.0 kg P $ha^{-1}$ $y^{-1}$), which approximates the amount of P that might have remained in the soil profile water. However, calculations for the 2.0 $ET_p$ effluent irrigation rate using the monthly average soil solution total P and corresponding water flux for each month showed an annual P flux in the soil profile of 131.0 kg P $ha^{-1}$ $y^{-1}$ (Aug 2004 to Aug 2005) and 99.0 kg P $ha^{-1}$ $y^{-1}$ (Sept 2005 to Aug 2006). These values are much higher than the previous estimates of 24.0 and 8.4 kg P $ha^{-1}$ $y^{-1}$, which used only the water fluxes and P concentrations in the drainage water on Aug 2005 and Aug 2006 multiplied by 12 months. The higher annual P flux in the second calculation method can be explained by the fluctuations in the concentration of P in the drainage water (3.2 to 8.1 mg $L^{-1}$) and the adjustments in the irrigation rates according to the estimated $ET_p$ resulting in variations in monthly water flux (Table 3.5). These results indicated that some downward movement of P may have occurred.

The fluctuations in the concentration of P in the drainage water can be explained by the variations in the amount and P content of the effluent used for irrigation (Fig. 3.10). Somewhat higher concentrations of Ca and micronutrients were also obtained during the second year (Sept 2005 to Aug 2006 period) than the first year (Jul 2004 to

Aug 2005) of the study presumably due to the reduced input of water to the lagoon during the second year (because of the reduction in cattle population) that resulted in a less diluted nutrient concentration. However, the trend in total P concentration in the effluent during the second year did not match the trend for total Ca. It could be explained by the possible precipitation of Ca with P in the lagoon (Fordham and Schwertmann, 1977), and also, due to the reduced input of P to the lagoon as a result of the reduction in animal population at the farm. Phosphorus is an important component of the diet of dairy cattle (Satter et al., 2005) and is typically added as a supplement to the dairy cattle ration.

Calculations also showed no water flux at the plots irrigated at the 0.5 $ET_p$ rate because the $ET_p$ exceeded the sum of the irrigation applied and the rainfall. The moisture content of the soil from the plots irrigated at the 0.5 $ET_p$ rate was also relatively low—an average of 13% at the 0 to 15 cm depth and 20% at the 15 to 30 cm depth.

**Table 3.5. Calculated monthly water flux in plots planted to tropical grasses and irrigated with dairy effluent, Waianae, O'ahu, Hawai'i.**

| | Irrigation | | Total Rainfall | $ET_p$ | Water Flux | | Water Flux | | Calculated P Flux‡ | |
|---|---|---|---|---|---|---|---|---|---|---|
| | 2.0 $ET_p$ | 0.5 $ET_p$ | | | 2.0 $ET_p$ | 0.5 $ET_p$ | 2.0 $ET_p$ | 0.5 $ET_p$ | 2.0 $ET_p$ | 0.5 $ET_p$ |
| | ----------mm $mo^{-1}$ | ---------- | | | | | ------L $mo^{-1}$ | ------ | kg $ha^{-1}$ | $mo^{-1}$ |
| **2004** | | | | | | | | | | |
| Sep | 270 | 120 | 4 | 300 | -26 | -176 | -345 | -2352 | -1.7 | -11.4 |
| Oct | 434 | 155 | 4 | 200 | 238 | -41 | 3185 | -547 | 11.4 | -2.0 |
| Nov | 420 | 150 | 187 | 192 | 415 | 145 | 5549 | 1937 | 14.6 | 5.1 |
| Dec | 434 | 155 | 125 | 171 | 388 | 109 | 5191 | 1459 | 10.2 | 2.9 |
| **2005** | | | | | | | | | | |
| Jan | 434 | 155 | 109 | 157 | 386 | 107 | 5166 | 1434 | 13.1 | 3.6 |
| Feb | 336 | 112 | 21 | 200 | 157 | -67 | 2107 | -890 | 4.4 | -1.9 |
| Mar | 372 | 124 | 27 | 248 | 151 | -97 | 2023 | -1295 | 4.3 | -2.7 |
| Apr | 390 | 98 | 15 | 296 | 109 | -184 | 1456 | -2457 | 3.1 | -5.2 |
| May | 1381 | 345 | 0 | 482 | 899 | -137 | 12022 | -1833 | 36.2 | -5.5 |
| Jun | 1336 | 334 | 0 | 499 | 837 | -165 | 11197 | -2212 | 28.0 | -5.5 |
| Jul | 690 | 159 | 0 | 579 | 112 | -419 | 1493 | -5611 | 5.9 | -22.2 |
| Aug | 690 | 159 | 0 | 641 | 50 | -481 | 664 | -6440 | 1.4 | -13.5 |
| Sep | 690 | 159 | 0 | 599 | 91 | -440 | 1221 | -5883 | 2.6 | -12.4 |
| Oct | 690 | 159 | 0 | 538 | 152 | -379 | 2036 | -5068 | 2.8 | -7.0 |
| Nov | 956 | 239 | 0 | 0 | 956 | 239 | 12786 | 3197 | 17.8 | 4.4 |
| Dec | 531 | 133 | 0 | 374 | 157 | -241 | 2100 | -3229 | 5.2 | -8.0 |
| **2006** | | | | | | | | | | |
| Jan | 531 | 133 | 29 | 430 | 131 | -267 | 1751 | -3579 | 6.0 | -12.2 |
| Feb | 637 | 159 | 19 | 355 | 302 | -176 | 4036 | -2359 | 6.4 | -3.7 |
| Mar | 637 | 159 | 331 | 290 | 679 | 201 | 9088 | 2693 | 14.3 | 4.2 |
| Apr | 637 | 159 | 19 | 328 | 328 | -150 | 4392 | -2003 | 19.8 | -9.0 |
| May | 637 | 159 | 5 | 467 | 176 | -302 | 2352 | -4043 | 11.0 | -18.9 |
| Jun | 617 | 154 | 0 | 499 | 118 | -345 | 1572 | -4616 | 7.6 | -22.4 |
| Jul | 637 | 159 | 0 | 560 | 77 | -401 | 1034 | -5361 | 4.3 | -22.5 |
| Aug | 637 | 159 | 0 | 620 | 17 | -461 | 231 | -6164 | 0.9 | -25.0 |

‡Based on the mean P concentration of soil solution collected at the 15 cm, 35 cm, 70 cm and 100 cm depths

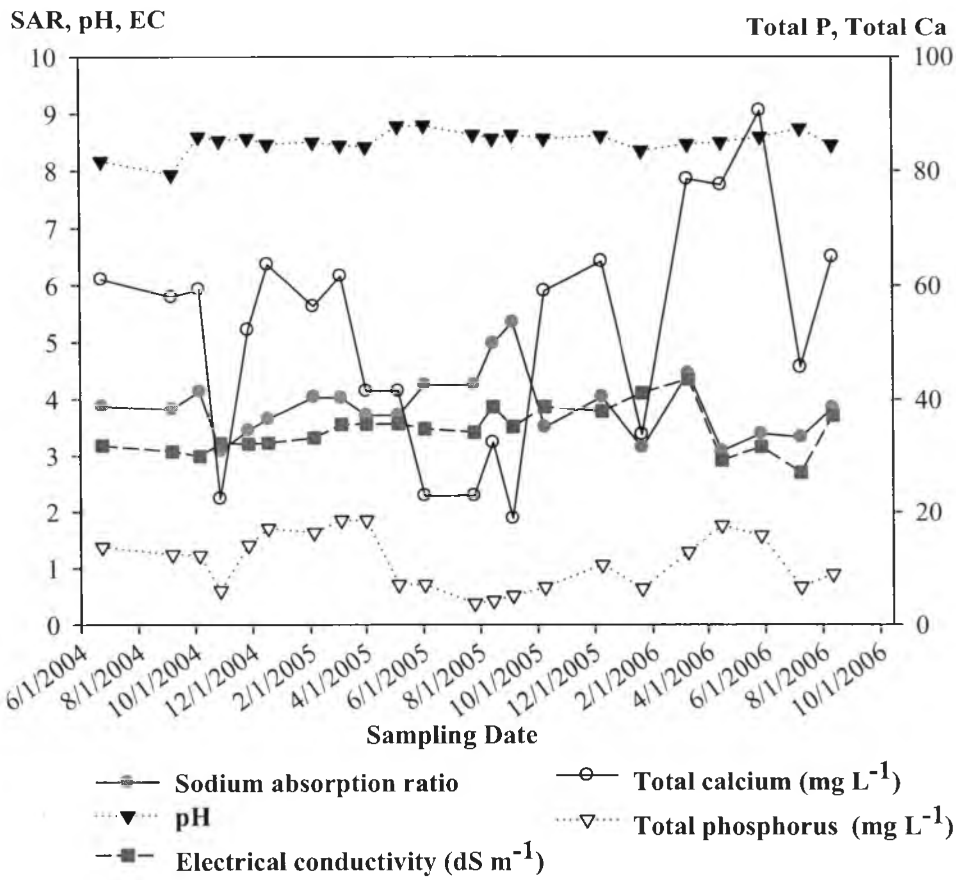



**Figure 3.10. Selected chemical properties of dairy effluent used for irrigating tropical grasses, Waianae, O'ahu, Hawai'i.**

## Phosphorus Sorption

Sorption-desorption reactions control the accumulation and movement of P in the soil profile. Quantitative descriptions of P sorption have commonly been made by fitting the Langmuir, Freundlich or Temkin equations into the data (Muneer and Lawrence, 2004; Villapando and Graetz, 2001). Both the Freundlich and Langmuir models fit the data well as indicated by the high $R^2$ values. However, convergence was not attained in many P sorption data fitted with the Langmuir model (data not shown) indicating that the Freundlich model fit the data better than the Langmuir equation. The Freundlich model was found to best describe the P sorption capacities of soils receiving cattle manure (Whalen and Chang, 2002).

Previous studies have shown that repeated application of manure and biosolids on soils could decrease the P sorption capacity of the soil (i.e., greater increase in soil solution P per unit of added P) (e.g., Sui and Thompson, 2000; Phillips, 2002; Holford et al., 1997). The soil samples taken before the effluent irrigation began (Jul 2003) had the highest P sorption capacity (Figs. 3.11 to 3.14). Consistent with our expectation, the P sorption capacity of the soils receiving effluent slightly decreased after a year of irrigation (Aug 2004) at both the 0 to 15 cm and 15 to 30 cm depths. In our case, however, the P sorption isotherm shifted again upward towards the initial P sorption curve for samples collected in Feb 2005 until Aug 2006, indicating a possible increase in the P sorption capacity of the effluent-irrigated soil. Some researchers who studied the impact of animal manure or effluent application on soil properties attributed the increased P sorption of the soil to the formation of Ca-P compounds (Robinson and Sharpley, 1996; Whalen and Chang, 2002), particularly in soils with very high Ca content and high pH (Eghball et al., 1996; Wang et al., 1995). A recent study explores the factors affecting Ca-P formation and degree of crystallinity in soils in Florida receiving manure applications (Cao, 2006). The changes in P sorption capacity of the soil in this study was not as prominent as that observed by other workers (Whalen and Chang, 2002; Wang et al., 1995) possibly due to the relatively short duration (2 years) of effluent application in this study.

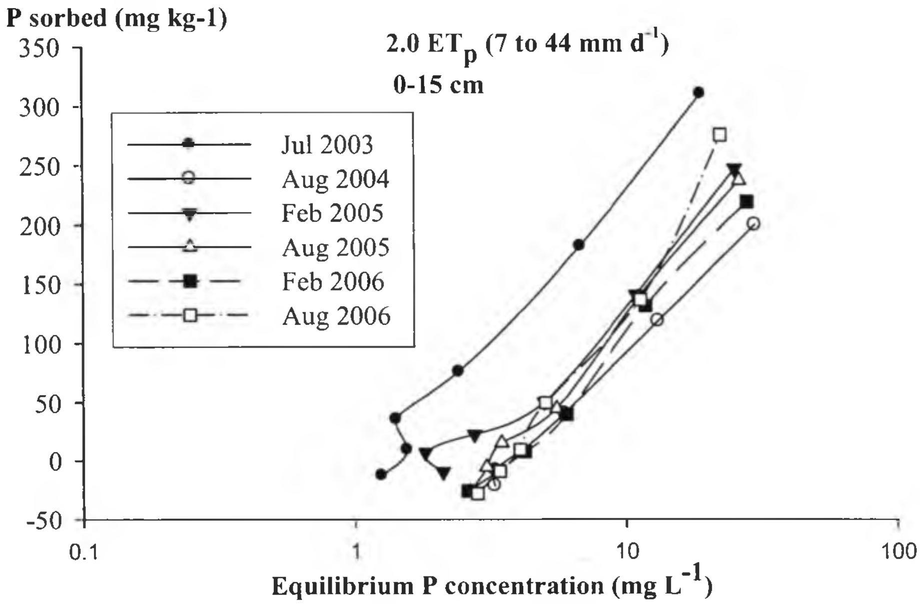



**Figure 3.11. Changes in phosphorus sorption by a soil (Mollisol) planted to tropical grasses and irrigated with dairy effluent at the 2.0 $ET_p$, 0 to 15 cm depth, Waianae, O'ahu, Hawai'i.**

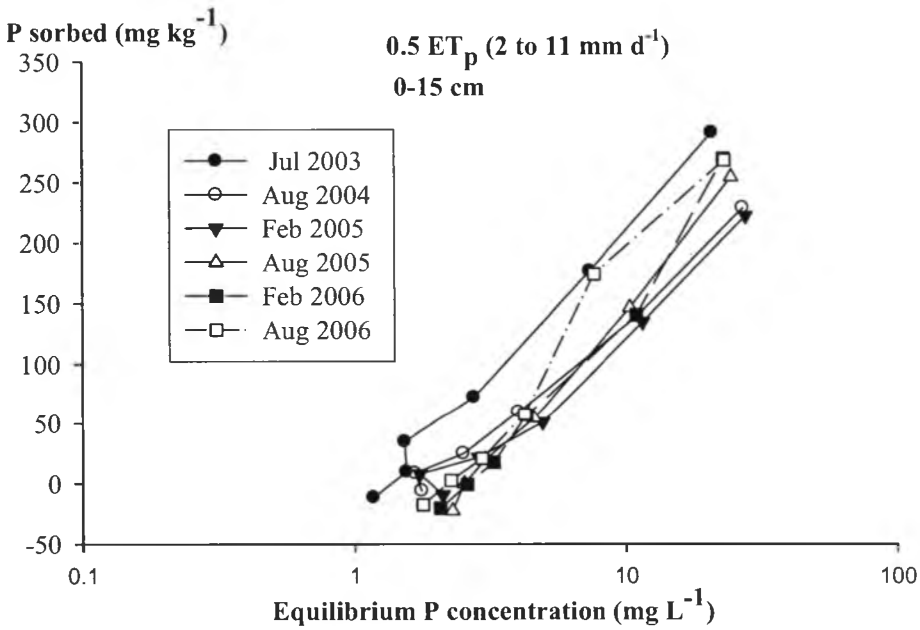



**Figure 3.12. Changes in phosphorus sorption curve for a soil (Mollisol) planted to tropical grasses and irrigated with dairy effluent at the 0.5 $ET_p$, 0 to 15 cm depth, Waianae, O'ahu, Hawai'i.**

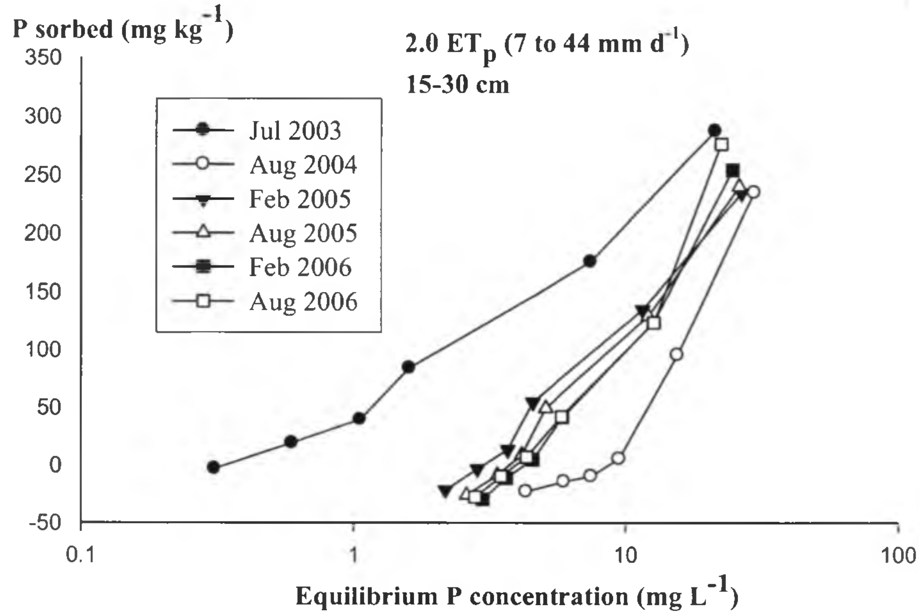



**Figure 3.13. Changes in phosphorus sorption curve for a soil (Mollisol) planted to tropical grasses and irrigated with dairy effluent at the 2.0 $ET_p$, 15 to 30 cm depth, Waianae, O'ahu, Hawai'i.**

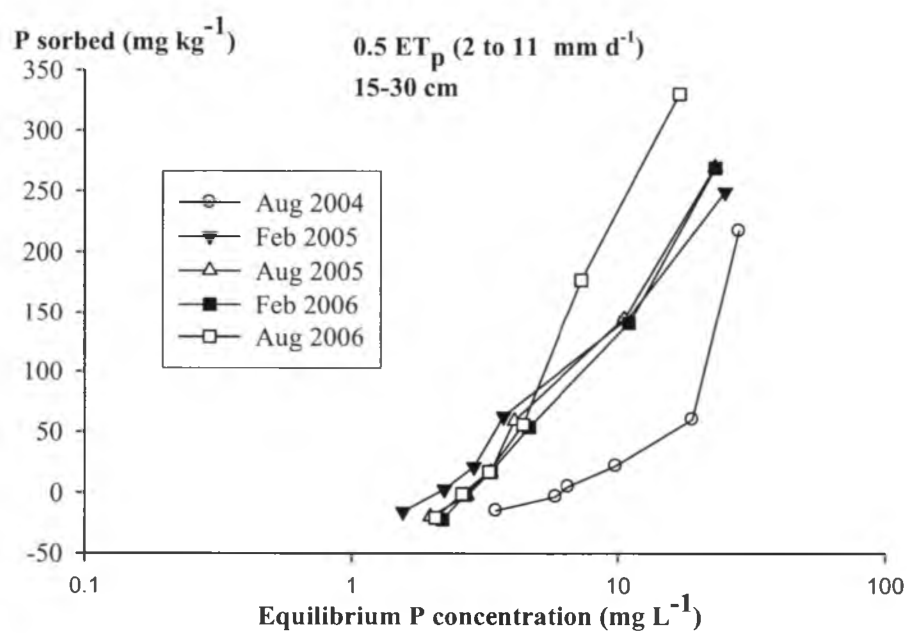



**Figure 3.14. Changes in phosphorus sorption curve for a soil (Mollisol) planted to tropical grasses and irrigated with dairy effluent at the 0.5 $ET_p$, 15 to 30 cm depth, Waianae, O'ahu, Hawai'i.**

**Precipitation of Phosphorus with Calcium and Magnesium**

The above trend in extractable soil P and soil solution total P bear an important relationship with the results of the extraction of P with acidic extractants and the trend in soil and soil solution pH. Higher extractable soil P was obtained when acidic extractants such as Modified Truog and HCl were used, indicating that some of the soil P pools are in a form, possibly as calcium precipitates, which was not extractable by the basic Olsen extractant (Figs. 3.15 and 3.16). It is, thus, hypothesized that some of the P in the soil may be precipitating with Ca (Wang et al., 1995), given the high amounts of Ca added to the soil from effluent (850 to 2900 kg Ca $ha^{-1}$ $y^{-1}$) plus the already high Ca content of the soil at the site (Fig. 3.17 to 3.18). The extractable soil Ca had also declined over time ($P$<0.01) especially at the 2.0 $ET_p$ irrigation rate, suggesting that Ca may have been forming into more difficultly available compound. A variety of calcium phosphates (octacalcium phosphate, dicalcium phosphate) and magnesium phosphates (struvite, trimagnesium phosphate) in liquid cattle manures were identified by Fordham and Schwertmann (1977).

The pH of the soil was already initially high (7.4 to 7.6 in Jul 2003) and with repeated irrigation with high pH effluent (7.9 to 8.8), the soil pH further increased up to 8.7 in some plots after three years (Figs. 3.19 and 3.20). The pH and total Ca content of the equilibrium P solutions in the P sorption experiment was also high. These conditions are favorable for the formation of various Ca-P compounds and $CaCO_3$.

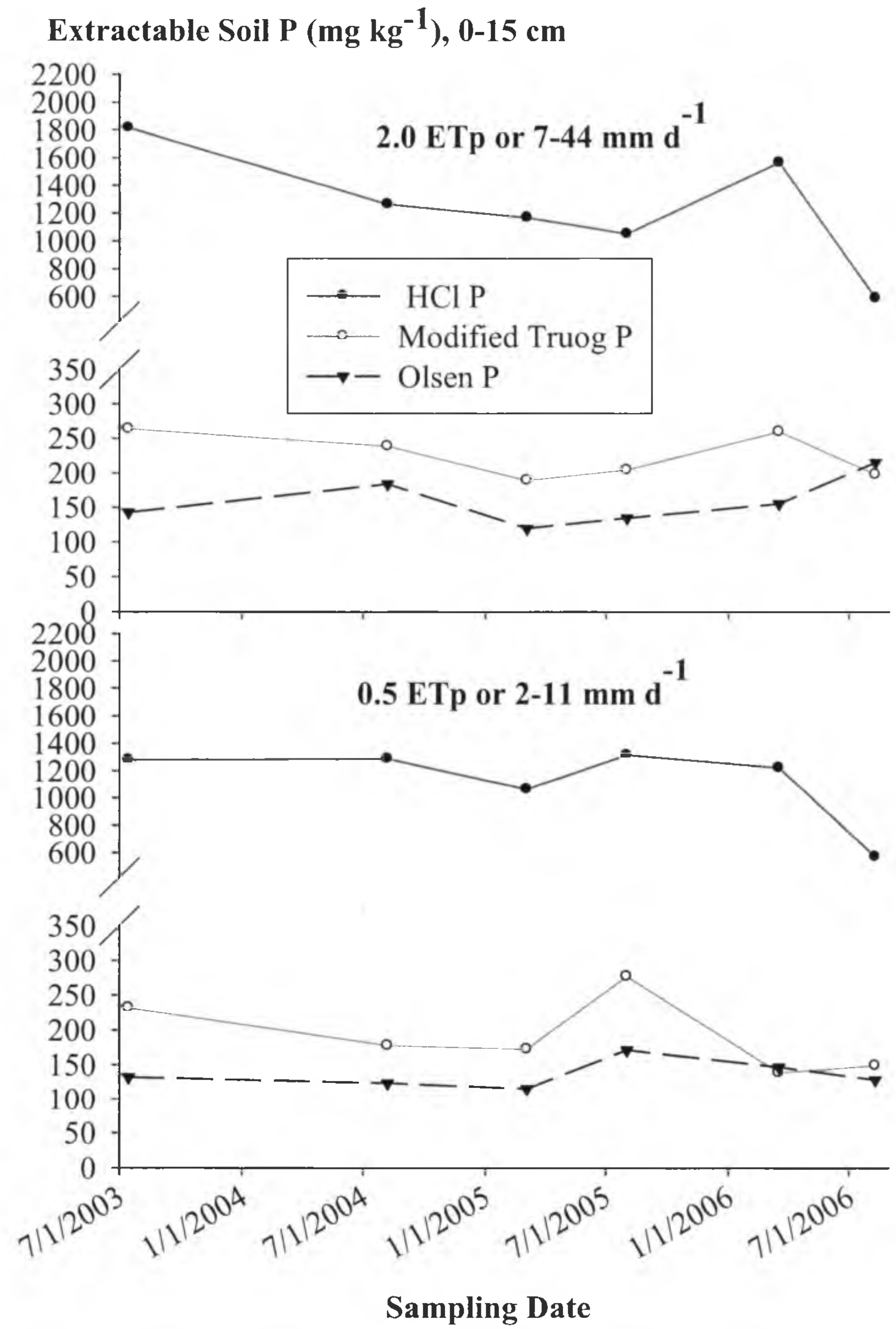



**Figure 3.15. Comparison of extractable soil P at the 0 to 15 cm depth using various extractants.**

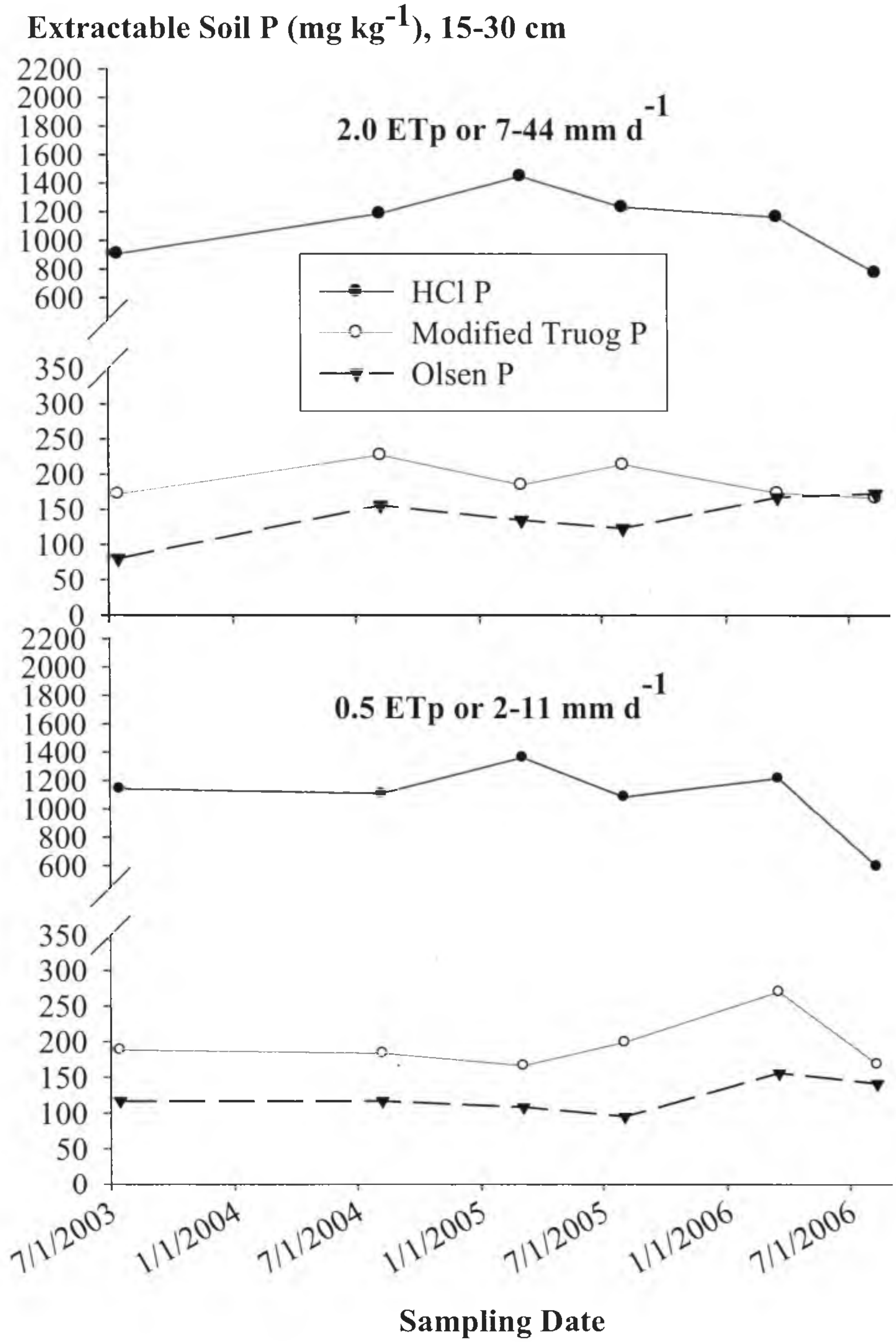
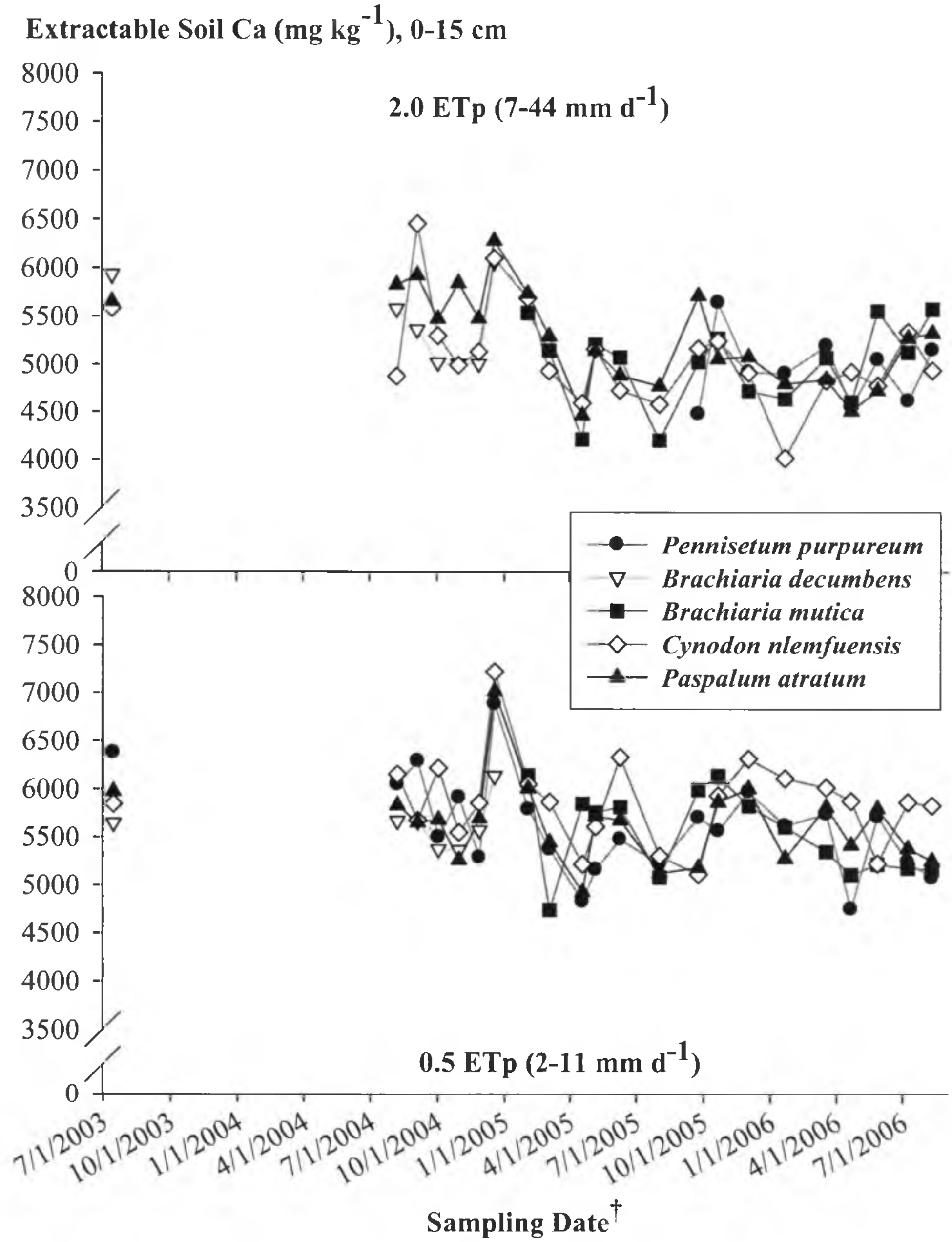
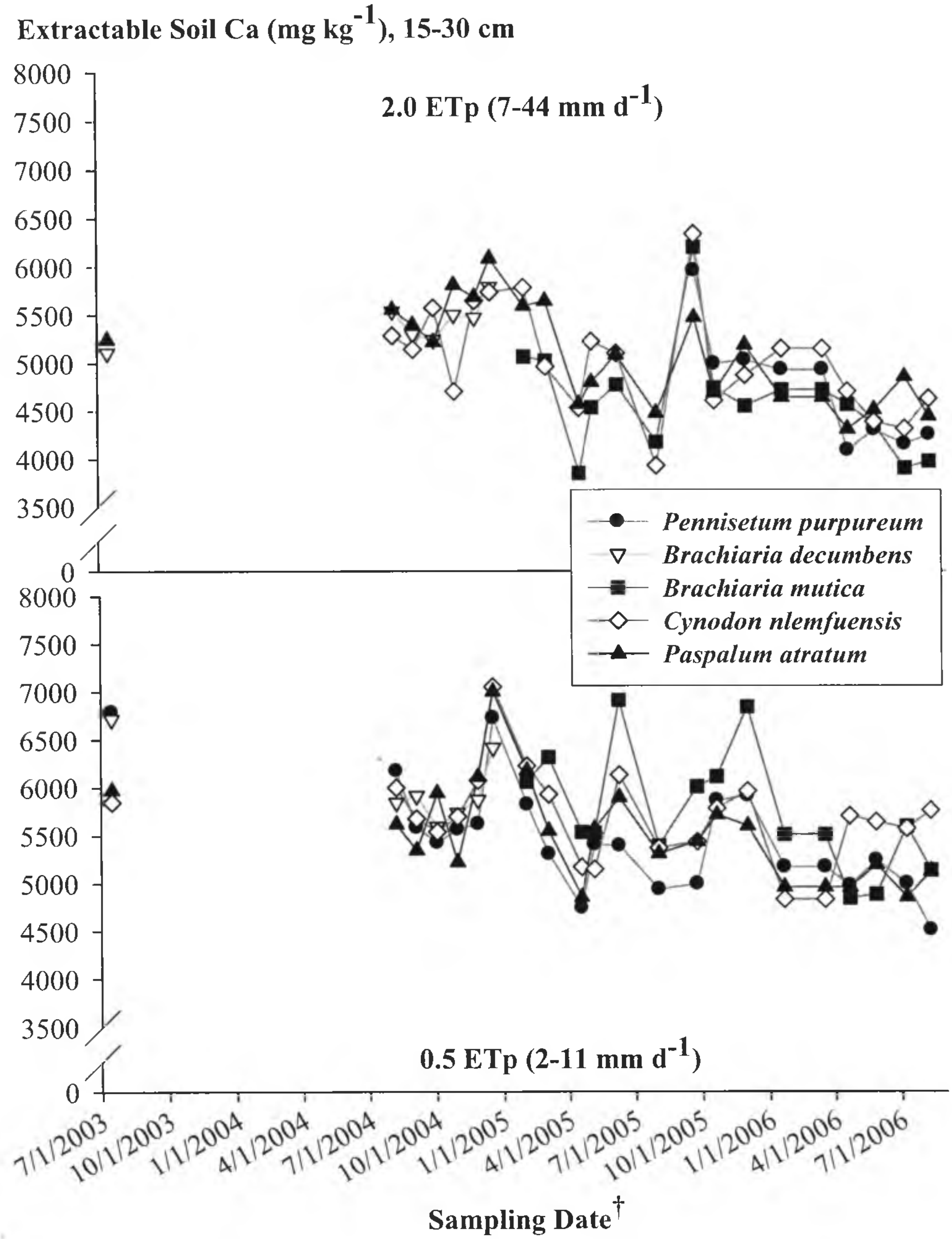



**Figure 3.16. Comparison of extractable soil P at the 15 to 30 cm depth using various extractants.**



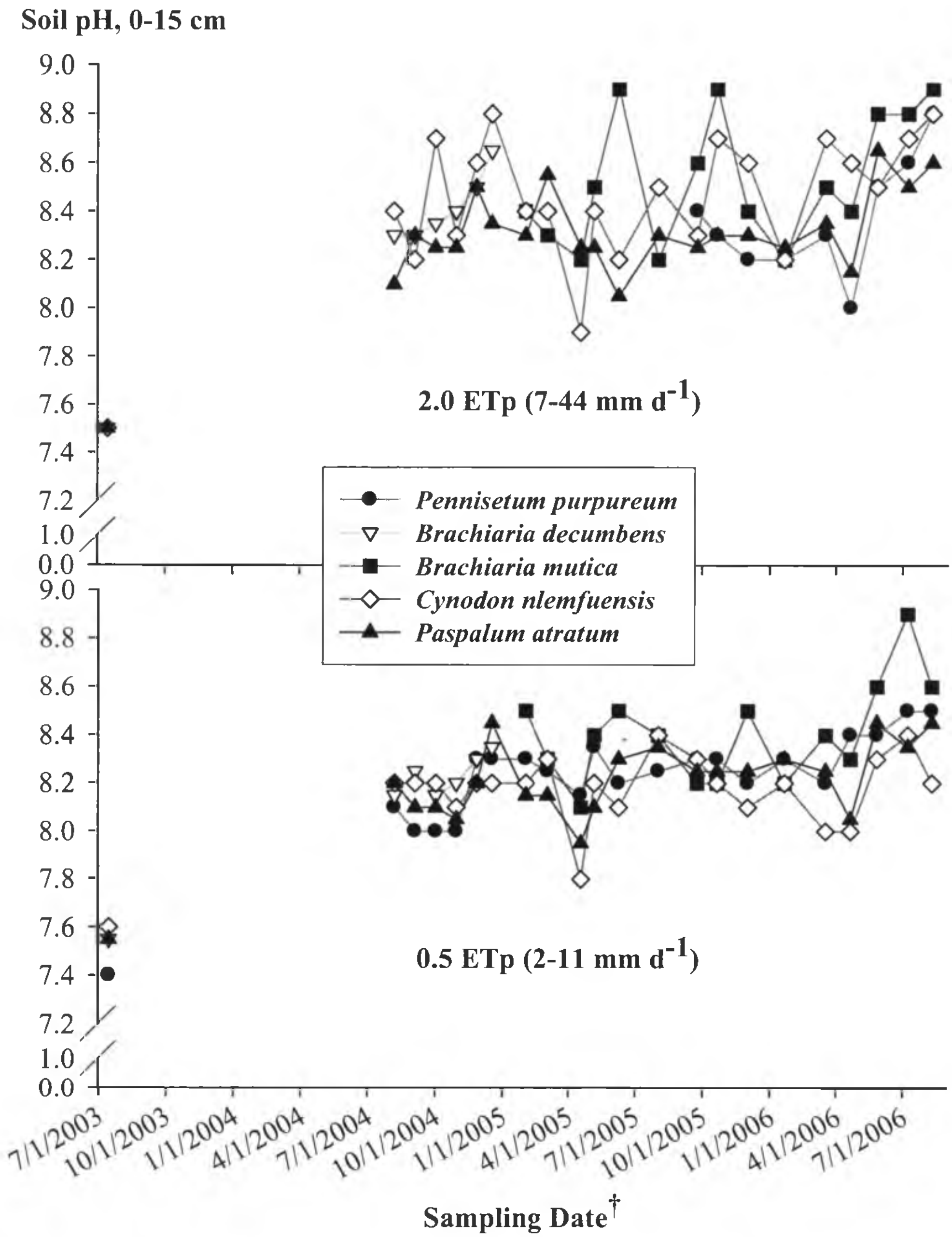

†Soil samples were collected before the irrigation was installed in July 2003; grass establishment period was from December 2003 to July 2004

**Figure 3.17. Extractable soil calcium at the 0 to 15 cm depth of soils from dairy effluent-irrigated plots at 0.5 $ET_p$ and 2.0 $ET_p$ rates, Waianae, O'ahu, Hawai'i.**



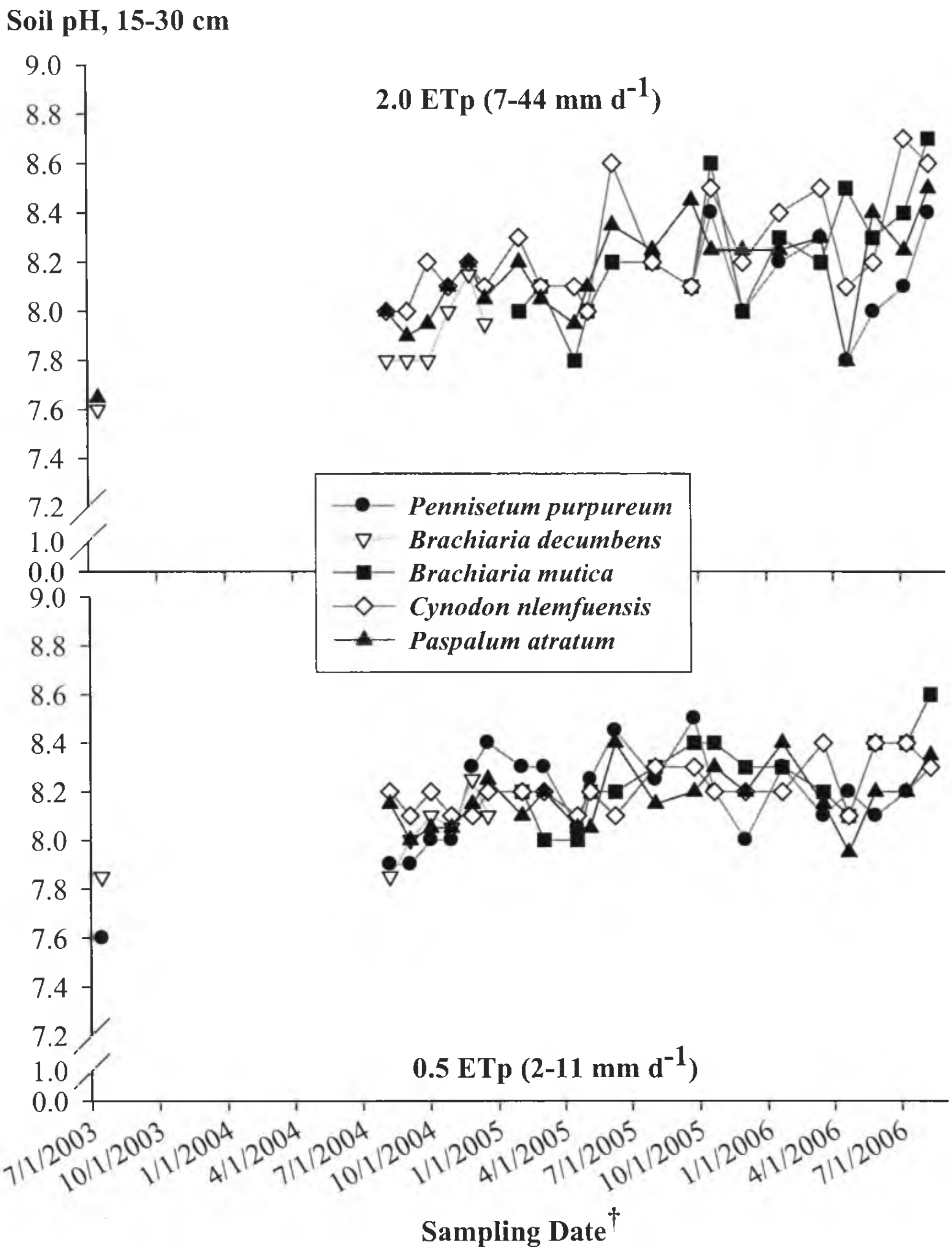

†Soil samples were collected before the irrigation was installed in July 2003; grass establishment period was from December 2003 to July 2004

**Figure 3.18. Extractable soil calcium at the 15 to 30 cm depth of soils from dairy effluent-irrigated plots at 0.5 $ET_p$ and 2.0 $ET_p$ rates, Waianae, O'ahu, Hawai'i.**




†Soil samples were collected before the irrigation was installed in July 2003; grass establishment period was from December 2003 to July 2004

**Figure 3.19. Soil pH at the 0 to 15 cm depth of soils from dairy effluent-irrigated plots at 0.5 $ET_p$ and 2.0 $ET_p$ rates, Waianae, O'ahu, Hawai'i.**




†Soil samples were collected before the irrigation was installed in July 2003; grass establishment period was from December 2003 to July 2004

**Figure 3.20. Soil pH at the 15 to 30 cm depth of soils from dairy effluent-irrigated plots at 0.5 $ET_p$ and 2.0 $ET_p$ rates, Waianae, O'ahu, Hawai'i.**

The phosphate potentials for various Ca-P compounds (Lindsay, 1979) of the soil samples collected in various times from plots that received the 2.0 $ET_p$ and 0.5 $ET_p$ irrigation rates (0 to 15 cm and 15 to 30 cm depths) were calculated (Tables 3.6 to 3.12). Assuming 15% or 20% ionic Ca and 45% inorganic P in the soil solution, the soils collected in Aug 2004 to Aug 2006 were undersaturated with respect to dicalcium phosphate dihydrate (DCPD), octacalcium phosphate (OCP), tricalcium phosphate (TCP) and hydroxyapatite (HA) (Tables 3.6 and 3.8). Soil samples collected in Aug 2006 were only oversaturated with respect to hydroxyapatite when 15% or 20% free Ca and 70% inorganic P in the soil solution were assumed (Tables 3.7 and 3.9, and Fig. 3.21). Thus, the formation of an insoluble apatite cannot be discounted. Calcium phosphate precipitation was reported as the major mechanism for reducing P availability to plants when the concentration of soil solution $Ca^{2+}$ exceeded 0.5*mM* (Tunesi et al., 1999). In our study, total Ca concentrations were between 1.0*mM* to 3.8*mM*, and calculations showed that the precipitation of Ca-P in the soil may occur if the free Ca in the soil solution was 15% to 25% of the total Ca and 100% phosphate activity (Tables 3.10 to 3.12). Struvite ($MgNH_4PO_4.6H_2O$) may not have precipitated in this effluent-irrigated soil (Table3.13 and Fig. 3.22). Phosphate potential calculations also showed that the soil was undersaturated with respect to bobierrite ($Mg_3(PO_4)_2.8H_2O$) (data not shown). These results were consistent with the finding that sparingly soluble Mg-P and Ca-P phases control the release of P in dairy manure-impacted soils (Josan et al., 2005).

Table 3.6. Calculated phosphate potential‡ for various calcium-phosphate compounds that may have formed in a soil planted to tropical grasses and irrigated with dairy effluent, Waianae, O'ahu, Hawai'i.

| Irrigation rate/ Sampling Depth | Soil pH | Solution EC | Ca activity | MCP | DCPD | OCP | TCP | HA |
|---|---|---|---|---|---|---|---|---|
| | | $dS\ m^{-1}$ | *mM* | -log $H_2PO_4^-$ | | | | |
| **2.0 $ET_p$, 0-15 cm** | | | | | | | | |
| Jul '03 | 7.5 | - | - | - | - | - | - | - |
| Aug '04 | 8.2 | 3 | 0.16 | 0.6 | -1.7 | -2.1 | -2.5 | -3.6 |
| Feb '05 | 8.4 | 4 | 0.10 | 0.6 | -1.7 | -2.1 | -2.6 | -3.6 |
| Aug '05 | 8.3 | 4.26 | 0.08 | 0.7 | -1.6 | -2.0 | -2.4 | -3.5 |
| Feb '06 | 8.4 | 6.16 | 0.09 | 0.7 | -1.7 | -2.1 | -2.5 | -3.6 |
| Aug '06 | 8.7 | 4.39 | 0.14 | 0.6 | -1.9 | -2.4 | -2.9 | -4.1 |
| **0.5 $ET_p$, 0-15 cm** | | | | | | | | |
| Jul '03 | 7.5 | - | - | - | - | - | - | - |
| Aug '04 | 8.2 | 4 | 0.16 | 0.6 | -1.7 | -2.1 | -2.5 | -3.6 |
| Feb '05 | 8.3 | 2 | 0.16 | 0.6 | -1.7 | -2.2 | -2.6 | -3.7 |
| Aug '05 | 8.3 | 3.42 | 0.09 | 0.7 | -1.6 | -2.0 | -2.4 | -3.5 |
| Feb '06 | 8.2 | 4.35 | 0.25 | 0.6 | -1.8 | -2.2 | -2.7 | -3.7 |
| Aug '06 | 8.4 | 3.69 | 0.20 | 0.6 | -1.8 | -2.3 | -2.8 | -3.9 |
| **2.0 $ET_p$, 15-30 cm** | | | | | | | | |
| Jul '03 | 7.6 | - | - | - | - | - | - | - |
| Aug '04 | 7.9 | 3 | 0.22 | 0.6 | -1.6 | -2.0 | -2.3 | -3.4 |
| Feb '05 | 8.2 | 3 | 0.07 | 0.7 | -1.5 | -1.9 | -2.3 | -3.3 |
| Aug '05 | 8.2 | 4 | 0.09 | 0.7 | -1.6 | -1.9 | -2.3 | -3.4 |
| Feb '06 | 8.3 | 7 | 0.11 | 0.6 | -1.7 | -2.1 | -2.5 | -3.6 |
| Aug '06 | 8.5 | 3 | 0.17 | 0.6 | -1.8 | -2.3 | -2.8 | -3.9 |
| **0.5 $ET_p$, 15-30 cm** | | | | | | | | |
| Jul '03 | 7.6 | - | - | - | - | - | - | - |
| Aug '04 | 8.0 | 2 | 0.12 | 0.6 | -1.6 | -1.9 | -2.3 | -3.3 |
| Feb '05 | 8.2 | 3 | 0.09 | 0.7 | -1.6 | -2.0 | -2.4 | -3.4 |
| Aug '05 | 8.2 | 4 | 0.08 | 0.7 | -1.6 | -1.9 | -2.3 | -3.4 |
| Feb '06 | 8.2 | 8 | 0.14 | 0.6 | -1.7 | -2.1 | -2.5 | -3.6 |
| Aug '06 | 8.4 | 3 | 0.23 | 0.6 | -1.9 | -2.4 | -2.8 | -3.9 |

‡ Assumed that free $Ca^{2+}$ is 15% of the total $Ca^{2+}$ activity and 45% inorganic P in the soil solution.

Table 3.7. Calculated phosphate potential‡ for various calcium-phosphate compounds that may have formed in a soil planted to tropical grasses and irrigated with dairy effluent, Waianae, O'ahu, Hawai'i.

| Irrigation rate/ Sampling Depth | Soil pH | Solution EC | Ca Activity | MCP | DCPD | OCP | TCP | HA |
|---|---|---|---|---|---|---|---|---|
| | | $dS\ m^{-1}$ | *mM* | log $H_2PO_4^-$ | | | | |
| **2.0 $ET_p$, 0-15 cm** | | | | | | | | |
| Jul '03 | 7.5 | - | - | - | - | - | - | - |
| Aug '04 | 8.2 | 3.3 | 0.16 | 0.9 | -2.6 | -3.3 | -3.9 | -5.6 |
| Feb '05 | 8.4 | 3.9 | 0.10 | 1.0 | -2.6 | -3.3 | -4.0 | -5.7 |
| Aug '05 | 8.3 | 4.3 | 0.08 | 1.0 | -2.5 | -3.1 | -3.8 | -5.4 |
| Feb '06 | 8.4 | 6.2 | 0.09 | 1.0 | -2.6 | -3.3 | -3.9 | -5.6 |
| Aug '06 | 8.7 | 4.4 | 0.14 | 0.9 | -2.9 | -3.8 | -4.6 | -6.3 |
| **0.5 $ET_p$, 0-15 cm** | | | | | | | | |
| Jul '03 | 7.5 | - | - | - | - | - | - | - |
| Aug '04 | 8.2 | 3.8 | 0.16 | 0.9 | -2.7 | -3.3 | -3.9 | -5.6 |
| Feb '05 | 8.3 | 2.3 | 0.16 | 0.9 | -2.7 | -3.4 | -4.1 | -5.7 |
| Aug '05 | 8.3 | 3.4 | 0.09 | 1.0 | -2.5 | -3.2 | -3.8 | -5.5 |
| Feb '06 | 8.2 | 4.4 | 0.25 | 0.9 | -2.8 | -3.5 | -4.1 | -5.8 |
| Aug '06 | 8.4 | 3.7 | 0.20 | 0.9 | -2.8 | -3.6 | -4.3 | -6.0 |
| **2.0 $ET_p$, 15-30 cm** | | | | | | | | |
| Jul '03 | 7.6 | - | - | - | - | - | - | - |
| Aug '04 | 7.9 | 2.9 | 0.22 | 0.9 | -2.5 | -3.1 | -3.7 | -5.3 |
| Feb '05 | 8.2 | 3.3 | 0.07 | 1.1 | -2.4 | -2.9 | -3.5 | -5.1 |
| Aug '05 | 8.2 | 3.6 | 0.09 | 1.0 | -2.5 | -3.0 | -3.6 | -5.3 |
| Feb '06 | 8.3 | 7.1 | 0.11 | 1.0 | -2.6 | -3.2 | -3.9 | -5.6 |
| Aug '06 | 8.5 | 3.3 | 0.17 | 0.9 | -2.9 | -3.7 | -4.4 | -6.1 |
| **0.5 $ET_p$, 15-30 cm** | | | | | | | | |
| Jul '03 | 7.6 | - | - | - | - | - | - | - |
| Aug '04 | 8.0 | 1.9 | 0.12 | 1.0 | -2.4 | -2.9 | -3.5 | -5.1 |
| Feb '05 | 8.2 | 2.9 | 0.09 | 1.0 | -2.5 | -3.0 | -3.7 | -5.3 |
| Aug '05 | 8.2 | 3.7 | 0.08 | 1.0 | -2.4 | -3.0 | -3.6 | -5.2 |
| Feb '06 | 8.2 | 7.7 | 0.14 | 0.9 | -2.6 | -3.2 | -3.9 | -5.5 |
| Aug '06 | 8.4 | 3.4 | 0.23 | 0.9 | -2.9 | -3.7 | -4.4 | -6.1 |

‡Assumed that free $Ca^{2+}$ is 15% of the total $Ca^{2+}$ activity and 70% inorganic P in the soil solution.

**Table 3.8. Calculated phosphate potential‡ for various calcium-phosphate compounds that may have formed in a soil planted to tropical grasses and irrigated with dairy effluent, Waianae, O‘ahu, Hawai‘i.**

| Irrigation rate/ Sampling Depth | Soil pH | Solution EC | Ca Activity | MCP | DCPD | OCP | TCP | HA |
|---|---|---|---|---|---|---|---|---|
| | | $dS\ m^{-1}$ | *mM* | log $H_2PO_4^-$ | | | | |
| **2.0 $ET_p$, 0-15 cm** | | | | | | | | |
| Jul '03 | 7.5 | - | - | - | - | - | - | - |
| Aug '04 | 8.2 | 3.3 | 0.21 | 0.6 | -2.0 | -2.2 | -2.6 | -3.7 |
| Feb '05 | 8.4 | 3.9 | 0.13 | 0.6 | -2.0 | -2.2 | -2.6 | -3.7 |
| Aug '05 | 8.3 | 4.3 | 0.11 | 0.6 | -2.0 | -2.1 | -2.5 | -3.6 |
| Feb '06 | 8.4 | 6.2 | 0.12 | 0.6 | -2.0 | -2.2 | -2.6 | -3.7 |
| Aug '06 | 8.7 | 4.4 | 0.19 | 0.6 | -2.2 | -2.5 | -3.0 | -4.2 |
| **0.5 $ET_p$, 0-15 cm** | | | | | | | | |
| Jul '03 | 7.5 | - | - | - | - | - | - | - |
| Aug '04 | 8.2 | 3.8 | 0.22 | 0.6 | -2.0 | -2.2 | -2.6 | -3.7 |
| Feb '05 | 8.3 | 2.3 | 0.21 | 0.6 | -2.1 | -2.3 | -2.7 | -3.8 |
| Aug '05 | 8.3 | 3.4 | 0.12 | 0.6 | -2.0 | -2.1 | -2.5 | -3.6 |
| Feb '06 | 8.2 | 4.4 | 0.34 | 0.5 | -2.1 | -2.3 | -2.7 | -3.8 |
| Aug '06 | 8.4 | 3.7 | 0.26 | 0.5 | -2.2 | -2.4 | -2.9 | -4.0 |
| **2.0 $ET_p$, 15-30 cm** | | | | | | | | |
| Jul '03 | 7.6 | - | - | - | - | - | - | - |
| Aug '04 | 7.9 | 2.9 | 0.29 | 0.5 | -2.0 | -2.0 | -2.4 | -3.5 |
| Feb '05 | 8.2 | 3.3 | 0.09 | 0.6 | -1.9 | -1.9 | -2.3 | -3.4 |
| Aug '05 | 8.2 | 3.6 | 0.11 | 0.6 | -1.9 | -2.0 | -2.4 | -3.5 |
| Feb '06 | 8.3 | 7.1 | 0.14 | 0.6 | -2.0 | -2.2 | -2.6 | -3.7 |
| Aug '06 | 8.5 | 3.3 | 0.22 | 0.6 | -2.2 | -2.4 | -2.9 | -4.0 |
| **0.5 $ET_p$, 15-30 cm** | | | | | | | | |
| Jul '03 | 7.6 | - | - | - | - | - | - | - |
| Aug '04 | 8.0 | 1.9 | 0.16 | 0.6 | -1.9 | -2.0 | -2.4 | -3.4 |
| Feb '05 | 8.2 | 2.9 | 0.12 | 0.6 | -1.9 | -2.0 | -2.4 | -3.5 |
| Aug '05 | 8.2 | 3.7 | 0.10 | 0.6 | -1.9 | -2.0 | -2.4 | -3.5 |
| Feb '06 | 8.2 | 7.7 | 0.19 | 0.6 | -2.0 | -2.2 | -2.6 | -3.7 |
| Aug '06 | 8.4 | 3.4 | 0.31 | 0.5 | -2.2 | -2.4 | -2.9 | -4.0 |

‡Assumed that free $Ca^{2+}$ is 20% of the total $Ca^{2+}$ activity and 45% inorganic P in the soil solution.

Table 3.9. Calculated phosphate potential‡ for various calcium-phosphate compounds that may have formed in a soil planted to tropical grasses and irrigated with dairy effluent, Waianae, O'ahu, Hawai'i.

| Irrigation rate/ Sampling Depth | Soil pH | Solution EC | Ca Activity | MCP | DCPD | OCP | TCP | HA |
|---|---|---|---|---|---|---|---|---|
| | | $dS\ m^{-1}$ | *mM* | log $H_2PO_4^-$ | | | | |
| **2.0 $ET_p$, 0-15 cm** | | | | | | | | |
| Jul '03 | 7.5 | - | - | - | - | - | - | - |
| Aug '04 | 8.2 | 3.3 | 0.21 | 0.9 | -3.2 | -3.4 | -4.1 | -5.7 |
| Feb '05 | 8.4 | 3.9 | 0.13 | 1.0 | -3.2 | -3.4 | -4.1 | -5.8 |
| Aug '05 | 8.3 | 4.3 | 0.11 | 1.0 | -3.0 | -3.2 | -3.9 | -5.6 |
| Feb '06 | 8.4 | 6.2 | 0.12 | 1.0 | -3.1 | -3.4 | -4.1 | -5.8 |
| Aug '06 | 8.7 | 4.4 | 0.19 | 0.9 | -3.5 | -3.9 | -4.7 | -6.5 |
| **0.5 $ET_p$, 0-15 cm** | | | | | | | | |
| Jul '03 | 7.5 | - | - | - | - | - | - | - |
| Aug '04 | 8.2 | 3.8 | 0.22 | 0.9 | -3.2 | -3.4 | -4.1 | -5.8 |
| Feb '05 | 8.3 | 2.3 | 0.21 | 0.9 | -3.2 | -3.5 | -4.2 | -5.9 |
| Aug '05 | 8.3 | 3.4 | 0.12 | 1.0 | -3.1 | -3.3 | -3.9 | -5.6 |
| Feb '06 | 8.2 | 4.4 | 0.34 | 0.8 | -3.3 | -3.6 | -4.3 | -6.0 |
| Aug '06 | 8.4 | 3.7 | 0.26 | 0.8 | -3.4 | -3.7 | -4.4 | -6.2 |
| **2.0 $ET_p$, 15-30 cm** | | | | | | | | |
| Jul '03 | 7.6 | - | - | - | - | - | - | - |
| Aug '04 | 7.9 | 2.9 | 0.29 | 0.8 | -3.1 | -3.2 | -3.8 | -5.4 |
| Feb '05 | 8.2 | 3.3 | 0.09 | 1.0 | -2.9 | -3.0 | -3.6 | -5.3 |
| Aug '05 | 8.2 | 3.6 | 0.11 | 1.0 | -3.0 | -3.1 | -3.8 | -5.4 |
| Feb '06 | 8.3 | 7.1 | 0.14 | 0.9 | -3.1 | -3.4 | -4.0 | -5.7 |
| Aug '06 | 8.5 | 3.3 | 0.22 | 0.9 | -3.4 | -3.8 | -4.5 | -6.3 |
| **0.5 $ET_p$, 15-30 cm** | | | | | | | | |
| Jul '03 | 7.6 | - | - | - | - | - | - | - |
| Aug '04 | 8.0 | 1.9 | 0.16 | 0.9 | -2.9 | -3.1 | -3.7 | -5.3 |
| Feb '05 | 8.2 | 2.9 | 0.12 | 1.0 | -3.0 | -3.2 | -3.8 | -5.4 |
| Aug '05 | 8.2 | 3.7 | 0.10 | 1.0 | -3.0 | -3.1 | -3.7 | -5.4 |
| Feb '06 | 8.2 | 7.7 | 0.19 | 0.9 | -3.1 | -3.4 | -4.0 | -5.7 |
| Aug '06 | 8.4 | 3.4 | 0.31 | 0.8 | -3.4 | -3.8 | -4.5 | -6.3 |

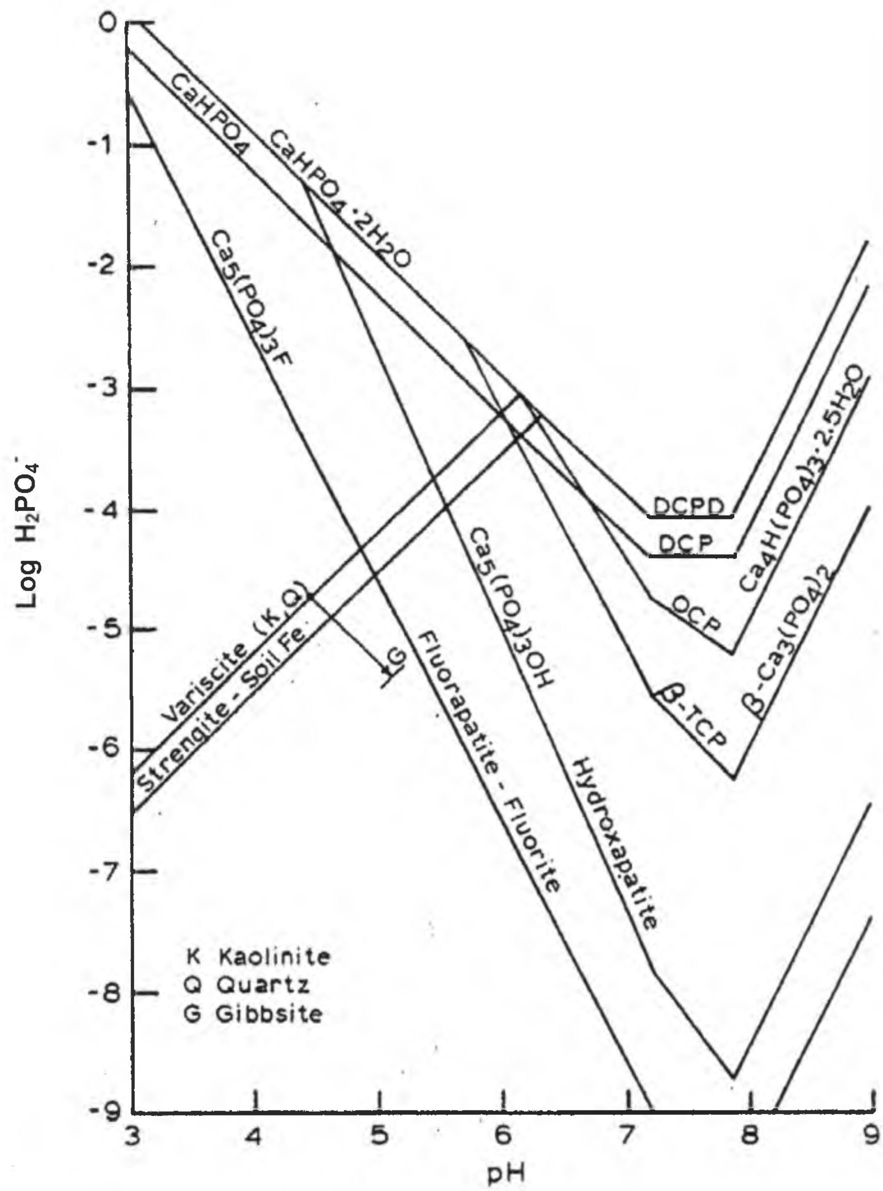
‡Assumed that free $Ca^{2+}$ is 20% of the total $Ca^{2+}$ activity and 70% inorganic P in the soil solution.

DCPD (-2.8 to -2.9)

OCP (-3.6 to -3.8)

TCP (-4.3 to -4.6)

HA (-6.0 to -6.3)




Soil pH, 0-15 cm = 8.7†, 8.4††
15-30cm = 8.5†, 8.4††

Source of diagram: Lindsay, 1979

†2.0 $ET_p$ (7 to 44 mm $d^{-1}$)
††0.5 $ET_p$ (2 to 11 mm $d^{-1}$)

**Figure 3.21. Solubility diagram for various calcium phosphate compounds showing the calculated phosphate potential (assuming 15% ionic Ca and 70% inorganic P) for soils collected at the 0 to 15 cm and 15 to 30 cm depths in Aug 2006 from effluent-irrigated plots (0.5 $ET_p$ and 2.0 $ET_p$), Waianae, O'ahu, Hawai'i.**

**Table 3.10. Calculated phosphate potential‡ for various calcium-phosphate compounds that may have formed in a soil planted to tropical grasses and irrigated with dairy effluent, Waianae, O'ahu, Hawai'i.**

| Irrigation rate/ Sampling Depth | Soil pH | Solution EC | Ca Activity | MCP | DCPD | OCP | TCP | HA |
|---|---|---|---|---|---|---|---|---|
| | | $dS\ m^{-1}$ | *mM* | log $H_2PO_4^-$ | | | | |
| **2.0 $ET_p$, 0-15 cm** | | | | | | | | |
| Jul '03 | 7.5 | - | - | - | - | - | - | - |
| Aug '04 | 8.2 | 3.3 | 0.16 | 1.3 | -3.8 | -4.7 | -5.6 | -8.0 |
| Feb '05 | 8.4 | 3.9 | 0.10 | 1.4 | -3.7 | -4.7 | -5.7 | -8.1 |
| Aug '05 | 8.3 | 4.3 | 0.08 | 1.5 | -3.6 | -4.5 | -5.4 | -7.7 |
| Feb '06 | 8.4 | 6.2 | 0.09 | 1.5 | -3.7 | -4.7 | -5.6 | -8.0 |
| Aug '06 | 8.7 | 4.4 | 0.14 | 1.4 | -4.2 | -5.4 | -6.5 | -9.1 |
| **0.5 $ET_p$, 0-15 cm** | | | | | | | | |
| Jul '03 | 7.5 | - | - | - | - | - | - | - |
| Aug '04 | 8.2 | 3.8 | 0.16 | 1.3 | -3.8 | -4.7 | -5.6 | -8.0 |
| Feb '05 | 8.3 | 2.3 | 0.16 | 1.3 | -3.9 | -4.8 | -5.8 | -8.2 |
| Aug '05 | 8.3 | 3.4 | 0.09 | 1.5 | -3.6 | -4.5 | -5.4 | -7.8 |
| Feb '06 | 8.2 | 4.4 | 0.25 | 1.2 | -4.0 | -4.9 | -5.9 | -8.3 |
| Aug '06 | 8.4 | 3.7 | 0.20 | 1.3 | -4.1 | -5.1 | -6.2 | -8.6 |
| **2.0 $ET_p$, 15-30 cm** | | | | | | | | |
| Jul '03 | 7.6 | - | - | - | - | - | - | - |
| Aug '04 | 7.9 | 2.9 | 0.22 | 1.3 | -3.6 | -4.4 | -5.2 | -7.5 |
| Feb '05 | 8.2 | 3.3 | 0.07 | 1.5 | -3.4 | -4.2 | -5.0 | -7.3 |
| Aug '05 | 8.2 | 3.6 | 0.09 | 1.5 | -3.5 | -4.3 | -5.2 | -7.5 |
| Feb '06 | 8.3 | 7.1 | 0.11 | 1.4 | -3.7 | -4.6 | -5.6 | -7.9 |
| Aug '06 | 8.5 | 3.3 | 0.17 | 1.3 | -4.1 | -5.2 | -6.3 | -8.7 |
| **0.5 $ET_p$, 15-30 cm** | | | | | | | | |
| Jul '03 | 7.6 | - | - | - | - | - | - | - |
| Aug '04 | 8.0 | 1.9 | 0.12 | 1.4 | -3.5 | -4.2 | -5.1 | -7.3 |
| Feb '05 | 8.2 | 2.9 | 0.09 | 1.4 | -3.5 | -4.4 | -5.2 | -7.6 |
| Aug '05 | 8.2 | 3.7 | 0.08 | 1.5 | -3.5 | -4.3 | -5.1 | -7.5 |
| Feb '06 | 8.2 | 7.7 | 0.14 | 1.3 | -3.7 | -4.6 | -5.5 | -7.9 |
| Aug '06 | 8.4 | 3.4 | 0.23 | 1.2 | -4.1 | -5.2 | -6.3 | -8.7 |

‡Assumed that free $Ca^{2+}$ is 15% of the total Ca activity and 100% inorganic P in the soil solution.

Table 3. 11. Calculated phosphate potential‡ for various calcium-phosphate compounds that may have formed in a soil planted to tropical grasses and irrigated with dairy effluent, Waianae, O'ahu, Hawai'i.

| Irrigation rate/ Sampling Depth | Soil pH | Solution EC | Ca Activity | MCP | DCPD | OCP | TCP | HA |
|---|---|---|---|---|---|---|---|---|
| | | $dS\ m^{-1}$ | *mM* | ------------------ log $H_2PO_4^-$ ------------------ | | | | |
| **2.0 $ET_p$, 0-15 cm** | | | | | | | | |
| Jul '03 | 7.5 | - | - | - | - | - | - | - |
| Aug '04 | 8.2 | 3.3 | 0.21 | 1.3 | -3.9 | -4.8 | -5.8 | -8.2 |
| Feb '05 | 8.4 | 3.9 | 0.13 | 1.4 | -3.9 | -4.9 | -5.9 | -8.3 |
| Aug '05 | 8.3 | 4.3 | 0.11 | 1.4 | -3.7 | -4.6 | -5.6 | -7.9 |
| Feb '06 | 8.4 | 6.2 | 0.12 | 1.4 | -3.8 | -4.8 | -5.8 | -8.2 |
| Aug '06 | 8.7 | 4.4 | 0.19 | 1.3 | -4.3 | -5.6 | -6.7 | -9.3 |
| **0.5 $ET_p$, 0-15 cm** | | | | | | | | |
| Jul '03 | 7.5 | - | - | - | - | - | - | - |
| Aug '04 | 8.2 | 3.8 | 0.22 | 1.3 | -3.9 | -4.9 | -5.8 | -8.2 |
| Feb '05 | 8.3 | 2.3 | 0.21 | 1.3 | -4.0 | -5.0 | -6.0 | -8.4 |
| Aug '05 | 8.3 | 3.4 | 0.12 | 1.4 | -3.7 | -4.7 | -5.6 | -8.0 |
| Feb '06 | 8.2 | 4.4 | 0.34 | 1.2 | -4.1 | -5.1 | -6.1 | -8.5 |
| Aug '06 | 8.4 | 3.7 | 0.26 | 1.2 | -4.2 | -5.3 | -6.3 | -8.8 |
| **2.0 $ET_p$, 15-30 cm** | | | | | | | | |
| Jul '03 | 7.6 | - | - | - | - | - | - | - |
| Aug '04 | 7.9 | 2.9 | 0.29 | 1.2 | -3.7 | -4.5 | -5.4 | -7.7 |
| Feb '05 | 8.2 | 3.3 | 0.09 | 1.4 | -3.5 | -4.3 | -5.2 | -7.5 |
| Aug '05 | 8.2 | 3.6 | 0.11 | 1.4 | -3.6 | -4.5 | -5.4 | -7.7 |
| Feb '06 | 8.3 | 7.1 | 0.14 | 1.3 | -3.8 | -4.8 | -5.7 | -8.1 |
| Aug '06 | 8.5 | 3.3 | 0.22 | 1.2 | -4.2 | -5.4 | -6.4 | -8.9 |
| **0.5 $ET_p$, 15-30 cm** | | | | | | | | |
| Jul '03 | 7.6 | - | - | - | - | - | - | - |
| Aug '04 | 8.0 | 1.9 | 0.16 | 1.3 | -3.6 | -4.4 | -5.2 | -7.5 |
| Feb '05 | 8.2 | 2.9 | 0.12 | 1.4 | -3.7 | -4.5 | -5.4 | -7.8 |
| Aug '05 | 8.2 | 3.7 | 0.10 | 1.4 | -3.6 | -4.4 | -5.3 | -7.7 |
| Feb '06 | 8.2 | 7.7 | 0.19 | 1.3 | -3.9 | -4.8 | -5.7 | -8.1 |
| Aug '06 | 8.4 | 3.4 | 0.31 | 1.2 | -4.3 | -5.4 | -6.4 | -8.9 |

‡Assumed that free $Ca^{2+}$ is 20% of the total Ca activity and 100% inorganic P in the soil solution.

Table 3.12. Calculated phosphate potential‡ for various calcium-phosphate compounds that may have formed in a soil planted to tropical grasses and irrigated with dairy effluent, Waianae, O'ahu, Hawai'i.

| Irrigation rate/ Sampling Depth | Soil pH | Solution EC | Ca Activity | MCP | DCPD | OCP | TCP | HA |
|---|---|---|---|---|---|---|---|---|
| | | $dS\ m^{-1}$ | *mM* | log $H_2PO_4^-$ | | | | |
| **2.0 $ET_p$, 0-15 cm** | | | | | | | | |
| Jul '03 | 7.5 | - | - | - | - | - | - | - |
| Aug '04 | 8.2 | 3.3 | 0.26 | 1.2 | -3.8 | -5.0 | -5.9 | -8.3 |
| Feb '05 | 8.4 | 3.9 | 0.16 | 1.3 | -3.7 | -5.0 | -6.0 | -8.4 |
| Aug '05 | 8.3 | 4.3 | 0.14 | 1.4 | -3.6 | -4.8 | -5.7 | -8.1 |
| Feb '06 | 8.4 | 6.2 | 0.14 | 1.3 | -3.7 | -5.0 | -5.9 | -8.4 |
| Aug '06 | 8.7 | 4.4 | 0.23 | 1.2 | -4.2 | -5.7 | -6.9 | -9.4 |
| **0.5 $ET_p$, 0-15 cm** | | | | | | | | |
| Jul '03 | 7.5 | - | - | - | - | - | - | - |
| Aug '04 | 8.2 | 3.8 | 0.27 | 1.2 | -3.8 | -5.0 | -6.0 | -8.4 |
| Feb '05 | 8.3 | 2.3 | 0.26 | 1.2 | -3.9 | -5.1 | -6.1 | -8.6 |
| Aug '05 | 8.3 | 3.4 | 0.15 | 1.3 | -3.6 | -4.8 | -5.8 | -8.2 |
| Feb '06 | 8.2 | 4.4 | 0.42 | 1.1 | -4.0 | -5.2 | -6.2 | -8.7 |
| Aug '06 | 8.4 | 3.7 | 0.33 | 1.2 | -4.1 | -5.4 | -6.5 | -9.0 |
| **2.0 $ET_p$, 15-30 cm** | | | | | | | | |
| Jul '03 | 7.6 | - | - | - | - | - | - | - |
| Aug '04 | 7.9 | 2.9 | 0.36 | 1.1 | -3.6 | -4.7 | -5.5 | -7.9 |
| Feb '05 | 8.2 | 3.3 | 0.11 | 1.4 | -3.4 | -4.5 | -5.3 | -7.7 |
| Aug '05 | 8.2 | 3.6 | 0.14 | 1.3 | -3.5 | -4.6 | -5.5 | -7.9 |
| Feb '06 | 8.3 | 7.1 | 0.18 | 1.3 | -3.7 | -4.9 | -5.9 | -8.3 |
| Aug '06 | 8.5 | 3.3 | 0.28 | 1.2 | -4.1 | -5.5 | -6.6 | -9.1 |
| **0.5 $ET_p$, 15-30 cm** | | | | | | | | |
| Jul '03 | 7.6 | - | - | - | - | - | - | - |
| Aug '04 | 8.0 | 1.9 | 0.20 | 1.3 | -3.5 | -4.5 | -5.4 | -7.7 |
| Feb '05 | 8.2 | 2.9 | 0.15 | 1.3 | -3.5 | -4.7 | -5.6 | -7.9 |
| Aug '05 | 8.2 | 3.7 | 0.13 | 1.4 | -3.5 | -4.6 | -5.5 | -7.8 |
| Feb '06 | 8.2 | 7.7 | 0.24 | 1.2 | -3.7 | -4.9 | -5.9 | -8.3 |
| Aug '06 | 8.4 | 3.4 | 0.39 | 1.1 | -4.1 | -5.5 | -6.6 | -9.1 |

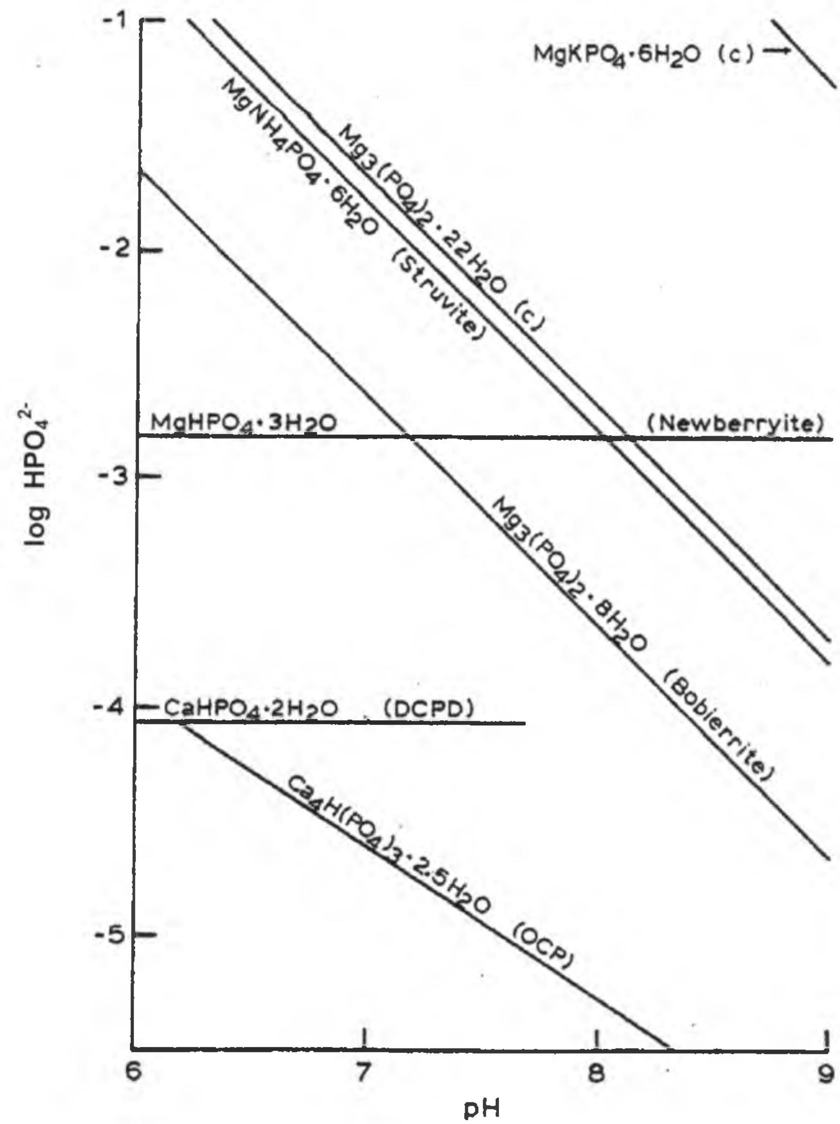
‡Assumed that free $Ca^{2+}$ is 25% of the total Ca activity and 100% inorganic P in the soil solution.

**Table 3.13. Calculated phosphate potential‡ for struvite that may have formed in a soil planted to tropical grasses and irrigated with dairy effluent, Waianae, O'ahu, Hawai'i.**

| Irrigation rate/ Sampling Depth | Soil pH | Solution EC | Ca Activity | Struvite Phosphate Potential | | |
|---|---|---|---|---|---|---|
| | | | | 100% $P_i$ | 70% $P_i$ | 45% $P_i$ |
| | | dS $m^{-1}$ | *mM* | log $HPO_4^{2-}$ | | |
| **2.0 $ET_p$, 0-15 cm** | | | | | | |
| Jul '03 | 7.5 | - | - | - | - | - |
| Aug '04 | 8.2 | 3.3 | 0.26 | -2.6 | -1.8 | -1.2 |
| Feb '05 | 8.4 | 3.9 | 0.16 | -3.8 | -2.6 | -1.7 |
| Aug '05 | 8.3 | 4.3 | 0.14 | -4.1 | -2.9 | -1.9 |
| Feb '06 | 8.4 | 6.2 | 0.14 | -4.8 | -3.4 | -2.2 |
| Aug '06 | 8.7 | 4.4 | 0.23 | -5.9 | -4.1 | -2.7 |
| **0.5 $ET_p$, 0-15 cm** | | | | | | |
| Jul '03 | 7.5 | - | - | - | - | - |
| Aug '04 | 8.2 | 3.8 | 0.27 | -2.0 | -1.4 | -0.9 |
| Feb '05 | 8.3 | 2.3 | 0.26 | -3.0 | -2.1 | -1.3 |
| Aug '05 | 8.3 | 3.4 | 0.15 | -3.9 | -2.7 | -1.8 |
| Feb '06 | 8.2 | 4.4 | 0.42 | -3.8 | -2.6 | -1.7 |
| Aug '06 | 8.4 | 3.7 | 0.33 | -4.8 | -3.4 | -2.2 |
| **2.0 $ET_p$, 15-30 cm** | | | | | | |
| Jul '03 | 7.6 | - | - | - | - | - |
| Aug '04 | 7.9 | 2.9 | 0.36 | -4.3 | -3.0 | -1.9 |
| Feb '05 | 8.2 | 3.3 | 0.11 | -3.9 | -2.7 | -1.8 |
| Aug '05 | 8.2 | 3.6 | 0.14 | -4.7 | -3.3 | -2.1 |
| Feb '06 | 8.3 | 7.1 | 0.18 | -4.5 | -3.1 | -2.0 |
| Aug '06 | 8.5 | 3.3 | 0.28 | -5.3 | -3.7 | -2.4 |
| **0.5 $ET_p$, 15-30 cm** | | | | | | |
| Jul '03 | 7.6 | - | - | - | - | - |
| Aug '04 | 8.0 | 1.9 | -3.2 | -2.3 | -1.5 | -3.2 |
| Feb '05 | 8.2 | 2.9 | -3.3 | -2.3 | -1.5 | -3.3 |
| Aug '05 | 8.2 | 3.7 | -3.8 | -2.6 | -1.7 | -3.8 |
| Feb '06 | 8.2 | 7.7 | -3.6 | -2.5 | -1.6 | -3.6 |
| Aug '06 | 8.4 | 3.4 | -4.6 | -3.2 | -2.1 | -4.6 |

‡Assumed 15% free Mg of the total Mg and 100%, 70% and 45% inorganic P of the total P in the soil solution.

Struvite (-2.7,
-2.4),
-2.2,
-4.6




Soil pH, 0-15 cm = 8.7†, 8.4††
15-30cm = 8.5†, 8.4††

Source of diagram: Lindsay, 1979

†2.0 $ET_p$ (7 to 44 mm $d^{-1}$)
††0.5 $ET_p$ (2 to 11 mm $d^{-1}$)

**Figure 3.22. Solubility diagram for various magnesium phosphate compounds showing the calculated phosphate potential for soils collected at the 0 to 15 cm and 15 to 30 cm depth in Aug 2006 from effluent-irrigated plots, Waianae, O‘ahu, Hawai‘i.**

It is also interesting to note that the P concentrations in tissues of the different grasses were mostly below the usual sufficient levels (Mills and Jones, 1996), despite being given a large cumulative effluent P application over time (Table 3.14). This observation suggested that either the external P requirement of the grasses is low (Sims et al., 2005) or that not all of the P is plant available (Pierzynski et al., 2005). The latter supported the hypothesis that some of the applied P may be precipitated by Ca.

**Table 3.14. Average tissue P concentration, annual P uptake and P removal of tropical grasses irrigated with dairy effluent, Waianae, O'ahu, Hawai'i, Aug 2004 to Aug 2006.**

| Rate | Grass Species | Tissue P Concentration | P Uptake | P Removal‡ |
|---|---|---|---|---|
| | | mg $kg^{-1}$ | kg $ha^{-1}$ $y^{-1}$ | % |
| 2.0 $ET_p$ (7 to 44 mm $d^{-1}$) | *P. purpureum* | 0.30 | 153 | 21 |
| | *B. decumbens*¶ | 0.27 | 21 | 14 |
| | *B. mutica* | 0.29 | 157 | 22 |
| | *C. nlemfuensis* | 0.32 | 124 | 19 |
| | *P. atratum* | 0.21 | 85 | 13 |
| 0.5 $ET_p$ (2 to 11 mm $d^{-1}$) | *P. purpureum* | 0.34 | 171 | 86 |
| | *B. decumbens*¶ | 0.42 | 41 | 77 |
| | *B. mutica* | 0.27 | 116 | 53 |
| | *C. nlemfuensis* | 0.40 | 133 | 65 |
| | *P. atratum* | 0.24 | 98 | 50 |

‡Annual P uptake divided annual cumulative applied P
¶Aug to Nov 2004 growing period only.

## General Discussion

Phosphorus accumulation, particularly on the surface soils, is among the primary concerns when applying effluent because this nutrient could be mobilized by soil erosion or dissolution in runoff water. Repeated application of animal effluent especially at high rates over a number of years is expected to eventually exhaust the P sorption sites and lead to concomitant increases in extractable soil P in the future, especially if plant uptake is low (Sanderson et al., 2001; Johnson et al., 2004; Redding, 2001). Contrary to expectations, the extractable soil P did not increase much over time despite the large cumulative P application through daily irrigation with dairy effluent for a year. This lack of large increases in soil P was ascribed to the already high extractable soil P level in the beginning of the experiment. The site was proximate to an area utilized for composting of solids from drained effluent lagoons. It was also adjacent to the effluent lagoon which was used as the source of irrigation water in the experiment and which probably, occasionally overflowed in the past during the rainy season prior to the implementation of the experiment. At the observed initial soil P level, no additional application of P from any source was recommended (Sharpley and Rekolainen, 1997). However, this soil condition was very common in areas utilized for dairy production. Thus, dairy producers need recommendations for practices that may help reduce the levels of soil P. This is the rationale for using the site in conducting this study.

The extractable soil P values, regardless of the treatment and sampling dates, were higher than the adequate P level (25 to 35 mg $kg^{-1}$) established for heavy soils in Hawai'i (Tamimi et al., 1997). This was expected for a soil with very low capacity to retain P compared with other soil orders (Fox and Searle, 1978). The extractable soil P (Figs. 3.3 and 3.4) in the effluent-irrigated plots was comparable to those reported for Mollisol (Hawi and Waialua) kikuyugrass (*Pennisetum clandestinum*) pastures (110 to 296 mg Modified Truog P $kg^{-1}$) on the Island of Hawai'i receiving dairy waste application (Mathews et al., 2005).

Where soil P accumulates, the likelihood of P being carried by runoff (Sharpley et al., 1994; Hart et al., 2004) or being leached in the soil profile (Simard et al., 2000;

Sharpley et al., 1994) increases. The soil at the site has low P sorption capacity (Figs. 3.10 to 3.13), thus, it is more likely for the applied P from effluent to be redistributed in the soil profile rather than accumulate at the application zone. However, there was a dense clay layer at the 30 to 70 cm zone, that may have restricted the downward movement of water and nutrients at this site.

During the 2-y period of effluent irrigation, the levels of soil solution P (ranging from 1.5 to 11 mg $L^{-1}$) had been mostly below those values reported for soils receiving animal wastes (7 to 8 mg P $L^{-1}$) (Pierzynski et al., 2005). These levels of soil solution P may still be high from an environmental perspective because the freshwater P concentration that may cause eutrophication ranges from 0.003 to 0.3 mg $L^{-1}$ (Pierzynski et al., 2005). On an agronomic perspective, however, although many grasses have low external P requirement (Fox, 1981), some species require higher P in soil solution for optimum growth (Pant et al., 2004). An example is the kikuyu grass in soils in Hawai‘i which requires >1 mg P $L^{-1}$ (Fox, 1969) compared to 0.2 mg P $L^{-1}$ considered as adequate for most crops (Fox, 1981). Also, recent findings on a short-term study on the external P requirement of leafy vegetables indicated that the extractable soil P (Modified Truog values) observed in this study was not excessive (J. Deenik, 2006, pers. comm.).

Although scientists usually employ a balance (input-output analysis) method for assessing the acceptability of a certain practice, this strategy does not seem to apply in this soil system. Despite the large amount of effluent P applied (620 kg P $ha^{-1}$ $y^{-1}$), not all of this P was accounted for by the flux and plant uptake. The calculated P flux in the soil profile may be relatively high (99 to 131 kg P $ha^{-1}$ $y^{-1}$), yet, its impact on the groundwater will depend on the vulnerability of the aquifer. In our case, this amount of soil solution P may not pose a serious threat to the groundwater, which is located 40 to 80 m below the surface mean sea level (R. Whittier, 2006, pers. comm.).

The change in the availability of micronutrients is one of the important impacts of effluent application. Given the relatively high extractable P in the soil, the possibility of pronounced antagonistic interactions of Zn with P (Alloway, 2004) may not be ruled out. The growth of the grasses improved and micronutrient concentrations in the forage tissues increased to acceptable levels when given supplemental fertilization with micronutrients such as Zn, Fe and Mn, in sulfate forms (Table 3.15). Recent results suggest that Cu may also become deficient. Micronutrient deficiency may also have been a consequence of the high soil pH, which resulted in their low plant availability (Moraghan and Mascagni, 1991). In addition, dairy effluent supplied only minimal quantities of micronutrients (Fig. 3.23).

**Table 3.15. Average micronutrient concentration in the tissues of grasses irrigated with dairy effluent, Waianae, O'ahu, Hawai'i.**

| Irrigation Rate | Grass Species | Tissue Micronutrient Concentration‡ | | | | |
|---|---|---|---|---|---|---|
| | | Fe | Mn | Zn | Cu | B |
| | | mg $kg^{-1}$ | | | | |
| 2.0 $ET_p$ (7 to 44 mm $d^{-1}$) | *P. purpureum* | 416 | 56 | 68 | 13 | 20 |
| | *B. decumbens*¶ | 1493 | 89 | 21 | 10 | 21 |
| | *B. mutica* | 207 | 81 | 66 | 11 | 15 |
| | *C. nlemfuensis* | 405 | 105 | 53 | 9 | 11 |
| | *P. atratum* | 1297 | 111 | 86 | 11 | 32 |
| 0.5 $ET_p$ (2 to 11 mm $d^{-1}$) | *P. purpureum* | 526 | 51 | 56 | 10 | 15 |
| | *B. decumbens*¶ | 1489 | 91 | 36 | 12 | 13 |
| | *B. mutica* | 266 | 64 | 62 | 10 | 12 |
| | *C. nlemfuensis* | 387 | 66 | 56 | 11 | 12 |
| | *P. atratum* | 1108 | 98 | 96 | 10 | 25 |

‡Average of two years

¶Aug to Nov 2004 growing period.

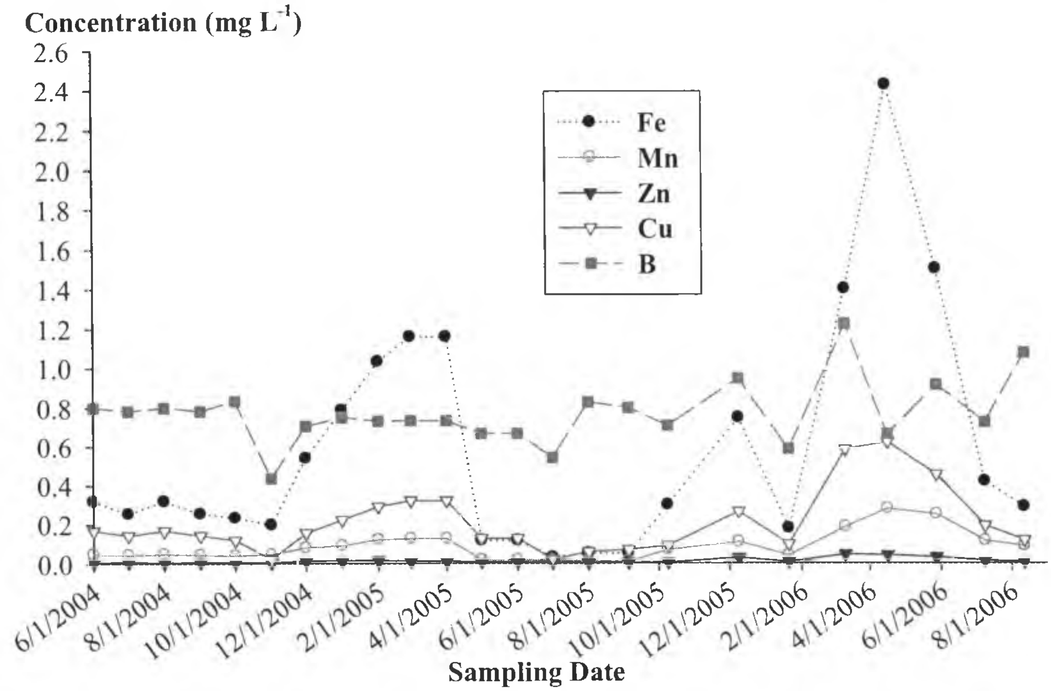



**Figure 3.23. Micronutrient concentration of dairy effluent that was used to irrigate the tropical grasses, Waianae, O'ahu, Hawai'i..**

## Conclusions

Effluent application in this type of soil at the 2.0 $ET_p$ rate appears to be acceptable as long as there are no adverse impacts of excessive nutrient accumulation and leaching through the soil profile. No considerable increases in extractable soil P and soil solution P over time were observed relative to the already high initial level of soil P and especially with the high biomass productivity of effluent-irrigated grasses. The high Ca content and pH of both the soil and the effluent may have resulted in the precipitation of P with Ca, possibly as hydroxyapatite, as indicated by the higher extractable soil P with the use of acidic extractants. The implication of Ca-P formation is the reduced pollution potential of effluent irrigation. The duration of effluent application was relatively short, thus, additional monitoring is necessary to determine the longer-term impacts of dairy effluent application on plant and soil properties, especially P. Specifically, longer-term monitoring is needed to ensure that P will not accumulate in the surface soil to become a pollutant to associated water bodies. Also, studies on the impacts of effluent application on other soil (high P sorption capacity, more permeability, low initial salt and P levels) and environmental conditions (higher rainfall, low $ET_p$, lower solar radiation) are needed.

# CHAPTER 4. MONITORING OF SALT ACCUMULATION IN A DAIRY EFFLUENT- IRRIGATED SOIL (MOLLISOL) IN A TROPICAL ISLAND ENVIRONMENT


## Abstract

In Hawai‘i and other island environments, many dairy producers are forced out of business due to the high costs of imported materials and their inability to comply with environmental regulations. Animal wastes and effluents have been identified as a non-point source of pollution. Dairy producers, thus, need an alternative effluent management strategy. Effluent may be used for irrigation to produce forage and recycle the water and nutrients. This paper discusses the effects of effluent irrigation on plant and soil properties, especially salinity and sodicity. Five tropical grasses—bana (*Pennisetum purpureum* S.), California (*Brachiaria mutica* S.), signal (*Brachiaria decumbens* S.), star (*Cynodon nlemfuensis* V.), and suerte (*Paspalum atratum* S.)—were planted in an augmented completely randomized design. Two rates of dairy effluent were applied through a subsurface (20 to 25 cm deep) drip irrigation based on the potential evapotranspiration ($ET_p$) rates at the site (Waianae)—2.0 $ET_p$ (7 to 44 mm $d^{-1}$) and 0.5 $ET_p$ (2 to 11 mm $d^{-1}$). Soil pH remained high over time, possibly due to the high pH and carbonate/bicarbonate content of the effluent. Soil pH increased from 7.4 to 7.8 in July 2003 to 8.2 to 8.9 in Aug 2006, possibly due to the high pH of the applied effluent. The pH of the soil and soil solution fluctuated relatively little over time indicating the high pH buffering capacity of the soil. Soil electrical conductivity of the saturated paste extract ($EC_{spe}$) declined from 18.0 dS $m^{-1}$ in July 2003 to 2.7 dS $m^{-1}$ in Aug 2006. The soil $EC_{spe}$ and soil solution EC values between Aug 2004 and Aug 2006 were less than the US

Salinity Laboratory's critical level for soils classified as saline (4.0 dS $m^{-1}$). Soil ESP declined despite repeated effluent application and means between Aug 2004 and Aug 2006 (ranging from 6.4 to 10.2%) remained lower than the 15% critical ESP value for classifying soils as sodic. The results suggested that effluent irrigation did not lead to excessive salinity increases and sodicity. Additional longer-term monitoring and soil dispersion studies are needed to determine the impacts of of prolonged applications of dairy effluent on plant and soil properties, especially on salinity and sodicity.

## Introduction

In Hawai‘i, the high net influx of nutrients from imported feeds and the consequent accumulation of large quantities of nutrient-rich waste products created an open nutrient cycle in most dairy production systems. The wastewater is usually collected in lagoons to allow the water to evaporate and the nutrients and salts to settle. Properly managed, settling lagoons generally reduce the amount of solids and nutrients/salts in the effluent. However, the nutrients, soluble solids, organic matter and microbiological contaminants contained in the effluent from nearly full lagoons can cause pollution of adjacent land and water bodies. It is crucial to find ways by which these nutrients could be re-utilized. Land application of animal wastes such as effluent onto pasture and in cut-and-carry forage production systems offers potential for recycling of nutrients from this resource while reducing costs associated with waste removal. The water and nutrients contained in the effluent may be effectively and sustainably utilized to improve the nutrient status of soils and to enhance grass productivity and quality. Its application, however, should not result in the contamination of associated water bodies and systems.

With the increased demands for efficient dairy production and resource management, the opportunity to utilize effluents to produce forage to recycle the water and nutrients in the effluent, is very attractive. However, limited information exists regarding effluent management for its sustainable utilization for crop and animal production systems in an island environment. It is therefore important to assess the effects of effluent irrigation on critical soil properties including salinity and sodicity. Irrigating with waters that contain high amounts of salts may result in salt accumulation in the soil and consequently, soil degradation.

Many studies indicated the potential negative effects of excessive salt accumulation from animal manure or effluent application (Assefa et al., 2004; Halliwell et al., 2001; Diez et al., 2001; Balks et al., 1998; Westerman et al., 1987; King et al., 1985; Pratt, 1984). Balks et al. (1998) reported that the ESP values of tree plantation soils irrigated with effluent high in Na, averaged for all depths as $27.4 \pm 3$ for low, $27.6 \pm 1$ for medium, and $15.8 \pm 1$ for the high rate. These ESP values were higher than that predicted from the SAR of the irrigation water used. (Medium rate referred to irrigation at the estimated water use of the plantation less the rainfall, while the low and high rates were 0.5 and 1.7 times, respectively, that of the estimated water use.). Diez et al. (2001) recorded higher electrical conductivity (EC) (2.8 to 5.9 dS $m^{-1}$) and Na (209 to 723 mg $L^{-1}$) levels in the soil solution at high rates of pig slurry application (180 $m^3$ $ha^{-1}$ equivalent to 486 kg N $ha^{-1}$). Westerman et al. (1987) found elevated Na concentrations in soils planted to 'Coastal' bermudagrass and irrigated with swine effluent. Plants also have differential tolerance to salinity (Kotuby-Amacher et al., 1997). Salinity tolerance has been reported for *B. decumbens* (Duke, 1983) and *P. purpureum* (Reynolds, 1995).

Whether animal effluent application would lead to soil degradation is determined by many factors including soil type (Uehara and Gillman, 1981; Rengasamy and Olsson, 1993), soil management practices (Diez et al., 2001; Balks et al., 1998), and effluent quality (Sumner, 1993). For example, many tropical soils such as those in Hawai‘i contain significant amounts of amorphous iron, silicon, aluminum oxides, which act as binding agents or coatings on soil aggregates (Uehara and Gillman, 1981). This prevents sodium-induced dispersion of the highly weathered Hawaiian soils such as Oxisols, which are mostly made up of 1:1 clays (kaolinites) and rich in amorphous oxides as well as some relatively highly weathered soil series of the Andisol (e.g., Honokaa), which consist of highly amorphous materials rich in Fe and Al oxides and organic matter. Soils dominated by 2:1 clays (smectites) do not have the same ability to resist dispersion (So and Aylmore, 1993). The chemical binding and proportion of the negative sites in the soil available for interacting with Na, water and other ions are affected by soil management practices, which induce changes in soil organic matter, soil pH and aggregation of the soil (Rengasamy and Olsson, 1993). Organic matter can also sometimes cause dispersion instead of preventing it, particularly certain organic compounds such as amino acids. In an experiment involving smectitic clay, Nelson et al. (1999) showed that certain organic compounds such as amino acids were present at highest levels in samples of “easily dispersed clay”. Animal effluents potentially contain significant amounts of proteins and amino acids (Muller, 1980).

The sodium absorption ration (SAR) and EC are commonly used to assess the suitability of water/wastewater for irrigation in terms of sodium hazard. In general, irrigation waters with $EC_{spe}$ of $<0.25$ dS m$^{-1}$ has low, between 0.25 to 0.75 dS m$^{-1}$ has medium, 0.75 to 2.25 dS $m^{-1}$ has high, and >2.25 dS $m^{-1}$ has very high salinity hazard (US Salinity Laboratory, 1954). In terms of sodium hazard, irrigation waters with an SAR value of 9.0 and EC of $\leq$0.168 dS $m^{-1}$ is low, EC of 0.168-2.25 dS $m^{-1}$ is moderate, and EC of >2.25 dS $m^{-1}$ is high. Studies showed that when the soil solution EC exceeds 0.3 dS $m^{-1}$, the tendency of the soil to become sodic increases with increasing clay (McNeal et al., 1968), montmorillonite ( McNeal and Coleman, 1966) and bulk density (Frenkel et al., 1978) and decreases with increasing sesquioxides (McNeal et al., 1968).

While the use of treated wastewater and effluent for irrigation has been encouraged in Hawai'i, there is limited information on its impact on soil quality such as salt accumulation. This paper discusses the impacts of dairy effluent application on salinity and sodicity of a soil planted to tropical forage grasses.

## Methodology

Five grass species—bana (*Pennisetum purpureum* S.), California (*Brachiaria mutica* S.), signal (*Brachiaria decumbens* S.), star (*Cynodon nlemfuensis* V.), and suerte (*Paspalum atratum* S.)—that have demonstrated tolerance to saline soil conditions were established in the experimental plots (Cumulic Haplustoll, Pulehu series), each measuring 13.4 $m^2$ (12 ft by 12 ft). Treatments were arranged in an augmented completely randomized design (Federer, 1956). Two rates of dairy effluent drip irrigation were applied daily (between 30 to 52 min) at the subsurface (20 to 25 cm deep) to the plots. Adjustments of irrigation rates were made depending on the rainfall and $ET_p$ that were calculated using the Reference Evapotranspiration Calculation and Software (Ref-ET v. 2.0) which uses the Penman equation in the $ET_p$ calculation (Allen, 2001). The $ET_p$ was calculated from data collected using the Hobo[†] weather station installed at the site. At the 2.0 $ET_p$ irrigation rate, cumulative amounts of applied nutrients were 620 kg P $ha^{-1}$ $y^{-1}$, 2900 kg Ca $ha^{-1}$ $y^{-1}$, 5900 kg Mg $ha^{-1}$ $y^{-1}$, 42000 kg K $ha^{-1}$ $y^{-1}$, and 15000 kg Na $ha^{-1}$ $y^{-1}$. At the 0.5 $ET_p$ rate, cumulative nutrients applied were 190 kg P $ha^{-1}$ $y^{-1}$, 850 kg Ca $ha^{-1}$ $y^{-1}$, 1700 kg Mg $ha^{-1}$ $y^{-1}$, 12000 kg K $ha^{-1}$ $y^{-1}$ and 4300 kg Na $ha^{-1}$ $y^{-1}$.

**Laboratory Analyses**

Soil pH was determined from a saturated paste using a pH meter (Oakton pH 510[†], Deerfield Beach, FL; Fischer Scientific Accumet 815[†], Hampton, NH). The EC was measured from the filtrate of a soil:water mixture (1:2) shaken for 45 min using an EC meter (Mettler Toledo NC 226[†], Columbus, OH). The methodology was a modification of the 1:1 soil-water ratio described by Hue et al. (1997). The saturated paste extract recommended by the US Salinity Laboratory (1954) was not used since no solution was obtained for Mollisol soils using regular filtration methods due to the high water holding capacity of this soil type. Vacuum filtration of the saturated paste extract method was also not practical due to the large numbers of soil samples for EC measurements. Both the 1:1 and 1:2 methods were modifications of the saturated paste extract method (US Salinity Laboratory, 1954) that was used to develop the criteria for classifying salt-affected soils. The conversion of the EC obtained from the 1:1 and 1:2 ratios into saturated paste extract equivalent EC was determined in a separate study. Thirty seven soil samples collected at various times during the 3-y study period, with $EC_{1:1}$ (2003 and 2004 samples) or $EC_{1:2}$ (2005 and 2006 samples) values ranging from 0.47 to 10.00 dS

[†] Reference herein to any specific product, by trade name, trademark, manufacturer, or distributor does not necessarily constitute or imply its endorsement or recommendation by the authors and publishers.

$m^{-1}$, were selected. The EC of the saturated paste (without vacuum filtration) was also obtained. Regression analyses were done to determine the equation for the conversion to saturated paste extract equivalent.

Soil cations such as Ca, K, Mg, and Na were extracted with ammonium acetate (1*M*, pH 7.0) and measured by inductively coupled plasma-atomic emission spectroscopy (ICP-AES) as described by Hue et al. (1997). The Exchangeable Sodium Percentage (ESP) of the soil, was calculated:

$$ESP = \frac{Na^{+}(cmol_c\, kg^{-1})}{\sum(Ca^{2+} + Mg^{2+} + K^{+} + Na^{+}, cmol_c\, kg^{-1})}\ x\,100 \qquad \text{(Eq. 1)}$$

to assess the impact of effluent application on Na accumulation. The Exchangeable Potassium Percentage (EPP) was also computed as an indication of the effect of K on soil.

The effluent Sodium Absorption Ratio (SAR) was calculated:

$$SAR = \frac{Na^{+}\,(mmol_c\ L^{-1})}{\sqrt{\dfrac{Ca^{2+}\,(mmol_c\ L^{-1}) + Mg^{2+}\,(mmol_c\ L^{-1})}{2}}} \qquad \text{(Eq. 2)}$$

The SAR and EC of the dairy effluent have both remained relatively low throughout the study period (3.1 to 5.3 and 2.7 to 4.4 dS $m^{-1}$, respectively), indicating the effluent's suitability for irrigation. Soil solution samples, collected using tension lysimeters, and dairy effluent were analyzed for pH and EC. The total cation concentration ($mmol_c\,L^{-1}$) in the soil solution was estimated by multiplying EC (dS $m^{-1}$) by 10, based on the relationship 1 dS $m^{-1}$ = 10 $mmol_c\,L^{-1}$ (Rhoades et al., 1992). Soil solution macro- and micro-nutrients (P, K, Ca, Mg, Na, Fe, Mn, Zn, Cu and B) were determined using the inductively coupled plasma-atomic emission spectroscopy (ICP-AES) method (Hue et al., 1997).

The leaching fraction (LF) was calculated using the Rhoades equation:

$$LF = \frac{EC_{iw}}{(5 * EC_e) - EC_{iw}} \qquad \text{(Eq. 3)}$$

where $EC_{iw}$ is the electrical conductivity of the irrigation water (effluent), $EC_e$ is the average salinity of the root zone which could be estimated from the electrical conductivity of the extract of saturated soil paste (Rhoades, 1974; Rhoades and Merrill, 1976), and 5 is a value obtained empirically (Tanji and Kielen, 2003). The values used for $EC_e$ were the EC of the drainage water past the root zone (70 cm and 100 cm depths) because no soil samples were collected below 30 cm depth. Assuming that the applied irrigation water mixes completely with the soil moisture, the actual water requirement (AWR) was calculated as:

$$AWR = \frac{ET}{(1 - LF)} \qquad \text{(Eq. 4)}$$

where ET is the evapotranspiration and LF is the leaching fraction (Ayers and Westcot, 1989). The calculations used the $ET_p$ values (average of 7 mm $d^{-1}$, minimum of 5 mm $d^{-1}$ and maximum of 21 mm $d^{-1}$).

The carbonate and bicarbonate contents of the effluent were obtained by titration with 0.01 *M* or 0.02 *N* $H_2SO_4$ (Clesceri et al., 1998) of triplicate samples (100-ml each) of effluent collected on Aug 2004, Sept 2005 and Aug 2006. The carbonate content and total alkalinity in mg $CaCO_3$ $L^{-1}$ equivalent were calculated based on the volume of the acid (0.01 *M* or 0.02 *N* $H_2SO_4$) used to bring the effluent's pH down to 8.3 and 4.3, respectively. The bicarbonate content was estimated as the difference between total alkalinity and carbonate content.

$$\text{Alkalinity}\,(\text{mg CaCO}_3\ \text{L}^{-1}) = \frac{\text{A} * \text{N} * 50000}{\text{ml of sample}} \qquad \text{(Eq. 5)}$$

where A is the mL of the acid used and N is the normality of the acid used.

**Statistical Analysis**

Best subset procedures were conducted using the MINITAB 14.13 (Minitab Inc., 2004) to determine the variables and interactions (grass species, irrigation rate, time and sampling depth) that were the best predictors of the dependent variable. The interaction was limited to second order to minimize the complexity of statistical analysis and interpretation of data. The model with the lowest Mallows $C_p$ value was chosen, and if any two or more models had the same Mallows $C_p$ value, the next criterion was the highest R-squared value. The main and interaction effects of the treatments as well as the dynamics of change in soil and soil solution properties (reduced model) were analyzed with Repeated-Measures ANOVA (compound symmetry was used as covariance structure) using PROC MIXED of SAS 9.1 (SAS Inst., 2004). For main effects and interactions with significant effects on soil and solution properties, least square (LS) means (which adjust for any missing data) were reported. Data were plotted using SigmaPlot 9.0 (SYSTAT Software Inc., 2004) and/or Excel 2000 (Microsoft Corporation, 1985-1999). To facilitate the data analysis and discussion, the "time" variable was used to refer to sampling dates, which in turn, corresponded to varying amounts of effluent irrigation over time at each of the two irrigation rates.

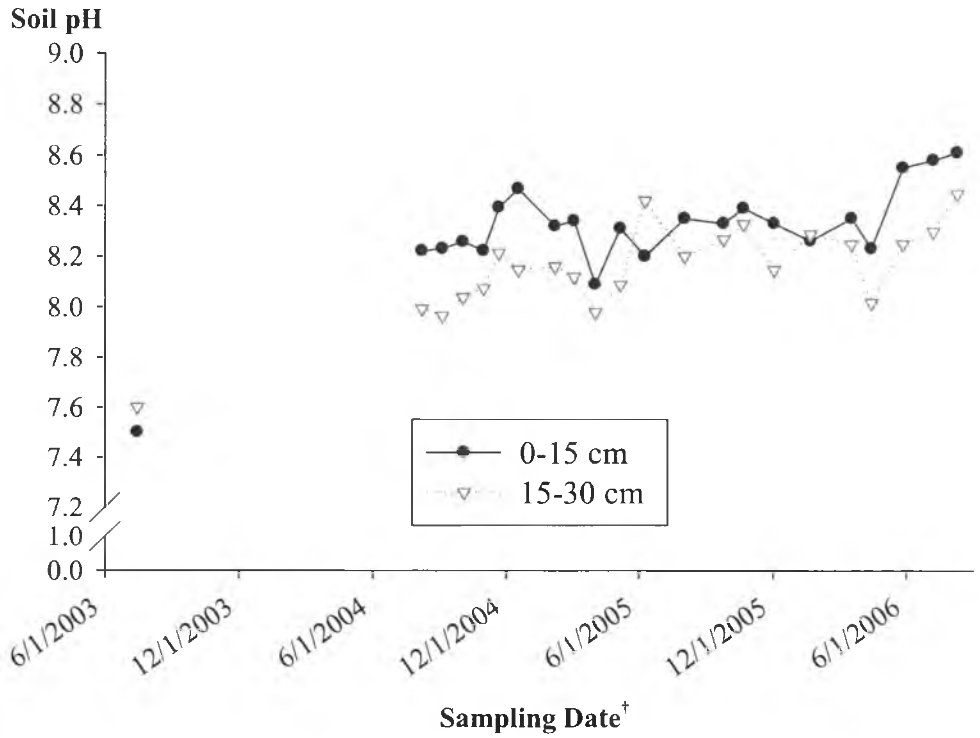
The soil EC data measured using four different methods (1:1, 1:2, saturated paste and saturated paste extract) were analyzed by ANOVA using Proc GLM. Regression analyses were also done in order to predict the approximate EC value for a saturated paste extract from EC measured using other (1:1, 1:2, saturated paste) methods. The regression equation for saturated paste extract vs. 1:1 was used to convert the EC values of 2003 and 2004 soil samples, while the regression equation for saturated paste extract vs. 1:2 was used to convert the EC values of 2005 and 2006 samples. The converted EC values were plotted and statistically analyzed using best subset regression and Repeated-Measures ANOVA of the Proc Mixed in SAS as described above.

## Results and Discussion

### Soil pH

Soil pH increased ($P$<0.01) between July 2003 (ranging from 7.4 to 7.8) and Aug 2004 (ranging from 7.8 to 8.4) (Fig 4.1). In general, the change in soil pH was greater at the 0 to 15 cm depth than at the 15 to 30 cm depth. Soil pH in July 2003 averaged $7.5\pm0.02$ at the 0 to 15 cm depth and $7.6\pm0.06$ at the 15 to 30 cm depth. By Aug 2004, soil pH at the 0 to 15 cm and 15 to 30 cm depths increased to an average of $8.2\pm0.03$ and $8.0\pm0.05$, respectively.

The main effect of time on soil pH between Aug 2004 and Aug 2006 was significant ($P$<0.01), as well as the interactions between sampling depth and time ($P$<0.01), grass ($P$<0.01) and irrigation rate ($P$<0.01) (Table 4.1). Soil pH generally increased over time, but a slightly declining pH was usually observed during the rainy season (Fig. 4.1). Although the initial (July 2003) soil pH was similar at the 15 to 30 cm depth (7.6) and 0 to 15 cm depth (7.5), subsequent samplings between Aug 2004 and Aug 2006 showed that soil pH was generally higher ($P$<0.01) in the 0 to 15 cm depth (average of 8.3$\pm$0.02) than in the 15 to 30 cm depth (average of 8.1$\pm$0.02) (Fig. 4.1). Among the grass species, the plots planted to *B. mutica* had the highest ($P$<0.01) soil pH (8.4$\pm$0.04) at the 0 to 15 cm depth, while the plots planted to *B. mutica* and *C. nlemfuensis* (8.2$\pm$0.04) had the highest soil pH at the 15 to 30 cm depth (Fig. 4.2). Plots irrigated at 2.0 $ET_p$ had higher ($P$<0.01) soil pH at the 0 to 15 cm depth (8.4$\pm$0.02) than at the 15 to 30 cm depth (8.1$\pm$0.02), but plots that received the 0.5 $ET_p$ had similar soil pH (8.3$\pm$0.04 and 8.2$\pm$0.05) at both depths (Fig. 4.3).




†Soil samples were collected before the irrigation was installed in July 2003; the grass establishment period was from December 2003 to July 2004

**Figure 4.1. Soil pH at the 0 to 15 cm and 15 to 30 cm depths of effluent-irrigated soil (Mollisol) planted to tropical grasses, O'ahu, Hawai'i.**

**Table 4.1. Proc Mixed ANOVA‡ statistics showing the Type III test of fixed effects on soil pH in dairy effluent-irrigated soil (Mollisol), Waianae, O'ahu, Hawai'i, Aug 2004 to Aug 2006.**

| Effect | Numerator DF | Denominator DF | F | *P* |
|---|---|---|---|---|
| Time | 20 | 371 | 14.84 | <.0001 |
| Irrigation rate | 1 | 371 | 2.57 | 0.1101 |
| Sampling depth | 1 | 371 | 118.07 | <.0001 |
| Grass x Irrigation rate | 4 | 371 | 2.39 | 0.0501 |
| Grass x Sampling depth | 4 | 371 | 3.85 | 0.0045 |
| Irrigation rate x Sampling depth | 1 | 371 | 39.26 | <.0001 |
| Time x Sampling depth | 20 | 371 | 3.79 | <.0001 |

‡Based on best subset regression results.

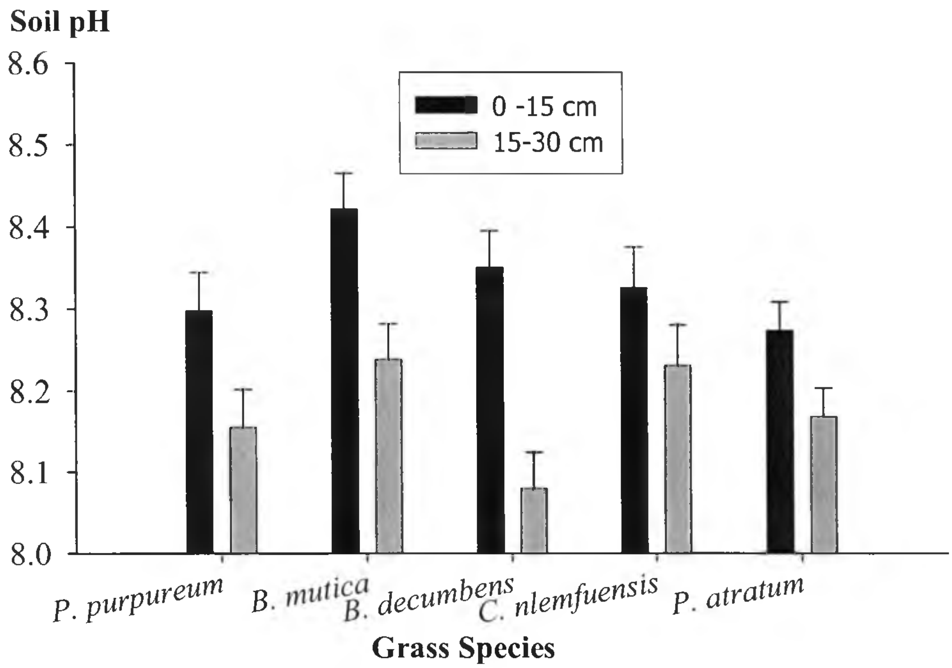



**Figure 4.2. Soil pH by grass species at the 0 to 15 and 15 to 30 cm depths of effluent-irrigated soil (Mollisol) planted to tropical grasses, O'ahu, Hawai'i.**

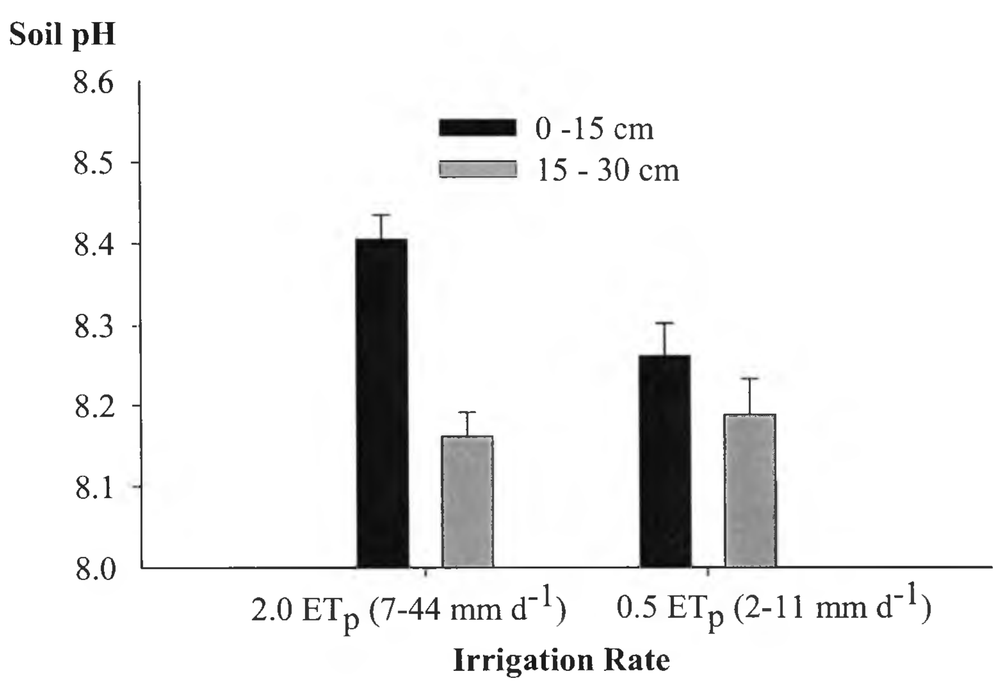



**Figure 4.3. Soil pH at the 0 to 15 cm and 15 to 30 cm depths of effluent-irrigated Mollisol soil (0.5 and 2.0 $ET_p$) planted to tropical grasses, O'ahu, Hawai'i.**

**Soil Solution pH**

Soil solution pH fluctuated between June 2004 to Aug 2006 ($P$<0.01) (Fig. 4.4 and Table 4.2). On average, soil solution pH was 8.4$\pm$0.1 in June 2004, 8.2$\pm$0.1 in July 2005 and 8.5$\pm$0.1 in Aug 2006. Additions of high pH effluent did not result in further increases in soil solution pH suggesting the high pH buffering capacity of the soil and the possible formation of calcium carbonate.

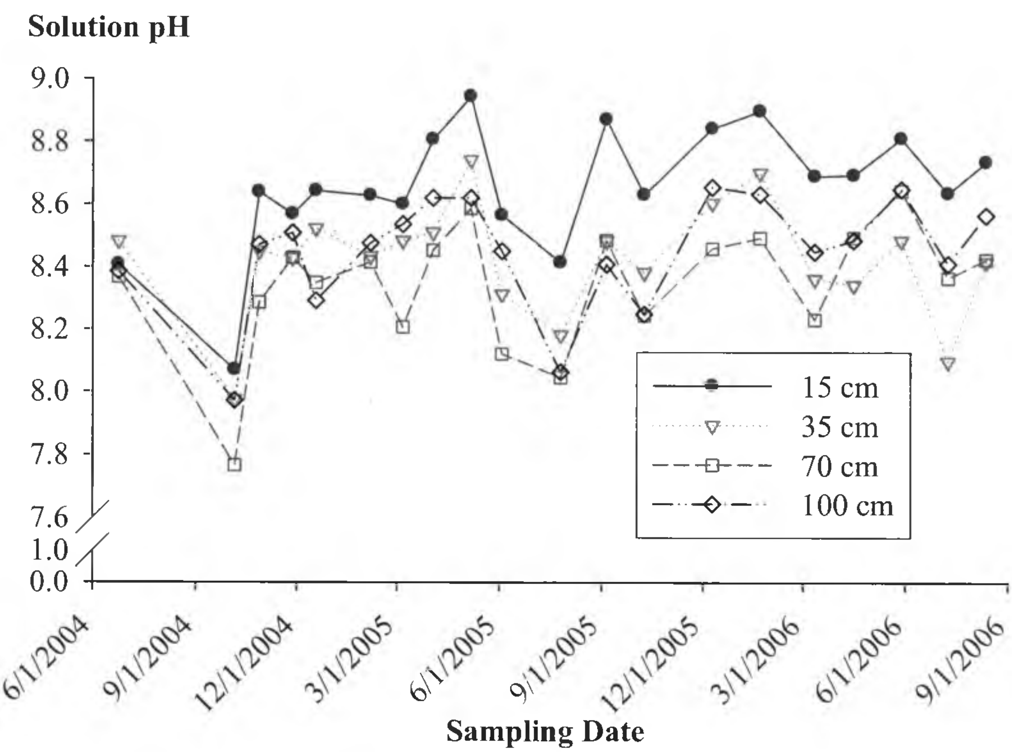



**Figure 4.4 . Soil solution pH at the 15 cm, 35 cm, 70 cm and 100 cm depths in effluent-irrigated soil (Mollisol) planted to tropical grasses, Waianae, O'ahu, Hawai'i.**

**Table 4.2. Proc Mixed ANOVA‡ statistics showing the Type III test of fixed effects on soil solution pH in dairy effluent-irrigated soil, Waianae, O'ahu, Hawai'i, June 2004 to Aug 2006.**

| Effect | Numerator DF | Denominator DF | F | *P* |
|---|---|---|---|---|
| Time | 19 | 41 | 10.27 | <.0001 |
| Grass | 3 | 5 | 0.20 | 0.8904 |
| Sampling depth | 3 | 12 | 50.62 | <.0001 |
| Grass x Time | 37 | 41 | 1.30 | 0.2064 |
| Grass x Sampling depth | 9 | 12 | 9.90 | 0.0003 |
| Irrigation rate x Sampling depth | 3 | 12 | 12.82 | 0.0005 |
| Time x Sampling depth | 57 | 267 | 1.17 | 0.2081 |
| Irrigation rate x Time | 19 | 41 | 2.12 | 0.022 |

‡Based on best subset regression results.

The average soil solution pH at the 15 cm sampling depth ($8.6 \pm 0.02$) was higher ($P<0.01$) than at the other three sampling depths ($8.3 \pm 0.04$ to $8.4 \pm 0.04$) (Fig. 4.4). In general, the lower pH of the subsurface soil and pH of the soil solution collected at the lower depths may be explained by the plant uptake of nutrients and the higher carbon dioxide pressure at the lower depth(s) as a result of root respiration and microbial activities possibly allowing for carbonic acid formation (Brady and Weil, 2002).

The generally high soil pH during the two years of effluent irrigation (2004 to 2006) may be attributed to the effluent's high pH (8.3 in July 2003 and 8.2 in June 2004) (Fig. 3.10), and due to its high carbonate and bicarbonate content (Table 4.3). The level of carbonate (55 to 62 mg $CaCO_3$ $L^{-1}$) was similar to and bicarbonate (1336 to 2010 mg $CaCO_3$ $L^{-1}$) was higher than those reported for dairy effluent in India (50 and 520 mg $CaCO_3$ $L^{-1}$, respectively) (Sharma et al., 2003). The soil used in this experiment had a high soil extractable Ca (5900 mg $kg^{-1}$ at the 0 to 15 cm depth and 4300 mg $kg^{-1}$ at the 15 to 30 cm depth), Mg (3100 and 2600 mg $kg^{-1}$ at the 0-15 cm and 15 to 30 cm depths, respectively) and Na (2600 mg $kg^{-1}$ at the 0 to 15 cm depth and 1800 mg $kg^{-1}$ at the 15 to 30 cm depth) contents before the effluent irrigation began. Each irrigation event, more Ca, Mg, Na and carbonates/bicarbonates were added to the soil and their accumulation may have resulted in higher soil pH.

**Table 4.3. Carbonate and bicarbonate content of dairy effluent, Oahu, Hawaii.**

| Sampling Date | pH | $CO_3^{2-}$ | $HCO_3^-$ |
|---|---|---|---|
| | | ----------------mg $CaCO_3$ $L^{-1}$ | ----------------- |
| Aug. 2004 | 8.60 | 56±1 | 1510±2 |
| Sept. 2005 | 8.58 | 55±1 | 2212±11 |
| Aug. 2006 | 8.57 | 62±1 | 1459±3 |

Whalen et al. (2000) and Sharpley and Moyer (2000) attributed the increased pH of soils amended with manure to increased bicarbonate as well as the increased carboxyl and phenolic hydroxyl groups from organic acids. The high soil pH may also be due to the large inputs of Ca (Sharpley and Moyer, 2000; Nair et al., 1995) and Mg (Nair et al., 1995). Contrary to our findings and the reported liming effect of manure and liquid animal wastes, a decreasing soil pH with increasing swine effluent application rate was observed in other studies (King et al., 1985; Whalen and Chang, 2002). The soil pH reduction was attributed to the loss of $Ca^{2+}$ and $Mg^{2+}$ from the surface layer due to effluent additions of $Na^+$, $K^+$ and $NH_4^+$ ( Whalen and Chang, 2002).

**Electrical Conductivity (EC)**

The soil EC was highest ($P$<0.01) when measured from the saturated paste extract (9.1) (Tables 4.4 and 4.5). The EC measured in this way was 2.1, 2.7 and 3.3 times higher than the EC measured in 1:1 extract (4.2 dS $m^{-1}$), 1:2 extract (2.7 dS $m^{-1}$) and saturated paste (3.6 dS $m^{-1}$), respectively. The lower EC readings for the 1:1 and 1:2 extracts could be attributed to the salt dilution effect (US Soil Salinity Laboratory, 1954).

The regression equations used to convert the soil EC measured using 1:1 and 1:2 extracts to saturated paste extract equivalent values were $EC_{spe} = 2.4375(EC_{1:1})-1.2955$ ($R^2 = 0.96$) and $EC_{spe} = 3.5928(EC_{1:2}) - 0.6775$ ($R^2 = 0.92$) (Figs. 4.5 and 4.6). The regression equation for converting 1:1 extract to saturated paste equivalent yielded results that were lower than those obtained using the equation $EC_{spe} = 3.00(EC_{1:1})$ developed by the US Salinity Laboratory (1954), and equation $EC_{spe} = 3.0(EC_{1:1})-0.77$ proposed by Franzen (2003) for soils of North Dakota representing coarse-, medium- and fine-textured soils. But our equation produced similar results as that suggested by Zhang et al. (2005)— $EC_{spe} = 1.85(EC_{1:1})$—who used soils from Texas and Oklahoma with a broad range of soil conditions and analyte concentrations.

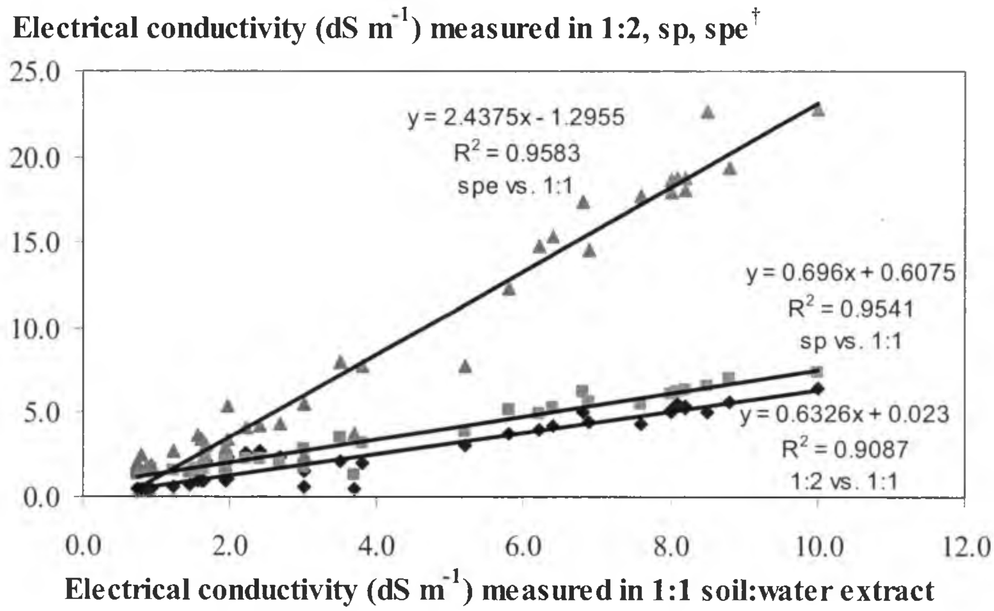
**Table 4.4. A Waller-Duncan comparison of means for various EC measurement methods in dairy effluent-irrigated soil (Mollisol), O'ahu, Hawai'i.**

| EC Determination Method | Mean‡ |
|---|---|
| Saturated paste extract | 9.07[a] |
| Saturated paste | 3.57[b] |
| 1:1 extract | 4.25[b] |
| 1:2 extract | 2.71[b] |

‡Means followed by the same letter are not statistically significant at the 5% level; Means of 37 soil samples collected between Apr 2003 and Aug 2006.

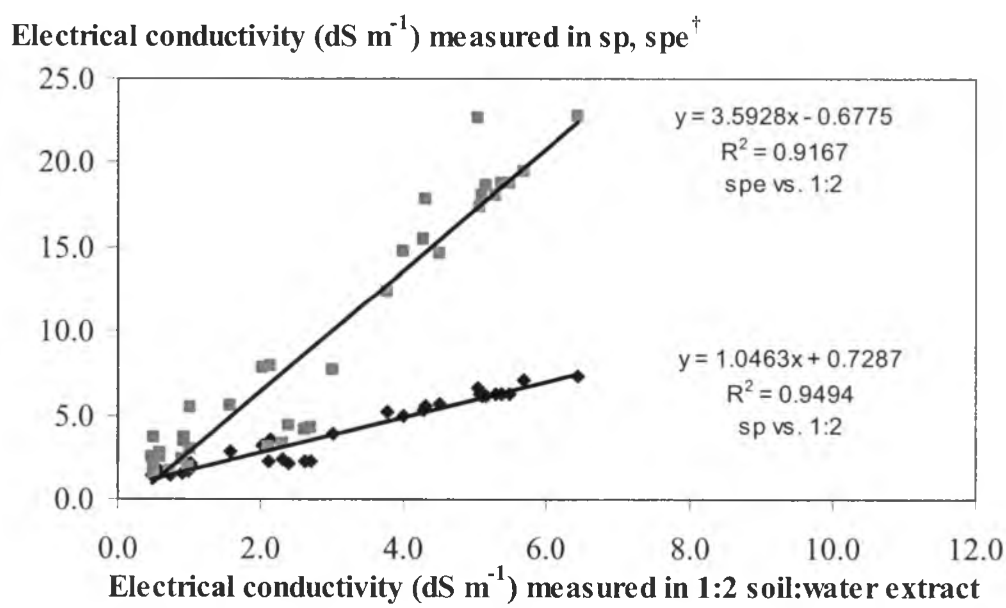
**Table 4.5. Proc GLM ANOVA statistics showing the Type III test of fixed effects (measurement method) on soil EC in dairy effluent-irrigated soil (Mollisol), Waianae, O'ahu, Hawai'i.**

| Effect | DF | SS | MSE | F | *P* |
|---|---|---|---|---|---|
| Treatment | 3 | 901.168557 | 300.389519 | 17.17 | <.0001 |
| Error | 144 | 2518.565795 | 17.490040 | | <.0001 |
| Total | 147 | 3419.734352 | | | 0.0003 |




†1:2 soil:water extract; sp - saturated paste; spe - saturated paste extract.

**Figure 4.5. Regression equations for EC measured from 1:2 extract, saturated paste and saturated paste extract vs. 1:1 extract in an effluent-irrigated soil (Mollisol) planted to tropical grasses, O'ahu, Hawai'i.**



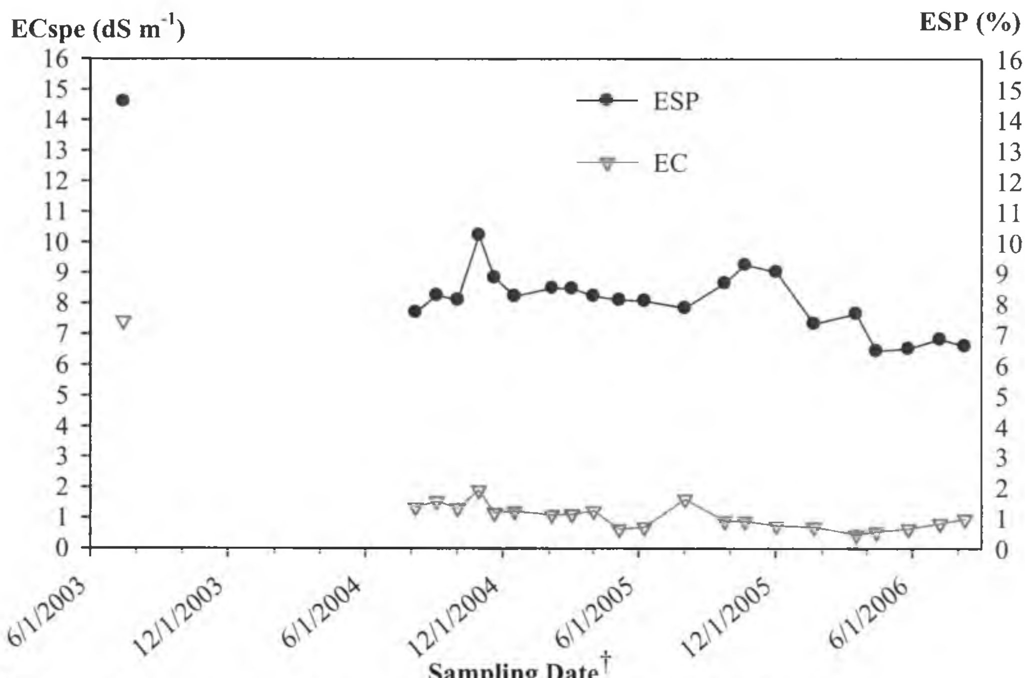

† sp - saturated paste; spe - saturated paste extract.

**Figure 4.6. Regression equations for EC measured from 1:2 extract, saturated paste and saturated paste extract vs. 1:1 extract in an effluent-irrigated soil (Mollisol) planted to tropical grasses, O'ahu, Hawai'i.**

The results of this study supported earlier findings that the $EC_{1:1}$ method may be used to obtain the saturated paste extract equivalent for soil salinity evaluations. Among the benefits are reduced cost and time to deliver the results for assessing salt impacts on soils and arriving at recommendations for remediation. However, this method may still be inconvenient for large number of soil samples with high amounts of smectitic clays. Such soils absorb large amounts of water and, consequently, the addition of water at a 1:1 ratio produces a mixture that is virtually a saturated paste. Thus, there was a need to further increase the ratio to 1:2 soil:water for relative ease in determining the EC of soils rich in smectites.

As with the relationship between $EC_{1:1}$ and $EC_{spc}$ ($R^2 = 0.96$), a high coefficient of determination ($R^2 = 0.92$) was also found between $EC_{1:2}$ and $EC_{spc}$. Hence, the $EC_{1:2}$ method may also be used in cases where the soil has a high amount of shrinking-swelling clays such as the soil in the study site. Follow-up studies involving a range of Hawaiian soils with different mineralogy, clay content, land use and EC values needs to be conducted to validate the regression equations obtained in this study for the conversion of EC obtained from more dilute solutions into saturated paste extract equivalent.

The trend in soil EC did not change after conversion of values into saturated paste extract equivalents using the equation developed in this study. The conversion changed the classification of the soil with regard to salinity only for soils collected in July 2003 due to the higher EC of these soil samples. Soil $EC_{spc}$ declined ($P$<0.01) from an average of 18.0$\pm$0.2 dS $m^{-1}$ in July 2003 to 2.7$\pm$0.3 dS $m^{-1}$ in Aug 2006 (Fig. 4.7 and Table 4.6). Some mixing of the topsoil and subsoil occurred during the installation of irrigation tubing, yet, the soil $EC_{spc}$ at the 0 to 15 cm depth (19.0$\pm$0.3 dS $m^{-1}$) and 15 to 30 cm depths ($16.0 \pm 0.4$ dS $m^{-1}$) were different ($P<0.05$). The largest drop in soil $EC_{spe}$ occurred between July 2003 and Aug 2004, with a change of 16.2 dS $m^{-1}$ ($P<0.01$) between the mean EC of the two sampling dates. This decline in $EC_{spe}$ may have resulted from freshwater irrigation (30 min, twice a day) during the grass establishment period (Dec 2003 to July 2004).




†Soil samples were collected before the irrigation was installed in July 2003; grass establishment period was from December 2003 to July 2004

**Figure 4.7. Soil $EC_{spe}$ and ESP from effluent-irrigated soil (Mollisol) planted to tropical grasses, O'ahu, Hawai'i.**

**Table 4.6. Proc Mixed ANOVA‡ statistics showing the Type III test of fixed effects on soil EC in dairy effluent-irrigated soil (Mollisol), Waianae, O'ahu, Hawai'i, Aug 2004 to Aug 2006.**

| Effect | Numerator DF | Denominator DF | F | *P* |
|---|---|---|---|---|
| Time | 20 | 380 | 20.07 | <.0001 |
| Sampling depth | 1 | 380 | 70.39 | <.0001 |
| Time x Sampling depth | 20 | 380 | 5.42 | <.0001 |

‡Based on best subset regression results.

The main effects and interaction of time and sampling depth on soil $EC_{spe}$ ($P$<0.01) as well as the main effects of time and sampling depth on soil solution EC ($P$<0.01) were significant (Tables 4.6 and 4.7). Between Aug 2004 and Aug 2006, soil $EC_{spe}$ and soil solution EC showed fluctuations ($P$<0.01) with average values ranged from 0.9 dS $m^{-1}$ to 5.1 dS $m^{-1}$ for soil $EC_{spe}$ (Fig. 4.7) and 2.2 to 5.8 dS $m^{-1}$ for soil solution EC (Fig. 4.8). The fluctuations were related to the rainfall pattern— rainfall tended to lower the soil $EC_{spe}$ (Fig. 4.8), suggesting some discharge effect (i.e., salt transfer from the solid phase to liquid phase) of rainfall (Caballero et al., 2001) and possibly, the consequent redistribution of electrolytes (Lado et al., 2005) The soil $EC_{spe}$ values between Aug 2004 and Aug 2006 were also lower than the US Salinity Laboratory's (1954) critical level (4.0 dS $m^{-1}$) for classifying soils as saline. The low $EC_{spe}$ maintained for the duration of the effluent irrigation may be ascribed to the freshwater flush after each irrigation event.

**Table 4.7. Proc Mixed ANOVA‡ statistics showing the Type III test of fixed effects on soil solution EC in dairy effluent-irrigated soil (Mollisol), Waianae, O'ahu, Hawai'i, June 2004 to Aug 2006.**

| Effect | Numerator DF | Denominator DF | F | *P* |
|---|---|---|---|---|
| Irrigation rate | 1 | 6 | 2.29 | 0.1806 |
| Time | 19 | 72 | 15.78 | <.0001 |
| Sampling depth | 7 | 24 | 6.38 | 0.0003 |
| Grass x Irrigation rate | 1 | 6 | 0.40 | 0.5523 |
| Grass x Time | 23 | 72 | 0.72 | 0.8117 |

‡Based on best subset regression results.

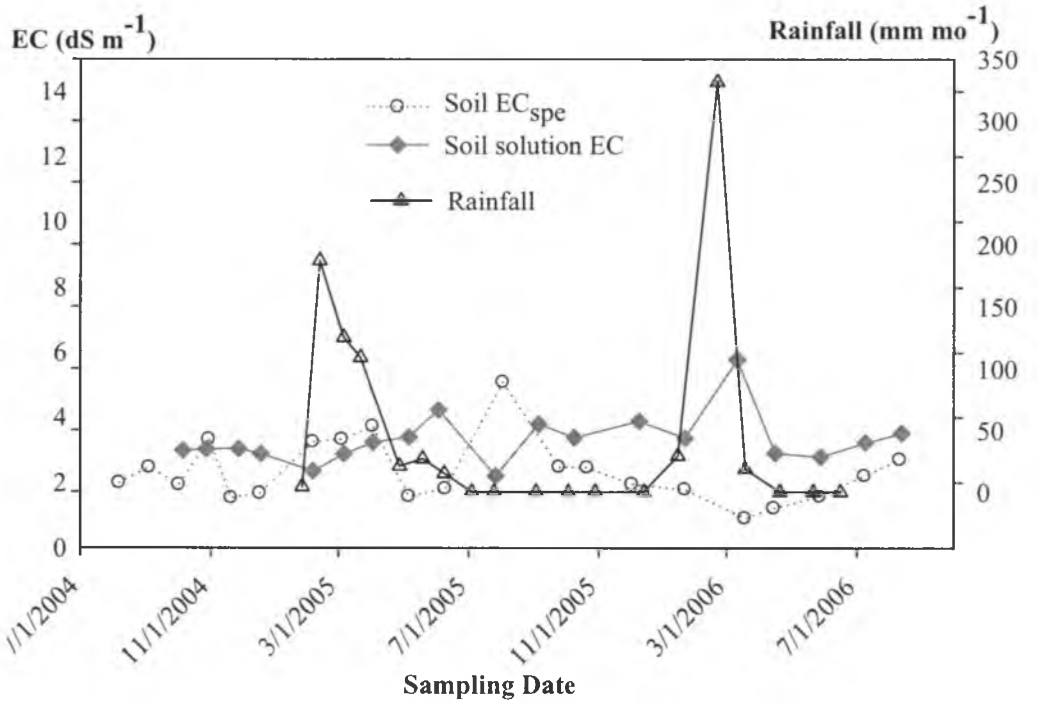
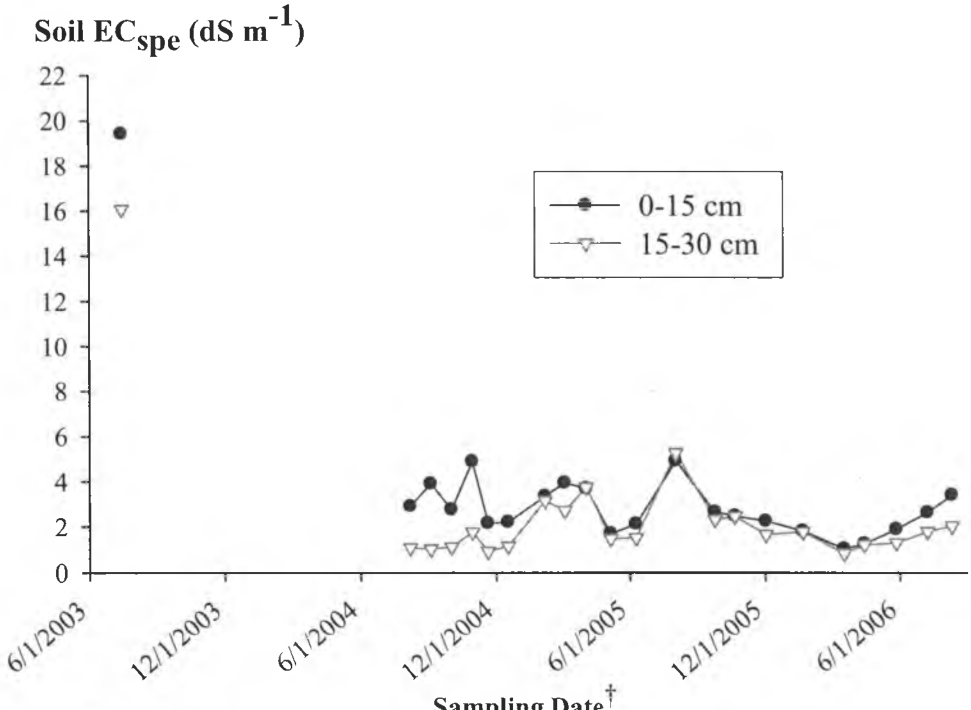



**Figure 4.8. Rainfall vs. soil and soil solution EC in effluent-irrigated soil (Mollisol) planted to tropical grasses, Waianae, O'ahu, Hawai'i.**

Soil $EC_{spe}$ was slightly higher in the 0 to 15 cm depth increment of soil (2.8±0.17 dS m$^{-1}$) than in the 15 to 30 cm depth (1.9±0.17 dS m$^{-1}$) ($P$<0.01) for the duration of effluent irrigation (Fig. 4.9). Similarly, the EC of the soil solution collected at the 15 cm depth (3.5±0.08 dS m$^{-1}$) was higher than those collected at the 35 cm, 70 cm and 100 cm depths (3.3±0.06 dS m$^{-1}$, 3.1±0.06 dS m$^{-1}$ and 3.0±0.1 dS m$^{-1}$, respectively) (Fig. 4.10).

These findings indicated that little dissolved salt should have moved downward due to the low hydraulic conductivity (1.6 to 5.1 cm $h^{-1}$) of the soil at the site (Foote et al., 1972). The precipitation of salts in carbonate forms may also be a possibility (Figs. 4.11 and 4.12; Table 4.8). Despite daily effluent application, the $EC_{spe}$ of the soil did not reach the value of 4.0 dS $m^{-1}$, often used to identify soils as saline (US Salinity Laboratory, 1954), demonstrating that even at the high rate of irrigation (corresponding to about 90,000 to 450,000 L $ha^{-1}$ $d^{-1}$), salt loading did not result in excessive increases in soil salinity.




†Soil samples were collected before the irrigation was installed in July 2003; grass establishment period was from December 2003 to July 2004

**Figure 4.9. Soil EC-saturated paste extract at the 0 to 15 cm and 15 to 30 cm depths from effluent-irrigated soil planted to tropical grasses, Waianae, O'ahu, Hawai'i.**

An interesting observation is the lack of significant differences in the EC of the plots that received the 0.5 and 2.0 $ET_p$ irrigation rates. This finding indicated that this system was highly buffered such that additions of salts to the plots regardless of the rate did not result in further increases in EC.

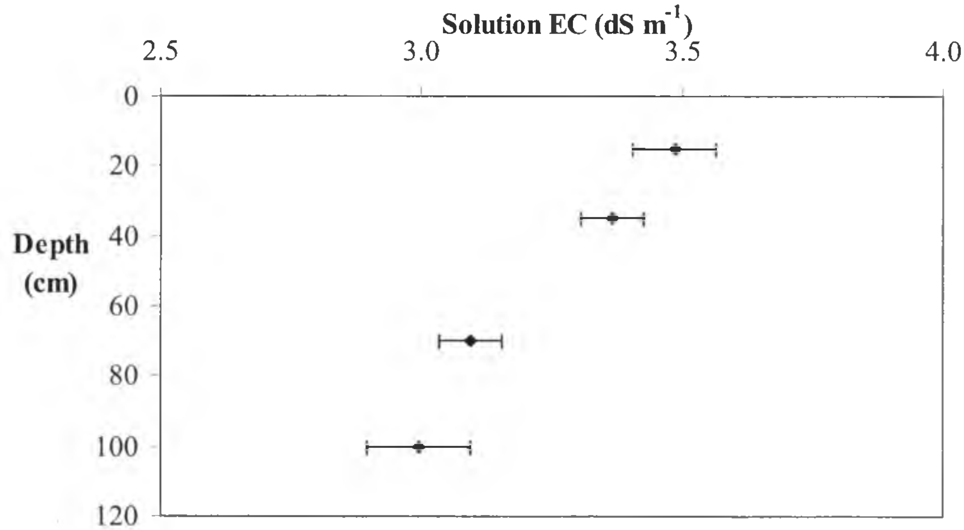



**Figure 4.10. Soil solution EC at the 15 cm, 35 cm, 70 cm and 100 cm depths from effluent-irrigated soil planted to tropical grasses, Waianae, O'ahu, Hawai'i.**

**Table 4.8. Soil pH and log activity of soil solution Ca, Mg and Na in effluent-irrigated soil (Mollisol) planted to tropical grasses, Waianae, O'ahu, Hawai'i.**

| Depth/ Irrigation Rate | Date | Soil pH | Ca‡ | Mg‡ | Na†† |
|---|---|---|---|---|---|
| | | | ----------log activity---------- | | |
| 0-15 cm | | | | | |
| 2.0 $ET_p$ | Aug '04 | 8.2 | -3.8 | -3.4 | -2.6 |
| (7 to 44 mm $d^{-1}$) | Feb '05 | 8.4 | -4.0 | -3.6 | -2.8 |
| | Aug '05 | 8.3 | -4.1 | -3.4 | -2.6 |
| | Feb '06 | 8.4 | -4.1 | -3.7 | -2.8 |
| | Aug '06 | 8.7 | -3.9 | -3.3 | -2.4 |
| 0.5 $ET_p$ | Aug '04 | 8.2 | -3.7 | -3.5 | -2.6 |
| (2 to 11 mm $d^{-1}$) | Feb '05 | 8.3 | -4.1 | -3.9 | -3.0 |
| | Aug '05 | 8.3 | -4.1 | -3.5 | -2.7 |
| | Feb '06 | 8.2 | -3.9 | -3.7 | -2.8 |
| | Aug '06 | 8.4 | -3.8 | -3.3 | -2.5 |
| 15-30 cm | | | | | |
| 2.0 $ET_p$ | Aug '04 | 7.9 | -3.8 | -3.4 | -2.6 |
| (7 to 44 mm $d^{-1}$) | Feb '05 | 8.2 | -3.8 | -3.5 | -2.7 |
| | Aug '05 | 8.2 | -4.1 | -3.5 | -2.6 |
| | Feb '06 | 8.3 | -3.7 | -3.3 | -2.5 |
| | Aug '06 | 8.5 | -3.7 | -3.3 | -2.5 |
| 0.5 $ET_p$ | Aug '04 | 8.0 | -3.9 | -3.7 | -2.9 |
| (2 to 11 mm $d^{-1}$) | Feb '05 | 8.2 | -4.0 | -3.6 | -2.8 |
| | Aug '05 | 8.2 | -4.1 | -3.5 | -2.6 |
| | Feb '06 | 8.2 | -3.8 | -3.5 | -2.6 |
| | Aug '06 | 8.4 | -3.6 | -3.3 | -2.5 |

‡Assumed 15% of total Ca and total Mg concentrations in the soil solution
††Assumed 100% of total Na concentration in the soil solution

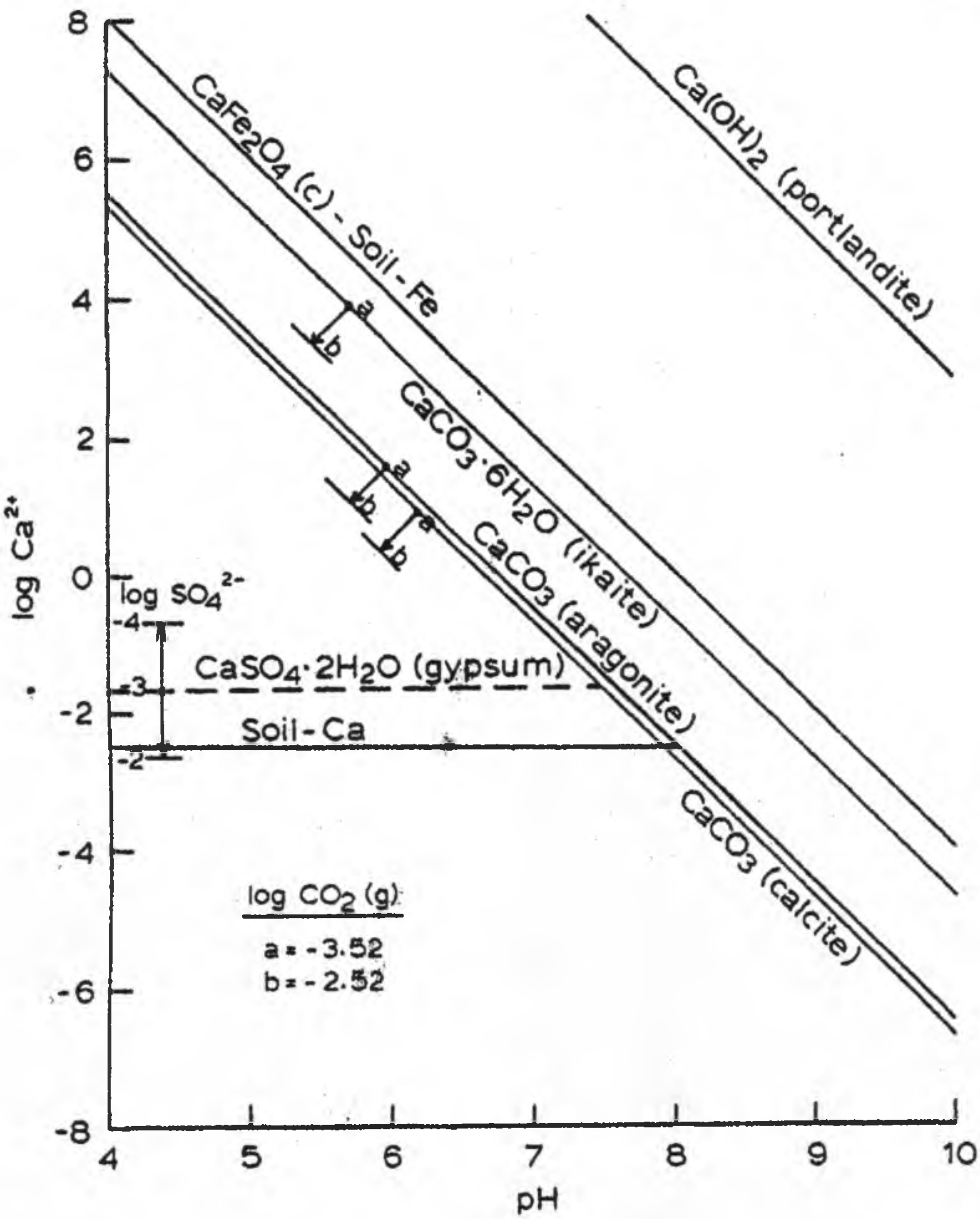



**Figure 4.11.** **Solubility diagram for various calcium compounds showing calculated Ca potential for soils collected at the 0-15 cm and 15-30 cm depths in Aug 2005 from effluent-irrigated soil (0.5 and 2.0 ETp), Waianae, O'ahu, Hawai'i.**

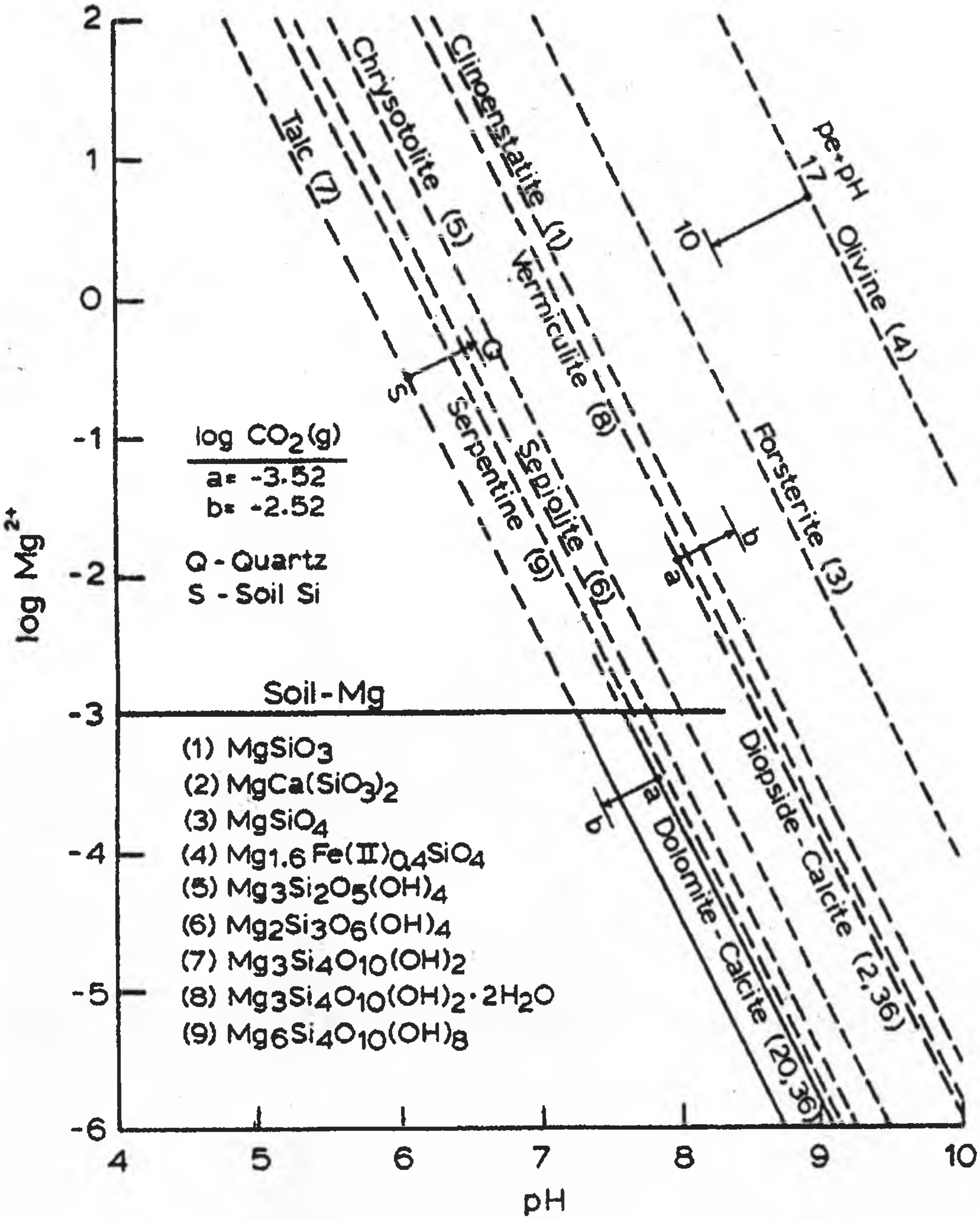



**Figure 4.12. Solubility diagram for various magnesium compounds showing calculated Mg potential for soils collected at the 0-15 cm and 15-30 cm depths in Aug 2005 from effluent-irrigated soil (0.5 and 2.0 ETp), Waianae, O'ahu, Hawai'i.**

**Indicators of Soil Swelling and Dispersion**

Clay swelling and dispersion are major concerns when irrigating with effluent, due to the high salt content of the wastewater and the consequent effects of salt accumulation on clay dispersion and swelling and slaking of aggregates (Shainberg, 1984). While clay swelling is not expected at ESP values below 10 to 15%, it increases considerably at ESP above 15% (McNeal and Coleman, 1966), whereas soil dispersion can occur even at ESP values $\leq 5\%$ (Oster et al., 1980; McIntyre, 1979) if the electrolyte concentration remains below a threshold level (Shainberg, 1984). The threshold concentration (or flocculation value) varies among soils and for smectite clay, these values are 0.25, 4, and 7 $mmol_c\ L^{-1}$ for ESP values of 0, 10 and 20%, respectively (Oster et al., 1980).

The soil at the experimental site (Pulehu series) has a moderately weathered smectite clay mineral fraction (2:1 clay minerals) and was already high in salts ($EC_{spc}$ = 11 to 26 dS $m^{-1}$) when the experiment began in July 2003. In addition, the dairy effluent used for irrigating the forage contains salts especially Na (total Na was 180 to 320 mg Na $L^{-1}$). Sodium chloride (NaCl) is an important component of a dairy cattle diet. When used repeatedly for irrigation, high sodium effluent may cause soil dispersion (Sumner, 1993). Thus, apart from constantly evaluating the SAR of the effluent used for irrigation and the EC of the soil solution, the changes in soil ESP and EPP were also monitored so that ameliorative actions could be taken before soil dispersion occurs, as happens in sodic soils. It is important to prevent soil dispersion before it occurs as it has negative impact on the soil by reducing infiltration rate and hydraulic conductivity (Levy et al., 1999) and crop growth (Rhoades et al., 1992). A greater cost is involved in reclaiming sodic soils

(Rengasamy, 2002 and irreversible changes in soil structure are possible (Lebron et al., 2002; Sumner, 1993).

The initial (July 2003) soil ESP was relatively high ranging from 13% to nearly 18%, with an average of 14.6$\pm$0.4% (Fig. 4.13 and Table 4.9). After the grass establishment period, soil ESP decreased to an average of 7.8$\pm$0.6% % in Aug 2004—, lower than the 15% critical level for classifying soils as sodic. The reduction in soil ESP could be explained by the possible leaching of salts due to freshwater sprinkler irrigation during the grass establishment phase.

Between Aug 2004 and Aug 2006, soil ESP fluctuated but generally followed a decreasing trend (Fig. 4.13). This decline in soil ESP may be explained by the large inputs of Ca, Mg and K relative to Na. The high initial Ca, Mg and K content of the soil coupled with the repeated addition of these nutrients through effluent irrigation may have resulted in relatively high concentrations of these nutrients in the soil solution, thereby preventing any increase in ESP. For example, at the 2.0 $ET_p$ rate, as much as 42000 kg K $ha^{-1}$ $y^{-1}$, 2900 kg Ca $ha^{-1}$ $y^{-1}$ and 5900 kg Mg $ha^{-1}$ $y^{-1}$ were applied, compared to only 15000 kg Na $ha^{-1}$ $y^{-1}$. At the 0.5 $ET_p$ rate, the cumulative amount added was 12000 kg K $ha^{-1}$ $y^{-1}$, 850 kg Ca $ha^{-1}$ $y^{-1}$ and 1700 kg Mg $ha^{-1}$ $y^{-1}$ compared to 4300 kg Na $ha^{-1}$ $y^{-1}$.

The SAR of the dairy effluent used for irrigation (3.1 to 5.3) (Fig. 3.10) remained at lower levels for the duration of the study than the critical value (9.0) set by the US Salinity Laboratory (1954) and its quality was, therefore, acceptable for irrigation. This low SAR value may explain the lack of increases in soil ESP in this effluent-irrigated soil (Mamedov et al., 2000). The SAR of the soil solution was also between 4 and 9 (Fig. 4.14)—these values are below 13 (Brady and Weil, 2002) or 15 (Tanji, 1990) which is the critical level for classifying soils as sodic. The total cation concentration of the soil remained high during the 2-y period of effluent irrigation (10 to 51 $mmol_c\ L^{-1}$) while the ESP was below 12% (Fig. 4.7). Thus, sodicity was not observed. Soil dispersion is expected at ESP of 10 to 20% if the electrolyte concentration in the soil is below 10 $mmolc\ L^{-1}$ (Felhendler et al., 1974). It is also possible that the high amount of Fe oxides in Hawaiian soils prevented soil dispersion (McNeal et al., 1968).

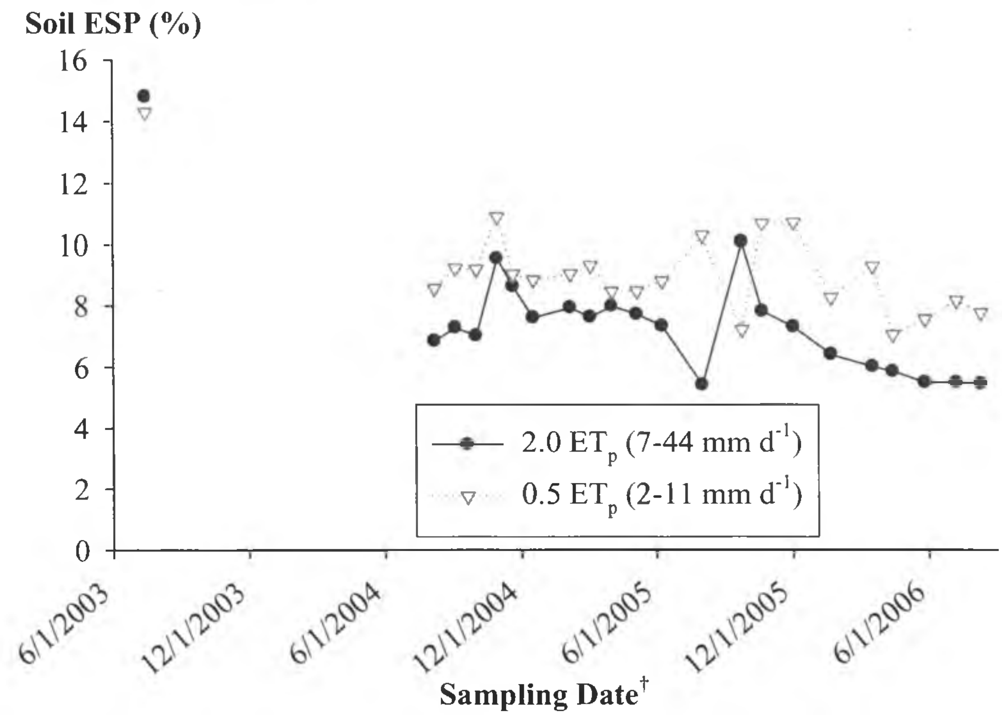
**Table 4.9. Proc Mixed ANOVA[‡] statistics showing the Type III test of fixed effects on soil ESP in dairy effluent-irrigated soil (Mollisol), Waianae, O'ahu, Hawai'i, June 2004-Aug 2006.**

| Effect | Numerator DF | Denominator DF | F | *P* |
|---|---|---|---|---|
| Irrigation rate | 1 | 298 | 10.61 | 0.0013 |
| Time | 20 | 298 | 4.70 | <.0001 |
| Sampling depth | 1 | 298 | 148.79 | <.0001 |
| Grass x Time | 63 | 298 | 1.13 | 0.2569 |
| Irrigation rate x Time | 20 | 298 | 3.28 | <.0001 |
| Time x Sampling depth | 20 | 298 | 6.03 | <.0001 |

‡Based on best subset regression results.

There was also a significant interaction between irrigation rate and time (P<0.01) as well as time and depth ($P$<0.01) (Figs. 4.13 and 4.15). The decline in soil ESP was more evident ($P$<0.01) in plots that received the 2.0 $ET_p$ rate (7.2±0.4%) than at those that received the 0.5 $ET_p$ rate (8.9±0.4%), due to the greater amounts of effluent applied at the 2.0 $ET_p$ irrigation rate (Fig. 4.13). Soil ESP also differed between the two sampling depths ($P$<0.01). Soil ESP was generally higher in the 0 to 15 cm depth (9.1±0.4%) than in the 15 to 30 cm depth (7.0±0.4%) (Fig. 4.15). The soil ESP averaged 15.3±0.5% in the 0 to 15 cm depth and 13.3±0.5% in the 15 to 30 cm depth in July 2003, 9.7±0.7% and

6.0±0.4%, respectively in Aug 2004, 8.5±1.3% and 7.5±1.0%, respectively, in Aug 2005 and 7.5±1.1 and 5.2±0.3%, respectively in Aug 2006 (Fig. 4.15).




†Soil samples were collected before the irrigation was installed in July 2003; grass establishment period was from December 2003 to July 2004

**Figure 4.13. Soil ESP of effluent-irrigated soil (0.5 and 2.0 $ET_p$) planted to tropical grasses, Waianae, Oahu, Hawai'i.**

Given the slightly higher soil ESP at the 0-15 cm depth, it is possible that there was some upward movement of Na from the effluent application zone particularly during summer when there was very high $ET_p$. This capillary rise of water commonly occurring in subsurface irrigation (FAO, 1996) may be more prevalent at the plots that received the 0.5 $ET_p$ rate, partly explaining the generally higher ESP in these plots. Calculations showed no water flux from the plots irrigated at the 0.5 $ET_p$ rate because the $ET_p$ exceeded the water from irrigation and rainfall—a condition that favors upward movement of water and salts.

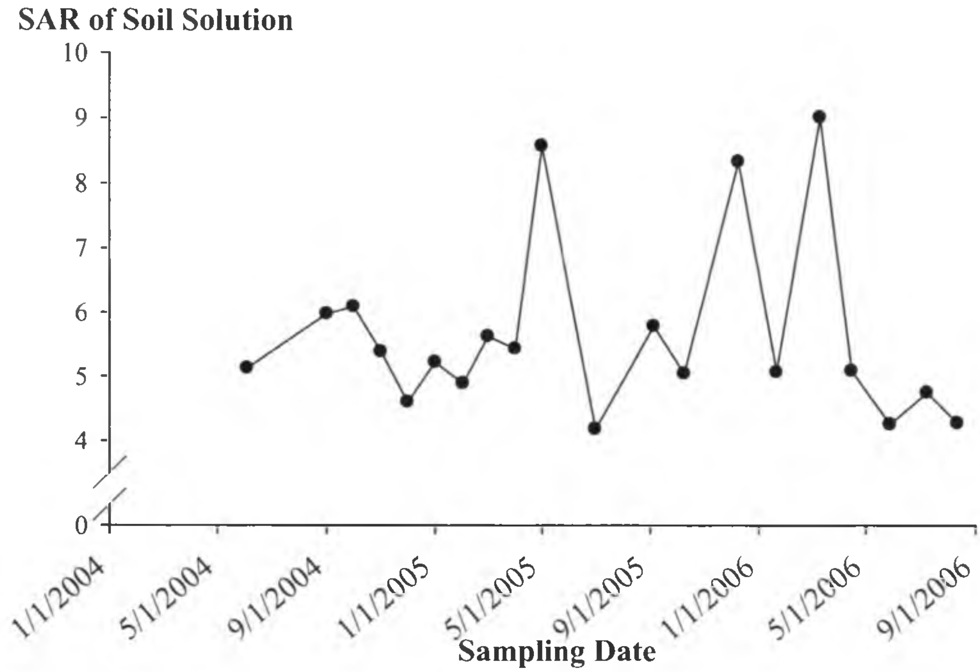
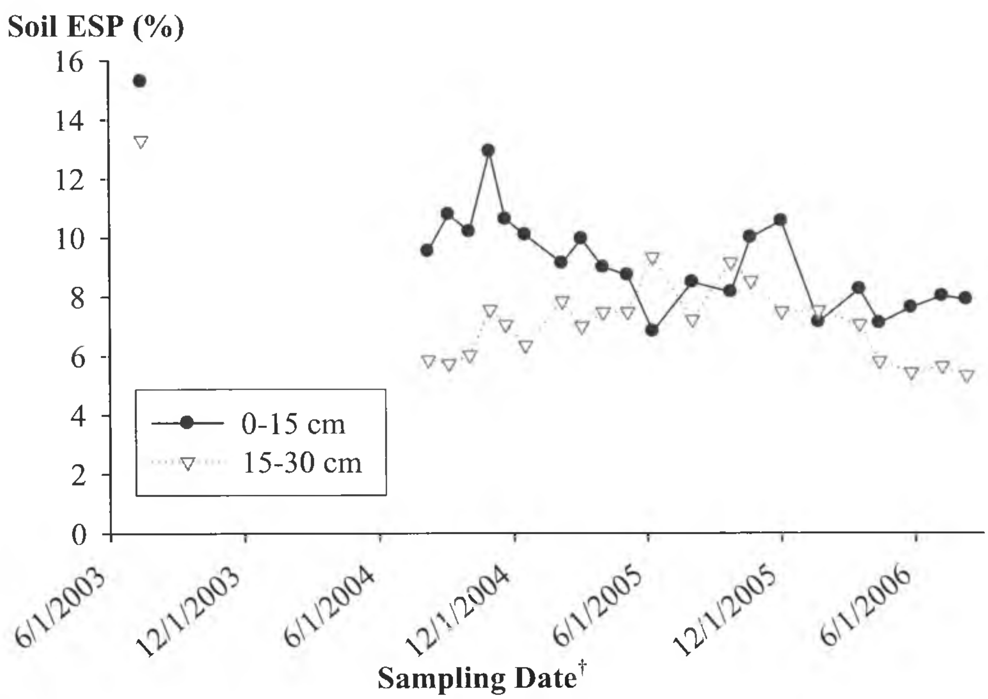



**Figure 4.14. Sodium absorption ratio of soil solution collected from effluent-irrigated soil planted to tropical grasses, Waianae, O'ahu, Hawai'i.**

Potassium, being a monovalent cation, may also cause dispersion, although at weaker tendency, than Na (Sumner, 1993). However, a study on liquid and solid manure application on Hanford soil in California revealed a greater potential salinity problem from K than from Na owing to the high capacity of this soil to retain K in non-exchangeable forms (Pratt, 1984). Application of large quantities of animal slurry manure was believed to cause soil dispersion, and consequently increase soil erodibility, by increasing exchangeable potassium percentage (Auerswald et al., 1996). But an earlier study indicated that with high EPP (up to 26%), there was improved aeration of the soil (a Mollisol with 40 to 60% smectite) and increased sorghum (*Sorghum vulgare*) yield (Ravina and Markus, 1975). Shainberg et al. (1987) reported that the effect of EPP on soil dispersion depends on the amount of clay. Soils containing high charge smectites did not disperse at EPP <22, whereas soils having low charge smectites dispersed with the effect on hydraulic conductivity similar to that of sodium. These seemingly contrasting reports make a discussion on soil EPP important.




†Soil samples were collected before the irrigation was installed in July 2003; grass establishment period was from December 2003 to July 2004

**Figure 4.15. Soil ESP at the 0-15 cm and 15-30 cm depths of effluent-irrigated soil planted to tropical grasses, Waianae, O'ahu, Hawai'i.**

The main effects of time and irrigation rate as well as interaction effects between irrigation rate and time, and time and sampling depth were significant ($P$<0.01) (Table 4.10). Soil EPP increased over time ($P$<0.01), from an average of 8.6$\pm$0.7% in July 2003, 13.4$\pm$0.7% in Aug 2004, 18.3$\pm$1.7% in Aug 2005 and 26.3$\pm$1.4 in Aug 2006 (Fig.4.16).

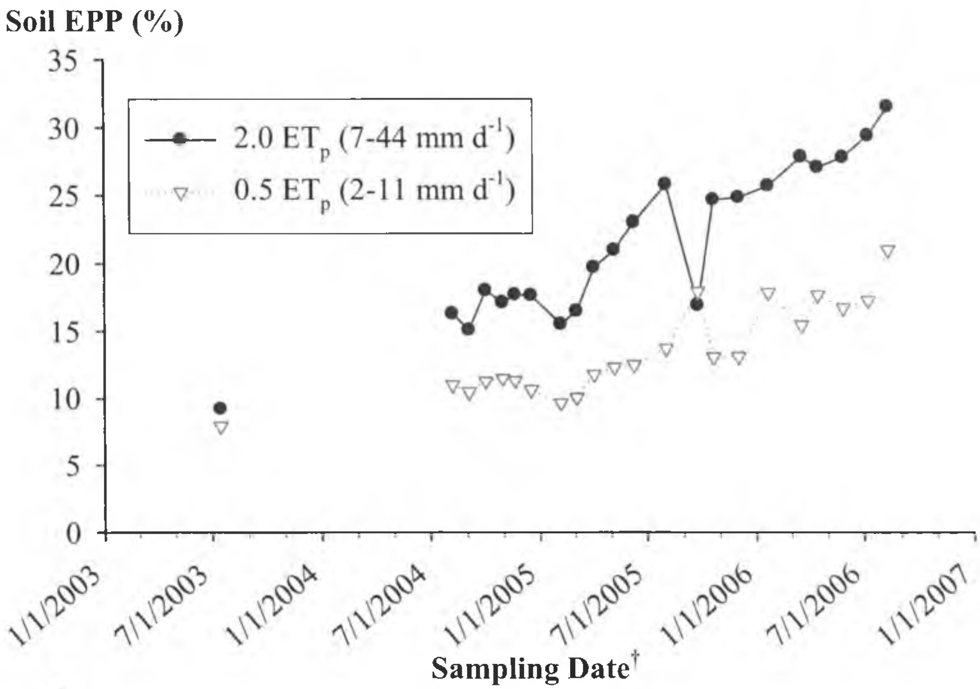
The K concentration of the effluent was relatively high (360 to 930 mg $L^{-1}$) and therefore, the cumulative amount of applied K was substantial.

**Table 4.10. Proc Mixed ANOVA‡ statistics showing the Type III test of fixed effects on soil EPP in dairy effluent-irrigated soil (Mollisol), Waianae, O'ahu, Hawai'i, June 2004-Aug 2006.**

| Effect | Numerator DF | Denominator DF | F | *P* |
|---|---|---|---|---|
| Grass | 4 | 297 | 3.22 | 0.0131 |
| Irrigation rate | 1 | 297 | 167.86 | <.0001 |
| Time | 20 | 297 | 41.47 | <.0001 |
| Grass x Irrigation rate | 4 | 297 | 1.68 | 0.1554 |
| Grass x Time | 59 | 297 | 1.34 | 0.0619 |
| Irrigation rate x Time | 20 | 297 | 9.65 | <.0001 |
| Time x Sampling depth | 21 | 297 | 4.49 | <.0001 |

‡Based on best subset regression results.

Soil EPP of plots irrigated at the 2.0 $ET_p$ rate (average of 21.4±0.4%) was also higher ($P$<0.01) than those irrigated at 0.5 $ET_p$ rate (average of 13±0.2%) (Fig. 4.16). Due to the much higher amount of K applied, plots that received the 2.0 $ET_p$ irrigation rate had a greater increase ($P$<0.01) in soil EPP over time (from an average of 9±1.3% in July 2003 to an average of 32±1.1% in Aug 2006) than those that received the 0.5 $ET_p$ irrigation rate (8±0.4% in July 2003 to 21±0.7% in Aug 2006). Regardless of grass species and irrigation rate, the increase in soil EPP was also greater at the 15-30 cm depth (average of 10.0% in July 2003 to 28.1% in Aug 2006) compared with the 0-15 cm depth (average of 7.8% in July 2003 to 24.5% in Aug 2006) (Fig. 4.17). These results indicate the importance of further work to assess the effects of high levels of K on soil properties that affect soil water movement.

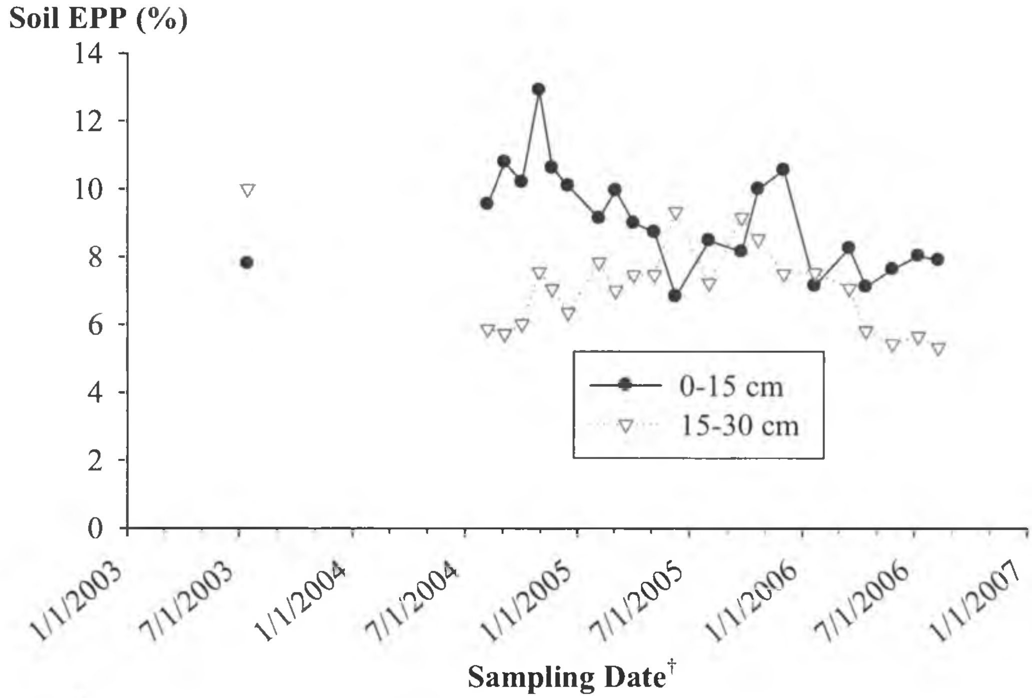



†Soil samples were collected before the irrigation was installed in July 2003; grass establishment period was from December 2003 to July 2004

**Figure 4.16. Soil EPP of effluent-irrigated soil (0.5 and 2.0 $ET_p$) planted to tropical grasses, Waianae, O'ahu, Hawai'i.**

### Leaching Fraction

The leaching fraction (LF) did not differ among the grasses (*P*>0.10) but differed between irrigation rates (*P*<0.05). Plots that received 2.0 $ET_p$ had somewhat lower leaching fraction than those that received 0.5 $ET_p$ (Table 4.11). This result would be expected as the plots that received the 0.5 $ET_p$ rate had much less water passing through the profile and possibly, some upward water movement occurred during the hot summer periods when evapotranspiration was high. Thus, the LF for the plots irrigated at 0.5 $ET_p$ ranged from 0.34 to 0.41 compared to 0.28 to 0.31 for plots irrigated at 2.0 $ET_p$. The actual water requirement to leach the salts was, thus, also higher for plots that received 0.5 $ET_p$. Comparisons of the amount of effluent water being applied at both 2.0 $ET_p$ and 0.5 $ET_p$ with the actual water requirement (AWR) to meet crop needs and additional water for leaching shows that the 0.5 $ET_p$ irrigation rate did not meet the AWR, except if the consumptive water use was assumed to be only 5 mm $d^{-1}$. Meiri et al. (1977) reported that the normal rate of evapotranspiration under field conditions was 6 mm $d^{-1}$. Despite these findings, salinity and sodicity did not occur on the plots irrigated at both irrigation rates.



†Soil samples were collected before the irrigation was installed in July 2003; grass establishment period was from December 2003 to July 2004

**Figure 4.17. Soil EPP at the 0-15 cm and 15-30 cm depths of effluent-irrigated soil planted to tropical grasses, Waianae, O'ahu, Hawai'i.**

**Table 4.11. Leaching fraction and actual water requirement of plots irrigated at 2.0 $ET_p$ and 0.5 $ET_p$ assuming various crop consumptive water use of tropical grasses receiving dairy effluent irrigation, Waianae, Oʻahu, Hawaiʻi.**

| Irrigation Rate | Grass Species | Leaching Eraction | Actual Water Requirement‡ | | |
|---|---|---|---|---|---|
| | | | 7 | 21 | 5 |
| | | | ---mm $d^{-1}$--- | | |
| *70 cm depth*†† | | | | | |
| 2.0 $ET_p$ | *B. mutica* | 0.28 | 9.86 | 29.59 | 7.04 |
| (7 to 44 mm $d^{-1}$) | *B. decumbens* | 0.33 | 10.63 | 31.89 | 7.59 |
| | *C. nlemfuensis* | 0.29 | 10.12 | 30.35 | 7.23 |
| | *P. atratum* | 0.31 | 10.57 | 31.70 | 7.55 |
| 0.5 $ET_p$ | *B. mutica* | 0.34 | 10.99 | 32.97 | 7.85 |
| (2 to 11 mm $d^{-1}$) | *C. nlemfuensis* | 0.39 | 11.51 | 34.53 | 8.22 |
| | *P. atratum* | 0.34 | 12.89 | 38.67 | 9.21 |
| *100 cm depth*†† | | | | | |
| 2.0 $ET_p$ | *B. mutica* | 0.38 | 12.35 | 37.05 | 8.82 |
| (7 to 44 mm $d^{-1}$) | *B. decumbens* | 0.31 | 10.30 | 30.89 | 7.35 |
| | *C. nlemfuensis* | 0.30 | 10.85 | 32.56 | 7.75 |
| | *P. atratum* | 0.29 | 10.02 | 30.06 | 7.16 |
| 0.5 $ET_p$ | *B. mutica* | 0.42 | 10.04 | 30.13 | 7.17 |
| (2 to 11 mm $d^{-1}$) | *C. nlemfuensis* | 0.41 | 11.81 | 35.43 | 8.44 |
| | *P. atratum* | 0.35 | 12.04 | 36.11 | 8.60 |

‡If the consumptive water use of the grasses was equal to the average (7 mm $d^{-1}$), highest (21 mm $d^{-1}$) and lowest (5 mm $d^{-1}$) $ET_p$ measured at the site.
††Calculated the LF and AW based on the EC of the soil solution collected at these depths.

## General Discussion

Despite repeated effluent applications resulting in very high amounts of applied nutrients, especially salts, there were no significant increases in the EC of the soil and soil solution as well as in soil ESP. This is contrary to the usual observations of salinity increase and Na accumulation when irrigating with animal wastes (Toze, 2004; Halliwell et al., 2001; Chang et al., 1991; Assefa et al., 2004). In the present study, the total cation concentration of the effluent used for irrigation (ranging from 27 to 44 $mmol_c\ L^{-1}$) and that of the soil (10 to 51 $mmol_c\ L^{-1}$) remained at a relatively high level for the duration of effluent irrigation. Dispersion may, thus, occur only at higher ESP values than what were observed for the duration of the study.

Predicting the field behavior of clay particles depends on the relationship between the total electrolyte concentration of the soil solution and the soil's ESP (Sumner, 1993). Studies have shown that a higher ESP is needed before soil degradation begins when irrigating with water having a higher total electrolyte concentration (Sumner, 1993; Balks et al., 1998). However, some researchers (Cameron et al., 2003; Sumner, 1993) have suggested that the use of a single critical ESP value is not reliable in assessing effluent application and may vary depending on soil conditions. In fact, a lower critical ESP value was proposed for classifying soils as sodic in Australia (McIntyre, 1979). Cameron et al. (2003) showed that a 50% reduction in hydraulic conductivity at sites receiving dairy factory effluent occurred at an ESP value of 13%, compared to the hydraulic conductivity at control sites with an ESP value of 3%.

In wastewater irrigated soils, the EC and ESP are generally higher at the subsurface due to active water uptake by the plant roots that leads to higher concentrations of cations left in the soil solution (Halliwell et al., 2001). Our results, however, showed that the subsurface soils have generally lower EC and ESP possibly because the irrigation was applied subsurface allowing for replenishment of water taken up by the plants. Also, between 0.43 to 1.72 mm $d^{-1}$ of freshwater is applied to the soil after each irrigation event to prevent clogging of the dripline.

The initial calculation of the leaching fraction (LF) used the standard formula: LF $= EC_{iw} \div EC_{dw}$ (US Salinity Laboratory, 1954). The LF data obtained appeared unreasonable (data not shown) indicating that this formula was not applicable to the conditions in this study. For one, this calculation assumed "uniform aereal application of irrigation water, no rainfall; no removal of salt in the harvested crop; and no precipitation of soluble constituents in the soil…and based on steady-state water-flow rates" (US Salinity Laboratory, 1954). In our study, not all of these assumptions were met. The effluent was applied subsurface; rainfall contributed water during at least three months of the year; some salts were removed by the harvested forage; and possibly, precipitation of some salts was occurring. The Rhoades equation seemed more appropriate to describe the LF under the conditions in this study.

Although salinity increase and dispersion were not observed in this experiment, it is necessary to maintain effluent applications such that there is adequate leaching of salts in the soil profile. Also, the detrimental effect of salts on soil properties was not likely if effluent irrigation continues to provide the required leaching fraction. The long-term effects of salt application may become evident after the effluent irrigation has stopped and land use has changed (Halliwell et al., 2001). Accordingly, the total electrolyte concentration may begin to decline once effluent irrigation stops and ESP may not return to pre-irrigation levels. Soil dispersion may occur even at low ESP values ($\leq$5.0) if the electrolyte concentration declines (Sumner, 1993). In sodic soils, mechanical disturbance such as cultivation followed by rainfall may increase the mobility of dispersed clay resulting in reduced permeability of the soil (Balks et al., 1998).

## Conclusions

Effluent irrigation is expected to result in salt accumulation and degradation of the soil physical properties. However, the evidence from this study suggests that the soil in this site can be irrigated with dairy effluent at twice the potential evapotranspiration rate as shown by the lack of significant increase in critical soil properties such as EC and ESP after a year of effluent application. Except for *B. decumbens*, all the other forage species grew well whether at the 0.5 or 2.0 $ET_p$ dairy effluent irrigation rates indicating relatively good salt tolerance of these species. However, the high soil pH led to micronutrient deficiencies in the forage making supplemental micronutrient fertilization necessary. Effluent irrigation at the 2.0 $ET_p$ rate met the leaching (based on Rhoades equation) and actual water requirements for the soil in this study. Contrary to our initial expectations, salinity increase and sodicity were not observed in this effluent-irrigated soil. However, the duration of the experiment was relatively short (3 years), thus, additional monitoring is needed to ensure that salinity and sodicity problems do not occur or can be ameliorated before they cause plant growth reduction and/or soil degradation. Further monitoring of soil properties is needed even after the cessation of the experiment because the effects of salts may become evident after the land has been converted for other agricultural uses. Soil dispersion studies in effluent irrigated soil is recommended to verify the lack of salinity and sodicity occurrence.

# CHAPTER 5. BIOMASS PRODUCTION AND NUTRIENT REMOVAL OF TROPICAL GRASSES IRRIGATED WITH DAIRY EFFLUENT IN A TROPICAL ISLAND ENVIRONMENT


## Abstract

Effluent lagoons in dairy production farms can potentially overflow and pollute associated land and water bodies. Dairy producers in isolated environments need to find ways to re-use effluents in a sustainable and environment-friendly ways. Effluent irrigation for forage production is an attractive alternative to recycle the nutrients from this resource. This study assessed the effects of dairy effluent irrigation on biomass production and nutrient uptake by tropical grasses. Five tropical grasses— bana (*Pennisetum purpureum* S.), California (*Brachiaria mutica* S.), signal (*Brachiaria decumbens* S.), star (*Cynodon nlemfuensis* V.), and suerte (*Paspalum atratum* S.)—were subsurface (20 to 25 cm) drip irrigated with dairy effluent at two rates based on the potential evapotranspiration ($ET_p$) at the site (Waianae, Hawai'i)—2.0 $ET_p$ (7 to 44 mm $d^{-1}$) and 0.5 $ET_p$ (2 to 11 mm $d^{-1}$). Treatments were arranged in an augmented completely randomized design. Highest dry matter yield and nutrient uptake were obtained for *Brachiaria mutica* (43 to 57 Mg $ha^{-1}$ $y^{-1}$) and *P. purpureum* (50 to 51 Mg $ha^{-1}$ $y^{-1}$). *Paspalum atratum* yielded about 38 Mg dry matter $ha^{-1}$ $y^{-1}$ at both irrigation rates. While *P. purpureum* removed the most K (1700 to 2000 kg $ha^{-1}$ $y^{-1}$), *P. atratum* removed the most Mg (200 to 270 kg $ha^{-1}$ $y^{-1}$) from the soil. Nutrient removal of grasses was 30 to 187% N, 13 to 86% P and 2 to14% K of the applied effluent. When bahia grass (*Paspalum notatum*) forage critical levels were used as the reference, average annual N concentration was in the deficient range (1.51 to 2.39%), P was nearly at adequate levels

(0.20 to 0.42%), and K was above the sufficient level (2.5 to 4.2%). The calculated carrying capacity was 8 to 11 cows $ha^{-1}$ $y^{-1}$ using the dry matter yields of *B. mutica* and *P. purpureum* and milk production of a 9 kg $d^{-1}$ of 4% fat corrected milk for a 607-kg lactating cattle with dry matter intake of 2.1% of body weight. Forage quality such as CP (10 to 16%), NDF (52 to 62%) and ADF (28 to 36%) were also at levels acceptable for feeding to lactating dairy cattle. Supplemental fertilization with micronutrients (Zn, Fe, Cu and Mn) was necessary to sustain adequate levels of micronutrients and biomass productivity. As with any other irrigation system, a limitation was the frequent inability to irrigate with effluent due to heavy rains typical of the rainy season.

## Introduction

In Hawai'i, an open nutrient cycle in the milk production system was created by the high net influx of nutrients from imported feeds and the accumulation of large quantities of nutrient-rich wastewater or effluent. The application of effluent for forage production can recycle the nutrients from this renewable resource. Ideally, plants suited for removing nutrients from the effluent should produce high quantities of dry matter, be tolerant to prolonged wet conditions, and recover quickly for frequent harvest (Macoon et al., 2002). Various grasses and crops have been used for phytoremediation, especially for N and P removal, from wastewater and lagoons. For example, star grass (*C. nlemfuensis*), Bermuda grass (*Cynodon dactylon*) and Johnson grass (*Sorghum halepense)* were used for nutrient removal from swine effluent (McLaughlin et al., 2004; Adeli et al., 2005). Switch grass was planted as filter strip to remove nutrients from dairy effluent (Sanderson et al., 2001). *Paspalum notatum* responds well when fertilized with domestic wastewater (Adjei and Rechcigl, 2002). Various forage systems have also been tested for the effects of dairy effluent or slurry application on nutrient removal, crop yield and forage quality (Macoon et al., 2002; Woodard et al., 2002).

Many tropical grass species have high biomass productivity and nutrient removal (especially when herbage is harvested frequently), persistence, acceptable nutritive quality, tolerance to harsh environmental conditions, ease of establishment, and resistance to pests and diseases (Pant et al., 2004; Takahashi et al., 1996, Duke, 1983). Few studies are available that examine the potential of tropical grasses for utilization of nutrients from effluent. One study on wastewater re-use conducted in Hawai'i to remove N from secondarily treated domestic sewage effluent used *B. mutica.* The rapid, dense and monocultural growth as well as the relatively high consumptive water use (4 mm $d^{-1}$) of this grass led to removal of about 69% of the effluent N at application rates ranging from 475 to 2600 kg $ha^{-1}$ $y^{-1}$ (Handley and Ekern, 1981).

*Pennisetum purpureum*, with its abundant and extensive fine root system, has high dry matter yield and a good capacity to take up nutrients from the soil. In Hawai'i, *P. purpureum* produces up to 336 Mg $ha^{-1}$ $y^{-1}$ of green forage (Takahashi et al., 1966) or approximately 70 Mg $ha^{-1}$ $y^{-1}$ of dry matter. In Brazil, some cultivars such as the 'Mineiro' yielded 21 Mg and 30 Mg dry matter $ha^{-1}$ $y^{-1}$ in two rainy seasons and 3.3 Mg and 6 Mg dry matter $ha^{-1}$ $y^{-1}$ in two dry seasons (Duke, 1983). Average dry matter yields of *P. purpureum* in Malaysia were 25 Mg $ha^{-1}$ $y^{-1}$ (Idris and Najib, 2004) and 25 Mg to 40 Mg $ha^{-1}$ $y^{-1}$ in Vietnam (Nguyen et al., 2004). In regions with over 125 cm annual rainfall, dry matter forage yields of *P. purpureum* ranged from 27 to 37 Mg kg $ha^{-1}$ (Duke, 1983). Bogdan's (1977) review concluded that in practice, dry matter yields of *P. purpureum* are more likely to be only 2 to 10 Mg $ha^{-1}$ $y^{-1}$ for low or no fertilizers, and 6 to 30 Mg $ha^{-1}$ $y^{-1}$ for well-fertilized farms. These yields are well below the 85 Mg $ha^{-1}$ $y^{-1}$ that can be achieved under optimum conditions such as fertilizing with about 900 kg N $ha^{-1}$ $year^{-1}$ and cutting every 90 days under natural rainfall of about 2000 mm per year (Vicente-Chandler et al., 1959). *Pennisetum purpureum* in Tobago, Caribbean reportedly removed 463 kg N, 96 kg P and 594 kg K $ha^{-1}$ $y^{-1}$ (Walmsley et al., 1978). A recent study on the nutrient uptake of crop-forage systems in the Andean hillsides showed that *P. purpureum* acquired greater amounts of N, P, and K from the soil, although shoot Ca and Mg uptake was similar with imperial grass (*Axonopus scoparius* cv. Imperial) (Zhiping et al., 2004).

In lowland paddy pastures in Thailand, *P. atratum* yielded 18 to 25 Mg dry matter $ha^{-1}$ $y^{-1}$, which was higher than the yield of *B. mutica* (Phoatong and Phaikaew, 2001). In northeast Thailand, *B. mutica* and *P. atratum* produced an average of 20 Mg $ha^{-1}$ dry matter in the first year (Hare et al., 1999). In the first wet season, no significant differences in production were found between the two species and no production differences between 45-day and 60-day cutting intervals. In the second wet season following establishment, *P. atratum* produced nearly 30 Mg $ha^{-1}$ dry matter, which was approximately 10 Mg $ha^{-1}$ greater than *B. mutica*.

Similar to other highly productive forage species, *B. decumbens* has high productivity and nutrient uptake, especially when produced through a cut and carry system (Vicente-Chandler et al., 1974; Mislevy, 2002; Pant et al., 2004). Different cultivars of *B. decumbens* produced 9.5 Mg dry matter $ha^{-1}$ $y^{-1}$ in Costa Rica (Bustamante et al., 1998), 7 to 24 Mg $ha^{-1}$ $y^{-1}$ in Brazil (Duke, 1983), and 33 Mg $ha^{-1}$ $y^{-1}$ in Australia

(Register of Australian Herbage Plant Cultivars, 1990). In Vicente-Chandler et al.'s (1974) study in Puerto Rico, *B. decumbens* yielded 28 Mg $ha^{-1}$ $y^{-1}$, which removed 390, 65, 470, 150, and 55 kg $ha^{-1}$ $yr^{-1}$ of N, P, K, Ca, and Mg, respectively. In another study, an average of 140 kg K $ha^{-1}$ was removed annually by tropical *B. decumbens* when no N was applied, compared with 500 kg K $ha^{-1}$ when 450 kg N $ha^{-1}$ $yr^{-1}$ was applied (Vicente-Chandler and Pearson, 1960).

In recent years, there has been a renewed interest in the use of wastewater for growing crops or irrigating golf courses and turf grasses, all motivated by environmental objectives. Little attention has been given to utilization of wastewater for forage production to increase efficiency in animal production system or for biomass production objective while conserving the environment. Also, most of the studies were conducted on U.S. continental soils and grass species were quite different from those grown in Hawaiian pastures. New information on the performance of different tropical grasses in soils receiving animal effluents must, therefore, be developed so that animal effluent application for forage production can be considered as a mode of re-using this important resource. This study assessed the effects of dairy effluent application on nutrient uptake, productivity and forage quality of the different tropical forages irrigated based on the $ET_p$.

## Methodology

The soil, climate, and other descriptions of the site as well as the treatment selection and experimental design were presented in Chapter 3. Laboratory analyses of dairy effluent, soil solution, and soil samples were also presented in Chapter 3.

**Plant Tissue Analyses**

The annual biomass production and nutrient uptake (average of 2-year annual data) were calculated from the harvest and tissue nutrient data. Nutrient uptake was calculated as the product of dry matter yield (kg $ha^{-1}$ $y^{-1}$) and tissue nutrient concentration (%). Nutrient removal as percent of the applied nutrient was calculated as the nutrient uptake divided by the annual applied nutrient. Seasonal changes (warm and cool season) in biomass production and nutrient uptake were also presented. The cool season was from October to March. During this period, rainfall usually exceeded $ET_p$, and temperature and solar radiation were generally lower compared to warm season.

The forage was analyzed for macro- (N, P, K, Ca, Mg, Na) and micronutrients (Fe, Mn, Zn, Cu). Forced-draft dried samples were ground and a 0.50 g sample was dry-ashed in a porcelain crucible for 4 to 6 h at 500°C in a muffle furnace (Hue et al., 1997). If the ashing appeared incomplete, the ash was cooled, digested in 1*M* nitric acid, evaporated to dryness and ashed again for one hour. Residues were dissolved in 25 mL of 1*M* hydrochloric acid The solution was then subjected to ICP spectrophotometry (Thermo Jarrell Ash[†], Franklin, MA). Ashed samples were analyzed by the azomethine-H colorimetric method for B determination (Wolf, 1974). Total N was analyzed using a LECO CN-2000[†] analyzer (LECO Corporation, St. Joseph, MI). A 0.25 g sample was mixed with 2 g of $Na_2SO_4$ and 7 mL of a digestion mixture of concentrated $H_2SO_4$, salicylic acid and selenium. After about 2 h, 3 to 4 drops of sodium thiosulfate solution were added. The mixture was left standing for 45 min, then 4 mL of 30% $H_2O_2$ was added. The mixture was digested at 410°C until a clear liquid was obtained. The cooled liquid was then diluted with water prior to analyses for N. Nitrate-N and $NH_4^+$-N were determined using an auto-analyzer using Scientific AP200† and Scientific AC200† spectrophotometers (Gentry and Willis, 1988).

The protein feeding value of the forage was estimated using the crude protein (CP). Crude protein content was estimated based on the standard conversion: CP (%) = N(%) x 6.25. Acid Detergent Fiber (ADF) and Neutral Detergent Fiber (NDF) were determined using the ANKOM Filter Bag Technique described in ANKOM Application Note 01/02 "Method for Determining Acid/Neutral Detergent Fiber" (ANKOM Technology Corp., Fairport, NY). In this method, 0.60 g of tissue sample (dried and ground) were weighed directly in the filter bag and placed in ANKOM 200 Fiber Analyzer for extraction. After extraction, the bags were weighed in the oven (102°C) for 2 to 4 h. The NDF was calculated:

$$\% NDF \; or \; \% ADF = \frac{W_3 - (W_1 * C_1)}{Sample\, weight} x100 \qquad \text{(Eq. 1)}$$

where $W_1$ is the tare weight of the bag (g), $W_3$ is the dried weight of the bag with fiber after the extraction process, and $C_1$ is the blank bag correction, which is the final oven dry weight divided by the original blank bag weight. The reagents for the NDF were a neutral detergent solution, α-amylase and sodium sulfite ($Na_2SO_3$), whereas for the ADF an acid detergent solution was used.

**Data Analysis**

The main treatments and interaction effects in annual dry matter, nutrient concentration, nutrient uptake and forage quality data were analyzed as an unbalanced ANOVA using PROC GLM of SAS 9.1 software (SAS Inst., 2004). For the monthly data, best subset procedures were conducted using MINITAB 14.13 software (Minitab Inc., 2004) to determine the variables that are the best predictors of the dependent variable. The main treatments, interaction effects and the dynamics in yield and nutrient uptake and removal (reduced model) were analyzed with Repeated Measures ANOVA (the autoregressive first order covariance structure was selected) using PROC MIXED SAS 9.1 software (SAS Inst., 2004).

Data were plotted using SigmaPlot 9.0 software (SYSTAT Software Inc., 2004). Correlation analyses were performed to determine if relationships exist between any two variables. The standard error of the mean of the replicated treatments is presented in the graphs and tables where applicable. To facilitate the data analysis and discussion, the "time" variable was used to refer to sampling dates that corresponded to the varying amount of effluent irrigation over time at each of the two irrigation rates.

## Results and Discussion

### Annual Dry Matter Production

The tropical grasses used in effluent irrigation exhibited relatively slow growth rates during the first 15 to 20 days, after which these species grew rapidly (Fig 5.1). Most of these grasses produced very high dry matter with effluent irrigation (Figs. 5.2 and 5.3 and Tables 5.1 and 5.5) that compared well or were higher than those commonly reported in the literature for these species. Dry matter production varied among the grasses ($P<0.01$), but not between irrigation rates ($P>0.10$). No interaction between the grass species and irrigation rate ($P>0.10$) was detected. *Brachiaria mutica* receiving the 2.0 $ET_p$ effluent irrigation rate outyielded all the other grass species, with dry matter production reaching nearly 60 Mg $ha^{-1}$ $y^{-1}$ (Table 5.1). *Pennisetum purpureum* irrigated at 2.0 $ET_p$ produced the second highest dry matter (51 Mg $ha^{-1}$ $y^{-1}$). Both grasses also yielded the highest dry matter at the 0.5 $ET_p$ rate (43 and 50 Mg $ha^{-1}$ $y^{-1}$, respectively). Averaging the respective dry matter production of these grasses at the two irrigation rates, dry matter production of *Pennisetum purpureum* and *Brachiaria mutica* were similar (50 Mg $ha^{-1}$ $y^{-1}$).

*Paspalum atratum* yielded the next highest dry matter (39 Mg $ha^{-1}$ $y^{-1}$), which was less than *P. purpureum* and *B. mutica* ($P$ <0.05). Although dry matter production of *P. atratum* was somewhat lower at the 2.0 $ET_p$ irrigation rate, there were no significant differences ($P$ >0.10) observed between *P. atratum* yields irrigated at the 0.5 $ET_p$ (40 Mg $ha^{-1}$ $y^{-1}$) and 2.0 $ET_p$ rates (38 Mg $ha^{-1}$ $y^{-1}$). The yields were not different from that of *C. nlemfuensis* ($P$ >0.10). At the 2.0 $ET_p$ rate, *C. nlemfuensis* produced 40 Mg dry matter $ha^{-1}$ $y^{-1}$ and at 0.5 $ET_p$ rate, 35 Mg $ha^{-1}$ $y^{-1}$.

At the 2.0 $ET_p$ and 0.5 $ET_p$ irrigation rates, *B. decumbens* initially grew well. However, this grass species eventually turned chlorotic after three months of irrigation and had much reduced growth that resulted in low dry matter production. By the fifth month, this grass was unproductive, suggesting that it cannot tolerate prolonged effluent irrigation and the high soil pH and salts at the site. *Brachiaria decumbens* was reportedly well-adapted to an acid conditions (Bogdan, 1977) and has low salinity tolerance (Deifel et al., 2006). The dry matter production was 8 Mg $ha^{-1}$ $y^{-1}$ at the 2.0 $ET_p$ irrigation rate, and 10 Mg $ha^{-1}$ $y^{-1}$ at the 0.5 $ET_p$ rate.

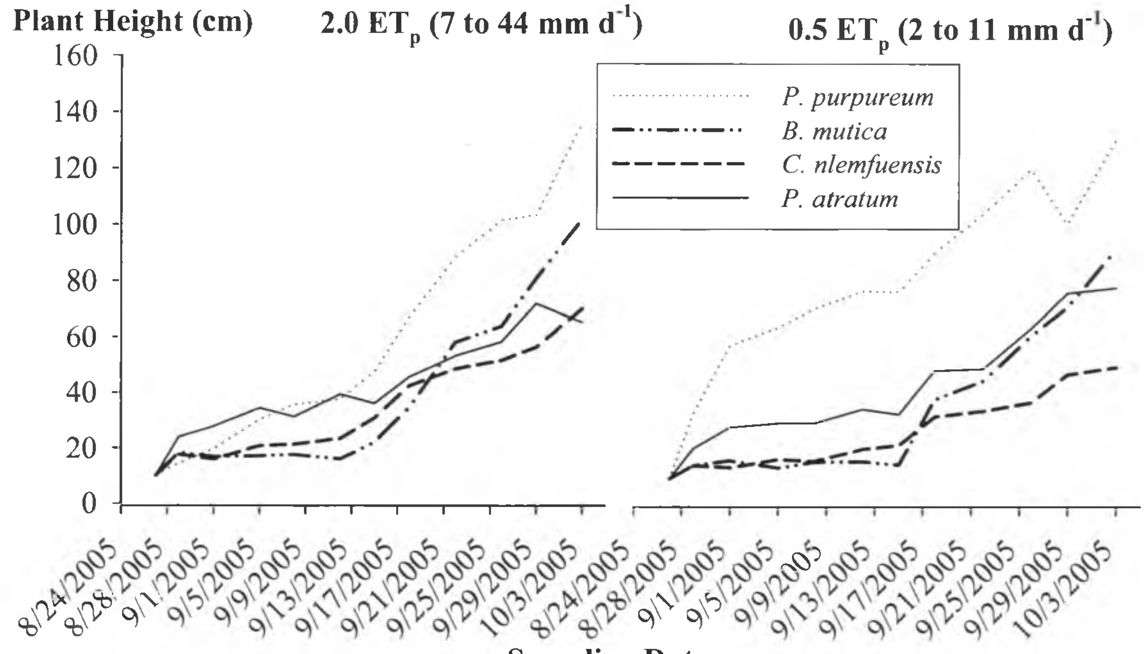
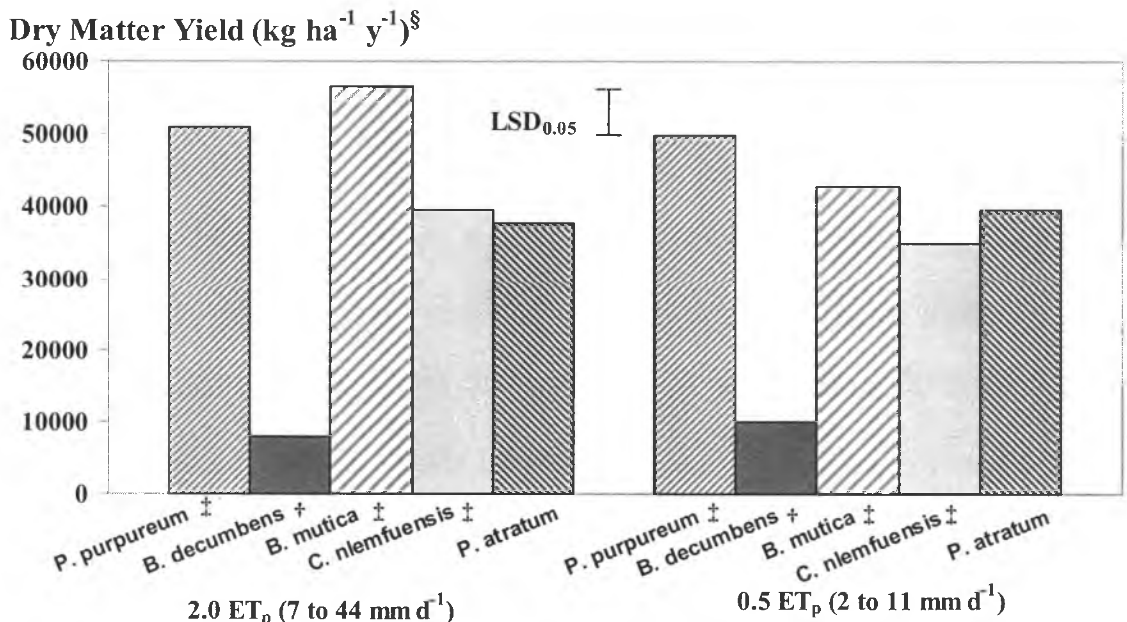



**Figure 5.1. Growth curves for tropical grasses irrigated with dairy effluent at 2.0 $ET_p$ and 0.5 $ET_p$ rates, O'ahu, Hawai'i.**




†Total of 4 monthly harvests in 2004; ‡Augmented treatments; §Average of 2-y annual data

**Figure 5.2. Average dry matter production of tropical grasses irrigated with dairy effluent, Waianae, O'ahu, Hawai'i.**

**N, P and K Uptake**

The N, P, and K uptake of the grasses differed between species ($P<0.01$) but not between irrigation rates ($P >0.10$). No significant interaction occurred between grasses and irrigation rate ($P >0.10$). The tissue N concentrations of the grasses (Table 5.1) were low compared with the adequate level (2.8%) proposed for *P. notatum* (Mills and Jones, 1993). Except for *B. decumbens* and *C. nlemfuensis* receiving 0.5 $ET_p$ irrigation rate, tissue P concentrations of all the other grasses were lower than the adequate level (0.40%) suggested for *P. notatum*. Tissue K concentrations of the grasses exceeded the adequate level (1.8%), with *P. purpureum* having the highest (3.7 to 4.2%). Among the grasses, *P. purpureum* and *B. mutica* had consistently the highest N, P, K uptake (Table 5.1 and Figs. 5.4 to 5.6). At the 2.0 ETp irrigation rate, *P. purpureum* removed 1190 kg N $ha^{-1}$ $y^{-1}$, 153 kg P $ha^{-1}$ $y^{-1}$ and 2008 kg K $ha^{-1}$ $y^{-1}$, while *B. mutica* removed 1262 kg N $ha^{-1}$ $y^{-1}$, 157 kg P $ha^{-1}$ $y^{-1}$ and 1922 kg K$ha^{-1}$ $y^{-1}$. At the 0.5 ETp irrigation rate, *P. purpureum* removed 934 kg N $ha^{-1}$ $y^{-1}$, 158 kg P $ha^{-1}$ $y^{-1}$ and 1730 kg K $ha^{-1}$ $y^{-1}$ while *B. mutica* removed 743 kg N $ha^{-1}$ $y^{-1}$, 116 kg P $ha^{-1}$ $y^{-1}$ and 1267 kg K $ha^{-1}$ $y^{-1}$. These rates were much higher than the nutrient uptake of *P. purpureum* reported in West Indies (Walmsley et al., 1978), Andean Hillsides (Zhiping et al., 2004) and Kenya (Odongo, 2002).

**Table 5.1. Annual dry matter production, average macronutrient concentration and annual total macronutrient uptake of various tropical grasses receiving dairy effluent, Waianae, O'ahu, Hawai'i, Aug 2004 to Aug 2006.**

| Irrigation Rate | Grass Species | Dry Matter | Concentration | | Uptake | | N:P Uptake Ratio |
|---|---|---|---|---|---|---|---|
| | | | N | P | N | P | |
| | | Mg ha$^{-1}$ y$^{-1}$ | ----------%---------- | | -----kg ha$^{-1}$ y$^{-1}$----- | | |
| 2.0 $ET_p$ (7 to 44 mm d$^{-1}$) | *P. purpureum* | 51 | 2.12 | 0.30 | 1190 | 153 | 7.38 |
| | *B. decumbens* | 8 | 1.60 | 0.27 | 114 | 21 | 6.47 |
| | *B. mutica* | 57 | 2.39 | 0.29 | 1262 | 157 | 8.26 |
| | *C. nlemfuensis* | 40 | 2.10 | 0.32 | 823 | 124 | 6.76 |
| | *P. atratum* | 38 | 1.51 | 0.20 | 551 | 85 | 8.14 |
| 0.5 $ET_p$ (2 to 11 mm d$^{-1}$) | *P. purpureum* | 50 | 1.89 | 0.33 | 934 | 158 | 5.78 |
| | *B. decumbens* | 10 | 2.06 | 0.42 | 194 | 41 | 4.98 |
| | *B. mutica* | 43 | 1.75 | 0.28 | 743 | 116 | 6.60 |
| | *C. nlemfuensis* | 35 | 1.86 | 0.40 | 667 | 133 | 5.06 |
| | *P. atratum* | 40 | 1.52 | 0.24 | 584 | 98 | 6.66 |

| Irrigation Rate | Grass Species | Concentration | | | Uptake | | |
|---|---|---|---|---|---|---|---|
| | | K | Ca | Mg | K | Ca | Mg |
| | | ----------------%---------------- | | | ------------kg ha$^{-1}$ y$^{-1}$------------ | | |
| 2.0 $ET_p$ (7 to 44 mm d$^{-1}$) | *P. purpureum* | 4.19 | 0.32 | 0.30 | 2008 | 182 | 155 |
| | *B. decumbens* | 2.62 | 0.25 | 0.44 | 198 | 18 | 33 |
| | *B. mutica* | 3.52 | 0.20 | 0.23 | 1922 | 110 | 125 |
| | *C. nlemfuensis* | 2.54 | 0.30 | 0.27 | 1022 | 112 | 106 |
| | *P. atratum* | 2.64 | 0.39 | 0.61 | 1051 | 139 | 210 |
| 0.5 $ET_p$ (2 to 11 mm d$^{-1}$) | *P. purpureum* | 3.65 | 0.37 | 0.32 | 1730 | 197 | 177 |
| | *B. decumbens* | 3.00 | 0.28 | 0.37 | 291 | 28 | 35 |
| | *B. mutica* | 3.05 | 0.23 | 0.29 | 1267 | 98 | 123 |
| | *C. nlemfuensis* | 2.52 | 0.39 | 0.28 | 874 | 133 | 99 |
| | *P. atratum* | 2.49 | 0.49 | 0.72 | 1035 | 182 | 266 |

**Table 5. 2. Average nutrient concentration and annual total micronutrient uptake of selected tropical grasses receiving dairy effluent, Waianae, O'ahu, Hawai'i, Aug 2004 to Aug 2006.**

| Irrigation Rate | Grass Species | Concentration | | | | | Uptake | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | Fe | Mn | Zn | Cu | B | Fe | Mn | Zn | Cu | B |
| | | ---------------mg $kg^{-1}$------------- | | | | | ----------kg $ha^{-1}$ $y^{-1}$----------- | | | | |
| 2.0 $ET_p$ (7 to 44 mm $d^{-1}$) | *P. purpureum* | 416 | 56 | 68 | 13 | 20 | 18 | 3 | 2.8 | 0.6 | 1.1 |
| | *B. decumbens* | 1503 | 92 | 21 | 10 | 21 | 11 | 1 | 0.1 | 0.1 | 0.2 |
| | *B. mutica* | 199 | 80 | 67 | 11 | 16 | 11 | 4 | 3.2 | 0.6 | 0.9 |
| | *C. nlemfuensis* | 405 | 105 | 53 | 9 | 11 | 13 | 4 | 2.2 | 0.3 | 0.4 |
| | *P. atratum* | 1244 | 107 | 88 | 11 | 32 | 32 | 3 | 2.3 | 0.4 | 1.0 |
| 0.5 $ET_p$ (2 to 11 mm $d^{-1}$) | *P. purpureum* | 615 | 56 | 51 | 11 | 16 | 28 | 3 | 2.6 | 0.6 | 0.9 |
| | *B. decumbens* | 1489 | 91 | 36 | 12 | 13 | 14 | 1 | 0.3 | 0.1 | 0.1 |
| | *B. mutica* | 252 | 63 | 68 | 10 | 12 | 11 | 3 | 2.6 | 0.4 | 0.5 |
| | *C. nlemfuensis* | 387 | 66 | 56 | 11 | 12 | 13 | 2 | 2 | 0.3 | 0.4 |
| | *P. atratum* | 1108 | 98 | 96 | 10 | 25 | 35 | 3 | 2.6 | 0.4 | 0.9 |

High K uptake of *P. purpureum* was observed in our study of tropical forage species fertilization conducted in Mealani Experiment Station, Island of Hawai'i (Table 5.3). Potassium is very important in dairy cattle diet, being a major mineral component of milk. In general, however, cattle require low amounts of K—a minimum of 1.0% or if under heat stress, up to 1.5%, of the total ration dry matter (National Research Council, 2001). Although the K concentration in the dry matter of this forage was relatively high (3.88%), it can be balanced with other components of the ration to achieve the acceptable K concentration. Some studies have implicated the high level of K in forage to milk fever or *parturient paresis*, an abnormal physiological event in lactating cattle characterized by rapid reduction in plasma Ca concentrations (Horst et al., 1997; Goff and Horst, 1997).

**Table 5.3. Dry matter yield, crude protein content and nutrient uptake of various tropical grasses grown with inorganic fertilization, Mealani Experiment Station, Kamuela, Hawaii.**

| Grass Species | Year | DM | CP | Nutrient Uptake | | |
|---|---|---|---|---|---|---|
| | | | | N | P | K |
| | | kg ha$^{-1}$ yr$^{-1}$ | % | --------kg ha$^{-1}$ yr$^{-1}$---------- | | |
| *B. mutica* | 1997 | 22002 | 6 | 202 | 37 | 177 |
| | 1998 | 6237 | 7 | 73 | 12 | 37 |
| | 2002 | 23544 | 13 | 492 | 50 | 692 |
| | 2003 | 17982 | 12 | 346 | 38 | 510 |
| *P. purpureum var. Bana* | 1997 | 19882 | 10 | 318 | 28 | 128 |
| | 1998 | 19835 | 12 | 366 | 37 | 84 |
| | 2002 | 40611 | 13 | 840 | 95 | 1395 |
| | 2003 | 32448 | 10 | 529 | 71 | 915 |
| *P. purpureum cv. Mott* | 1997 | 10680 | 12 | 207 | 22 | 223 |
| | 1998 | 9017 | 12 | 171 | 23 | 119 |
| | 2002 | 27122 | 17 | 735 | 76 | 1119 |
| | 2003 | 13190 | 14 | 289 | 33 | 457 |
| *P. glaucum x P. purpureum cv. Mott* | 1997 | 12569 | 9 | 176 | 28 | 197 |
| | 1998 | 8707 | 9 | 129 | 25 | 104 |
| | 2002 | 19027 | 15 | 448 | 49 | 603 |
| | 2003 | 23000 | 11 | 397 | 55 | 636 |
| *P. clandestinum* | 1997 | 9018 | 8 | 113 | 18 | 89 |
| | 1998 | 3022 | 9 | 43 | 6 | 34 |
| | 2002 | 11218 | 15 | 272 | 29 | 411 |
| | 2003 | 12030 | 10 | 201 | 29 | 376 |
| *C. nlemfuensis* | 1997 | 13956 | 7 | 154 | 24 | 172 |
| | 1998 | 2721 | 10 | 42 | 5 | 36 |
| | 2002 | 10596 | 13 | 229 | 24 | 247 |
| | 2003 | 9707 | 14 | 210 | 20 | 198 |
| *P. atratum* | 1997 | 11037 | 8 | 138 | 11 | 99 |
| | 1998 | 2473 | 8 | 31 | 4 | 22 |
| | 2002 | 20021 | 9 | 302 | 33 | 425 |
| | 2003 | 18065 | 9 | 255 | 28 | 359 |
| *P. virgatum* | 1997 | 24825 | 8 | 322 | 30 | 127 |
| | 1998 | 7704 | 8 | 102 | 14 | 38 |
| | 2002 | 23829 | 11 | 420 | 50 | 494 |
| | 2003 | 22301 | 11 | 403 | 42 | 444 |

The N, P, and K uptake of *P. atratum* and *C. nlemfuensis* were substantially lower ($P$ <0.05) compared with *P. purpureum* and *B. mutica* (Table 5.1). Although *P. atratum* appeared to grow better at the 2.0 $ET_p$ irrigation rate, it was also observed to be sensitive to prolonged effluent-saturated soil conditions. This is in contrast with its adaptation to wet, lowland areas in Thailand (Phoatong and Phaikaew, 2001). Although the N, P and K uptake of *C. nlemfuensis* were not different from that of *P. atratum*, the former produced slightly higher average N and P uptake (745 and 129 kg $ha^{-1}$ $y^{-1}$, respectively), while it was less than *P. atratum* in terms of K uptake (948 kg $ha^{-1}$ $y^{-1}$). *Cynodon nlemfuensis* is highly responsive to N fertilization (Sotomayor-Rios et al., 1976; Garay et al., 2004) and a concern is that prussic or hydrocyanic acid, which may be harmful to the cattle, is produced when excess N is supplied. In Florida, poisoning did not occur on cattle that graze on *C. nlemfuensis* (Mislevy, 2002).

*Brachiaria decumbens* yielded the least dry matter yield and N, P and K uptake, and exhibited the poorest growth especially at the 2.0 $ET_p$ application rate (Figs. 5.2 to 5.6). The average N, P and K uptake of *B. decumbens* was 155, 31 and 244 kg $ha^{-1}$ $y^{-1}$, respectively. This grass has very low tolerance to waterlogged conditions and high soil salinity (Humphreys and Partridge, 1995). The declining growth of *B. decumbens* resulted in relatively poor groundcover and severe weed competition. At the 0.5 $ET_p$ irrigation rate, *B. decumbens* grew better as indicated by its higher dry matter yield and nutrient uptake than at the 2.0 $ET_p$ rate (Figs. 5.2 to 5.6). This species is known to be drought-tolerant and adapt well in quick-drying soils (Humphreys and Partridge, 1995; Fukumoto and Lee, 2002).

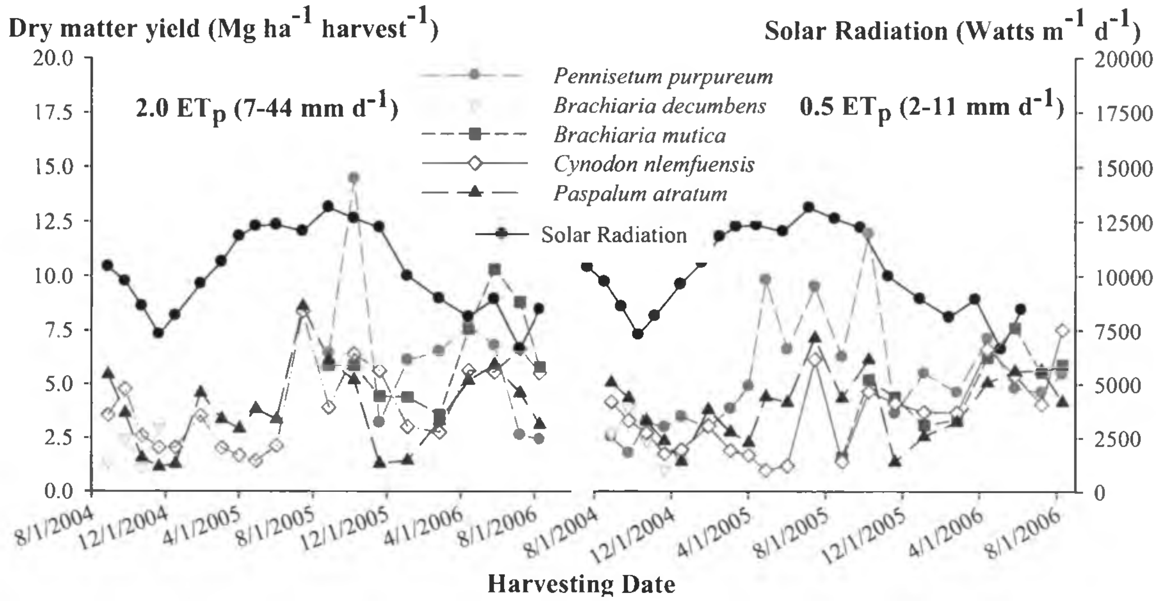



**Figure 5.3. Dry matter yield (kg ha$^{-1}$ y$^{-1}$) of tropical grasses irrigated with dairy effluent and solar radiation (Watts m$^{-2}$), Waianae, Hawai'i, Aug 2004 to Aug 2006.**

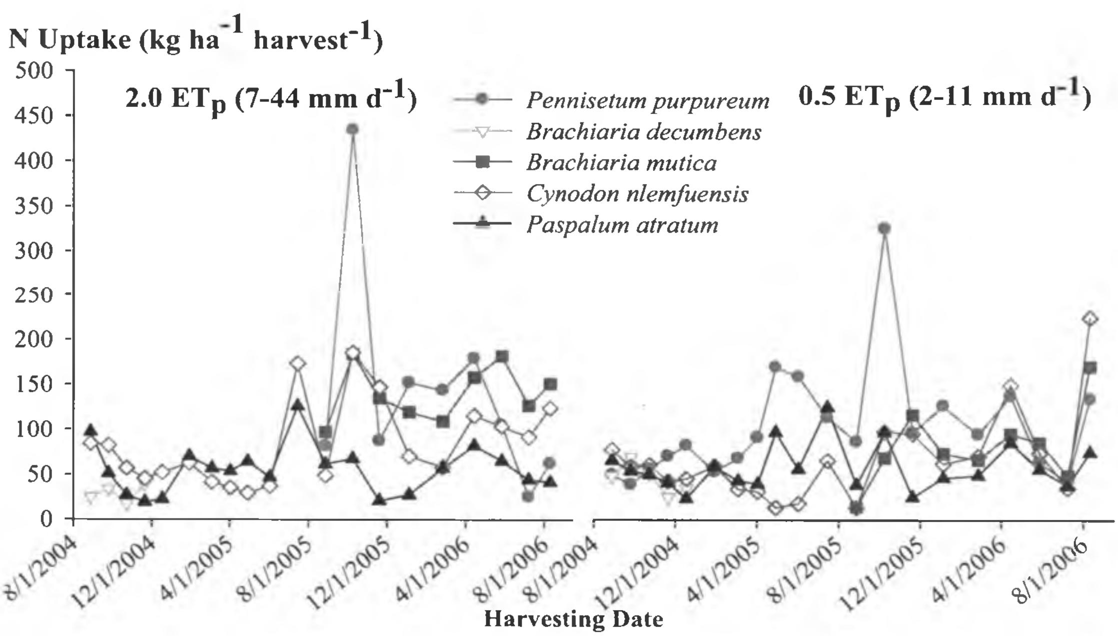



**Figure 5.4. Nitrogen uptake of tropical grasses irrigated with dairy effluent, Waianae, O'ahu, Hawai'i, Aug 2004 to Aug 2006.**

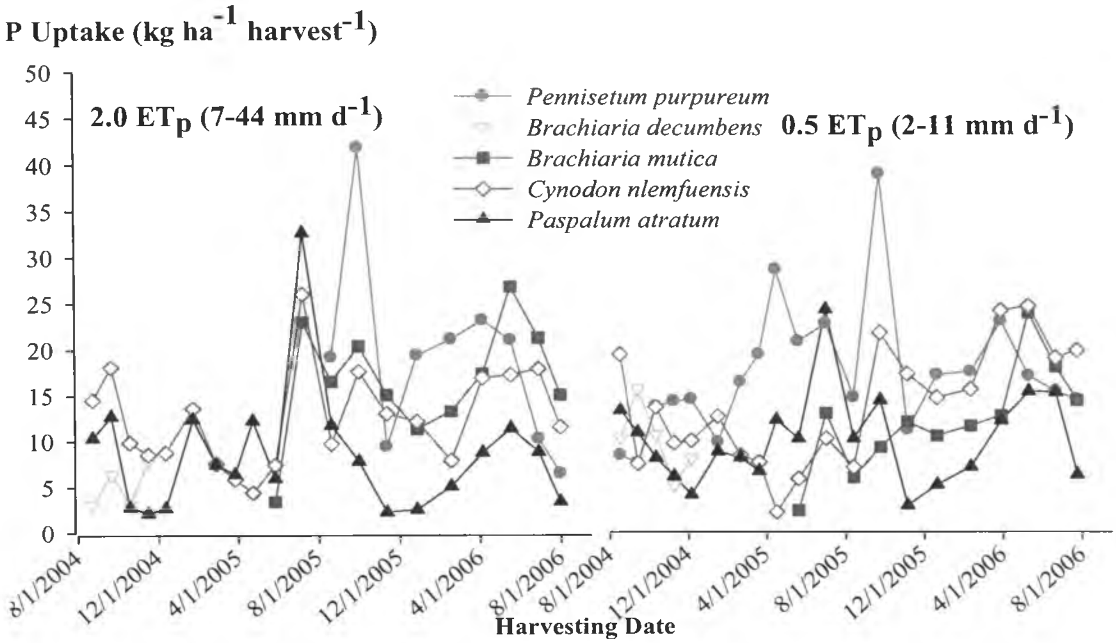



**Figure 5.5. Phosphorus uptake of tropical grasses irrigated with dairy effluent, Waianae, O'ahu, Hawai'i, Aug 2004 to Aug 2006.**

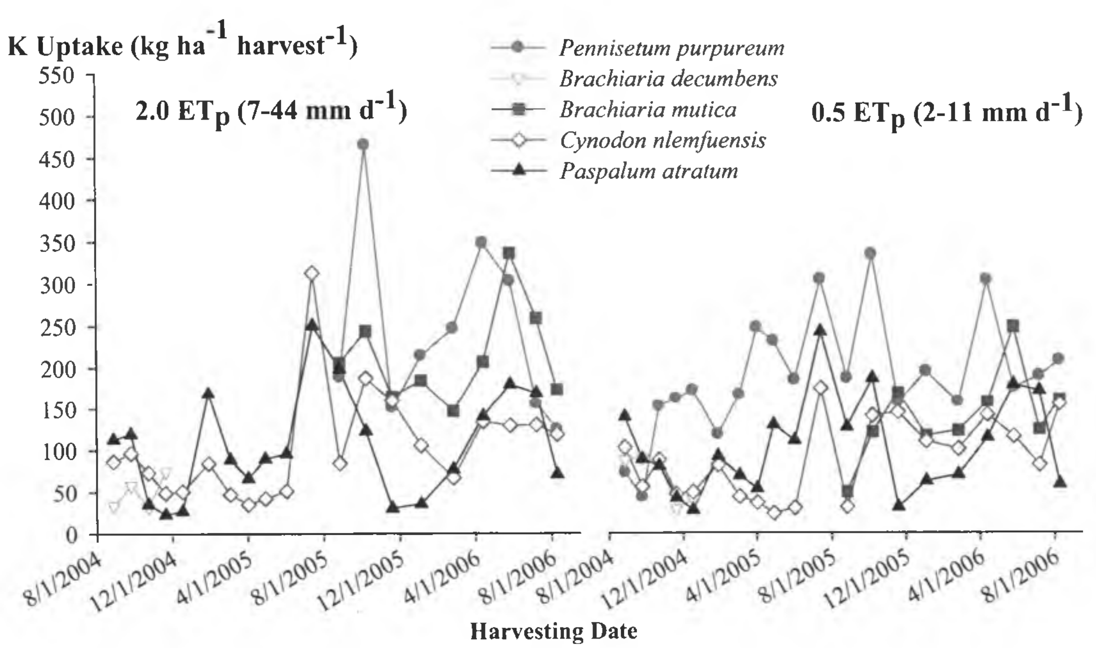



**Figure 5.6. Potassium uptake of tropical grasses irrigated with dairy effluent, Waianae, O'ahu, Hawai'i, Aug 2004 to Aug 2006.**

**Uptake of Other Nutrients**

Micronutrient availability appeared to limit the productivity of the grasses beginning on the fourth month after the application of effluent treatments during the first year as indicated by leaf chlorosis. When micronutrients (Fe, Mn, Zn, Cu) were applied as foliar fertilizer, the chlorosis became very minimal and there were improvements in the micronutrient levels in the forage. Uptake of Ca, Mg, Mn, Zn, Cu and B differed between grasses ($P < 0.01$) (Tables 5.1 and 5.2 and Figs. 5.7 to 5.9). The main effect of irrigation rate and its interaction with the grass species was not significant ($P > 0.10$). Adequate levels of other nutrients aside from N, P and K for *P. notatum* forage were (Mills and Jones, 1993): Ca, 0.52%; Mg, 0.32%; Fe, 100 mg $kg^{-1}$; Mn, 105 mg $kg^{-1}$; Zn, 31 mg $kg^{-1}$; Cu, 11 mg $kg^{-1}$, and B, 9 mg $kg^{-1}$ (Mills and Jones, 1993). Using these levels as a point of comparison, tissue concentrations in all the grasses were sufficient for Ca, Mg, B and Cu and deficient to adequate for Mn and Zn. Although the Fe content of the tissues differed among the grasses ($P < 0.05$), the Fe uptake was not different among the grasses ($P > 0.10$). The Ca and Fe content of the tissues (average of 2.5 to 4.2% and 230 to 1500 mg $kg^{-1}$, respectively) were much higher than the adequate level possibly due to contamination of tissue samples by the soil despite efforts to minimize such contamination. The sufficient or excessive levels of micronutrients could also be explained by the supplementary foliar micronutrient application.

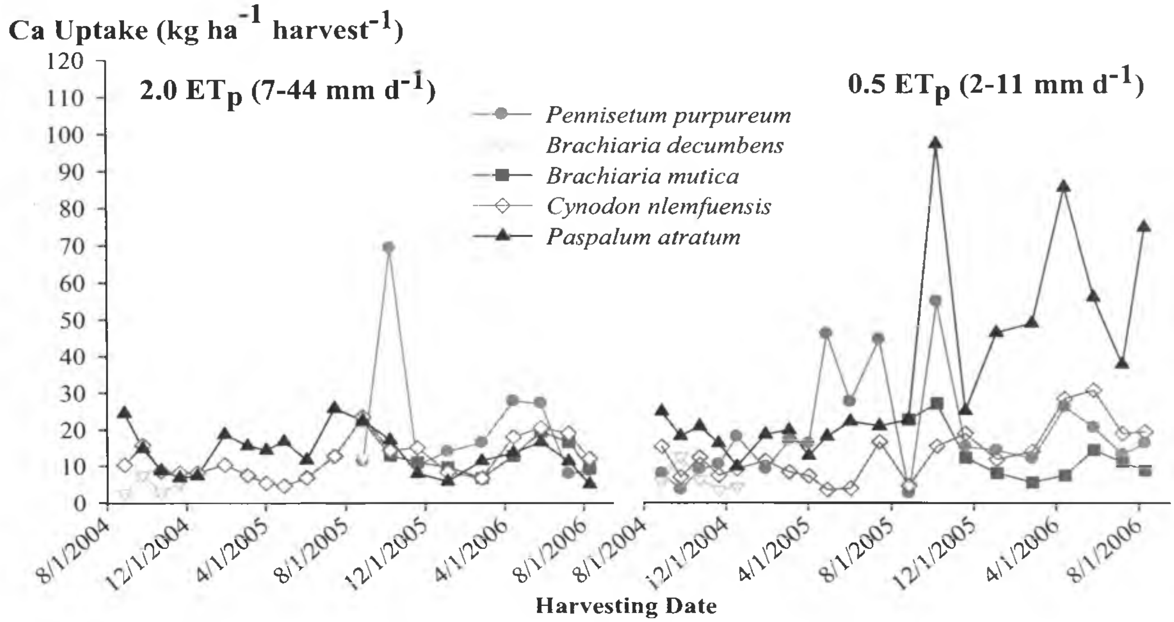



**Figure 5.7. Calcium uptake of tropical grasses irrigated with dairy effluent, Waianae, O'ahu, Hawai'i, Aug 2004 to Aug 2006.**

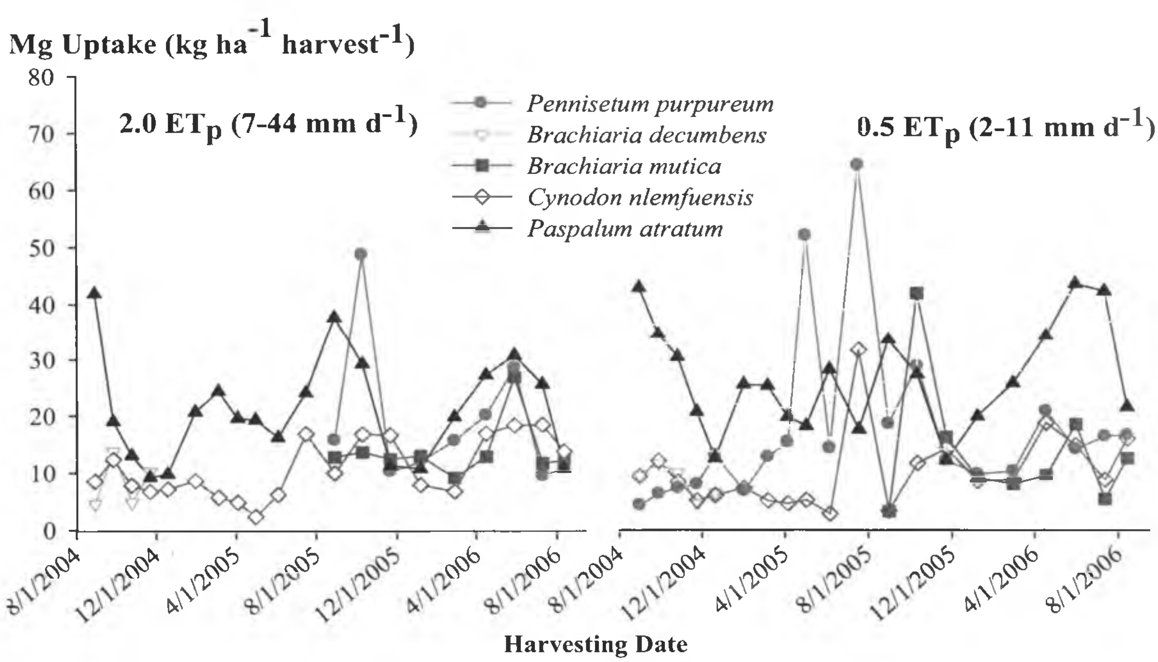



**Figure 5.8. Magnesium uptake of tropical grasses irrigated with dairy effluent, Waianae, O'ahu, Hawai'i, Aug 2004 to Aug 2006.**

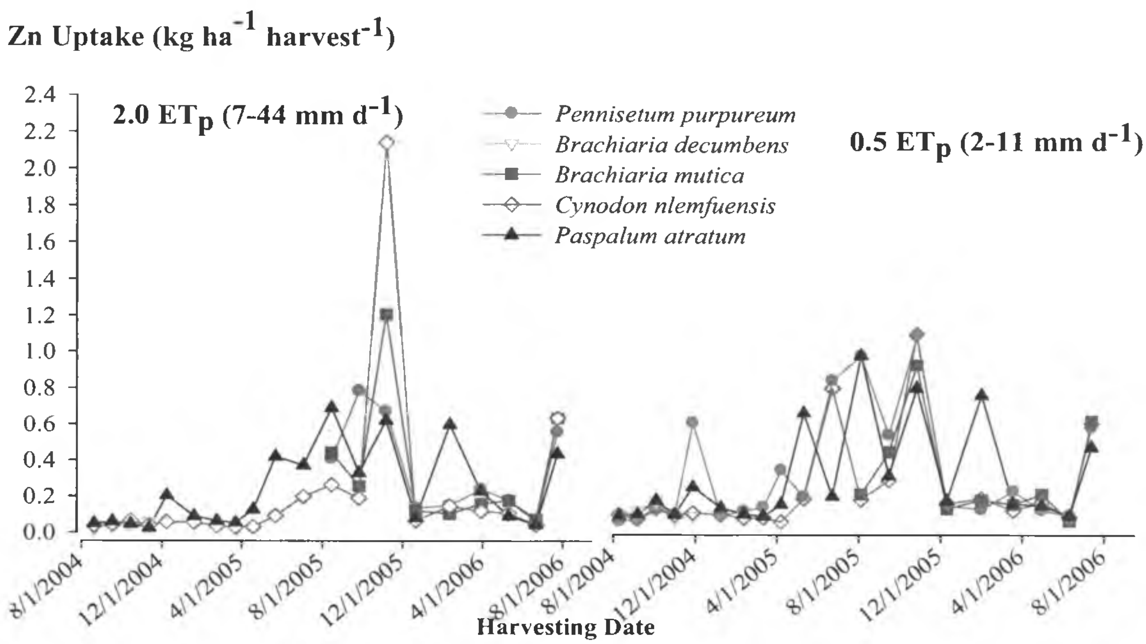



**Figure 5.9. Zinc uptake of tropical grasses irrigated with dairy effluent, Waianae, O'ahu, Hawai'i, Aug 2004 to Aug 2006.**

Among the grasses, *P. atratum* had the highest uptake of Mg at 210 and 266 kg ha$^{-1}$ y$^{-1}$ for the 2.0 ET$_p$ and 0.5 ET$_p$ irrigation rates, respectively (Table 5.1). This grass species and *P. purpureum* receiving 0.5 ET$_p$ irrigation rate had the highest Ca uptake (224 and 226 kg ha$^{-1}$ y$^{-1}$, respectively). *Paspalum atratum* had an average annual Mg concentration in the dry matter of 0.61% for the 2.0 ET$_p$ irrigation rate and 0.72% for the 0.5 ET$_p$ rate, and Ca concentration of 0.39% and 0.49%, respectively. The recommended Mg concentration in a dairy cattle ration is from 0.25% to 0.35%, and slightly higher for early lactating, high producing cows (0.45%) (Harris et al., 1994), but the Mg level in the forage should not be lower than 0.20% to prevent grass tetany (Voisin, 1963). Grass tetany is exhibited by cattle fed with very low Mg feed, relative to other salts such as K, Ca and Na (Voisin, 1963). The high Mg content of *P. atratum* forage is, therefore, favorable for avoiding the occurrence of grass tetany in dairy cattle.

The uptake of Mg, Ca and nitrate may be suppressed by the high levels of soil solution K (Barber, 1995), but no strong correlation existed between them in this study (Pearson correlation coefficients <0.13). There was no correlation between nutrient concentration in soil and uptake of nutrients (Pearson correlation coefficients <0.23) and nutrient concentration in soil solution and uptake of nutrients (Pearson correlation coefficients <0.24).

**Nutrient Removal**

The tropical grasses differed in their ability to remove nutrients ($P < 0.01$) expressed as percent of applied nutrient from effluent (i.e., total annual nutrient uptake divided by cumulative annual nutrient applied). The grasses irrigated at 0.5 $ET_p$ removed more nutrients (nearly four times) than when irrigated at 2.0$ET_p$, primarily because the cumulative amount of nutrients applied at 2. $ET_p$ was much larger ($P < 0.01$) (Table 5.3). The nutrient removal of grasses irrigated at the 0.5 $ET_p$ rate ranged from 67% to 187% for N and 50 to 86% for P. Grasses irrigated at 2.0 $ET_p$ removed 30 to 55% of the N and 13 to 22% of the P applied (Table 5.3).

Among the grasses irrigated at 0.5 $ET_p$, *P. purpureum* consistently removed the highest amounts of N (187%) and P (86%), followed by *B. decumbens* (145% N and 77% P) (Table 5.3). *Brachiaria mutica* removed the highest percentage of applied nutrient (55% N and 22% P) among the grasses that received the 2.0 $ET_p$ rate. Removal percentages of K (2 to14%), Ca (3 to 26%) and Mg (2 to18%) were rather low, implying a possible accumulation or leaching of these nutrients in the soil. This has important implications on the possible Ca-P precipitation and eventual accumulation of these nutrients in the soil, and K leaching in the soil profile.

**Table 5.4. Macronutrient removal (as percent of applied nutrients from effluent) of various tropical grasses receiving dairy effluent, Waianae, O'ahu, Hawai'i, Aug 2004 to Aug 2006.**

| Irrigation Rate | Grass Species | Macronutrient Removal | | | | |
|---|---|---|---|---|---|---|
| | | N | P | K | Ca | Mg |
| | | % | | | | |
| 2.0 $ET_p$ (7 to 44 mm $d^{-1}$) | *P. purpureum* | 52 | 21 | 4 | 4 | 2 |
| | *B. decumbens* | 30 | 14 | 3 | 3 | 3 |
| | *B. mutica* | 55 | 22 | 4 | 3 | 2 |
| | *C. nlemfuensis* | 51 | 19 | 2 | 3 | 2 |
| | *P. atratum* | 38 | 13 | 2 | 4 | 3 |
| 0.5 $ET_p$ (2 to 11 mm $d^{-1}$) | *P. purpureum* | 187 | 86 | 14 | 20 | 9 |
| | *B. decumbens* | 145 | 77 | 11 | 12 | 8 |
| | *B. mutica* | 67 | 53 | 8 | 7 | 5 |
| | *C. nlemfuensis* | 107 | 65 | 6 | 12 | 5 |
| | *P. atratum* | 115 | 50 | 8 | 19 | 14 |

When irrigated with effluent, a considerable amount of P and K along with other nutrients were applied to the soil. One study reported that an increasing rate of K application to *C. nlemfuensis* resulted in a more efficient P removal and a combined application of 10 kg P $ha^{-1}$ $yr^{-1}$ and 93 kg K $ha^{-1}$ $yr^{-1}$ gave a maximum (161% of applied P) P removal from the soil (Pant et al., 2004). Their study also showed that the P applications above 10 kg $ha^{-1}$ $yr^{-1}$ reduced the efficiency of P removal by *C. nlemfuensis* (in a sandy, silicious Spodosol). In the present study, all grasses including *C. nlemfuensis* were supplied with P at much greater amounts than 10 kg $ha^{-1}$ $yr^{-1}$ and very high amounts of K from effluent at the ratio of 1 P to 64 - 69 K on a kg $ha^{-1}$ $yr^{-1}$ basis, and may explain the lower P removal of this grass (65% for the 0.5 $ET_p$ rate and 19% for the 2.0 $ET_p$ rate) (Table 5.3) compared with the reported 161% (Pant et al., 2004).

**Table 5.5. Micronutrient removal (as percent of applied nutrients from effluent) of various tropical grasses receiving dairy effluent, Waianae, O'ahu, Hawai'i, Aug 2004 to Aug 2006.**

| Irrigation Rate | Grass Species | Micronutrient Removal | | | | |
|---|---|---|---|---|---|---|
| | | Fe | Zn | Mn | Cu | B |
| | | % | | | | |
| 2.0 $ET_p$ (7 to 44 mm $d^{-1}$) | *P. purpureum* | 29 | 28 | 153 | 3 | 2 |
| | *B. decumbens* | 354 | 110 | 129 | 5 | 2 |
| | *B. mutica* | 18 | 43 | 175 | 3 | 1 |
| | *C. nlemfuensis* | 49 | 83 | 167 | 3 | 1 |
| | *P. atratum* | 121 | 78 | 236 | 4 | 2 |
| 0.5 $ET_p$ (2 to 11 mm $d^{-1}$) | *P. purpureum* | 358 | 215 | 992 | 17 | 6 |
| | *B. decumbens* | 1206 | 367 | 758 | 20 | 4 |
| | *B. mutica* | 64 | 81 | 476 | 6 | 3 |
| | *C. nlemfuensis* | 161 | 155 | 617 | 9 | 3 |
| | *P. atratum* | 470 | 250 | 901 | 12 | 6 |

The percent nutrient removal values for micronutrients such as Fe, Mn, and Zn were relatively high (Table 5.4) due to the supplemental foliar fertilization of these micronutrients. Micronutrient deficiencies were observed on the grasses probably due to the high soil pH, resulting in very low plant availability of micronutrients and the need for supplemental fertilization (Plate 5.1). The low Zn removal may also be related to the possible antagonistic interactions between P and Zn as the extractable soil P was very high at the beginning and throughout the study (Fig. 3.14).

**Plate 5.1. Micronutrient deficiency in *P. purpureum*.**

**Seasonal Changes in Dry Matter Production and Nutrient Uptake**

Field plant growth and productivity is very dependent upon the weather conditions that vary with season. Strong seasonal variations in forage productivity and nutrient uptake were observed in this study (Figs. 5.2 to 5.6). The dry matter yield and nutrient uptake of the tropical grasses changed over time ($P$ <0.01). A significant interaction was observed between grass and season ($P$ <0.01) for dry matter yield and uptake of N, K and Mg. There was no significant interaction between grass and season for the uptake of P, Ca and micronutrients (Fe, Mn, Zn, Cu, B) ($P$ >0.10). The dry matter and N, P and K uptake of the grasses followed the trend of solar radiation (Figs. 5.2 to 5.6).

The variation in dry matter yield and nutrient uptake can also be related to the dynamics of the amount and nutrient content of effluent applied. In general, the fluctuations over time of dry matter yield and nutrient uptake of the grasses coincided with the fluctuations in nutrients of the effluent (Fig. 3.10). Higher dry matter yields and nutrient uptake were obtained in Jun-Aug 2005 (Figs. 5.2 to 5.3), a warmer period (April-September), with higher temperature, solar radiation and $ET_p$ (Fig. 3.3). The warmer conditions coupled with adequate water supplied by effluent irrigation were highly favorable to high photosynthetic rates and improved growth.

It was also noted that *B. mutica*, *C. nlemfuensis* and *P. atratum* flowered during the winter period, when the photoperiod is shorter, suggesting that these grasses are short-day or thermoperiodic plants (Plate 5.2).

**Plate 5.2. Flowering of *C. nlemfuensis* and *B. mutica***

**Forage Quality**

***Crude Protein***

Crude protein is a measure of the ability of forage to meet the protein requirement of the livestock. Since protein is one of the most costly supplements for livestock, high protein forages are desirable. In this study, the average CP contents of the effluent-irrigated grasses were between 10% and 16% during the 2-y period (Fig. 5.4; Table 5.5).

The CP content required for forage intended for feeding a lactating dairy cow (50-70 lb milk $d^{-1}$ or 22-32 kg milk $d^{-1}$) is $\geq$15% (National Research Council, 2001). The CP contents of the grasses were mostly below 15% possibly due to somewhat long cutting intervals (28 to 52 days). Chin's (1995) study on forage fertilization in Malaysia involving *B. decumbens, P. purpureum* and *P. maximum* had shown that pastures that are well-fertilized with N and cut early (17-21 days) results in high levels of CP (>15%). The CP content of the forage was not always below 15% during the two-year period (Fig. 5.10), but rather changed over time ($P$<0.01). During certain months of the year (between Sept and Mar), the CP content usually falls within the range of 14% to 20%. This period coincides with the period of high rainfall and robust growth of the grasses. Averaged across species and irrigation rate, all other grass species, except for *B. decumbens* and *P. atratum*, had generally higher average CP content when irrigated at 2.0 $ET_p$ rate ($P$ <0.01).

**Table 5.6. Forage quality of tropical grasses irrigated with dairy effluent, Waianae, O'ahu, Hawai'i, Aug 2004 to Aug 2006.**

| Irrigation Rate | Grass Species | Dry Matter | CP | NDF | ADF |
|---|---|---|---|---|---|
| | | | % | | |
| 2.0 $ET_p$ (7 to 44 mm $d^{-1}$) | *P. purpureum* | 19 | 13.3 | 58 | 33 |
| | *B. decumbens* | 27 | 9.8 | 56 | 28 |
| | *B. mutica* | 19 | 15.5 | 58 | 33 |
| | *C. nlemfuensis* | 28 | 13.1 | 62 | 32 |
| | *P. atratum* | 24 | 9.5 | 59 | 34 |
| 0.5 $ET_p$ (2 to 11 mm $d^{-1}$) | *P. purpureum* | 19 | 12.3 | 57 | 34 |
| | *B. decumbens* | 22 | 12.9 | 52 | 30 |
| | *B. mutica* | 20 | 11.59 | 62 | 36 |
| | *C. nlemfuensis* | 29 | 11.61 | 61 | 32 |
| | *P. atratum* | 23 | 9.50 | 59 | 34 |

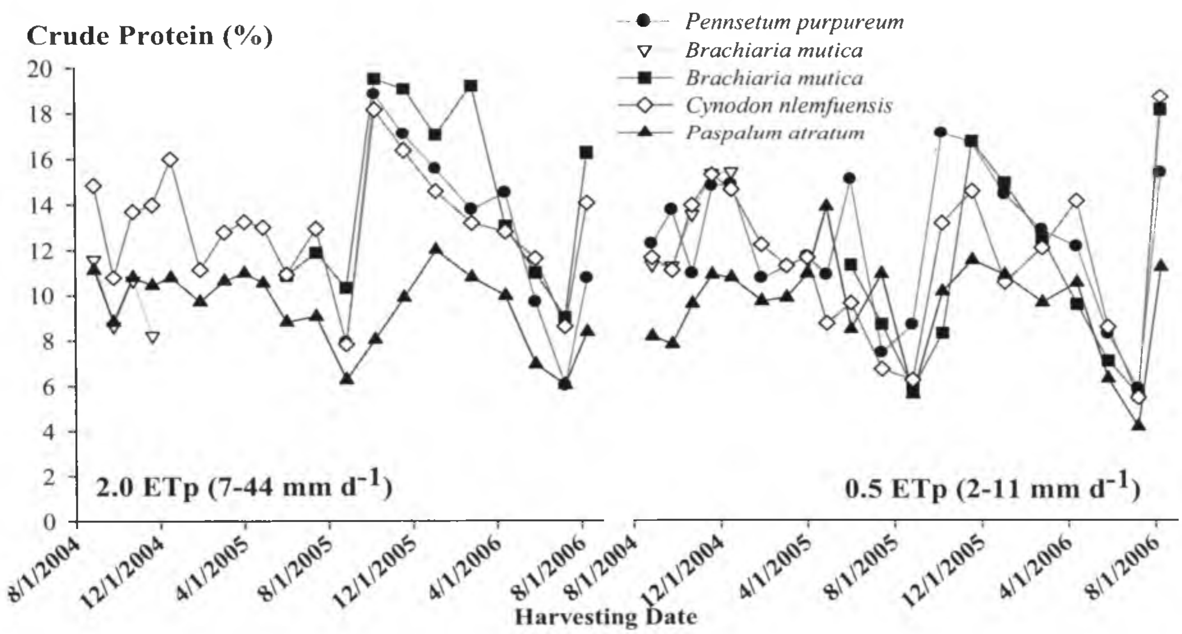



**Figure 5.10. Crude protein content of tropical grasses irrigated with dairy effluent, Waianae, O'ahu, Hawai'i, Aug 2004 to Aug 2006.**

*Neutral Detergent Fiber*

The energy value of forage can be evaluated using the NDF. The NDF is composed of cellulose, hemicellulose, lignin, silica, insoluble CP and ash. The recommended minimum dietary NDF values are 25 to 29% during early lactation and 32 to 34% during mid- to late lactation period (NRC, 2001) to allow cattle to eat more forage. The minimum forage NDF should be 15 to 19%, although a higher minimum NDF value is required for NDF from forage and for diets not fed as total mixed rations. A study, however, indicated that increasing NDF from 32% to 48% did not reduce the dry matter intake and that a minimum of 60% NDF has little effects on milk production (Sarwar et al., 1992). Higher NDF limits the intake and results in dairy cattle's reduced milk production, poor body condition, and difficulty in rebreeding (Schroeder, 2004).

The NDF of forage depends on the crop species, the form the forage is stored (Ruiz et al., 1995), and environmental factors (Buxton and Fales, 1994).

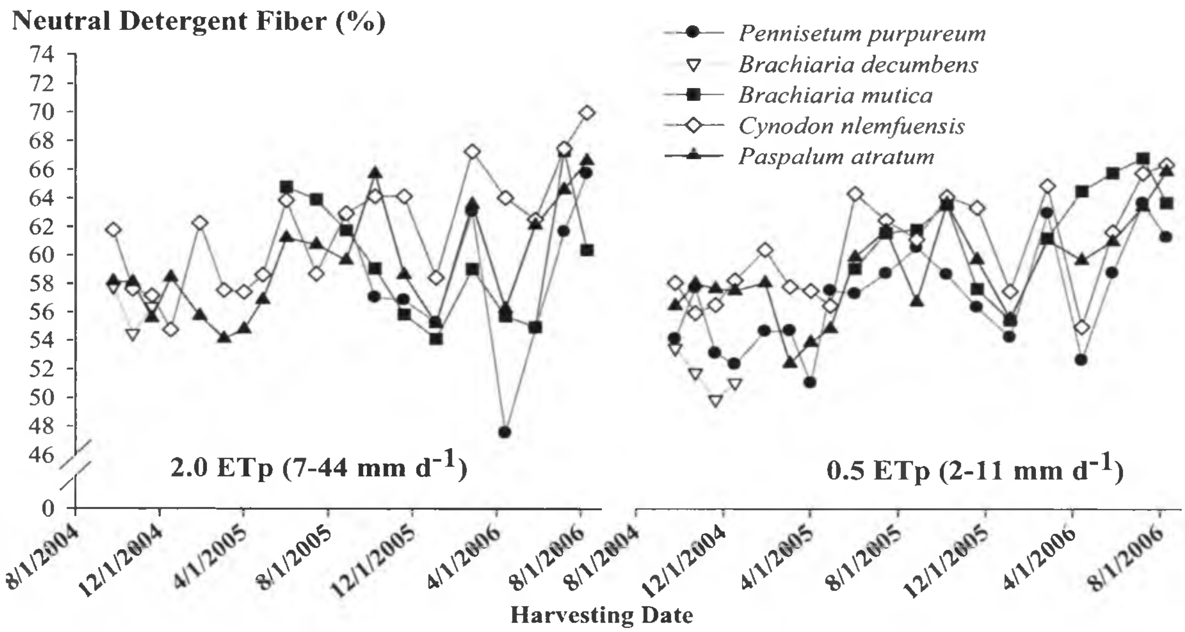



**Figure 5.11. Neutral detergent fiber of tropical grasses irrigated with dairy effluent, Waianae, O'ahu, Hawai'i, Aug 2004-Aug 2006.**

The NDF of the forage averaged to 52 to 62% over the 2-y period (Table 5.5), with significant main effect of time ($P$<0.01) and interaction effect of grass species and irrigation rate ($P$<0.01). The grasses had generally higher NDF during the June to November period when solar radiation and temperature were usually higher (Fig. 5.11). Lignin synthesis is enhanced by higher temperatures resulting in lower digestibility (Nelson and Moser, 1994; Buxton and Fales, 1994). Most of the grasses (*P. atratum, C. nlemfuensis* and *B. mutica*) also flower in Oct-Nov, which could explain the somewhat lower NDF. *Pennisetum purpureum* and *B. mutica* receiving the 2.0 $ET_p$ irrigation rate had average NDF value of 58% while those receiving the 0.5 $ET_p$ rate had 57% and 62%, respectively (Table 5.5). On the other hand, *C. nlemfuensis* and *B. decumbens* irrigated at 2.0 $ET_p$ had average NDF of 62% and 56%, respectively while those irrigated at 0.5 $ET_p$ rate had 61% and 52%, respectively. *Paspalum atratum* receiving dairy effluent at 2.0 $ET_p$ and 0.5 $ET_p$ rates had similar NDF (59%). The NDF values of the grasses were higher than the recommended minimum values of the forage intended for lactating cow (NRC, 2001), but mostly lower than the range (60 to 75%) reported for many cool-season and tropical grasses (Nelson and Moser, 1994). For example, the NDF of these grasses were lower than that reported by Ruiz et al. (1995) for silages made from *P. purpureum* (68%) and *C. dactylon* (74.5%).

***Acid Detergent Fiber***

The ADF is composed of the least digestible parts of cell walls such as cellulose, lignin, silica, ash and insoluble CP. The recommended minimum forage ADF is 17 to 21%, but as with NDF, a higher minimum is required for forage ADF depending on various factors that also affect NDF such as particle size of forage, feeding methods, supplements, among others (NRC, 2001). The ADF of the grasses irrigated with effluent changed with season ($P<0.01$) and differed among grass species ($P<0.01$) and irrigation rate ($P<0.01$). As with the NDF, relatively higher ADF (34% to 35%) were obtained during the period when solar radiation and temperature were higher (June to November) (Fig. 5.12) resulting in higher lignin synthesis (Nelson and Moser, 1994; Buxton and Fales, 1994). *Pennisetum purpureum*, *B. mutica* and *P. atratum* had higher ADF (33 to 36%) than *B. decumbens* (29%) and *C. nlemfuensis* (32%) (Table 5.5). Slightly lower ADF was recorded for tissues of grass species receiving 0.5 ETp irrigation rate (28% to

33%) than those receiving 2.0 ETp (30% to 36%). Again, these differences in ADF may be due to the favorable effect of water stress on certain species.

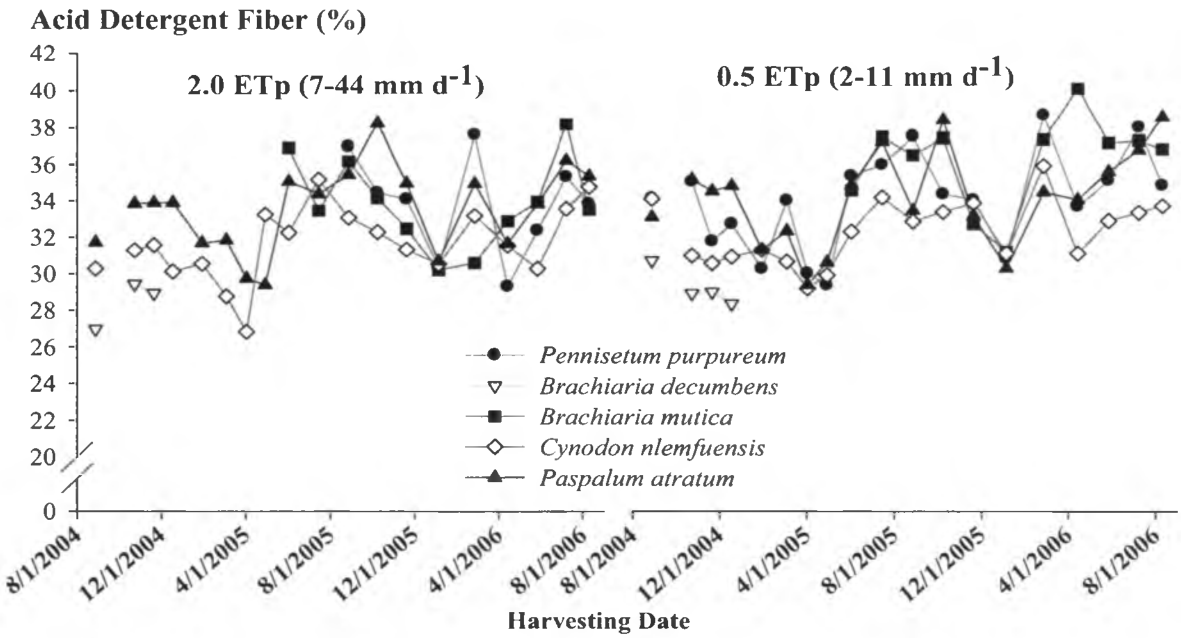



**Figure 5.12. Acid detergent fiber of tropical grasses irrigated with dairy effluent, Waianae, O'ahu, Hawai'i, Aug 2004 to Aug 2006.**

### General Discussion

All of the tropical grasses, except *B. decumbens*, irrigated with effluent had very high dry matter production that compared well or were higher than those commonly reported in the literature. While N uptake of the grasses was high, the P uptake was relatively low due partly to the low plant availability of the applied P. The low P availability was possibly due to Ca-P precipitation in this high pH, high Ca soil. The external P requirement of forage grasses is also generally low (Sims et al., 2005; Pant et al., 2004). The amount of K removed was also generally low despite the high amount of applied effluent K indicating that the amount applied was possibly much more than the plants require for optimum plant growth. The concentration of K in the tissue was at a sufficient level, using the adequate level for *P. notatum*.

The predicted nutrient uptake from intensively managed, irrigated tropical pasture grasses were reported to approach 750 to 1500 kg N $ha^{-1}$ $yr^{-1}$ and 75 to 150 kg P $ha^{-1}$ $yr^{-1}$ (Yost et al., 1999, unpublished). The N and P uptake of the grasses, except *B. decumbens*, was mostly within this predicted range of nutrient removal. The nutrient uptake of the grasses may have been higher if there was consistently high N supplied from the effluent, and can be expected to result in greater plant P uptake. Higher plants generally require an N:P ratio of 6:1 to 10:1 in the soil (Sharpley and Halvorson, 1994), whereas the effluent used for irrigation in this study had an N:P ratio of only 3.7:1. Only trace amounts of total Kjehldal N were found in the soil that resulted in very low ratio of soil N and extractable P. This very low ratio may explain why the average concentrations of N and P in the tissues were mostly at the deficient levels (Table 5.1) when compared with the usual adequate nutrient level suggested for *P. notatum* (Mills and Jones, 1996). Despite the deficient P level in the forage tissues, P concentrations (2.4 to 4.3 g $kg^{-1}$) were all within the typical range of tissue P concentrations (1.5 to 4.0 g $kg^{-1}$) reported in heavily manured soils or certain other high P soils with relatively low P sorption capacities (Mathews et al., 2005).

Overall results suggest that effluent irrigation at twice the $ET_p$ is acceptable, especially for irrigating *B. mutica* and *P. purpureum* for forage production in this high $ET_p$ site. The only exception was *B. decumbens* that grew better at the low irrigation rate, being a drought-tolerant grass. Where the objective is to maximize the utilization of effluent at a smaller land area available that is usually the case in Hawai'i, irrigation at 2.0 $ET_p$ appears to be acceptable because of the high dry matter production and nutrient uptake of most of the grasses.

Among the grasses, *B. mutica* and *P. purpureum* had the highest dry matter yields and the greatest potential to take up nutrients from the effluent. The two species were good choices for forage production, if the prime objective was to effectively remove nutrients from the soil. The calculated carrying capacity was 8 to 11 cows $ha^{-1}$ $y^{-1}$ using the dry matter yields of *B. mutica* and *P. purpureum* and milk production of a 9 kg $d^{-1}$ of 4% fat corrected milk (NRC, 2001) for a 607-kg cattle (Wilkerson et al., 1997) with dry matter intake of 2.1% of body weight (NRC, 2001).Potassium uptake and tissue K concentration of *P. purpureum* was relatively high and should be considered when formulating dairy cattle rations to avoid excessive intake of K. The imbalance of K and Mg could lead to the dairy cattle metabolic disorder known as grass tetany.

The P content of the forage ranged from 0.20 to 0.42 %--these values meet the dietary P requirement of dairy cattle according to National Research Council (2001) guidelines. Thus, farmers feeding their cattle with this forage may reduce or eliminate the supplementation of ration with imported P supplements for sustained milk production of the cattle. In terms of other forage quality parameters, an adjustment in the dietary formulation with allowance for higher proportions from concentrates should be made when feeding lactating cows with the tropical grasses in order to meet the NRC requirements for NDF and ADF.

## Conclusions

Tropical grasses, except *B. decumbens*, planted in this soil of the tropics could be irrigated with dairy effluent at 2.0 $ET_p$ rate. At this irrigation rate, higher dry matter yields and nutrient uptake were obtained than at the 0.5 $ET_p$ rate. Forage quality such as CP, NDF and ADF were also at levels acceptable for feeding to lactating cattle. *Brachiaria mutica* and *P. purpureum* appeared to be the best choices for forage production, especially if the prime objective was to effectively remove nutrients from the soil in order to maximize the application of liquid effluent without serious nutrient accumulation in the soil. However, the K concentration of *P. purpureum* was relatively high and should be taken into account when formulating the cattle ration to avoid metabolic disorders due to excessive K intake that may create an imbalance in K, Ca and Mg. Due to the high soil pH, supplemental application of micronutrients such as Zn, Cu, Fe and Mn were found to be necessary to maintain acceptable tissue levels of these nutrients. Effluent irrigation to produce forage is an effective means of closing the open nutrient cycle in the milk production system.

# CHAPTER 6. DEVELOPMENT OF A DECISION-AID STRUCTURE FOR MANAGING SOILS RECEIVING DAIRY EFFLUENT

## Abstract

A decision-aid tool can help dairy producers improve their management of the soil, plant and environment in systems involving effluent application. A decision-making structure was developed to support effluent application by evaluating selected options for effluent management. Modeling of selected effluent management options (area needed for effluent application, number of cattle that can be raised, basis for choosing the rate of effluent application, and nutrient concentration in the leachate) was carried out using a rule-based decision-making software and the information obtained from the field experiment. Scenarios to evaluate the selected effluent management options involved assumptions on the potential evapotranspiration ($ET_p$) rates, rainfall amounts, irrigation rates, effluent N and P contents, and nutrient uptake of the grasses—*P. purpureum* and *B. mutica*. A flowchart illustrated the pathways of and the agronomic and environmental factors involved in the decision-making process for effluent management. Modeling results showed that the area needed for effluent N and P application and the number of animals to utilize the effluent N and P for forage production is largely determined by the total applied phosphorus. If the $ET_p$ remains at a high level (e.g., 21 m $d^{-1}$), irrigating based on the consumptive water use of the crop results in excessive amount of applied nutrients. If the $ET_p$ is low (e.g., 5 m $d^{-1}$), irrigating based on crop water requirement may result in inadequate application of nutrients especially N, which would necessitate inorganic fertilizer application. Nonetheless, irrigating based on the $ET_p$ is an acceptable practice when using salt-rich wastewater such as dairy effluent to meet the leaching

requirement. Irrigating the grasses based on the crop nutrient requirement results in much less water applied thereby requiring freshwater supplementation, if rainfall is not enough, to meet crop water requirement and leaching fraction. High rainfall increases the volume of lagoon effluent and dilutes the concentrations of nutrients that could leach through the soil profile. Current models need improvements for more realistic estimates. One important question to address in future modeling work is how to provide the leaching fraction needed to ensure maintaining soil properties while minimizing the amount of nutrients that could leach to the groundwater.

## Introduction

Decision support systems have been increasingly used as a tool for various aspects of soil, plant and environmental management. For example, the Nutrient Management Support System (NuMaSS) allows for diagnosis of soil constraints and provides recommendations for suitable management practices based on agronomic, economic and environmental information for specified locations (Osmond et al., 2002). It has three support systems components, namely, Acid Decision Support System (ADSS), Nitrogen Decision Support System (NDSS) and the Phosphorus Decision Support System (PDSS). Another example is the Decision Support System for Agrotechnology Transfer (DSSAT), which has also been used by many plant/soil scientists to formulate decisions regarding crop management (Jones et al., 1998). It combines crop, soil and weather data to run simulations in the computer and obtain results much faster than actual agronomic experiments. Many models and nutrient management tools have also been developed for animal waste management ranging from manually-filled worksheets to computerized worksheets and programs, e.g., NRCS' Animal Waste Management Software, Purdue University's Manure Management Planner, among others. Some simple programs are available at the internet.

## Importance of Understanding the Components of Decision-Making Structures

Being able to understand the processes that are significant in terms of their relative influence on the risk of nutrient losses to the environment is an important component of decision-making. This exercise is also relevant for a scientist to understand the various factors to consider in making important decisions such as regarding effluent management. It is important for soil scientists and researchers to gain an understanding of the processes in designing a decision-aid system to enable them to understand which factors are most crucial. It will enable for identification of soil processes or factors that may limit the land application of nutrients such as those from animal effluents.

Process-based decision-making structures are useful for understanding the intricacies of a software model that quantitatively predict, for example, the likelihood of nutrient transport from a point where it is applied, or the possible accumulation of nutrients at the site of application. Once the decision-making structure is converted into a software, it can be used in qualitatively or quantitatively assessing both temporal and spatial likelihood of nutrient accumulation and losses. This information is very important in identifying areas that need management decisions. Decision-making tools are appropriate for research because on-farm or field experiments are difficult to control, take time to obtain results, and often, costly. Although best management practices already provide some guidelines on proper ways to manage animal wastes, their use is limited by the fact that these guidelines do not provide advice that is specific to a farm or the conditions in the farm.

### Characteristics of a Good Decision-Aid Tool

Some of the basic criteria for "good decision-making are efficiency, effectiveness, equity…and flexibility" (Gough and Ward, 1994). To achieve these, it is important to search for means that will improve the ability of the decision maker to evaluate alternatives and come up with a good judgment. This is where decision-aid tools and models are useful. Management system models that will allow users of natural resources and policy/decision-makers to make science-based judgments on the relative merits of alternative practices are important. A good decision-aid tool is one that integrates knowledge from various sources and enables the users to select beneficial options for managing the problem identified. It must also consider the environmental conditions and the ecological state of the farm, location or system being assessed or modeled. A useful decision-making structure is one that adequately simulates the situation it represents and is useful in exploring solutions and opportunities. A decision-aid tool should require as few data as possible without compromising the results and is user-friendly.

### Applications of Decision-Aid Tool to Management of Animal Wastes/Effluents

Dairy producers, farm managers, livestock agents and even researchers can benefit from decision-aid tools to educate and convince the producers to adopt sound practices for effluent utilization. Decision-aid tools can be used to evaluate whether current practices are sustainable and to identify practice(s) that need(s) improvement to minimize waste of resource inputs and reduce the risk of environmental pollution.

Decision-aid tools enable researchers, farm managers and decision-makers to create scenarios to evaluate previously unexplored options or to quantitatively evaluate competing options.

While many researchers on continental U.S. believe that agricultural research results in Hawaii are not representative of mainland conditions, these results often can represent extreme or unusual values, which are relevant to be considered nationally or internationally. The proposed decision-making structure and models are, therefore, intended for dairy producers, extension agents, policy regulators and researchers in Hawaii and other places (nationally and internationally) where animal effluent management is a concern.

## Objectives

The objectives of this modeling exercise were to:

1. Propose a decision-making structure that facilitates the evaluation of effluent management options for consideration by dairy producers, and
2. Model selected aspects of dairy effluent management using the available and user-friendly software.

## Methodology

The development of the decision-aid tool involved reviewing the literature on information pertinent to effluent management. Livestock agents were also consulted for additional unwritten decision-making considerations that may be of interest to them and to dairy producers. The evaluation of selected alternatives used the information obtained from the field experiment, which, in turn, were used to create scenarios using a DecisionPro[†] software (Vanguard Software Corp., 1992-2002). DecisionPro is a modeling tool that allows users to enter the formula or variable at each node to perform a series of calculations that forms into a tree structure making it easier to communicate the analysis to others. Calculations and input data are defined by the user. Scenarios either predicted an outcome based on given conditions, or proposed a condition that will allow the producer to achieve a desired outcome. DecisionPro software was chosen due to its availability and user-friendliness allowing even a novice modeler to create models to analyze a problem logically. It uses plain English, simple mathematical formulas defined by the user. The analysis is presented in a hierarchical tree-type layout making it easier for the modeler to communicate the ideas clearly or for the user to understand how a solution to the problem was derived.

A flowchart showing the various decision-making aspects in effluent management was created. Four aspects were selected for DecisionPro modeling:

- Area needed for effluent application,
- Number of cattle that can be raised,
- Basis for choosing the effluent application rate – Nutrients or Irrigation water, and
- Nutrient concentration in the leachate.

---

[†]Reference herein to any specific product, by trade name, trademark, manufacturer, or distributor does not necessarily constitute or imply its endorsement or recommendation by the authors and publishers.

These aspects were chosen because these are practical questions that are of interest to a dairy producers and agents, and due to availability of field data to evaluate these options. The assumptions and types of input and output data are shown in Table 6.1.

The area needed to utilize the N and P from the applied effluent was evaluated by specifying the number of animals, effluent production per cow per day, effluent N and P content and forage production data such as dry matter yield and tissue nutrient concentration for the top two producing forages (*P. purpureum* with 51 Mg dry matter $ha^{-1}$ $y^{-1}$ and *B. mutica* with 57 Mg dry matter $ha^{-1}$ $y^{-1}$) (Table 6.1). The effluent production per cow per day was expressed in unit of gallons because it is the unit commonly used by dairy farmers. The model does the conversion from gallons to liters in the formula window. The formula or input data for each node is entered by the user at the tree editor's formula window and the model automatically creates a flow diagram of the input and output. Using the same model as the area requirement, the number of dairy cattle that a farmer can raise was modeled by specifying the land area available for effluent irrigation at the Goal Seek option (Tools menu of the model) and allowing the model to automatically calculate the number of cattle.

Evaluation of the basis for effluent irrigation rate (nutrient or crop water requirement) was done by modeling different scenarios. The evaluation for the basis of effluent application rate involved determining how much excess N or P occurs or the irrigation rate whereby excess N and P is minimized as well as the concentrations of these nutrients in the leaching water at a given potential evapotranspiration ($ET_p$), irrigation rate, effluent N and P content, dry matter yield and tissue nutrient concentration of the two selected grasses. Modeling of effluent application rate was based on crop water requirement, the specified $ET_p$ and the irrigation rate. Using the same model, the effluent application rate based on crop nutrient requirement was determined. This was done by setting the amount of excess N and P to zero in the Goal Seek option in the Tools menu.

Modeling of the nutrient concentration in the water leaching through the soil profile considered different scenarios for $ET_p$, rainfall, irrigation rate and effluent N and P contents (Table 6.1). The amount of nutrient available for leaching was determined as the difference between the total applied nutrient from the irrigation water (effluent plus rainfall) and nutrient uptake of the grass.

For each model, a table of inputs and outputs of the model was automatically created by the software when the user clicked on each node and specified whether the node was an input or an output. Examples of the different models were presented in the Results and Discussion.

(NOTE: Leave 7 PAGES BLANK FOR 6 TABLES; see attached file)

**Table 6.1. Assumptions in modeling selected aspects of dairy effluent management: area requirement for effluent application.**

**Table 6.2. Assumptions in modeling selected aspects of dairy effluent management: number of animals.**

**Table 6. 3. Assumptions in modeling selected aspects of dairy effluent management: estimating requirements on the basis of effluent application rate.**

**Table 6.3. Assumptions in modeling selected aspects of dairy effluent management (cont.).**

**Table 6.3. Assumptions in modeling selected aspects of dairy effluent management (cont.).**

**Table 6. 4. Assumptions in modeling selected aspects of dairy effluent management: nutrient concentrations in the water leaching through the soil profile.**

## Results and Discussion

### Decision-Aid Structure for Effluent Irrigation

Various factors, agronomic and environmental, have to be considered when irrigating with effluent such that it is difficult for a dairy producer to make immediate decision(s). Effluent management may require a step-by-step assessment of the various issues involved in the decision making process for effluent utilization (Fig. 6.1). The first step in our proposed decision-making is to identify the problem(s) and recognize the potential impacts of the problem. The next step is to determine alternative solutions preferably those that are affordable and locally available to the dairy producer. After alternatives were identified, an assessment of the feasibility of implementing the proposed solution(s) follows. Modeling some aspects of the decision making process, provided data are available, usually saves time and effort in arriving at a solution to the problem.

### Significance of the Information in the Decision-Aid Structure

Determining the potential of the lagoon to overflow requires data on climatic conditions of the site such as rainfall, $ET_p$ and solar radiation. Usually this is a required step in the design of the lagoon – in order to contain a 24-hr maximum rainfall that could occur every 10 years (U.S.-EPA, 1995). The information helps in assessing the potential of lagoon(s) for holding the effluent as well as the irrigation potential of the site. If the effluent is being used for irrigation, information about the climate is used to assess when irrigation is possible, when storage is needed, potential for nutrient losses to the environment, dilution of effluent and soil nutrients, and for programming/scheduling of application. Also, evapotranspiration and solar radiation data are indicators of the potential productivity of the crop grown since they determine the photosynthetic potential of the plants.

Apart from climatic data, information about soil properties such as soil type, mineralogy, macro- and micronutrient contents, and phosphorus sorption capacity are needed to determine soil constraints and opportunities for effluent irrigation. The soil analysis data indicate the capacity of the soil to support crop growth and accumulate nutrients. Some hydrological parameters such as slope gradient and shape, drainage class (hydraulic conductivity and infiltration rate), proximity to water body and depth to water table are important for assessing the environmental risks, especially runoff, associated with effluent irrigation at the site. Soil data also help determine the potential for leaching of nutrients especially nitrate, phosphorus and potassium. Knowing the concentration of nutrients in the leaching water will assist in assessing the possibilities of contaminating the groundwater. Soil analysis also allows for evaluating the need for ameliorative actions (e.g., amendment application or supplemental fertilization) in the event of nutrient deficiency or toxicity.

**Note: Leave 5 pages blank for this figure (see attached file).**

**Figure 6.1. Decision-making diagram for effluent management.**

Knowing the area available for application helps to determine the capacity of the farm for utilization of the wastewater and whether there will be excess supply of wastewater. Information on the number of animals raised helps assess the potential supply of wastewater and nutrients for irrigation. Analyzing the effluent for nutrient composition, alkalinity, and salt content are important in assessing its acceptability for irrigation and potential impacts on the soil. The volume and quality of wastewater produced are needed to evaluate the nutrient loading to the soil as well as the likelihood of polluting both the land irrigated with wastewater and the associated water bodies.

The forage species to be grown and its water requirement are needed to determine the potential of the site for nutrient utilization. In particular, plant information such as dry matter production and nutrient uptake allows for the assessment of the impacts of effluent application on forage yield and quality as well as the ability of the system to remove nutrients supplied by effluent. These variables are affected by the management of the grasses such as harvesting frequency and height.

In the decision-making structure presented, much of the information may not be readily known to or available from the dairy producer. Examples of such information include soil properties (type, mineralogy, nutrient content, P sorption capacity, hydraulic conductivity and infiltration rate), $ET_p$ and solar radiation, and depth to groundwater. Simple models that require the least amount of input data, yet provide numbers that can be used to justify the selection of a waste management option are useful to dairy farmers and livestock agents. An illustration of this kind of model is the modeling scheme using the DecisionPro software.

**Evaluating Selected Management Options for Effluent Irrigation using DecisionPro Software**

Land availability, especially in island environments, has been a major constraint for the active use of effluent for irrigation. Due to the limited holding capacity and potential of lagoon(s) to overflow, it is necessary that dairy producers find alternative use(s) of the effluent such as irrigation for forage production. It is, therefore, of interest to a dairy producer to know how much area is needed to utilize the nutrients and water in the effluent to grow the forage, or how many animals can be raised given the limited land available for effluent application. These are also key questions in developing a Nutrient Management Plan as required by the NRCS, U.S.-EPA and policy regulators. Design of effluent application systems could be useful in reducing the design costs of a lagoon since a smaller lagoon could be acceptable if there were a consistent use of the effluent for irrigation.

***Area Needed for Effluent Irrigation***

A sample model for determining the area needed to utilize the N and P from effluent for forage production is shown in Fig. 6.2. One of the assumptions in the model was that irrigation was supplied to meet crop nutrient requirements, without nutrient losses taking place. Dairy effluent containing 52 mg N $L^{-1}$ and 16 mg P $L^{-1}$ may be used to irrigate *P. purpureum* and *B. mutica* in an area up to 3 ha to utilize the N and P generated from 500 cattle assuming an effluent production of 45 gal $cow^{-1}$ $d^{-1}$ or 170 L $cow^{-1}$ $d^{-1}$ (Table 6.5). The area requirement increases to 15 ha to consume the effluent N and P generated from 2500 cattle. Assuming the same effluent production per cow and effluent N and P concentrations of 24 mg $L^{-1}$ and 11 mg $L^{-1}$, respectively, a dairy producer needs up to 2 ha for *P. purpureum* and *B. mutica* to utilize the N and P produced by 500 cattle and up to 10 ha (*P. purpureum* and *B. mutica*) to use the N and P produced by 2500 cattle (Table 6.5). These numbers on area requirement may be highly underestimated because the model did not consider the additional water from rainfall and runoff during the rainy season.

Inputs:

| | |
|---|---|
| P Conc. in percent | 0.34 |
| N Conc. in percent | 1.97 |
| Number of Animals | 500 |
| kg per ha per y | 51000 |
| Effluent per cow | 45 |
| percent N in effluent | 24 |

Outputs:

| | |
|---|---|
| kg N per ha per y | 1005 |
| kg P per ha per y | 173 |
| Effluent N in kg per y | 746 |
| Effluent P in kg per y | 342 |
| P Area Reqt | 2 |
| N Area Reqt | 1 |
| Area in Ha | 2 |

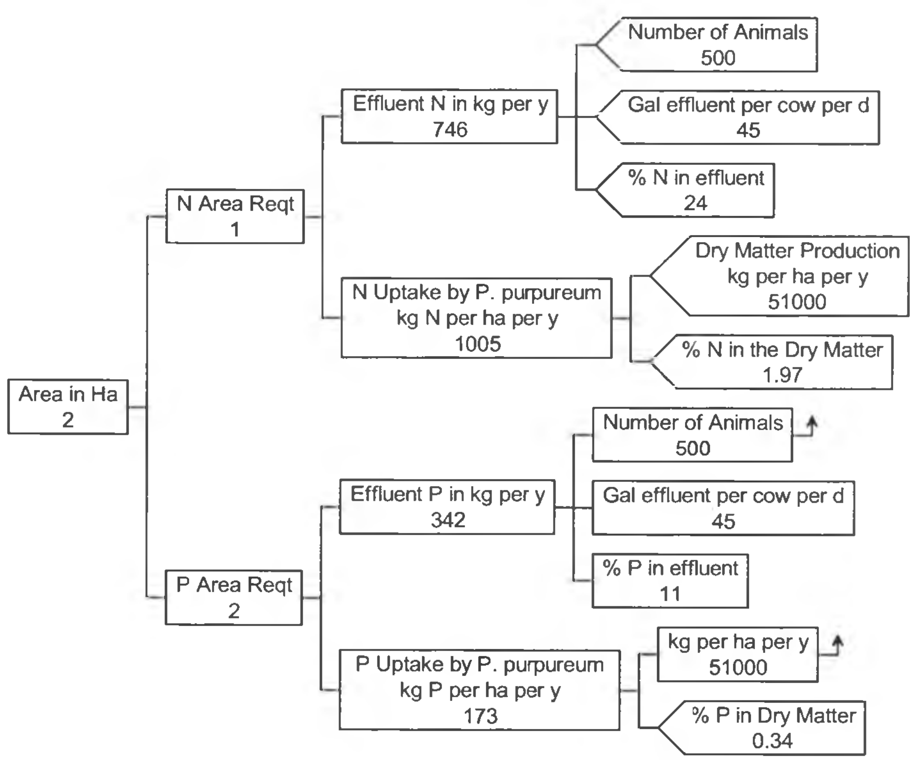



**Figure 6.2. An example of a DecisionPro model to determine the area needed for effluent irrigation with a given number of cattle.**

**Table 6.5. Input and output data to determine the area required to utilize the effluent N and P for forage production.**

**Note: this table is a landscape format (see attached file)**

*Number of Animals to be Raised*

Based on the above modeling results, the area needed if application was based on the nutrient requirement of the grass can be as high as 15 ha or much more (if the runoff and rainfall contribution was considered) to utilize for forage production the nutrients from the effluent produced by 2500 cattle (Table 6.6). However, in certain cases, the dairy farmer may have limited land area for effluent application and may be interested to know the limit on cattle population to raise for milk production (Fig. 6.3). At a rate of 170 L $cow^{-1}$ $d^{-1}$ of effluent (24 mg N $L^{-1}$ and 11 mg P $L^{-1}$) production and 5 ha of land available to use for effluent application, a dairy producer can raise up to 1400 cattle to irrigate *P. purpureum* and *B. mutica* (Table 6.6). But if the concentrations of N and P in effluent are higher (52 mg N $L^{-1}$ and 16 mg P $L^{-1}$), fewer cattle can be raised (870 to 970). Ten times as many animals can be raised if the land available for effluent application was 50 ha (Table 6.7). In all cases, P was the nutrient that determines the amount of effluent that can be applied. However, these estimates may be overestimated because the additional water from rainfall and runoff during the rainy season was not considered.

**Table 6.6. Number of animals that can be raised for a 5-ha land available for effluent application.**

| INPUTS | | | | OUTPUTS |
|---|---|---|---|---|
| **Grass Species** | **Dry Matter Production** | **Dry Matter Nutrient Concentration** | | **Number of Animals Required** |
| | | **N** | **P** | |
| | Mg $ha^{-1}$ $y^{-1}$ | -----------%----------- | | |
| 52 mg N $L^{-1}$ and 16 mg P $L^{-1}$ | | | | |
| *P. purpureum* | 51 | 1.97 | 0.34 | 870 |
| *B. mutica* | 57 | 2.48 | 0.29 | 970 |
| 24 mg N $L^{-1}$ and 11 mg P $L^{-1}$ | | | | |
| *P. purpureum* | 51 | 1.97 | 0.34 | 1300 |
| *B. mutica* | 57 | 2.48 | 0.29 | 1400 |

**Table 6.7. Number of animals that can be raised for a 50-ha land available for effluent application.**

| INPUTS | | | | OUTPUTS |
|---|---|---|---|---|
| **Grass Species** | **Dry Matter Production** | **Dry Matter Nutrient Concentration** | | **Number of Animals Required** |
| | | **N** | **P** | |
| | Mg $ha^{-1}$ $y^{-1}$ | -----------%----------- | | |
| 52 mg N $L^{-1}$ and 16 mg P $L^{-1}$ | | | | |
| *P. purpureum* | 51 | 1.97 | 0.34 | 8500 |
| *B. mutica* | 57 | 2.48 | 0.29 | 9700 |
| 24 mg N $L^{-1}$ and 11 mg P $L^{-1}$ | | | | |
| *P. purpureum* | 51 | 1.97 | 0.34 | 13000 |
| *B. mutica* | 57 | 2.48 | 0.29 | 14000 |

Inputs:

| | |
|---|---|
| Number of Animals | 1300 |
| gal effluent per cow per d | 45 |
| percent N in effluent | 24 |
| kg dry matter per ha per y | 51000 |
| percent P in dry matter | 0.34 |

Outputs:

| | |
|---|---|
| kg P per ha per y | 173 |
| P in Effluent in kg per y | 889 |
| kg N per ha per y | 1005 |
| Effluent N in kg per y | 1940 |
| Area Avail. based on N | 2 |
| Area Avail. based on P | 5 |
| Area in Ha | 5 |

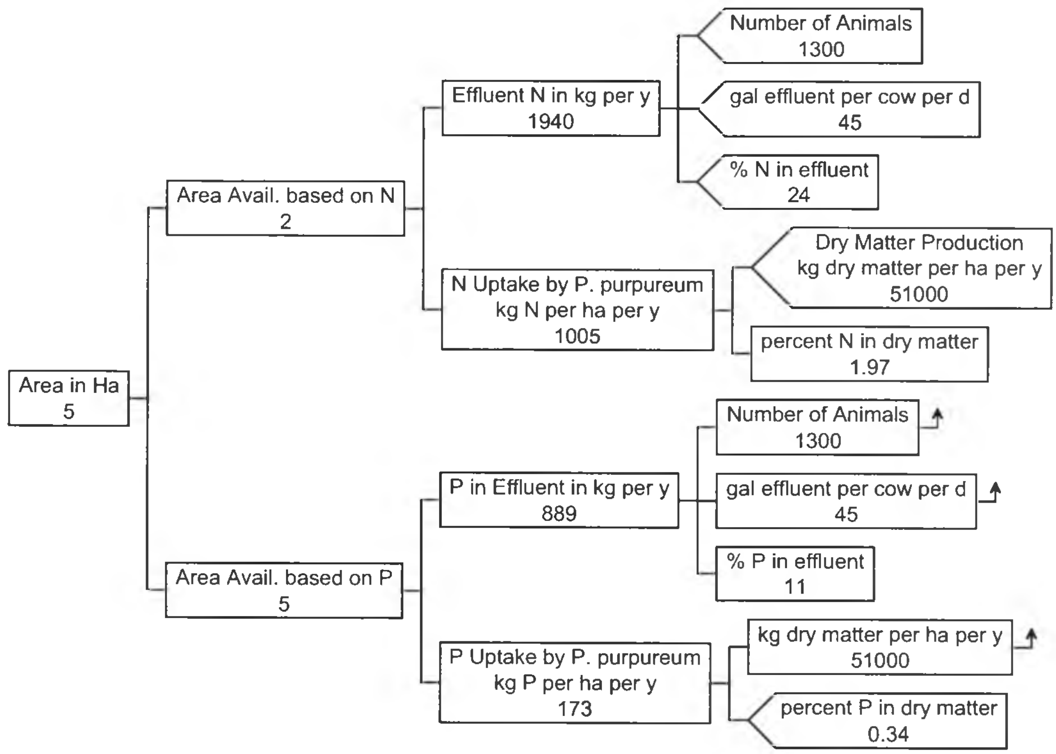



**Figure 6.3. An example of a DecisionPro model to determine the number of animals that can be raised with a given land area available for effluent irrigation.**

## Estimating Requirements on the Basis of Effluent Irrigation Rate

Once the farmer has decided to use the effluent for forage irrigation, the next major question is whether to maximize the application of effluent based on the nutrients or on the water requirements of the crops. Examples of models to evaluate these options are shown in Figs. 6.4 and 6.5. Nutrient-based land application of wastewater and manures has been implicated on the delivery of high levels of nutrients and pathogens to the soil and water bodies causing pollution of these natural resources (Hansen et al., 2002; McDowell and Sharpley, 2001). Nitrogen-based manure application, for example, can lead to excessive application of P with consequent accumulation (Eghball and Power, 1999; Ferguson et al., 2005; Toth et al., 2006; Golden et al., 2006) or leaching (Eghball, 2002 and 2003), as well as $NO_3$-N leaching (Ferguson et al., 2005). Meanwhile, P-based application can result in inadequate supply of N for the crop, thus, would require additional N fertilization (Eghball, 2002).

Dairy effluent usually contains less nutrients per given volume than the manure, thus, nutrient-based application may result in tremendous amount of water applied. For example, dairy cow manure typically contains 5.0 kg solids $Mg^{-1}$ (0.5% w/w) whereas liquid pit dairy cow manure contains 3.0 kg solids $m^3$ (0.3% w/v) (Mullins et al., 2005). Effluents normally contain more water (99 to 99.9%) than liquid pit manure (90 to 95%) (Kohl, 2004). Effluent application based on crop water requirements is probably more beneficial in areas where the rainfall is less and $ET_p$ is high. This will allow the farmer to save on the cost of water for irrigation. In areas where the rainfall is high, effluent irrigation that exceeds the consumptive water use of the crop may result in nutrient losses to the environment through leaching and/or runoff/erosion, especially if soil properties are conducive for these processes.

Inputs:

| | |
|---|---|
| Rate | 1 |
| mm per d | 5 |
| kg DM per ha per y | 51000 |
| percent N | 1.97 |

Outputs:

| | |
|---|---|
| effluent N kg per L | 2.4e-5 |
| L effluent per ha per y | 18250000 |
| kg of N per ha per y | -567 |
| kg per ha per y N | 1005 |
| kg N per ha per y | 438 |

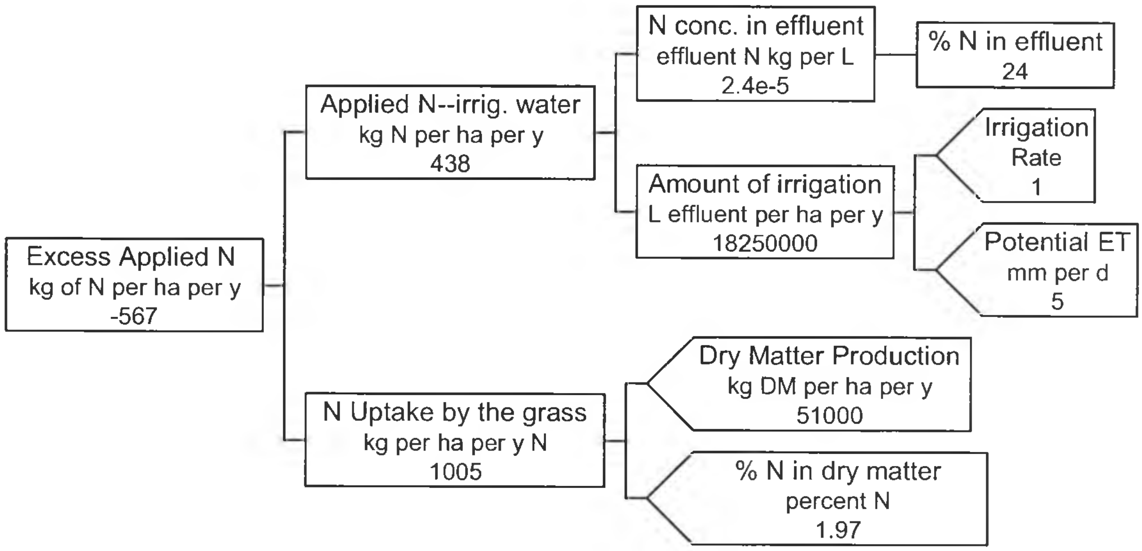



**Figure 6.4. An example of a DecisionPro model to determine the excess applied nutrient with effluent irrigation based on crop water requirement.**

***Excess N and P if $ET_p$ is 5 mm $d^{-1}$***

If the $ET_p$ was low (5 mm $d^{-1}$) and the concentrations of N and P in the effluent were 24 to 52 mg N $L^{-1}$ and 11 to 16 mg P $L^{-1}$, respectively, irrigating *P. purpureum* and *B. mutica* at 1.0 $ET_p$ resulted in a deficit in applied N (60 to 980 kg $ha^{-1}$ $y^{-1}$) and a surplus in applied P (30 to 130 kg $ha^{-1}$ $y^{-1}$) (Tables 6.8 and 6.10). Irrigating these grasses at 0.5 $ET_p$ resulted in much greater deficit in applied nutrients ranging from 530 to 1200 kg N $ha^{-1}$ $y^{-1}$ and 20 to 70 kg P $ha^{-1}$ $y^{-1}$. At 2.0 $ET_p$ irrigation rate, there was an excess applied N of 480 to 890 kg $ha^{-1}$ $y^{-1}$ if the effluent contained 52 mg N $L^{-1}$ and 16 mg P $L^{-1}$, and a deficit of 130 to 540 kg $ha^{-1}$ $y^{-1}$ if the effluent contained 24 mg N $L^{-1}$ and 11 mg P

$L^{-1}$. Regardless of effluent N and P concentration, irrigating these grasses at 2.0 $ET_p$ resulted in excess applied P of 230 to 420 kg $ha^{-1}$ $y^{-1}$.

Meeting the N requirement of the crop required irrigating at 1.0 to 3.2 $ET_p$ and minimizing excess P entailed irrigating at 0.6 to 0.9 $ET_p$ (Tables 6.9 and 6.11). Thus, in these cases, irrigating at less than 5 mm $d^{-1}$ (1.0 $ET_p$) would necessitate the application of inorganic fertilizer N.

Inputs:

| | |
|---|---|
| Rate | 2.29 |
| mm per d | 5 |
| kg DM per ha per y | 51000 |
| percent N | 1.97 |

Outputs:

| | |
|---|---|
| effluent N kg per L | 2.4e-5 |
| L effluent per ha per y | 41865500 |
| kg of N per ha per y | 0.07 |
| kg per ha per y N | 1004.7 |
| kg N per ha per y | 1004.77 |

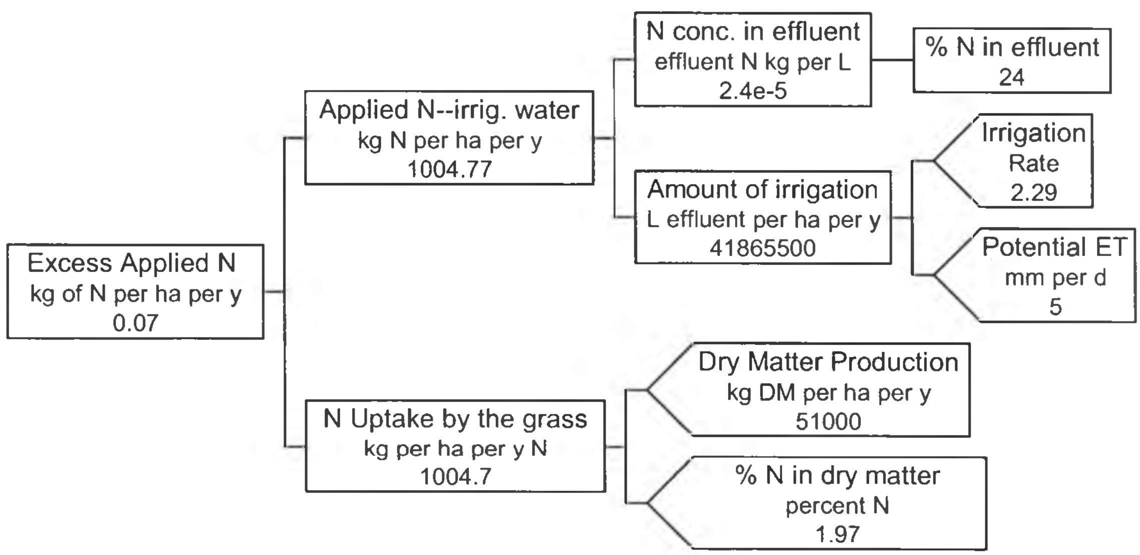



**Figure 6.5. An example of a DecisionPro model to determine the excess applied nutrient with effluent irrigation based on crop nutrient requirement.**

*Excess N and P if $ET_p$ is 21 mm $d^{-1}$*

If dairy effluent was used to irrigate *P. purpureum* at a rate corresponding to the highest recorded $ET_p$ of 21 mm $d^{-1}$ (1.0 $ET_p$) in Waianae, there was an estimated excess nutrients in the soil of 2980 kg N $ha^{-1}$ $y^{-1}$ and 1050 kg P $ha^{-1}$ $y^{-1}$ if the effluent had 52 mg N $L^{-1}$ and 16 mg P $L^{-1}$ (Table 6.8). If the irrigation rate was 0.5 $ET_p$ (10.5 mm $d^{-1}$), the amounts of excess nutrients was approximately 990 kg N $ha^{-1}$ $y^{-1}$ and 440 kg P $ha^{-1}$ $y^{-1}$. Irrigating at 2.0 $ET_p$ led to very high amounts of excess N and P (6970 and 2280 kg $ha^{-1}$ $y^{-1}$, respectively). Assuming a lower effluent N and P concentrations of 24 mg $L^{-1}$ and 11 mg $L^{-1}$, respectively, the excess applied N at an irrigation rate of 1.0 $ET_p$ was 830 kg $ha^{-1}$ $y^{-1}$ and P was 670 kg $ha^{-1}$ $y^{-1}$. The excess applied nutrients were much higher if the irrigation rate was at 2.0 $ET_p$ (2670 kg N $ha^{-1}$ $y^{-1}$ and 1510 kg P $ha^{-1}$ $y^{-1}$). At such low effluent N level, there was an estimated deficit in applied N amounting to 90 kg $ha^{-1}$ $y^{-1}$, while excess P was about 250 kg $ha^{-1}$ $y^{-1}$ if the irrigation rate was only 0.5 $ET_p$. To match the N and P requirement of *P. purpureum* (i.e., minimize the excess N and P applied), the irrigation rate had to be reduced to $\leq$0.6 $ET_p$ (Table 6.9).

Table 6.8. Modeling results if *P. purpureum* is irrigated with dairy effluent based on crop water requirement.

| | INPUTS | | | OUTPUTS | |
|---|---|---|---|---|---|
| $ET_p$ | Concentration of Nutrients in Effluent | | Irrigation Rate | Excess Applied Nutrient | |
| | N | P | | N | P |
| mm $d^{-1}$ | -----------mg $L^{-1}$------------- | | -----x $ET_p$------ | ----------kg $ha^{-1}$ $y^{-1}$---------- | |
| 5 | 52 | 16 | 0.5 | -530 | -30 |
| | | | 1.0 | -60 | 120 |
| | | | 2.0 | 890 | 410 |
| | 24 | 11 | 0.5 | -790 | -70 |
| | | | 1.0 | -570 | 30 |
| | | | 2.0 | -130 | 230 |
| 21 | 52 | 16 | 0.5 | 990 | 440 |
| | | | 1.0 | 2980 | 1050 |
| | | | 2.0 | 6970 | 2280 |
| | 24 | 11 | 0.5 | -85 | 250 |
| | | | 1.0 | 835 | 670 |
| | | | 2.0 | 2670 | 1510 |
| 80 | 52 | 16 | 0.5 | 6590 | 2160 |
| | | | 1.0 | 14180 | 4500 |
| | | | 2.0 | 29360 | 9170 |
| | 24 | 11 | 0.5 | 2500 | 1430 |
| | | | 1.0 | 6000 | 3040 |
| | | | 2.0 | 13010 | 6250 |

Table 6.9. Modeling results if *P. purpureum* is irrigated with dairy effluent based on crop nutrient requirement (i.e. zero or minimum excess nutrient).

| | INPUTS | | OUTPUTS | |
|---|---|---|---|---|
| $ET_p$ | Concentration of Nutrients in Effluent | | Irrigation Rate Required | |
| | N | P | | |
| ------mm $d^{-1}$------ | ----------------------mg $L^{-1}$----------------------- | | --------x $ET_p$--------- | |
| | | | N-based | P-based |
| 5 | 52 | 16 | 1.0 | 0.6 |
| | 24 | 11 | 2.0 | 0.9 |
| 21 | 52 | 16 | 0.25 | 0.30 |
| | 24 | 11 | 0.6 | 0.2 |
| 80 | 52 | 16 | 0.07 | 0.04 |
| | 24 | 11 | 0.14 | 0.05 |

**Table 6.10. Modeling results if *B. mutica* is irrigated with dairy effluent based on crop water requirement.**

| INPUTS | | | | OUTPUTS | |
|---|---|---|---|---|---|
| $ET_p$ | Concentration of Nutrients in Effluent | | Irrigation Rate | Excess Applied Nutrient | |
| | N | P | | N | P |
| mm $d^{-1}$ | -----------mg $L^{-1}$------------- | | -----x $ET_p$------ | ----------kg $ha^{-1}$ $y^{-1}$---------- | |
| 5 | 52 | 16 | 0.5 | -940 | -20 |
| | | | 1.0 | -470 | 130 |
| | | | 2.0 | 480 | 420 |
| | 24 | 11 | 0.5 | -1200 | -70 |
| | | | 1.0 | -980 | 35 |
| | | | 2.0 | -540 | 240 |
| 21 | 52 | 16 | 0.5 | 580 | 450 |
| | | | 1.0 | 2600 | 1100 |
| | | | 2.0 | 6560 | 2290 |
| | 24 | 11 | 0.5 | -500 | 260 |
| | | | 1.0 | 400 | 700 |
| | | | 2.0 | 2270 | 1520 |
| 80 | 52 | 16 | 0.5 | 6180 | 2170 |
| | | | 1.0 | 13770 | 4510 |
| | | | 2.0 | 28950 | 9180 |
| | 24 | 11 | 0.5 | 2090 | 1440 |
| | | | 1.0 | 5590 | 3050 |
| | | | 2.0 | 12600 | 6260 |

**Table 6.11. Modeling results if *B. mutica* is irrigated with dairy effluent based on crop nutrient requirement (i.e. zero or minimum excess nutrient).**

| INPUTS | | | OUTPUTS | |
|---|---|---|---|---|
| $ET_p$ | Concentration in Effluent | | Irrigation Rate Required | |
| | N | P | | |
| ------mm $d^{-1}$------ | -----------------------mg $L^{-1}$------------------------ | | ------x $ET_p$------- | |
| | | | N-based | P-based |
| 5 | 52 | 16 | 1.5 | 0.6 |
| | 24 | 11 | 3.2 | 0.8 |
| 21 | 52 | 16 | 0.4 | 0.1 |
| | 24 | 11 | 0.8 | 0.2 |
| 80 | 52 | 16 | 0.09 | 0.04 |
| | 24 | 11 | 0.20 | 0.05 |

Irrigating *B. mutica* with dairy effluent containing 52 mg N $L^{-1}$ and 16 mg P $L^{-1}$ at 1.0 $ET_p$ (21 mm $d^{-1}$) also led to excess application of N (2600 kg $ha^{-1}$ $y^{-1}$) and P (1100 kg $ha^{-1}$ $y^{-1}$) (Table 6.10). Doubling the irrigation rate (42 mm d-1) led to excess application of N (6560 kg $ha^{-1}$ $y^{-1}$) and P (2290 kg $ha^{-1}$ $y^{-1}$). At 0.5 $ET_p$, substantial excess applied N (580 kg $ha^{-1}$ $y^{-1}$) and P (450 kg $ha^{-1}$ $y^{-1}$) was also expected. Assuming that N and P concentrations in the effluent were low (24 mg $L^{-1}$ and 11 mg $L^{-1}$, respectively), the excess applied N was 2270 kg $ha^{-1}$ $y^{-1}$ and 400 kg $ha^{-1}$ $y^{-1}$ for 2.0 and 1.0, respectively and excess applied P was 1520 kg $ha^{-1}$ $y^{-1}$ and 700 kg $ha^{-1}$ $y^{-1}$, respectively. At 0.5 $ET_p$ irrigation rate, a deficit in applied N (500 kg $ha^{-1}$ $y^{-1}$) and a surplus in applied P (260 kg $ha^{-1}$ $y^{-1}$) was expected. Similar to *P. purpureum*, *B. mutica* required effluent irrigation of less than 1.0 $ET_p$ even at a lower N and P concentrations in the effluent to minimize excess applied nutrients (Table 6.11).

***Excess N and P if $ET_p$ is 80 mm $d^{-1}$***

Large amounts of N and P (2090 to 29360 kg $ha^{-1}$ $y^{-1}$) were expected to be applied in excess to the soil irrigated with effluent at 0.5, 1.0 and 2.0 $ET_p$ (80 mm $d^{-1}$) rates regardless of the concentrations of these nutrients in the effluent and irrespective of the grass species grown (Tables 6.8 and 6.10). To minimize the excess applied nutrients, the irrigation rate had to be reduced to 0.04 to 0.20 $ET_p$. (Tables 6.9 and 6.11). In this case, the crop water requirement was probably met if the rainfall was high enough or if freshwater supplementation was possible.

## Concentrations of Nutrient Leached

The above modeling of excess nutrients assumed no N and P contributions from rainfall and other sources. The scenarios created to determine the nutrient concentrations in the leaching water involved three $ET_p$ rates (5, 21 and 80 mm $d^{-1}$), three rainfall regimes (0, 110, and 290 mm $mo^{-1}$), three irrigation rates based on the $ET_p$ (0.5, 1.0 and 2.0) and two concentrations of effluent N (52 and 24 mg $L^{-1}$) and P (16 and 11 mg $L^{-1}$). Because modeling results were similar for both *P. purpureum* and *B. mutica*, only the results for *P. purpureum* were discussed here.

***$ET_p$ of 5 mm $d^{-1}$***

Regardless of N and P concentrations in the effluent, the N concentrations in the leaching water were negative at irrigation rates of 0.5 $ET_p$ and 1.0 $ET_p$, indicating a deficit in the amounts of applied N (Table 6.12). Negative N concentrations (deficit in applied N) also occurred if the irrigation rate is 2.0 $ET_p$ and the effluent used for irrigation contains 24 mg N $L^{-1}$ and 11 mg P $L^{-1}$, whether the rainfall is 0, 110 or 290 mm $mo^{-1}$. The concentrations of applied P were negative (deficit in applied P) in all cases where the irrigation rate was 0.5 $ET_p$ regardless of rainfall amount and effluent P concentration. The P concentration in the leaching water ranged from 0.5 to 7.0 mg $L^{-1}$ if the irrigation rate was 1.0 $ET_p$, and from 3 to 11 mg $L^{-1}$ if the irrigation rate was 2.0 $ET_p$.

Inputs:

| | |
|---|---|
| percent N in effluent | 24 |
| Rate | 1 |
| mm per d | 21 |
| kg DM per ha per y | 51000 |
| mg per kg | 1.97 |
| L rain per ha per y | 0 |

Outputs:

| | |
|---|---|
| L per ha per y | 76650000 |
| kg N per ha per y | 1839.6 |
| kg per L | 2.4e-5 |
| kg per ha per y N | 1005 |
| L water per ha per y | 76650000 |
| kg per ha per y | 835 |
| mg N per L | 11 |

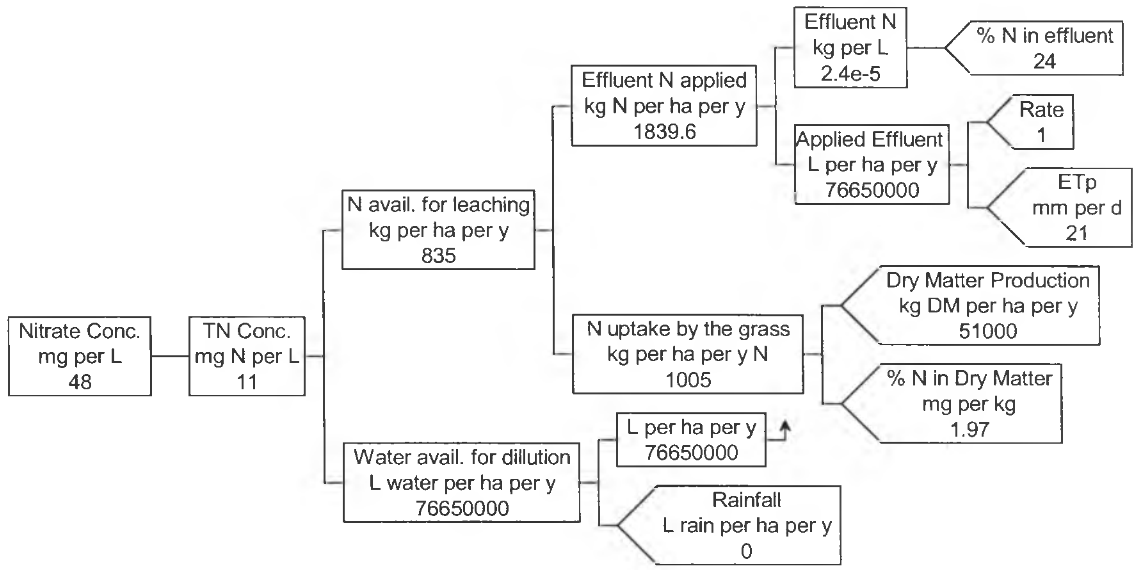



**Figure 6.6. An example of a DecisionPro model to determine the nutrient concentration in the water leaching through the soil profile.**

### *$ET_p$ is 21 mm $d^{-1}$*

Irrigating with effluent containing 52 mg N $L^{-1}$ and 16 mg P $L^{-1}$ at 0.5 $ET_p$ rate resulted in N concentrations in the leaching water ranging from 14 to 26 mg $L^{-1}$, while P concentrations were from 6 to 12 mg $L^{-1}$ (Table 6.13). If the irrigation rate is 1.0 $ET_p$, the concentration of N was between 27 to 39 mg $L^{-1}$ (0, 110 and 290 mm mo rainfall), while that of P was between 9 to 14 mg $L^{-1}$ (0, 110 and 290 mm mo rainfall). At 2.0 $ET_p$, the N concentration ranged from 37 to 45 mg $L^{-1}$ and P ranged from 12 to 15 mg $L^{-1}$.

Table 6.12. N and P concentrations in the water leaching through the soil profile in a soil planted to *P. purpureum* receiving dairy effluent at various irrigation rates based on $ET_p$ (5 mm $d^{-1}$), rainfall regimes, and effluent nutrient concentrations.

| INPUTS | | | | OUTPUTS | | | |
|---|---|---|---|---|---|---|---|
| | | | | Concentration of Nutrients in Water Leaching Through the Soil Profile | | | |
| Rainfall | Irrigation Rate | Nutrient Concentration in Effluent | | 100% Leaching Rate | | 13% Leaching Rate | |
| | | N | P | N | P | N | P |
| mm $mo^{-1}$ | x $ET_p$ | ----mg $L^{-1}$--- | | ----------mg $L^{-1}$-------- | | ---------mg $L^{-1}$------- | |
| 0 | 0.5 | 52 | 16 | -58 | -3 | -7.5 | -0.4 |
| | 1.0 | | | -3 | 7 | -0.4 | 0.9 |
| | 2.0 | | | 25 | 11 | 3.3 | 1.4 |
| | 0.5 | 24 | 11 | -86 | -8 | -11.2 | -1.0 |
| | 1.0 | | | -31 | 2 | -4.0 | 0.3 |
| | 2.0 | | | -4 | 6 | -0.5 | 0.8 |
| 110 | 0.5 | 52 | 16 | -24 | -1 | -3.1 | -0.1 |
| | 1.0 | | | -2 | 4 | -0.3 | 0.5 |
| | 2.0 | | | 18 | 8 | 2.3 | 1.0 |
| | 0.5 | 24 | 11 | -35 | -3 | -4.6 | -0.4 |
| | 1.0 | | | -18 | 0.9 | -2.3 | 0.1 |
| | 2.0 | | | -3 | 5 | -0.4 | 0.7 |
| 290 | 0.5 | 52 | 16 | -12 | -0.6 | -1.6 | -0.1 |
| | 1.0 | | | -1 | 2 | -0.1 | 0.3 |
| | 2.0 | | | 13 | 6 | 1.7 | 0.8 |
| | 0.5 | 24 | 11 | -18 | -2 | -2.3 | -0.3 |
| | 1.0 | | | -11 | 0.5 | -1.4 | 0.1 |
| | 2.0 | | | -2 | 3 | -0.3 | 0.4 |

Regardless of the amount of rainfall, irrigating with effluent containing 24 mg N $L^{-1}$ and 11 mg P $L^{-1}$ at 0.5 $ET_p$ rate resulted in a deficit in N applied, hence, the negative values for N concentrations in the leaching water. At 1.0 $ET_p$ rate, the N concentration ranges from 8 to 11 mg $L^{-1}$, while P concentration ranged from 6 to 9 mg $L^{-1}$. Higher N (14 to 17 mg $L^{-1}$) and P (8 to 10 mg $L^{-1}$) concentrations were expected at the 2.0 $ET_p$ irrigation rate.

Table 6.13. N and P concentrations in the leaching water in a soil planted to *P. purpureum* receiving dairy effluent at various irrigation rates based on $ET_p$ (21 mm $d^{-1}$), rainfall regimes, and effluent nutrient concentrations.

| INPUTS | | | | OUTPUTS | | | |
|---|---|---|---|---|---|---|---|
| | | | | Concentration of Nutrients in Water Leaching Through the Soil Profile | | | |
| Rainfall | Irrigation Rate | Nutrient Concentration in Effluent | | 100% Leaching Rate | | 13% Leaching Rate | |
| | | N | P | N | P | N | P |
| mm $mo^{-1}$ | x $ET_p$ | ----mg $L^{-1}$--- | | ----------mg $L^{-1}$-------- | | ----------mg $L^{-1}$-------- | |
| 0 | 0.5 | 52 | 16 | 26 | 12 | 3.4 | 1.6 |
| | 1.0 | | | 39 | 14 | 5.1 | 1.8 |
| | 2.0 | | | 45 | 15 | 5.9 | 2.0 |
| | 0.5 | 24 | 11 | -2 | 7 | -0.3 | 0.9 |
| | 1.0 | | | 11 | 9 | 1.4 | 1.2 |
| | 2.0 | | | 17 | 10 | 2.2 | 1.3 |
| 110 | 0.5 | 52 | 16 | 19 | 9 | 2.5 | 1.2 |
| | 1.0 | | | 33 | 12 | 4.3 | 1.6 |
| | 2.0 | | | 42 | 14 | 5.5 | 1.8 |
| | 0.5 | 24 | 11 | -2 | 5 | -0.3 | 0.7 |
| | 1.0 | | | 9 | 7 | 1.2 | 0.9 |
| | 2.0 | | | 16 | 9 | 2.1 | 1.2 |
| 290 | 0.5 | 52 | 16 | 14 | 6 | 1.8 | 0.8 |
| | 1.0 | | | 27 | 9 | 3.5 | 1.2 |
| | 2.0 | | | 37 | 12 | 4.8 | 1.6 |
| | 0.5 | 24 | 11 | -1 | 3 | -0.1 | 0.4 |
| | 1.0 | | | 8 | 6 | 1.0 | 0.8 |
| | 2.0 | | | 14 | 8 | 1.8 | 1.0 |

***$ET_p$ is 80 mm $d^{-1}$***

At irrigation rates of 0.5, 1.0 and 2.0 $ET_p$ and given a rainfall of 0, 110 and 290 mm $mo^{-1}$, the nutrient concentrations in the leaching water ranged from 36 to 50 mg N $L^{-1}$ and 12 to 16 mg P $L^{-1}$, respectively, if the effluent had 52 mg N $L^{-1}$ and 16 mg P $L^{-1}$ (Table 6.14). The nutrient concentrations in the leaching water ranged from 14 to 22 mg N $L^{-1}$ and 8 to 11 mg P $L^{-1}$, if the effluent had 24 mg N $L^{-1}$ and 11 mg P $L^{-1}$.

Table 6.14. N and P concentrations in the leaching water in a soil planted to *P. purpureum* receiving dairy effluent at various irrigation rates based on $ET_p$ (80 mm $d^{-1}$), rainfall regimes, and effluent nutrient concentrations.

| INPUTS | | | | OUTPUTS | | | |
|---|---|---|---|---|---|---|---|
| | | | | Concentration of Nutrients in Water Leaching Through the Soil Profile | | | |
| Rainfall | Irrigation Rate | Nutrient Concentration in Effluent | | 100% Leaching Rate | | 13% Leaching Rate | |
| | | N | P | N | P | N | P |
| mm $mo^{-1}$ | x $ET_p$ | ----mg $L^{-1}$--- | | ----------mg $L^{-1}$-------- | | ----------mg $L^{-1}$-------- | |
| 0 | 0.5 | 52 | 16 | 45 | 15 | 5.9 | 2.0 |
| | 1.0 | | | 49 | 15 | 6.4 | 2.0 |
| | 2.0 | | | 50 | 16 | 6.5 | 2.1 |
| | 0.5 | 24 | 11 | 17 | 10 | 2.2 | 1.3 |
| | 1.0 | | | 21 | 10 | 2.7 | 1.3 |
| | 2.0 | | | 22 | 11 | 2.9 | 1.4 |
| 110 | 0.5 | 52 | 16 | 41 | 14 | 5.3 | 1.8 |
| | 1.0 | | | 47 | 15 | 6.1 | 2.0 |
| | 2.0 | | | 49 | 15 | 6.4 | 2.0 |
| | 0.5 | 24 | 11 | 16 | 9 | 2.1 | 1.2 |
| | 1.0 | | | 20 | 10 | 2.6 | 1.3 |
| | 2.0 | | | 22 | 11 | 2.9 | 1.4 |
| 290 | 0.5 | 52 | 16 | 36 | 12 | 4.7 | 1.6 |
| | 1.0 | | | 43 | 14 | 5.6 | 1.8 |
| | 2.0 | | | 48 | 15 | 6.2 | 2.0 |
| | 0.5 | 24 | 11 | 14 | 8 | 1.8 | 1.0 |
| | 1.0 | | | 18 | 9 | 2.3 | 1.2 |
| | 2.0 | | | 21 | 10 | 2.7 | 1.3 |

In general, the N and P concentrations in the leaching water were always lower at higher amounts of rainfall because of more water available for dilution. At the same time, however, high amounts of rainfall can exacerbate leaching (Lado et al., 2005). Leaching may be a concern under the assumption that all of the excess applied N and P leached because the predicted concentrations of N regardless of the amount of rainfall exceeded the limit for the concentration of N in groundwater (10 mg N $L^{-1}$ or 44 mg $NO_3$ $L^{-1}$) set by the EPA. In reality, not all of the N may leach if plant uptake was high (Korsaeth et al., 2003), the $ET_p$ was high (Guillard and Kopp, 2004), and if positively charged zone was present in the subsoil (Deenik, 1997), among other factors. There may also be losses through ammonia volatilization and denitrification (Wilkison et al., 2000; Baker and Timmons, 1994) as well as immobilization in soil organic matter (Wilkison et al., 2000; Baker and Timmons, 1994) and ammonium sorption by the soil (Fernando et al., 2005). Reinhart (2000) found that about 166 kg N $ha^{-1}$ leached out of the total leachable N of 1342 kg $ha^{-1}$, or about 13% leaching rate. Assuming that only 13% of the excess N was leached, then, the N concentrations in the leachate (highest value was 6.5 mg $L^{-1}$) were lower than the EPA standard.

Not all of the excess applied P also leached in actual conditions because some of the P may be immobilized by microorganisms (Gagnon and Simard, 1999), sorbed by the soil (Siddique and Robinson, 2004) or move with runoff (Eghball et al., 1996) during rainy periods. At 100% N leaching rate, the predicted concentrations of P were mostly higher than that reported for soils receiving fertilizers or organic wastes (7 to 8 mg P $L^{-1}$) (Pierzynski et al., 2005) and those measured in soil solutions in the field experiment (Figs. 3.6 to 3.9). Assuming that only 13% of the P leached, the highest predicted concentration of P in the leachate was 2.1 mg $L^{-1}$.

## General Discussions

These results show that irrigating with effluent to meet the crop water requirement results in considerable amount of excess nutrients applied, except when the $ET_p$ is low that a deficit in N application may occur. If irrigation was based on nutrient requirement of the grasses, irrigating based on N requirement resulted in excessive application of P, whereas irrigating based on P requirement resulted in inadequate application of N. The total applied P, thus, limits the amount of effluent that can be used for irrigation. Also, additional N fertilization was required if the irrigation was based on crop water requirement when the $ET_p$ was low (e.g., 5 mm $d^{-1}$) and when effluent irrigation was based on crop P requirement. Both irrigation objectives can result in either excessive application or inadequate application of nutrients depending on the irrigation rate.

In cases where excess applied nutrients were expected, these nutrients can be lost to the environment through leaching and runoff/erosion. Ideally, nutrient losses to the environment can be minimized by applying water and nutrients to match the plant uptake rates (Assouline, 2002). However, achieving this balance may be difficult with effluent irrigation because of the low nutrient content of effluent per volume (Mullins et al., 2005; Kohl, 2004). The crop water requirement was not met if irrigation rates were to satisfy the crop nutrient requirement, and *vice versa*. For example, a much lower than 1.0 $ET_p$ irrigation rate was required to match the P needs of the crop without applying an excess amount of this nutrient. But in this case, the leaching requirement may also not be achieved. Meeting the leaching requirement is an important concern when irrigating with salt-rich wastewater. Thus, supplemental freshwater irrigation is needed if rainfall is not enough to supply the additional water needed to meet the crop water requirements and/or leaching requirements. If both alternatives are not possible, the farmer must be prepared to implement practices that will minimize salt accumulation such as effluent clean-up prior to irrigation (e.g., multi-soil layer systems) or amendment application (e.g., gypsum).

The above modeling approach with Decision Pro has limitations. The model can overpredict or underpredict the output because constant rainfall, $ET_p$ and nutrient concentration in the effluent were assumed. In reality, these factors vary throughout the year. Lower nutrient concentrations in the effluent and in the soil solution are expected during the rainy season due to dilution from rainfall. However, it may also result in higher nutrient concentrations because of the sediment-loaded runoff which may be enriched with nutrients as well as some degree of agitation of the lagoon by the water from runoff and wind movement. In this study, for example, the average effluent N and P concentrations were 34 mg $L^{-1}$ and 15 mg $L^{-1}$, respectively during the rainy season and 4 mg $L^{-1}$ and 6 mg $L^{-1}$ during the dry season. The model also assumed fixed dry matter production and nutrient concentrations in the forage, which in reality, may vary with changes in $ET_p$, rainfall, soil and other conditions.

An accurate estimate of the land area required for effluent application, the limit on animal population and the potential runoff and leaching losses of nutrients need to consider other factors such as the amount of rainfall falling into the lagoon and the volume of runoff from the surrounding areas. This is part of the requirements for the National Pollutant Discharge Elimination System (NPDES) regulations that dairy producers have to comply with (US-EPA, 1995). During the study period, we observed that the volume of effluent in the lagoon dramatically increased after each heavy rainfall event, usually occurring during the Nov-Mar period. In the current model, the estimated area requirement were, therefore, not realistic and greatly underestimated because those factors were not considered. This is one aspect of the model that needs improvement so that the information can be used by farmers and agents in developing a Nutrient Management Plan required under the NPDES regulation.

The model also cannot take into account some of the soil characteristics such as P sorption capacity, water retention capacity, degree of aggregation, and shrink-swell characteristic, among others. Future modeling work may consider assigning probabilities for these factors to take them into account in the model. Phosphorus applied in soils with high P sorption capacity is not all plant-available. Soils with high water-holding capacity and shrink-swell characteristics change in volume with changes in water content resulting in swelling when wet and cracking when dry, and cracks could serve as avenues for preferential flow of water and solutes (Bronswijk, 1989; Tanton et al., 1988). Soils that are well-aggregated behave like sands in terms of water movement (Uehara and Gillman, 1980). These conditions could contribute to nutrient movement to the lower soil profiles that could lead to nutrient losses to the groundwater.

There are other questions in the minds of a dairy producer that need to be incorporated in the decision-making process. Among the major questions are the economic feasibility of putting up and maintaining the irrigation system and the net benefits that can be derived from the practice. Should there be an accumulation of or deficit in the amount of applied nutrients and salts, the producer is interested to know what ameliorative actions are needed and the costs associated with it.

## Conclusions

A decision-aid tool will help dairy producers make important decisions with regard to effluent management. Developing this kind of tool is a valuable exercise to gain an understanding of the various factors that warrant considerations for improving management. The total applied P sets the limit on the number of animals to be raised and determines how much area is needed to utilize the effluent for forage production. Achieving a balance between crop water needs and nutrient requirement is difficult with effluent irrigation due to the generally low nutrient content of effluent. High rainfall increases the volume of wastewater in the lagoon and increases the amount of available water for diluting the nutrients that potentially leach through the soil profile. Improvements in the current model need to be done to obtain more realistic estimates of the area requirement, animal population and amounts of nutrients that can be lost to the environment as well as to address other practical concerns of a dairy farmer. How to maximize the leaching fraction while minimizing the amount of nutrients that can leach to the groundwater is an important effluent management question that can be addressed in future modeling work. More studies are needed to develop an algorithm, similar to those used in the NuMaSS and PDSS, for a model that applies to animal effluent application. Future work will also involve compiling the individual models into a single user interface that can be run by the user in a personal computer or via the internet.

# CHAPTER 7. GENERAL SUMMARY, CONCLUSIONS AND RECOMMENDATIONS

## Summary

Livestock and dairy producers are under pressure to increase the efficiency, productivity and viability of their businesses while complying with environmental regulations. Dairy operations generate large quantities of effluent, which are stored in lagoons to allow solids to settle and the water to evaporate. Lagoons, however, have finite storage capacity and can overflow, thereby potentially contributing to the pollution of land and associated water bodies. Dairy producers, thus, urgently need management options by which wastewaters can be utilized for a more sustainable and environment-friendly dairy production. This study assessed the effects of effluent irrigation on plant and soil (Cumulic Haplustoll) properties. Five tropical grasses—bana (*Pennisetum purpureum* S. cv. HA 5690), California (*Brachiaria mutica* S.), signal (*Brachiaria decumbens* S.), star (*Cynodon nlemfuensis* V.), and suerte (*Paspalum atratum* S.)—were planted in Waianae, Hawai‘i. Two rates of dairy effluent were supplied daily through a subsurface (20 to 25 cm depth) drip irrigation system based on the potential evapotranspiration ($ET_p$)—2.0 $ET_p$ (7 to 44 mm $d^{-1}$) and 0.5 $ET_p$ (2 to 11 mm $d^{-1}$)—in the site. Treatments were arranged in an augmented completely randomized design. Selected soil samples collected between July 2003 and August 2006 were used in the P sorption experiment and for extraction with the recommended extractant (Olsen) and acidic extractants (Modified Truog and HCl).

Daily irrigation with effluent for two years, even at the 2.0 $ET_p$ rate did not result in significant increases in Olsen extractable soil P and soil solution total P, which may be ascribed to the relatively high levels at the beginning of the experiment (81 to 176 mg $kg^{-1}$ and 3 to 9 mg $L^{-1}$, respectively). At the 2.0 $ET_p$ irrigation rate, a relatively low flux of P (95 to 131 kg $ha^{-1}$ $y^{-1}$) was observed. Some of the soil P may be precipitating with Ca, given the high amounts of Ca added to the soil from effluent (850 to 2900 kg Ca $ha^{-1}$ $y^{-1}$) and the already high Ca content of the soil (5700 to 6400 mg $kg^{-1}$) in the site prior to effluent irrigation. The pH of the soil also remained high (7.4 to 8.9) for the duration of the study. Calcium-phosphate precipitation was suggested by the higher amounts of acid extractable soil P than the Olsen extractable soil P. The extractable soil Ca also declined with time, further supporting the hypothesis of Ca-P formation. Calculation of the phosphate potential indicated that hydroxyapatite may be forming in this soil. An important implication of the Ca-P formation is the reduced likelihood of P being dissolved in runoff water and thus, lower potential of soil P to be transferred to water bodies. No significant changes in the P sorption capacity of this effluent-irrigated soil were observed possibly due to the relatively short duration (2 years) of the effluent application.

Soil pH generally increased over time from 7.4 to 7.8 in July 2003 to 8.2 to 8.9 in Aug 2006. Soil solution pH also remained high over time. The high pH of the soil could be explained by the high pH (7.9 to 8.9), carbonate (55 to 62 mg $CaCO_3$ $L^{-1}$) and bicarbonate (1460 to 2210 mg $CaCO_3$ $L^{-1}$) content of the effluent used for irrigation. The fluctuations in the pH of the soil and soil solution indicated that this effluent-irrigated soil has a high pH buffering capacity.

The saturated paste extract method of determining soil electrical conductivity ($EC_{spe}$) gave the highest EC values due to less dilution of electrolyte in the solution compared to the 1:1 and 1:2 soil:water method of EC determination. In the effluent-irrigated plots, the soil $EC_{spe}$ declined from 18.0 dS $m^{-1}$ in July 2003 to 2.7 dS $m^{-1}$ in Aug 2006 possibly due to freshwater irrigation the grasses during the establishment period and the freshwater flush in the dripline after each irrigation event during the 2-y period of effluent irrigation. The soil $EC_{spe}$ and soil solution EC values between Aug 2004 and Aug 2006 were lower than the U.S. Salinity Laboratory's critical level for soils classified as saline (4.0 dS $m^{-1}$). No differences in the EC values of soils receiving 0.5 $ET_p$ and 2.0 $ET_p$ were observed which may be explained by the high buffering capacity of the soil.

Soil ESP decreased despite repeated effluent application. The mean ESP values between Aug 2004 and Aug 2006 (ranging from 6.4 to 10.2%) remained lower than the 15% critical ESP value for classifying soils as sodic. The SAR (3.1 to 5.3) of the dairy effluent used for irrigation remained at low levels for the duration of the study, which explains the lack of increases in soil ESP. The SAR of the soil solution (4 to 9) also remained below 13 or 15--the critical level for classifying soils as sodic. The soil EPP increased with repeated effluent irrigation regardless of the rate of application owing to the high amounts of effluent K applied. The leaching fraction ranged from 0.28 to 0.38 at the 2.0 $ET_p$ application rate and 0.34 to 0.42 at the 0.5 $ET_p$ application rate. The 0.5 $ET_p$ application rate does not usually meet the actual water requirements. Results suggest that effluent irrigation did not result in excessive increases in salinity and sodicity.

Regarding plant responses, highest dry matter yields and nutrient uptake were obtained for *B. mutica* (43 to 57 Mg $ha^{-1}$ $y^{-1}$) and *P. purpureum* (50 to 51 Mg $ha^{-1}$ $y^{-1}$).

*Paspalum atratum* yielded about 38 Mg kg dry matter ha$^{-1}$ y$^{-1}$ when irrigated at 0.5 and 2.0 $ET_p$ rates. While *P. purpureum* removed the most K (1700 to 2000 kg ha$^{-1}$ y$^{-1}$), *P. atratum* removed the most Mg (200 to 270 kg ha$^{-1}$ y$^{-1}$) from the soil. Nutrient removal of grasses was 30 to 187% N, 13 to 86% P and 2 to14% K in the applied effluent. When bahia grass (*Paspalum notatum*) forage critical levels were used as the reference, average annual N concentration was in the deficient range (1.51 to 2.39%), P was nearly at adequate levels (0.20 to 0.42%), and K was above the sufficient level (2.5 to 4.2%). With the dry matter yields of *B. mutica* and *P. purpureum* and milk production of 9 kg d$^{-1}$ of a 607-kg cattle with a dry matter intake of 2.1% of body weight and 4% fat corrected milk, the carrying capacity of this cut-and-carry system was 8 to 11 cows ha$^{-1}$ y$^{-1}$. Forage quality such as CP (10 to 16%), NDF (52 to 62%) and ADF (28 to 36%) were slightly different than the NRC requirements (>15% CP, minimum of 25 to 34% NDF, and minimum of 17 to 21% ADF) but these levels are acceptable for feeding to lactating dairy cattle especially when adjustments on the ration are made.

A decision-making structure was presented. Using the information from the field experiment, some effluent management questions were modeled to provide dairy producers with information that will help them improve their management of this resource. Scenarios involving different potential evapotranspiration ($ET_p$) rates, rainfall regimes, irrigation rate, effluent N and P contents, and nutrient uptake of the grasses—*P. purpureum* and *B. mutica*—were evaluated. Modeling results showed that the total applied phosphorus is the determining factor in deciding how many animals may be raised and how much area may be utilized to produce the forage. An estimate of the area required was highly underestimated while and the cattle population limit was likely overestimated by the current models because the rainfall and runoff contributions were not considered in estimating the total effluent production. If the $ET_p$ remains at a high level, irrigating based on the consumptive water use of the crop results in excessive amount of nutrients applied. Nonetheless, irrigating based on the $ET_p$ is an acceptable practice when using salt-rich wastewater such as dairy effluent to meet the leaching requirement. Irrigating the grasses based on the crop nutrient requirement results in much less water applied thereby requiring freshwater supplementation, if rainfall is not enough, to meet crop water requirement and leaching fraction. High rainfall increases the quantity of effluent in the lagoon and the available water for diluting the concentrations of nutrients in the effluent and those that could leach through the soil profile.

## Conclusions

In general, effluent irrigation to produce the forage is an effective means of closing the open nutrient cycle in the milk production system. *Brachiaria mutica* and *P. purpureum* were the best choices for forage production, especially if the prime objective was to effectively remove nutrients from the soil in order to maximize the application of liquid effluent without serious nutrient accumulation in the soil. Results of this study suggest that high productivity forage grown in this type of soil can be irrigated with dairy effluent at 2.0 $ET_p$ (7 to 44 mm $d^{-1}$)—a rate that is substantially higher than that designed for most irrigation objectives. This is necessary to meet the leaching requirement that is important when irrigating with salt-rich wastewaters.

## Recommendations

*Brachiaria mutica* and *P. purpureum* were the most productive species when irrigated with dairy effluent. However, the K concentration of *P. purpureum* was relatively high and should be taken into account when formulating the cattle ration to avoid metabolic disorders due to excessive K intake that creates an imbalance in K, Ca and Mg. Supplemental fertilization with micronutrients (zinc, iron, copper and manganese) to the grasses was necessary to sustain adequate levels of micronutrients and biomass productivity.

The interpretation of soil EC and ESP values is not straightforward. Care must be exercised in interpreting the soil EC values measured using the 1:1 and 1:2 soil:water extracts and directly measured from saturated paste because the criteria for classifying soils as saline was developed based on saturated paste extract method. Also, there is a need to conduct more studies on the critical ESP values for various soils in Hawaii. There is disagreement on the critical ESP values for classifying soils as sodic due to the differences in the conditions whereby those standards were developed.

Regarding the irrigation system, a split application of effluent in multiple events during the day may minimize the saturation of the soil with effluent, and reduce the loss of applied effluent through evaporation especially during the hottest parts of the day. This strategy may also be more practical for prolonging the lifespan of the irrigation pump.

Long-term repeated applications of lagoon effluents, however, can have numerous negative agronomic and environmental effects. Additional monitoring is, thus, needed to determine the longer-term impacts of dairy effluent application on plant and soil properties in these conditions. Studies on the impacts of effluent application on other soil (high P sorption capacity, more permeability, low initial salt and P levels) and environmental conditions (higher rainfall, low $ET_p$, lower solar radiation) are needed.

Measurements of hydraulic conductivity and infiltration rate also need to be done to support the information provided by the ESP and EPP. As with any other irrigation system, a limitation to this system is the limited opportunity to irrigate with effluent during the rainy season.

More modeling work needs to be done to address the limitations of the current model. Improvements in the model may include accounting for the fluctuations (monthly or seasonal) in the concentrations of nutrients in the effluent, $ET_p$, rainfall, and nutrient uptake and dry matter production of the grasses. The improved model may also incorporate soil properties such as P sorption capacity, aggregation (infiltration rate and hydraulic conductivity), water saturation and water holding capacity, among others. Future modeling work may focus on how to maximize the leaching fraction while minimizing the amount of nutrients that could leach to the groundwater. An algorithm similar to the available nutrient management software (NuMaSS, PDSS, etc.) needs to be developed for effluent management.

# Appendices

## Appendix Tables

**Table A.1. Least square means of soil solution total P for various treatment interactions.**

Grass x Sampling depth

| Grass Species | Sampling Depth | P | Grass Species | P |
|---|---|---|---|---|
| | cm | mg $L^{-1}$ | | mg $L^{-1}$ |
| *B. mutica* | 15 | 4.09 | *C. nlemfuensis* | 4.03 |
| | 35 | 4.16 | | 3.29 |
| | 70 | 3.13 | | 3.29 |
| | 100 | 3.83 | | 5.49 |
| *B. decumbens* | 15 | 3.61 | *P. atratum* | 4.13 |
| | 35 | 3.87 | | 3.88 |
| | 70 | 3.98 | | 3.59 |
| | 100 | 3.90 | | 3.27 |

Grass x Time

| Grass | Date | P |
|---|---|---|
| | | mg $L^{-1}$ |
| *B. mutica* | 1/2/05 | 3.29 |
| | 8/12/06 | 4.38 |
| *B. decumbens* | 6/24/04 | 6.36 |
| | 12/18/04 | 3.07 |
| *C. nlemfuensis* | 6/4/04 | 4.98 |
| | 9/5/05 | 2.19 |
| | 8/12/06 | 5.50 |
| *P. atratum* | 6/4/04 | 6.30 |
| | 9/7/04 | 3.96 |
| | 8/12/06 | 4.51 |

Irrigation rate x Sampling depth

| Irrigation rate | Sampling Depth | P |
|---|---|---|
| | cm | mg $L^{-1}$ |
| *2.0 ETp (7 to 44 mm $d^{-1}$)* | 15 | 4.55 |
| | 35 | 4.17 |
| | 70 | 3.42 |
| | 100 | 4.26 |
| *0.5 ETp (2 to 11 mm $d^{-1}$)* | 15 | 3.54 |
| | 35 | 3.60 |
| | 70 | 3.61 |
| | 100 | 3.50 |

Table A.2. Annual excess or deficit in applied macronutrients to the soil receiving dairy effluent, Waianae, Oahu, Hawaii.

| | | N | P | K | Ca | Mg | Na |
|---|---|---|---|---|---|---|---|
| **Aug 2004 to Aug 2005** | | | | | | | |
| *2.0 ETp (7 to 44 mm d$^{-1}$)* | *B. decumbens* | 262 | 124 | 7371 | 626 | 1221 | 2761 |
| | *C. nlemfuensis* | 431 | 476 | 40779 | 2750 | 5749 | 14900 |
| | *P. atratum* | 484 | 490 | 40516 | 2672 | 5591 | 14933 |
| | *P. purpureum* | -668 | -3 | 9867 | 620 | 1487 | 4238 |
| *0.5 ETp (2 to 11 mm d$^{-1}$)* | *B. decumbens* | -60 | 12 | 2460 | 206 | 421 | 1001 |
| | *C. nlemfuensis* | -152 | 76 | 11145 | 746 | 1608 | 4246 |
| | *P. atratum* | -325 | 67 | 10705 | 627 | 1400 | 4249 |
| **Aug 2005 to Aug 2006** | | | | | | | |
| *2.0 ETp (7 to 44 mm d$^{-1}$)* | *P. purpureum* | 1085 | 568 | 45495 | 4030 | 7248 | 15622 |
| | *B. mutica* | 1012 | 563 | 45582 | 4101 | 7279 | 15384 |
| | *C. nlemfuensis* | 1378 | 606 | 46471 | 4097 | 7288 | 15601 |
| | *P. atratum* | 1868 | 670 | 46676 | 4122 | 7238 | 15628 |
| *0.5 ETp (2 to 11 mm d$^{-1}$)* | *P. purpureum* | 81 | 64 | 13239 | 1130 | 2181 | 4818 |
| | *B. mutica* | 361 | 101 | 13689 | 1205 | 2186 | 4667 |
| | *C. nlemfuensis* | -90 | 62 | 13967 | 1145 | 2209 | 4822 |
| | *P. atratum* | 630 | 140 | 14085 | 1164 | 2083 | 4810 |
| **Average (Aug 2004 to Aug 2006)** | | | | | | | |
| *2.0 ETp (7 to 44 mm d$^{-1}$)* | *P. purpureum* | 1085 | 568 | 45495 | 4030 | 7248 | 15622 |
| | *B. decumbens* | 262 | 124 | 7371 | 626 | 1221 | 2761 |
| | *B. mutica* | 1012 | 563 | 45582 | 4101 | 7279 | 15384 |
| | *C. nlemfuensis* | 905 | 541 | 43625 | 3423 | 6519 | 15251 |
| | *P. atratum* | 1176 | 580 | 43596 | 3397 | 6415 | 15281 |
| *0.5 ETp (2 to 11 mm d$^{-1}$)* | *P. purpureum* | -294 | 31 | 11553 | 875 | 1834 | 4528 |
| | *B. decumbens* | -60 | 12 | 2460 | 206 | 421 | 1001 |
| | *B. mutica* | 361 | 101 | 13689 | 1205 | 2186 | 4667 |
| | *C. nlemfuensis* | -121 | 69 | 12556 | 945 | 1909 | 4534 |
| | *P. atratum* | 152 | 104 | 12395 | 896 | 1742 | 4529 |

**Table A.3. Annual excess or deficit in applied micronutrients to the soil receiving dairy effluent, Waianae, Oahu, Hawaii.**

| | | Fe | Mn | Zn | Cu | B |
|---|---|---|---|---|---|---|
| **Aug 2004 to Aug 2005** | | | | | | |
| *2.0 ETp (7 to 44 mm $d^{-1}$)* | *B. decumbens* | -8 | 0 | 0 | 1 | 9 |
| | *C. nlemfuensis* | 3 | -1 | 0 | 8 | 48 |
| | *P. atratum* | -23 | -1 | -2 | 8 | 47 |
| | *P. purpureum* | -36 | -3 | -3 | 2 | 13 |
| *0.5 ETp (2 to 11 mm $d^{-1}$)* | *B. decumbens* | -13 | -1 | 0 | 0 | 3 |
| | *C. nlemfuensis* | -11 | -1 | -1 | 2 | 13 |
| | *P. atratum* | -50 | -3 | -2 | 2 | 13 |
| **Aug 2005 to Aug 2006** | | | | | | |
| *2.0 ETp (7 to 44 mm $d^{-1}$)* | *P. purpureum* | 43 | 7 | -1 | 20 | 58 |
| | *B. mutica* | 50 | 6 | -1 | 20 | 58 |
| | *C. nlemfuensis* | 53 | 6 | -2 | 21 | 58 |
| | *P. atratum* | 42 | 8 | -1 | 21 | 58 |
| *0.5 ETp (2 to 11 mm $d^{-1}$)* | *P. purpureum* | 5 | 1 | -2 | 6 | 18 |
| | *B. mutica* | 6 | 1 | -2 | 6 | 18 |
| | *C. nlemfuensis* | 9 | 1 | -2 | 6 | 18 |
| | *P. atratum* | 5 | 0 | -2 | 6 | 18 |
| **Average (Aug 2004 to Aug 2006)** | | | | | | |
| *2.0 ETp (7 to 44 mm $d^{-1}$)* | *P. purpureum* | 43 | 7 | -1 | 20 | 58 |
| | *B. decumbens* | -8 | 0 | 0 | 1 | 9 |
| | *B. mutica* | 50 | 6 | -1 | 20 | 58 |
| | *C. nlemfuensis* | 28 | 3 | -1 | 14 | 53 |
| | *P. atratum* | 9 | 3 | -1 | 14 | 52 |
| *0.5 ETp (2 to 11 mm $d^{-1}$)* | *P. purpureum* | -15 | -1 | -2 | 4 | 15 |
| | *B. decumbens* | -13 | -1 | 0 | 0 | 3 |
| | *B. mutica* | 6 | 1 | -2 | 6 | 18 |
| | *C. nlemfuensis* | -1 | 0 | -2 | 4 | 16 |
| | *P. atratum* | -23 | -1 | -2 | 4 | 15 |

**Table A.4. Phosphorus sorption coefficients for Freundlich and Langmuir equations for soils† receiving dairy effluent, Oahu, Hawaii.**

| | Freundlich††: $Y = ax^{1/b}$ | | | | | |
|---|---|---|---|---|---|---|
| | $R^2$ | a | b | $R^2$ | a | b |
| **2.0 ETp** *(7 to 44 mm $d^{-1}$)*, **0-15 cm** | | | | **2.0 $ET_p$** *(7 to 44 mm $d^{-1}$)*, **15-30 cm** | | |
| Jul '03 | 0.93 | 29.28 | 1.23 | 0.97 | 47.90 | 1.68 |
| Aug '04 | 0.90 | 4.42 | 0.88 | 0.94 | 0.19 | 0.47 |
| Feb '05 | 0.96 | 10.56 | 1.01 | 0.92 | 7.45 | 0.94 |
| Aug '05 | 0.90 | 6.28 | 0.89 | 0.92 | 4.38 | 0.81 |
| Feb'06 | 0.89 | 4.87 | 0.86 | 0.96 | 2.10 | 0.66 |
| Aug '06 | 0.94 | 3.40 | 0.70 | 0.94 | 1.48 | 0.59 |
| **0.5 ETp, 0-15 cm** | | | | **0.5 $ET_p$, 15-30 cm** | | |
| Jul '03 | 0.93 | 27.88 | 1.27 | - | - | - |
| Aug '04 | 0.95 | 15.46 | 1.21 | 0.99 | 0.01 | 0.33 |
| Feb '05 | 0.95 | 11.45 | 1.11 | 0.94 | 12.53 | 1.06 |
| Aug '05 | 0.92 | 9.26 | 0.95 | 0.94 | 8.15 | 0.89 |
| Feb'06 | 0.95 | 7.22 | 0.86 | 0.95 | 6.48 | 0.84 |
| Aug '06 | 0.86 | 16.47 | 1.11 | 0.92 | 9.10 | 0.78 |
| | Langmuir¶: $Y = abx/(1+bx)$ | | | | | |
| | $R^2$ | a | b | $R^2$ | a | b |
| **2.0 ETp, 0-15 cm** | | | | **2.0 $ET_p$, 15-30 cm** | | |
| Jul '03 | 0.95 | 743 | 0.04 | 0.99 | 402 | 0.11 |
| Aug '04 | | did not converge | | | did not converge | |
| Feb '05 | 0.96 | 1784 | 0.01 | 0.92 | 6882 | 0.00 |
| Aug '05 | | did not converge | | | did not converge | |
| Feb'06 | | did not converge | | | | |
| Aug '06 | | did not converge | | 0.97 | 1111 | 0.00 |
| **0.5 ETp, 0-15 cm** | | | | **0.5 $ET_p$, 15-30 cm** | | |
| Jul '03 | 0.95 | 629 | 0.04 | | | |
| Aug '04 | 0.96 | 585 | 0.02 | | did not converge | |
| Feb '05 | 0.96 | 796 | 0.01 | 0.95 | 1111 | 0.01 |
| Aug '05 | 0.92 | 4272 | 0.00 | | did not converge | |
| Feb'06 | | did not converge | | | did not converge | |
| Aug '06 | 0.88 | 882 | 0.02 | | did not converge | |

†Average P sorption data of two subsamples per plot and two plots per irrigation rate per sampling date.

††a – a constant related to sorption maxima; b – another constant independent of a

¶a – constant related to sorption maxima; b – energy related to the binding strength

## Appendix Figures

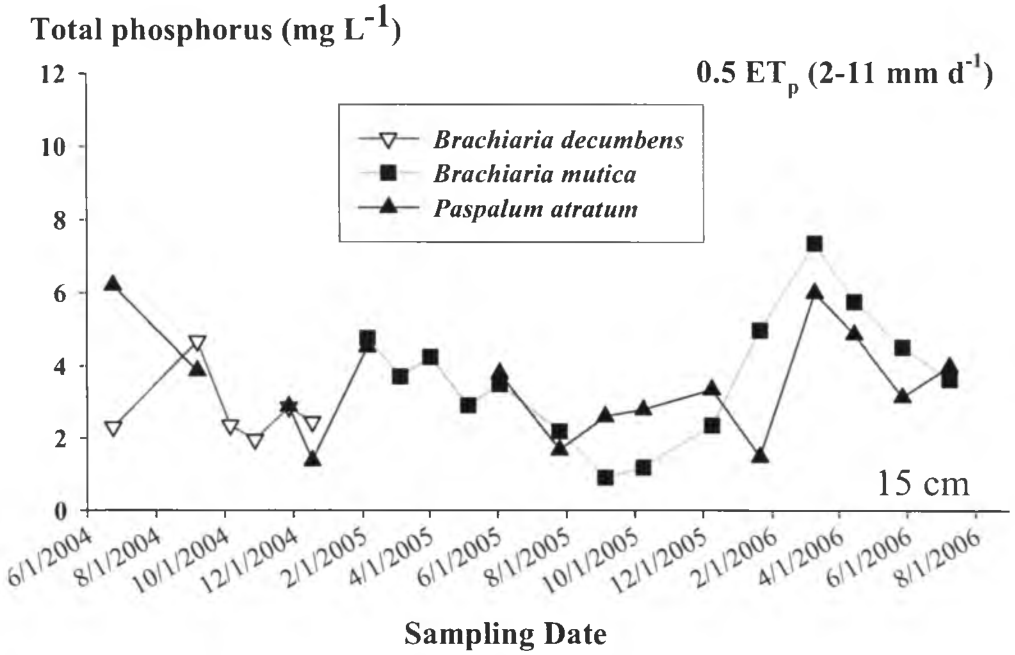



**Figure A.1. Soil solution total phosphorus collected at 15 cm depth from plots irrigated at 0.5 $ET_p$ rate, Waianae, O'ahu, Hawai'i.**

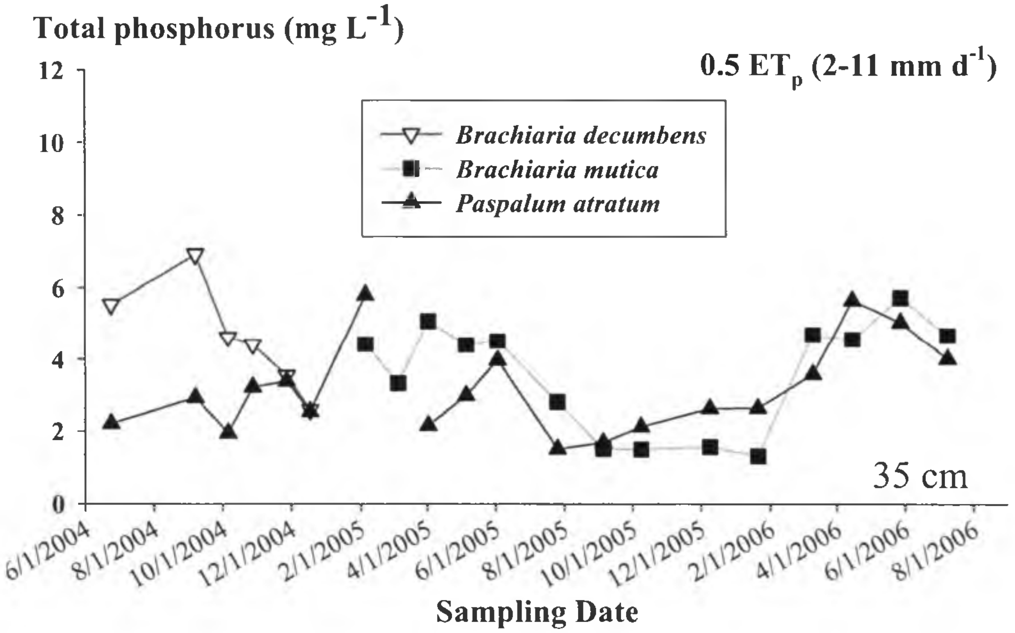



**Figure A.2. Soil solution total phosphorus collected at 35 cm depth from plots irrigated at 0.5 $ET_p$ rate, Waianae, O'ahu, Hawai'i.**

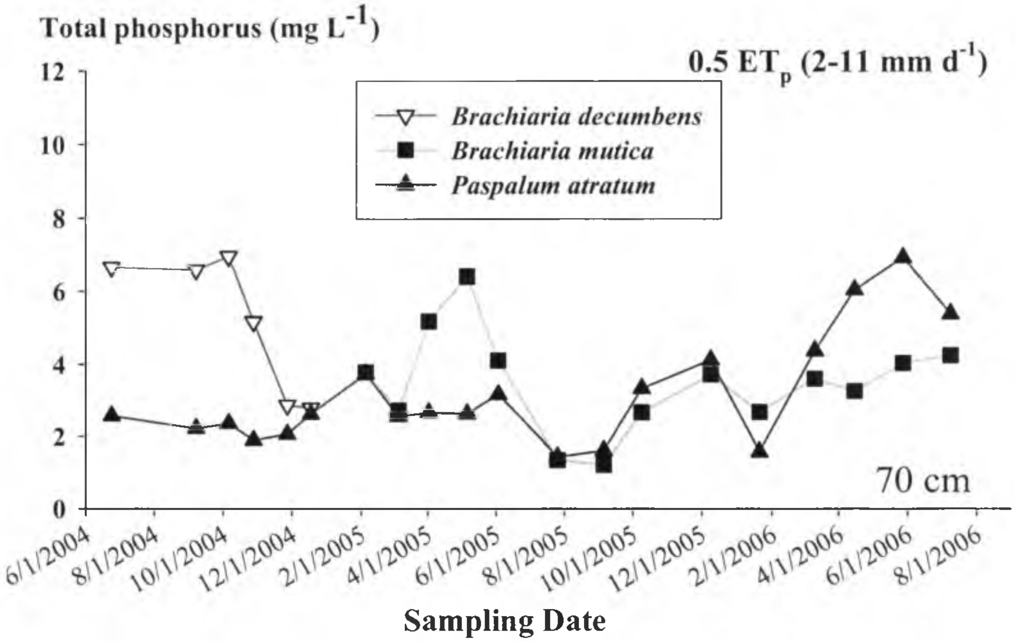



**Figure A.3. Soil solution total phosphorus collected at 70 cm depth from plots irrigated at 0.5 ET$_p$ rate, Waianae, O'ahu, Hawai'i.**

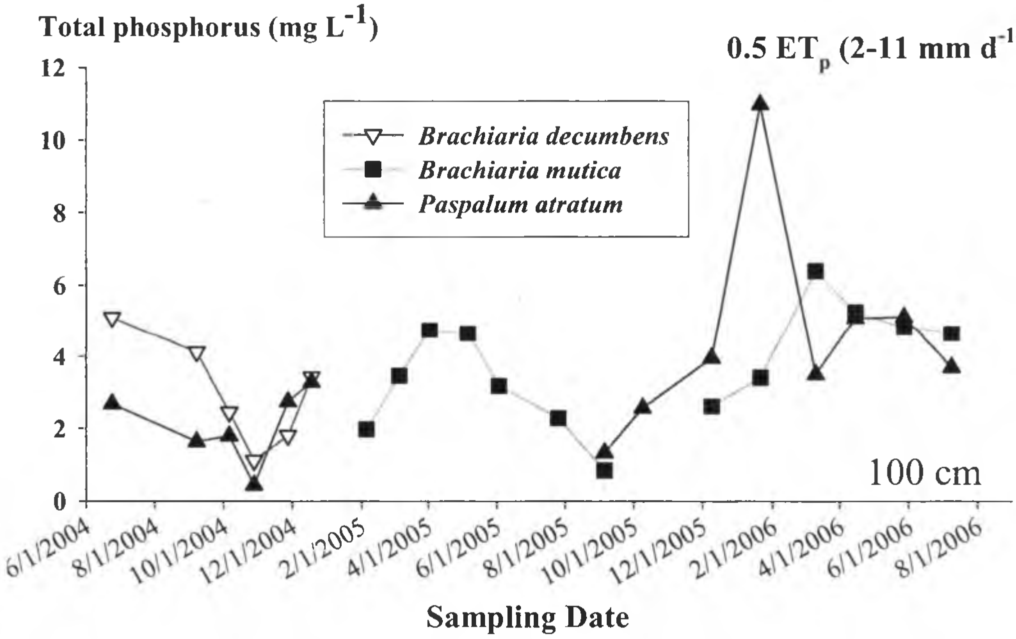



**Figure A.4. Soil solution total phosphorus collected at 100 cm depth from plots irrigated at 0.5 ET$_p$ rate, Waianae, O'ahu, Hawai'i.**

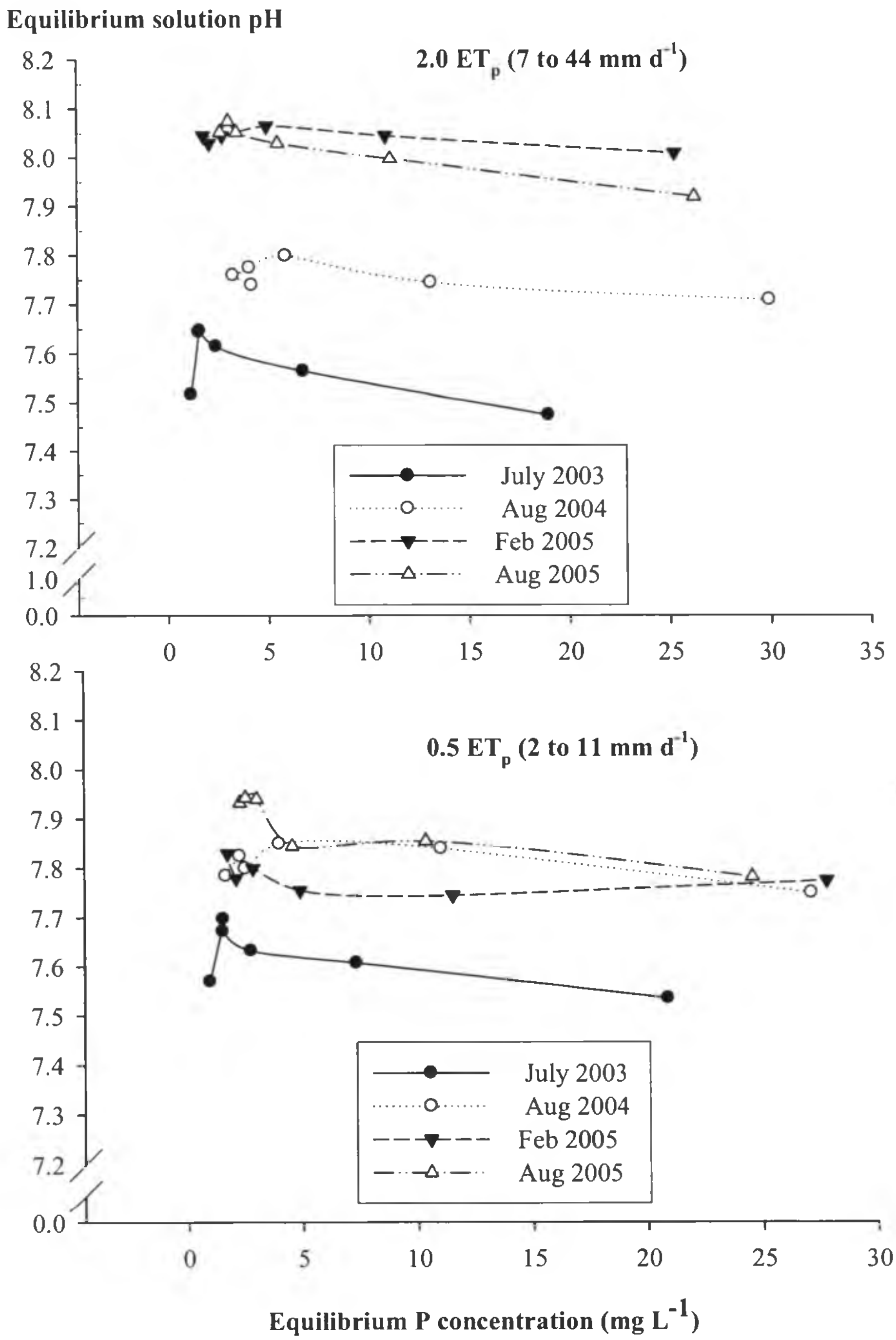



**Figure A.5. Equilibrium solution pH of soil samples from the plots irrigated at 2.0 $ET_p$ and 0.5 $ET_p$, Waianae, Oahu, Hawaii.**

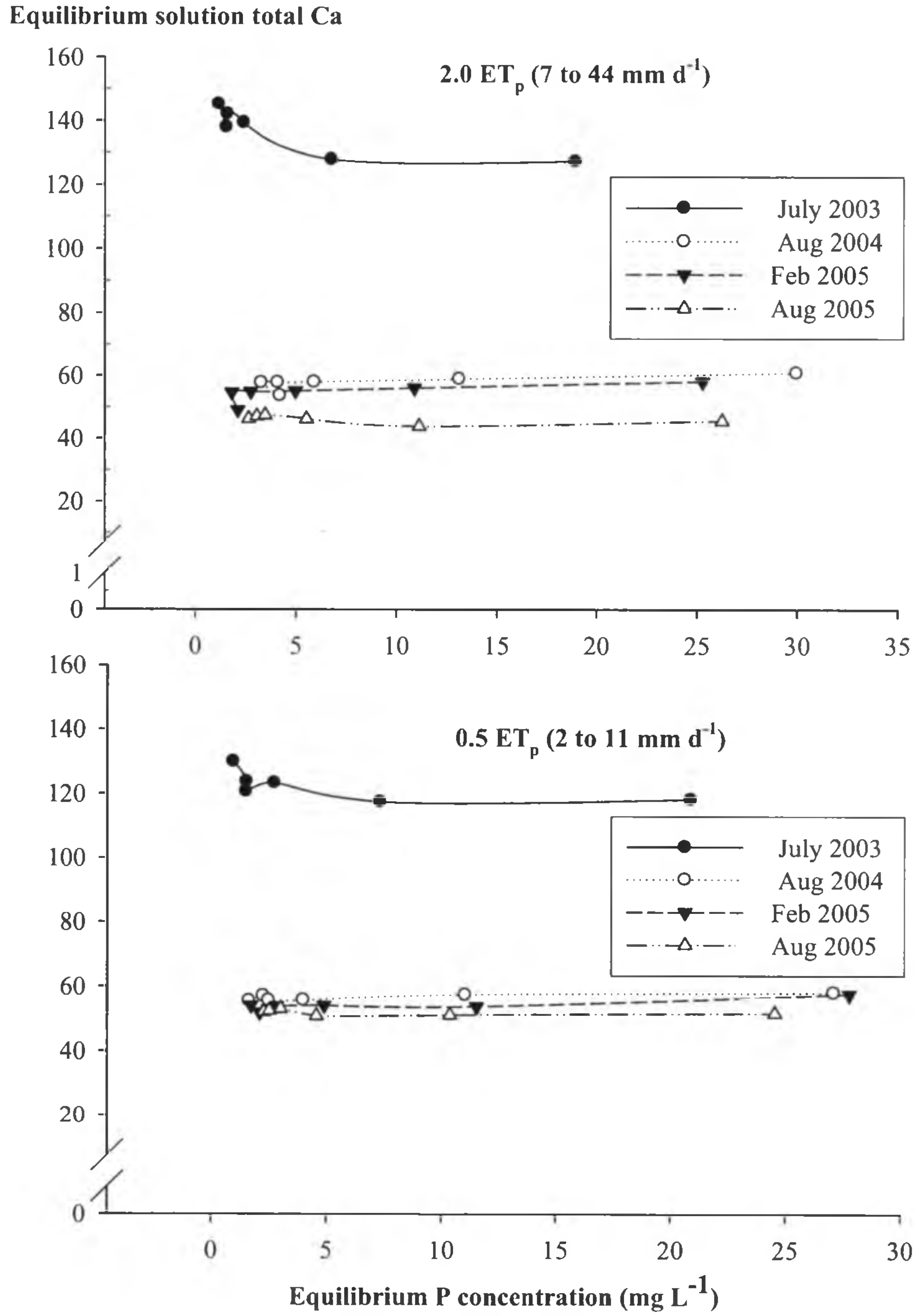
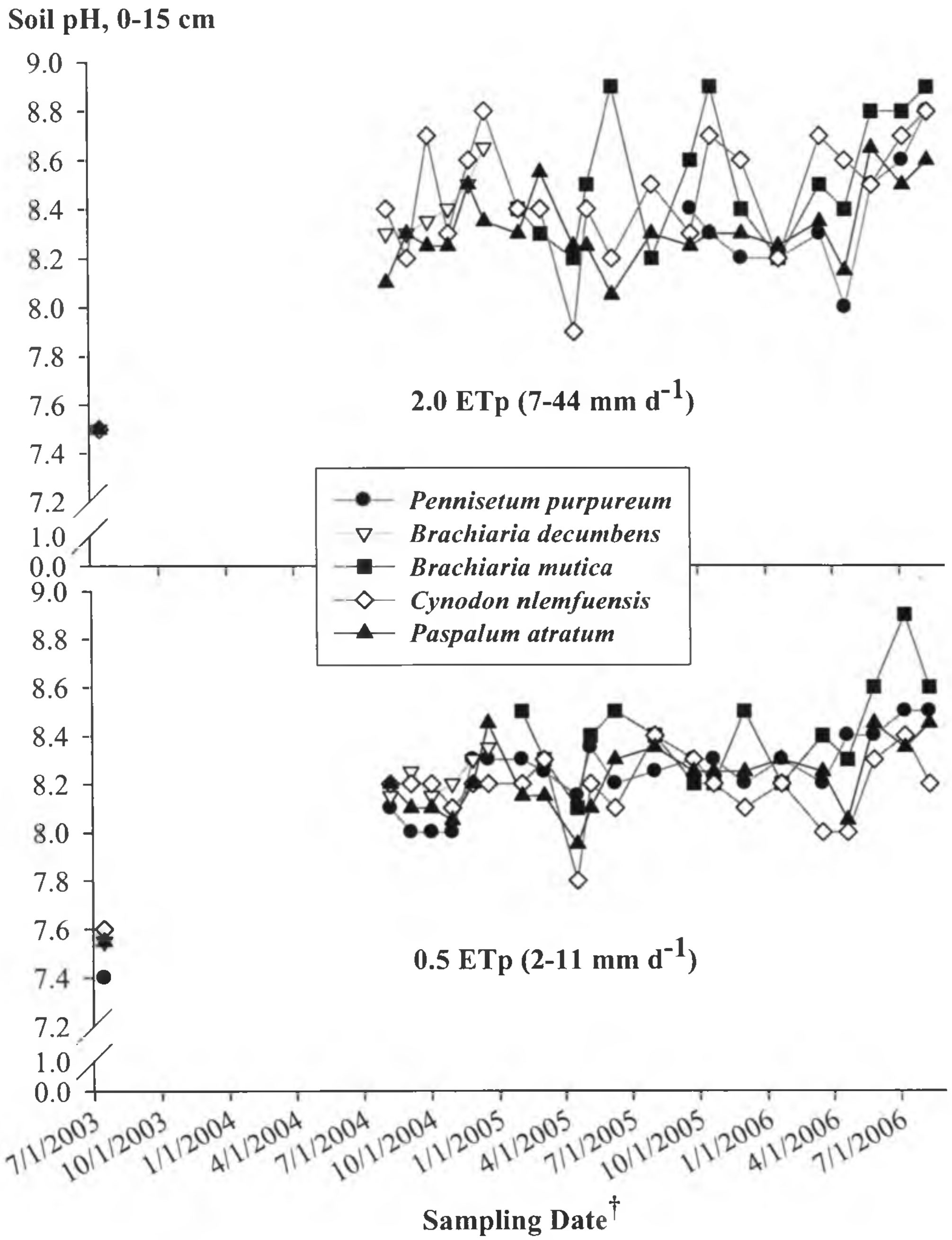
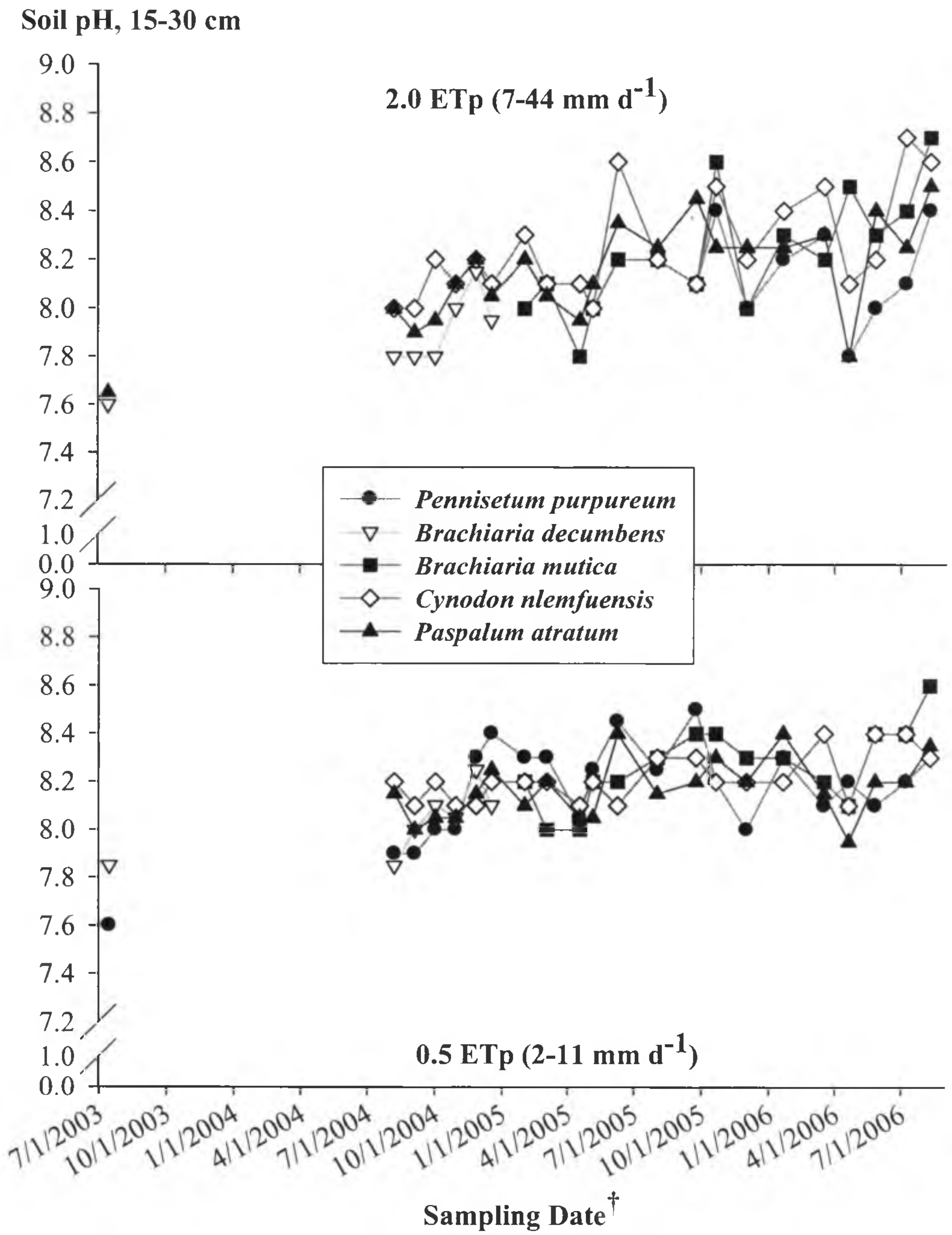



**Figure A.6. Equilibrium solution total Ca of soil samples from the plots irrigated at 2.0 ET$_p$ and 0.5 ET$_p$, Waianae, Oahu, Hawaii.**




†Soil samples were collected before the irrigation was installed in July 2003; grass establishment period was from December 2003 to July 2004

**Figure A.7. Soil pH at the 0-15 cm depth of effluent-irrigated soil (Mollisol) planted to tropical grasses, Waianae, O'ahu, Hawai'i.**




†Soil samples were collected before the irrigation was installed in July 2003; grass establishment period was from December 2003 to July 2004

**Figure A.8. Soil pH by grass species and irrigation rate at the 15-30 cm depth of effluent-irrigated soil (Mollisol) planted to tropical grasses, Waianae, O'ahu, Hawai'i.**

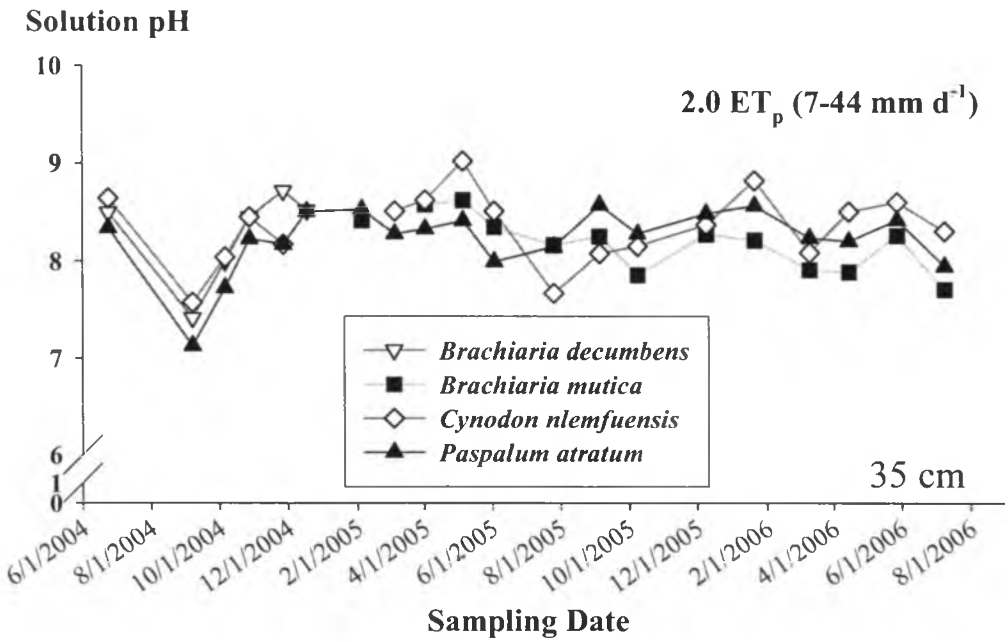



**Figure A.9. Soil solution pH at the 15 cm depth of effluent-irrigated soil (Mollisol) at the 2.0 $ET_p$ rate, Waianae, O'ahu, Hawai'i.**

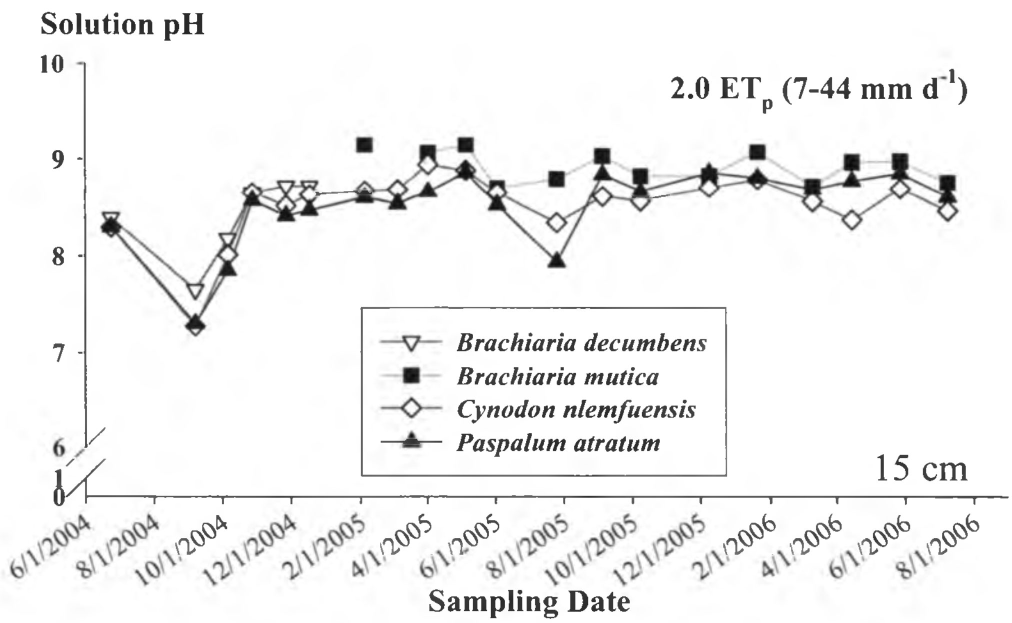



**Figure A.10. Soil solution pH at the 35 cm depth of effluent-irrigated soil (Mollisol) at the 2.0 $ET_p$ rate, Waianae, O'ahu, Hawai'i.**

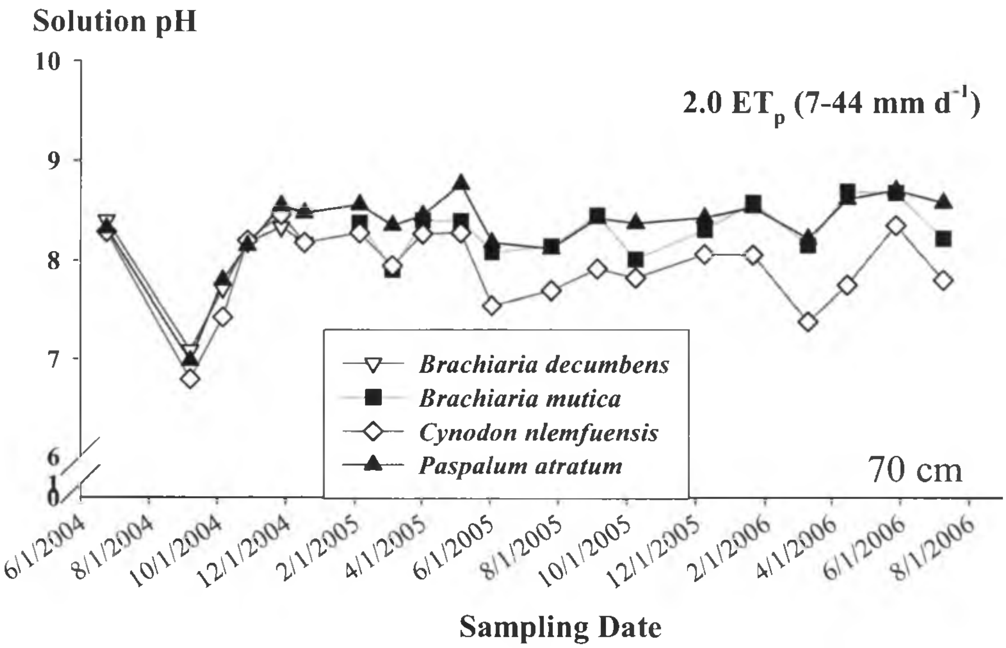



**Figure A.11.** Soil solution pH at the 70 cm depth of effluent-irrigated soil (Mollisol) at the 2.0 ET$_p$ rate, Waianae, O'ahu, Hawai'i.

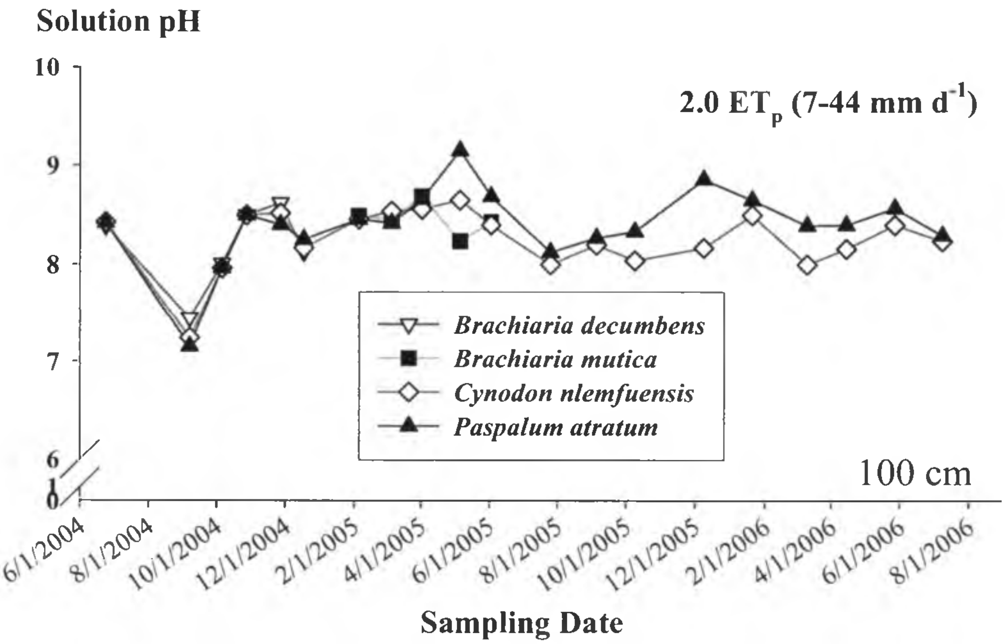



**Figure A.12.** Soil solution pH at the 100 cm depth of effluent-irrigated soil (Mollisol) at the 2.0 ET$_p$ rate, Waianae, O'ahu, Hawai'i.

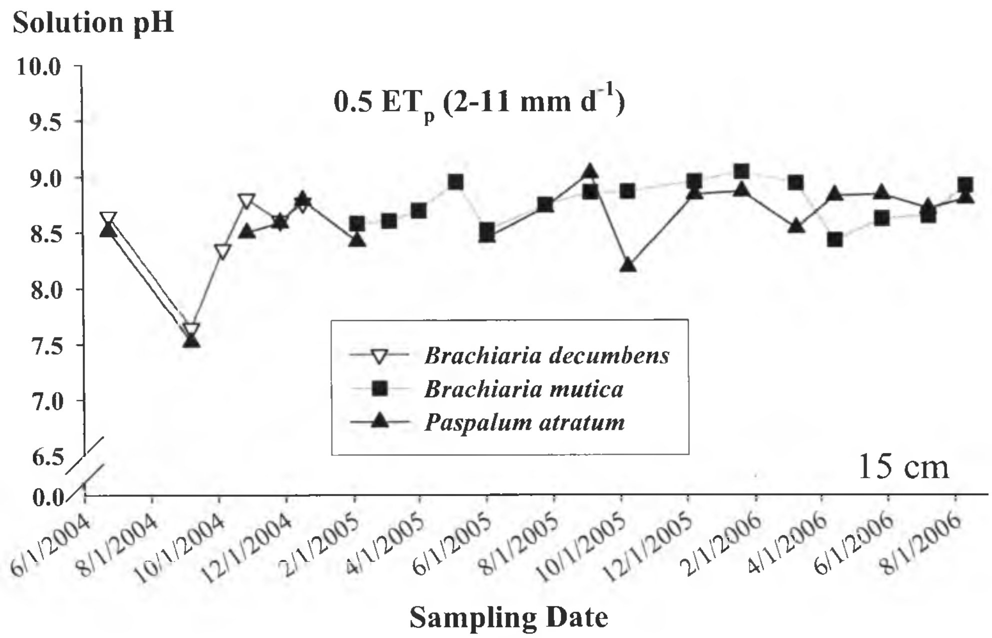



**Figure A.13. Soil solution pH at the 15 cm depth of effluent-irrigated soil (Mollisol) at the 0.5 ET$_p$ rate, Waianae, O'ahu, Hawai'i.**

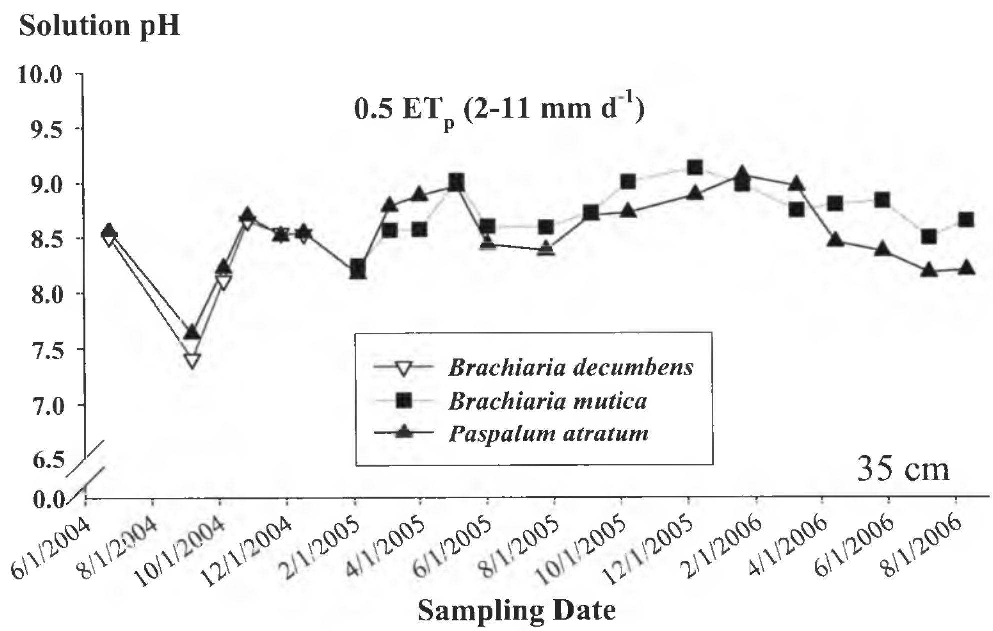



**Figure A.14. Soil solution pH at the 35 cm depth of effluent-irrigated soil (Mollisol) at the 0.5 ET$_p$ rate, Waianae, O'ahu, Hawai'i.**

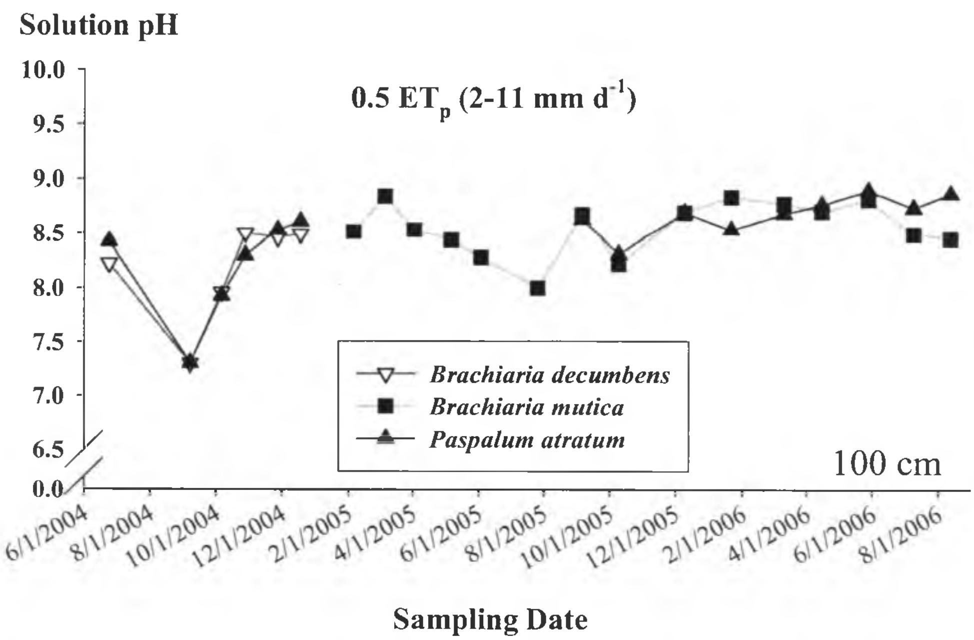



**Figure A.15.** **Soil solution pH at the 70 cm depth of effluent-irrigated soil (Mollisol) at the 0.5 $ET_p$ rate, Waianae, O'ahu, Hawai'i.**

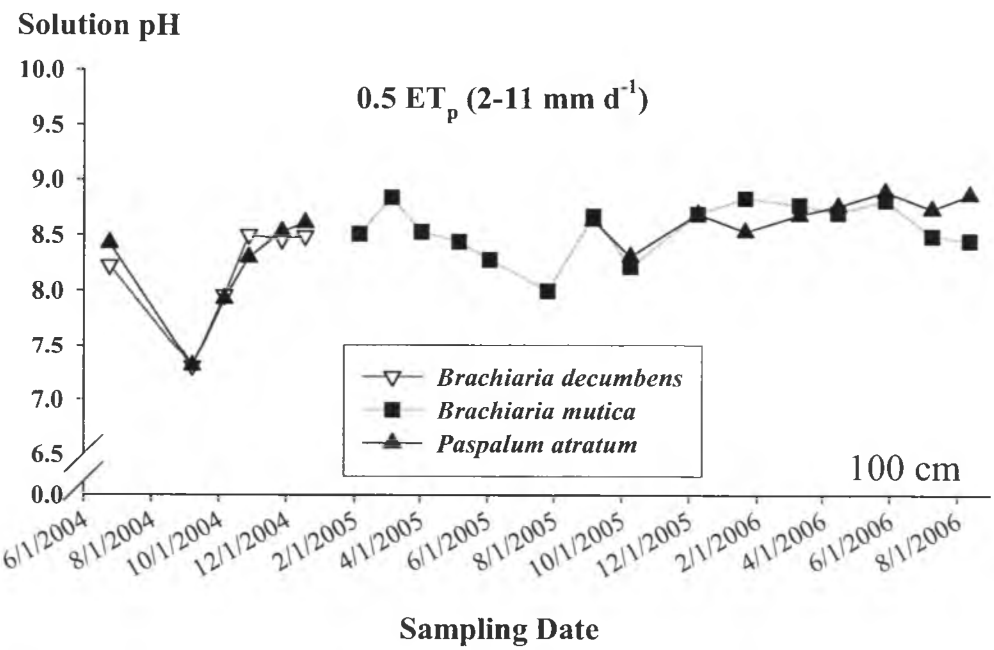
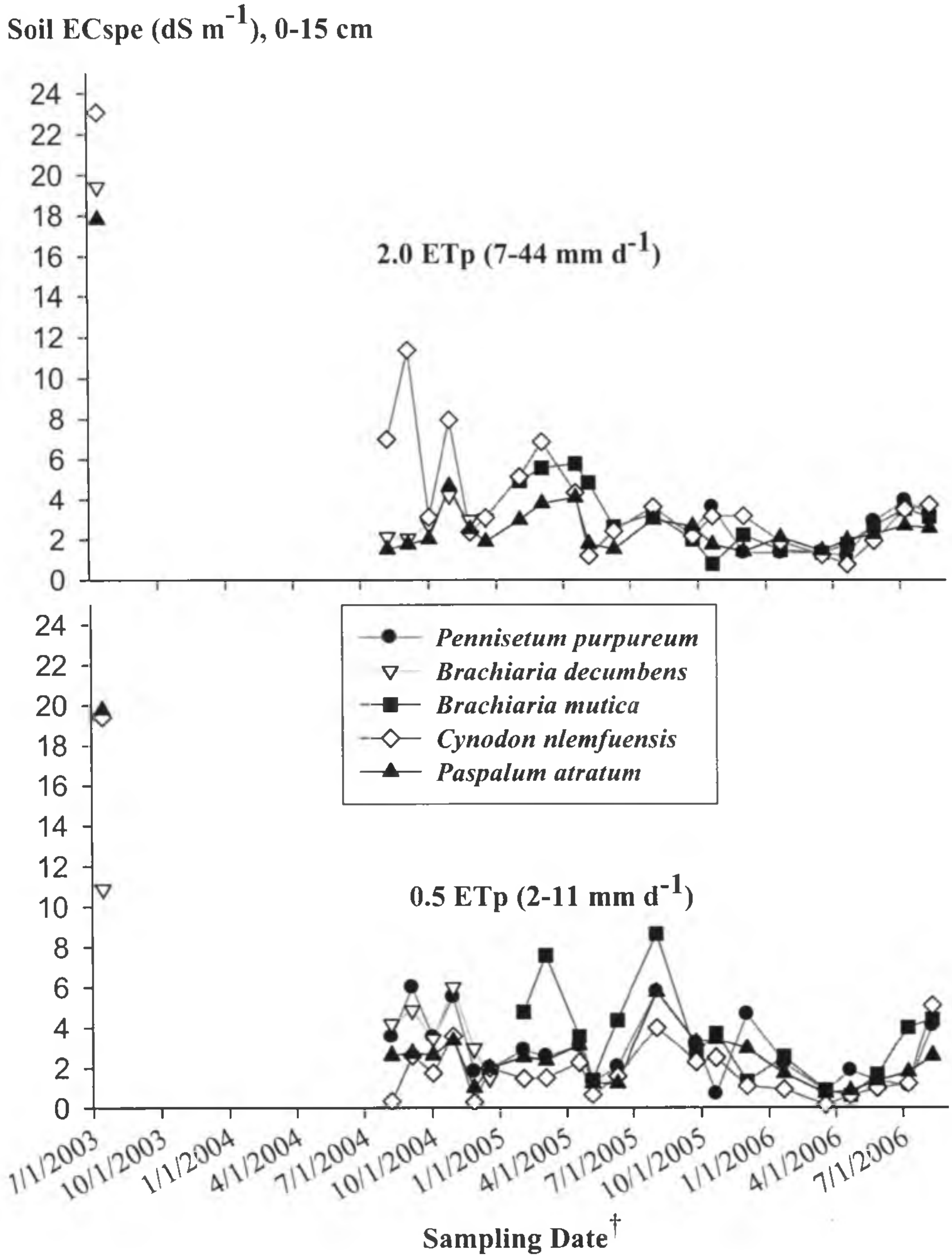
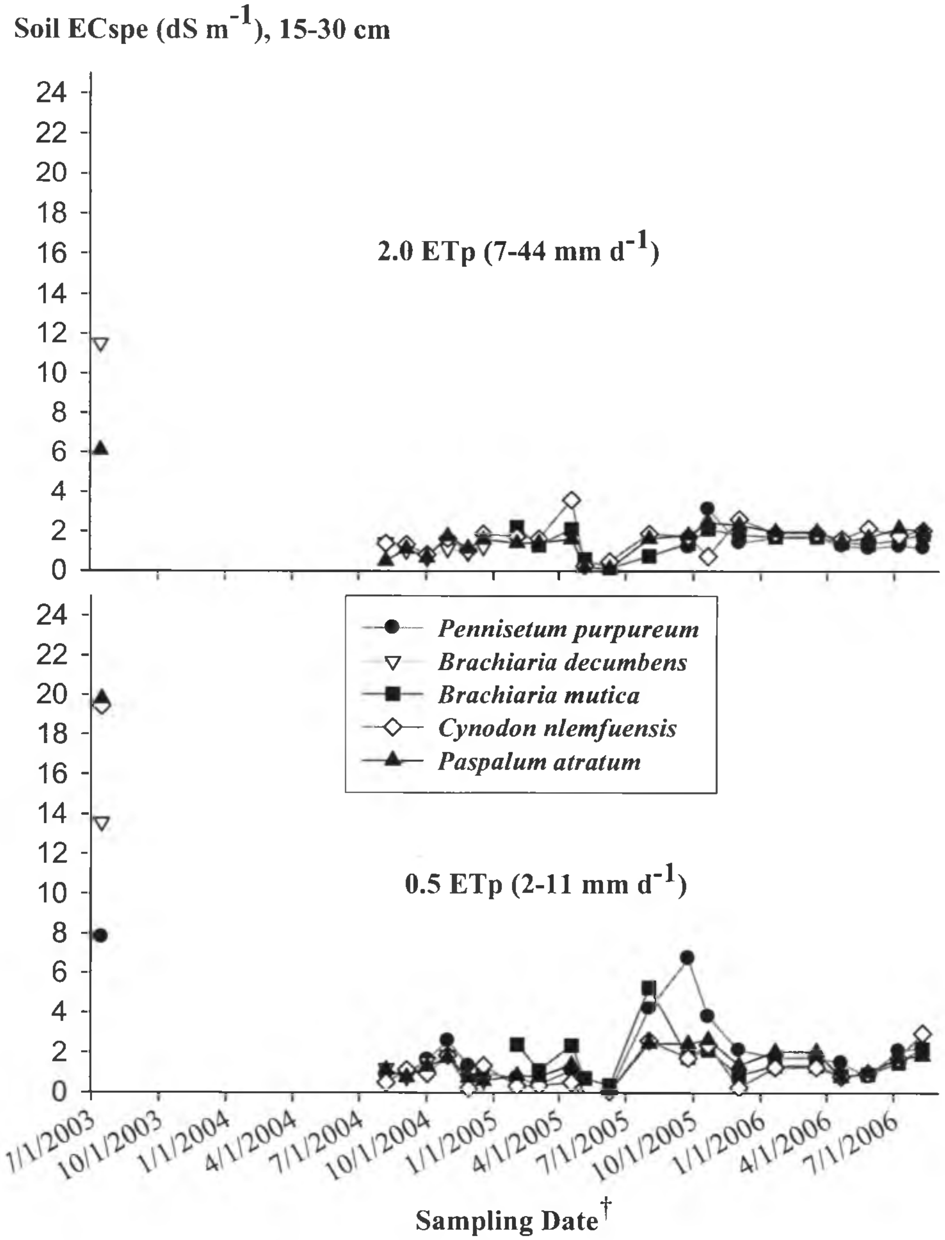



**Figure A.16.** **Soil solution pH at the 100 cm depth of effluent-irrigated soil (Mollisol) at the 0.5 $ET_p$ rate, Waianae, O'ahu, Hawai'i.**




†Soil samples were collected before the irrigation was installed in July 2003; grass establishment period was from December 2003 to July 2004

**Figure A.17. Soil $EC_{spe}$ at the 0-15 cm depth of effluent-irrigated soil (Mollisol) planted to tropical grasses, Waianae, O'ahu, Hawai'i.**




†Soil samples were collected before the irrigation was installed in July 2003; grass establishment period was from December 2003 to July 2004

**Figure A.18.** **Soil $EC_{spe}$ at the 0-15 cm depth of effluent-irrigated soil (Mollisol) planted to tropical grasses, Waianae, O'ahu, Hawai'i.**

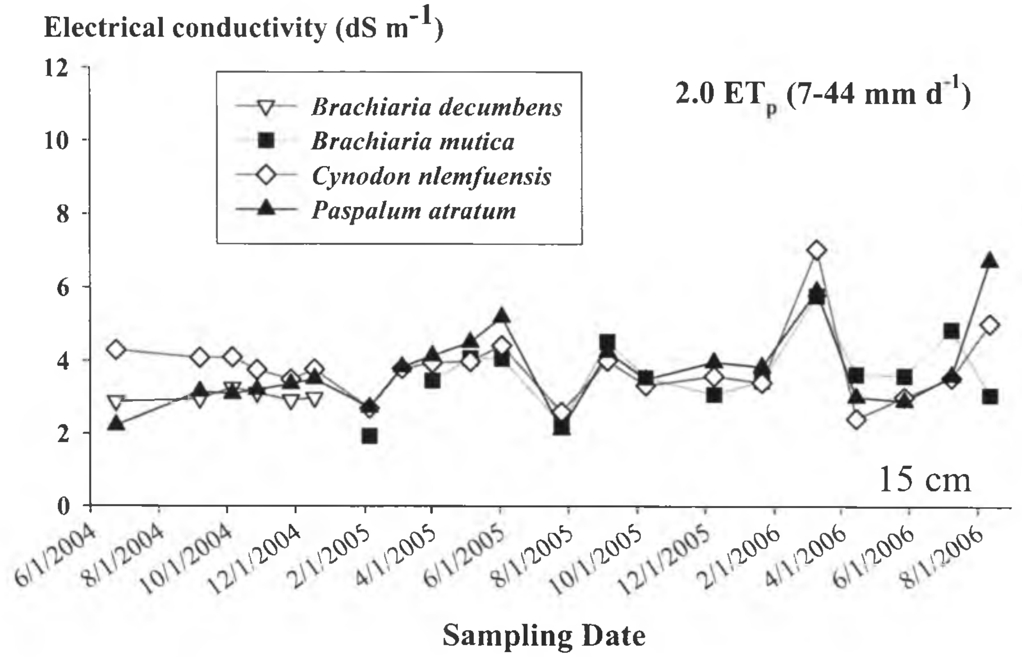



**Figure A.19. Soil solution EC at the 15 cm depth of effluent-irrigated soil (Mollisol) at the 2.0 ET$_p$ rate, Waianae, O'ahu, Hawai'i.**

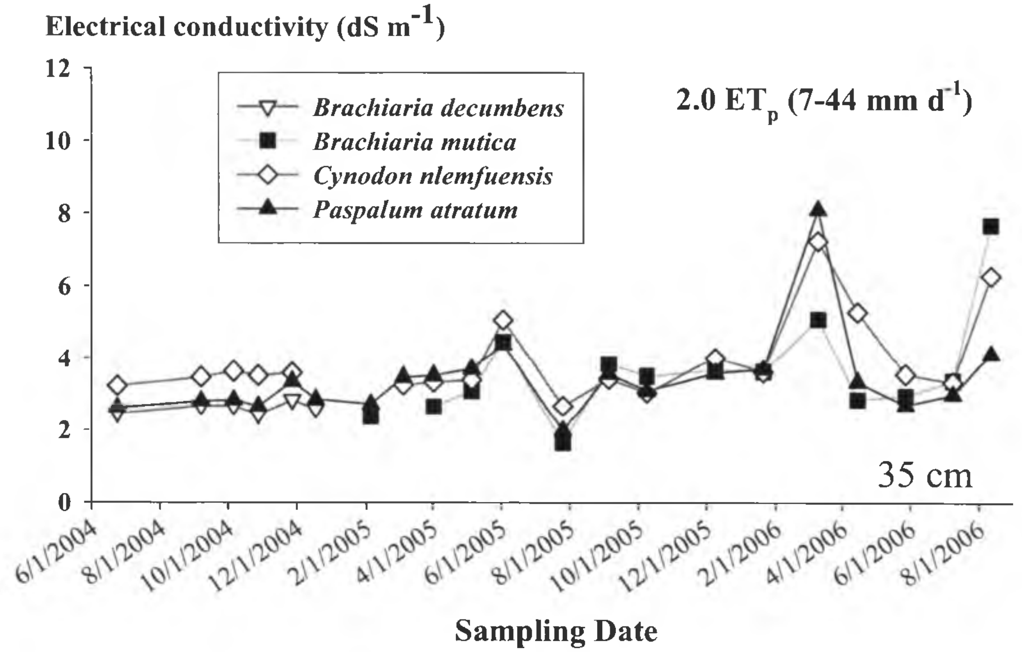



**Figure A.20. Soil solution EC at the 35 cm depth of effluent-irrigated soil (Mollisol) at the 2.0 ET$_p$ rate, Waianae, O'ahu, Hawai'i.**

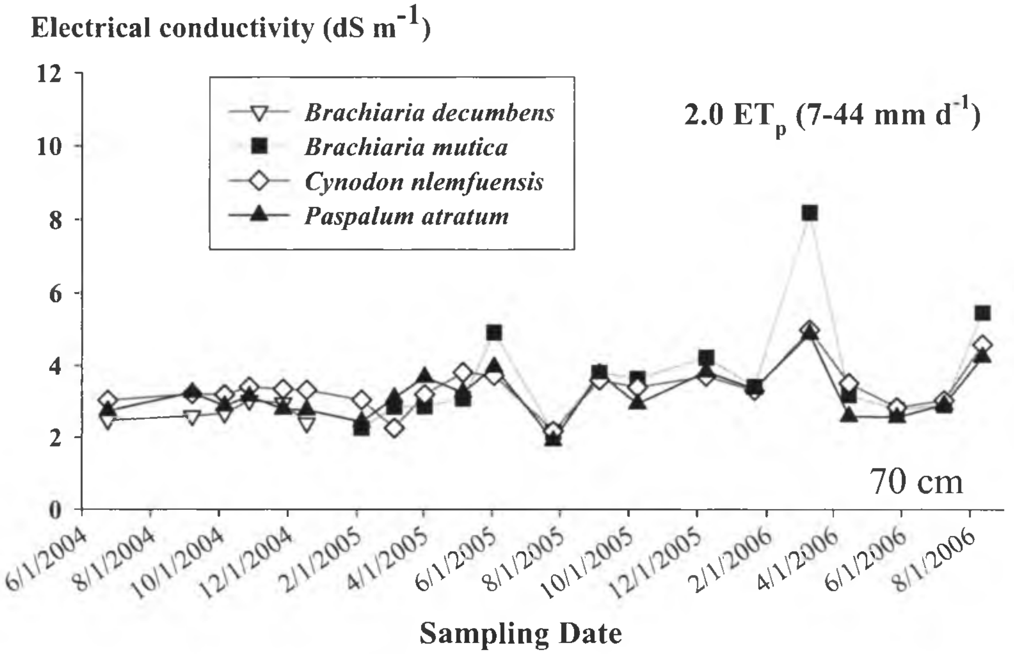



**Figure A.21.** Soil solution EC at the 70 cm depth of effluent-irrigated soil (Mollisol) at the 2.0 $ET_p$ rate, Waianae, O'ahu, Hawai'i.

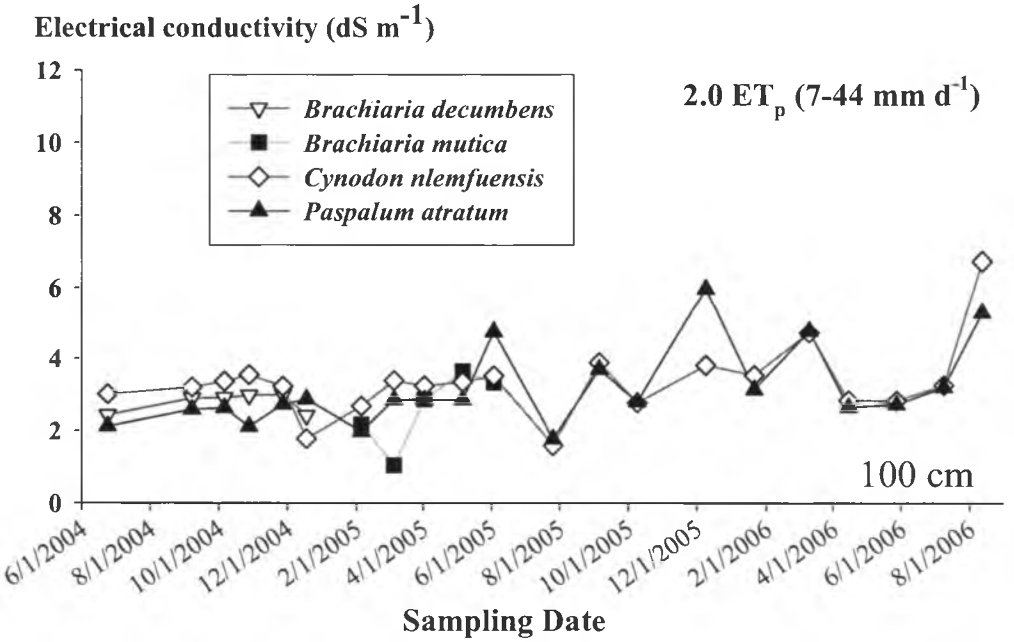



**Figure A.22.** Soil solution EC at the 70 cm depth of effluent-irrigated soil (Mollisol) at the 2.0 $ET_p$ rate, Waianae, O'ahu, Hawai'i.

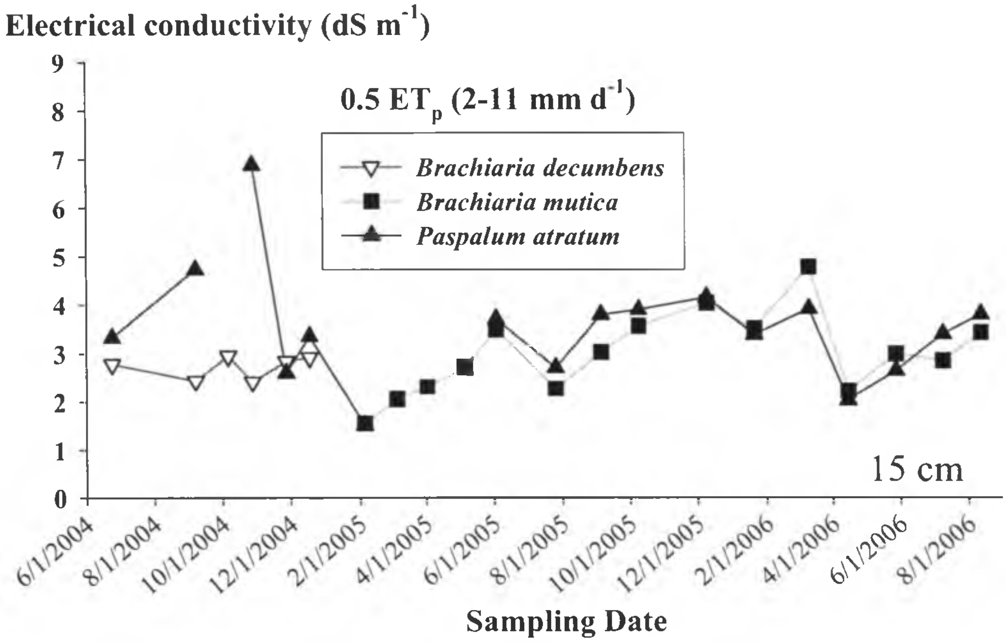



**Figure A.23. Soil solution EC at the 15 cm depth of effluent-irrigated soil (Mollisol) at the 0.5 $ET_p$ rate, Waianae, O'ahu, Hawai'i.**

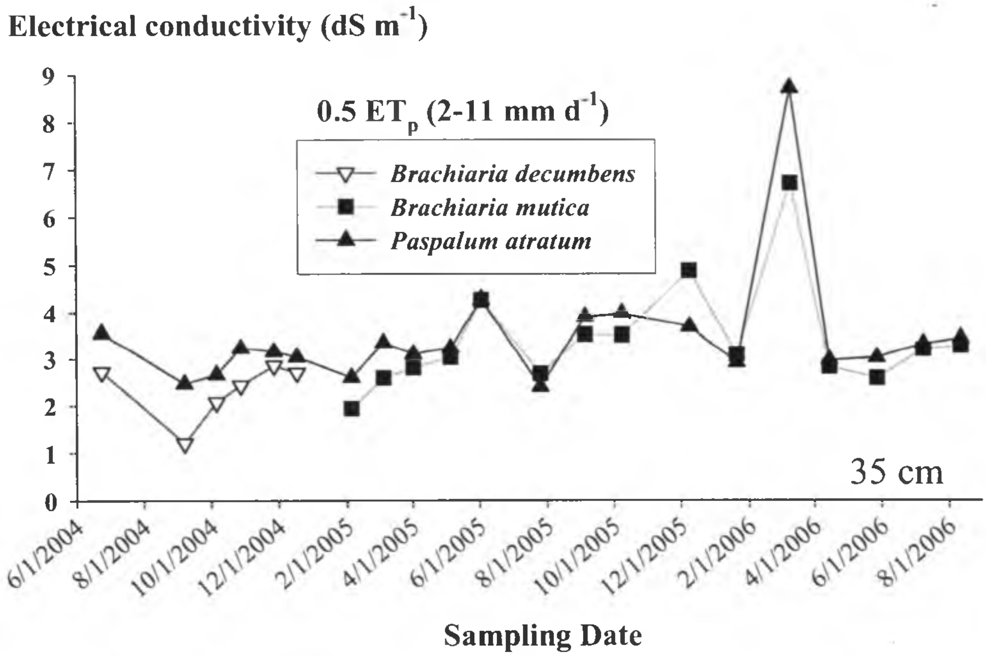



**Figure A.24. Soil solution EC at the 35 cm depth of effluent-irrigated soil (Mollisol) at the 0.5 $ET_p$ rate, Waianae, O'ahu, Hawai'i.**

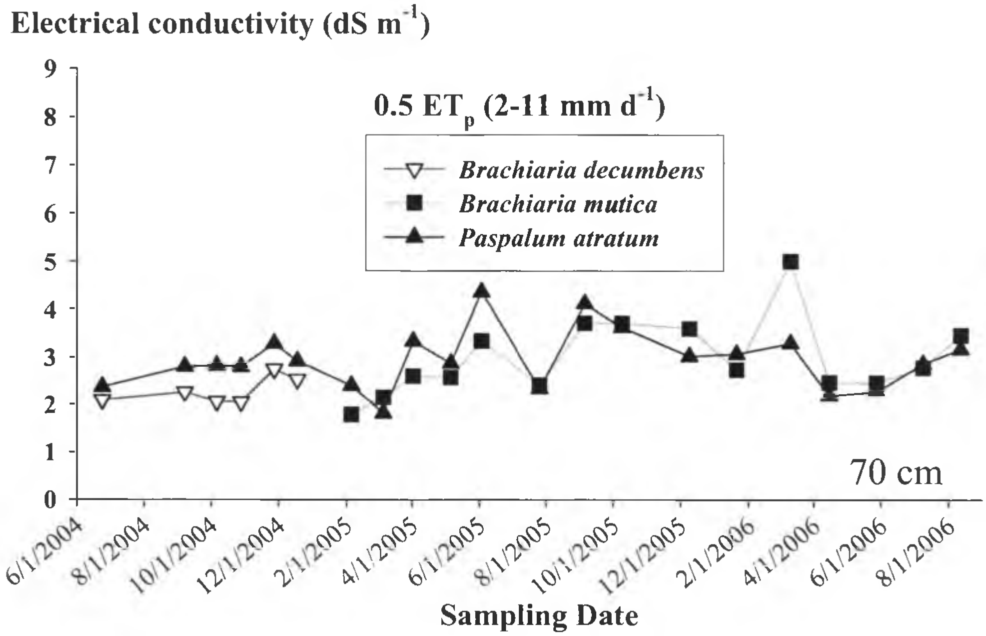



**Figure A.25. Soil solution EC at the 70 cm depth of effluent-irrigated soil (Mollisol) at the 0.5 $ET_p$ rate, Waianae, O'ahu, Hawai'i.**

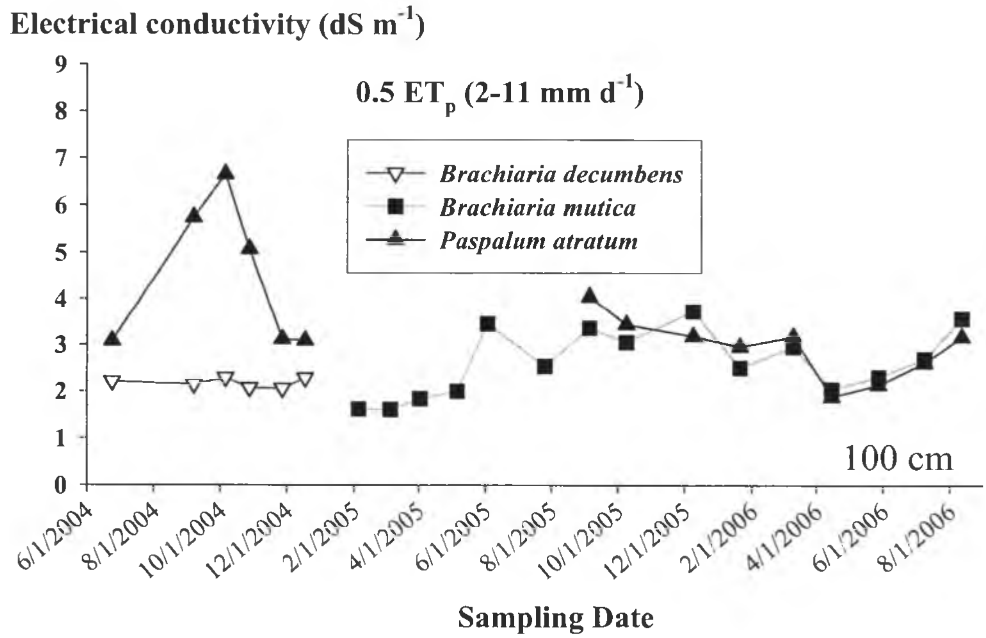
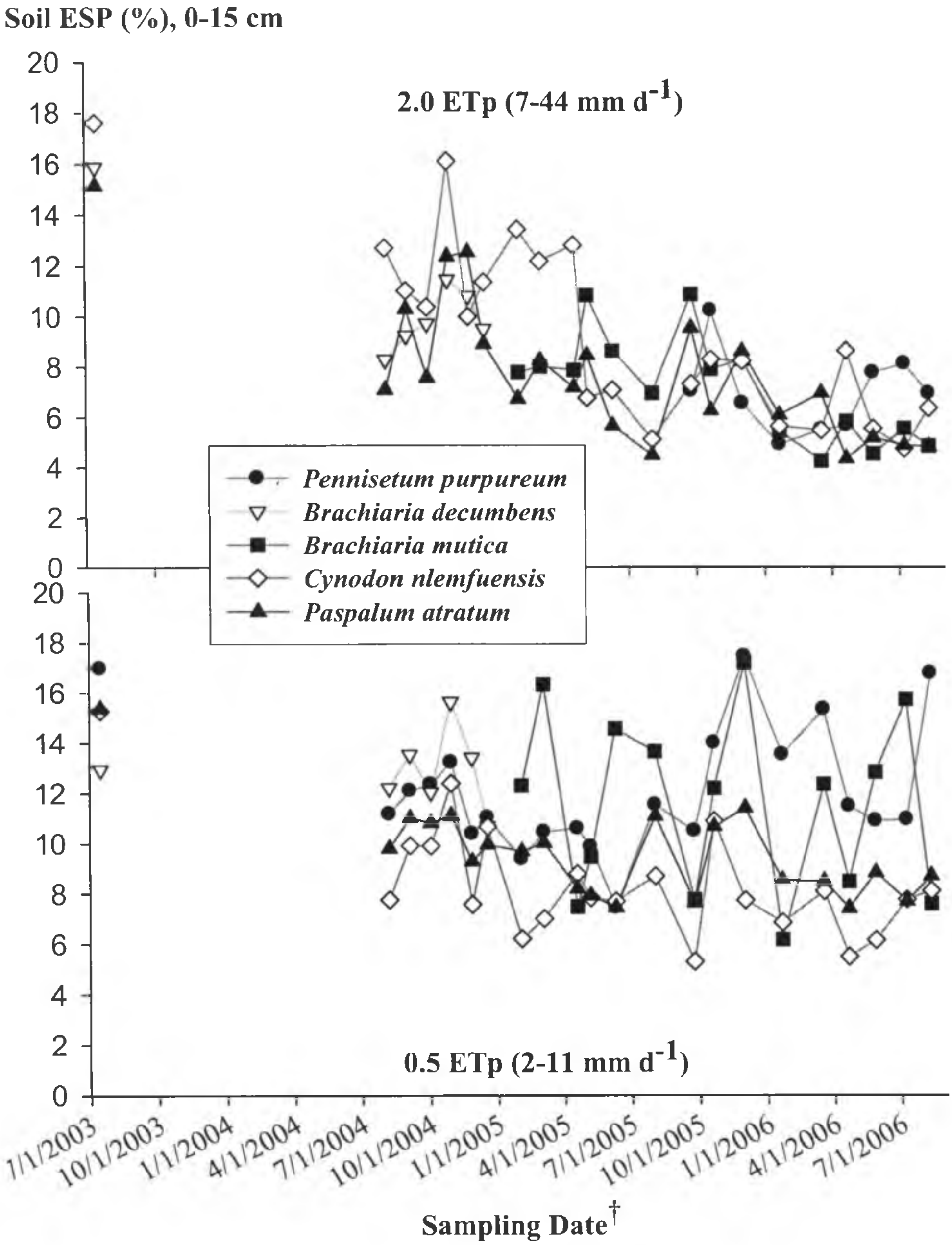
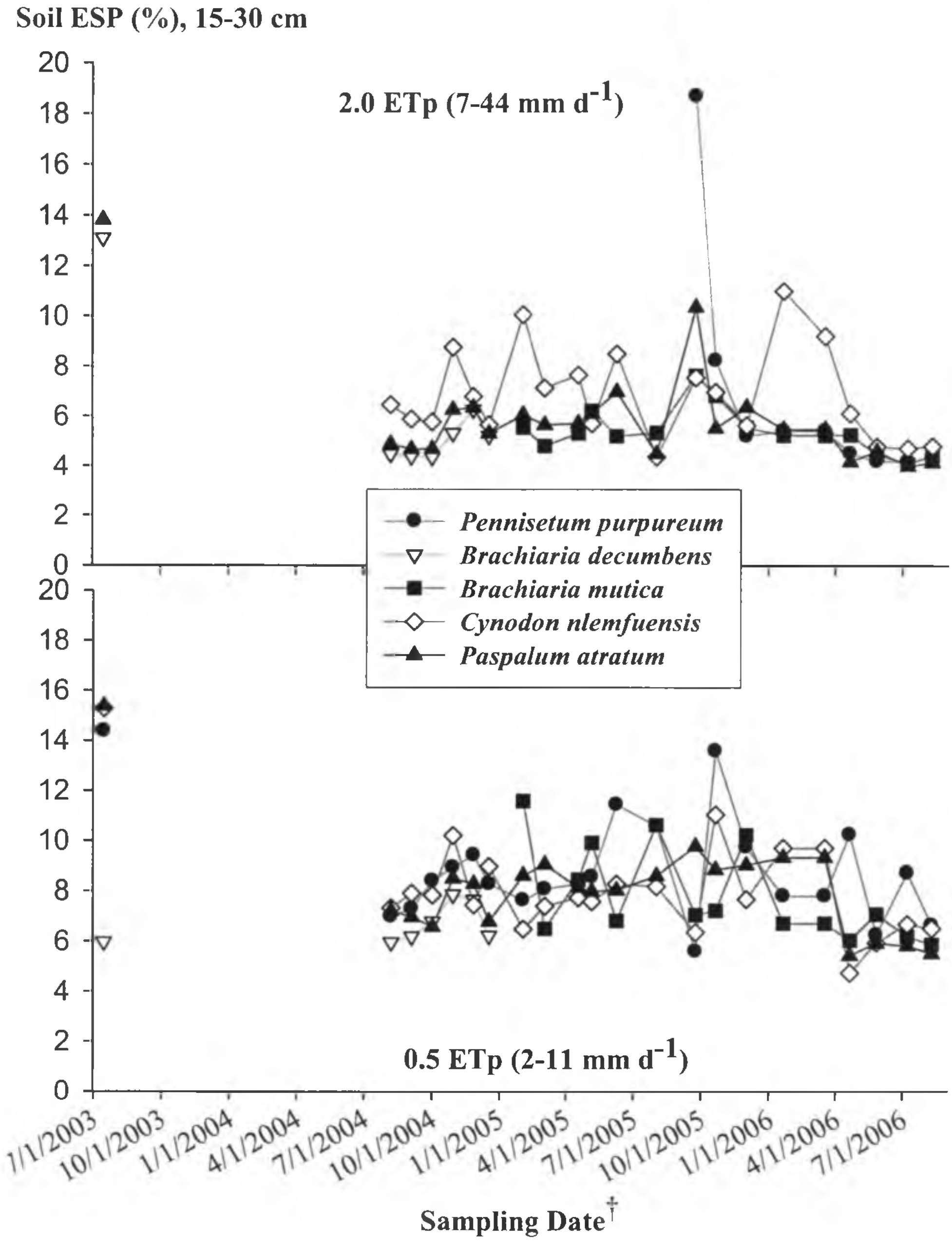



**Figure A.26. Soil solution EC at the 35 cm depth of effluent-irrigated soil at the 0.5 $ET_p$ rate, Waianae, O'ahu, Hawai'i.**



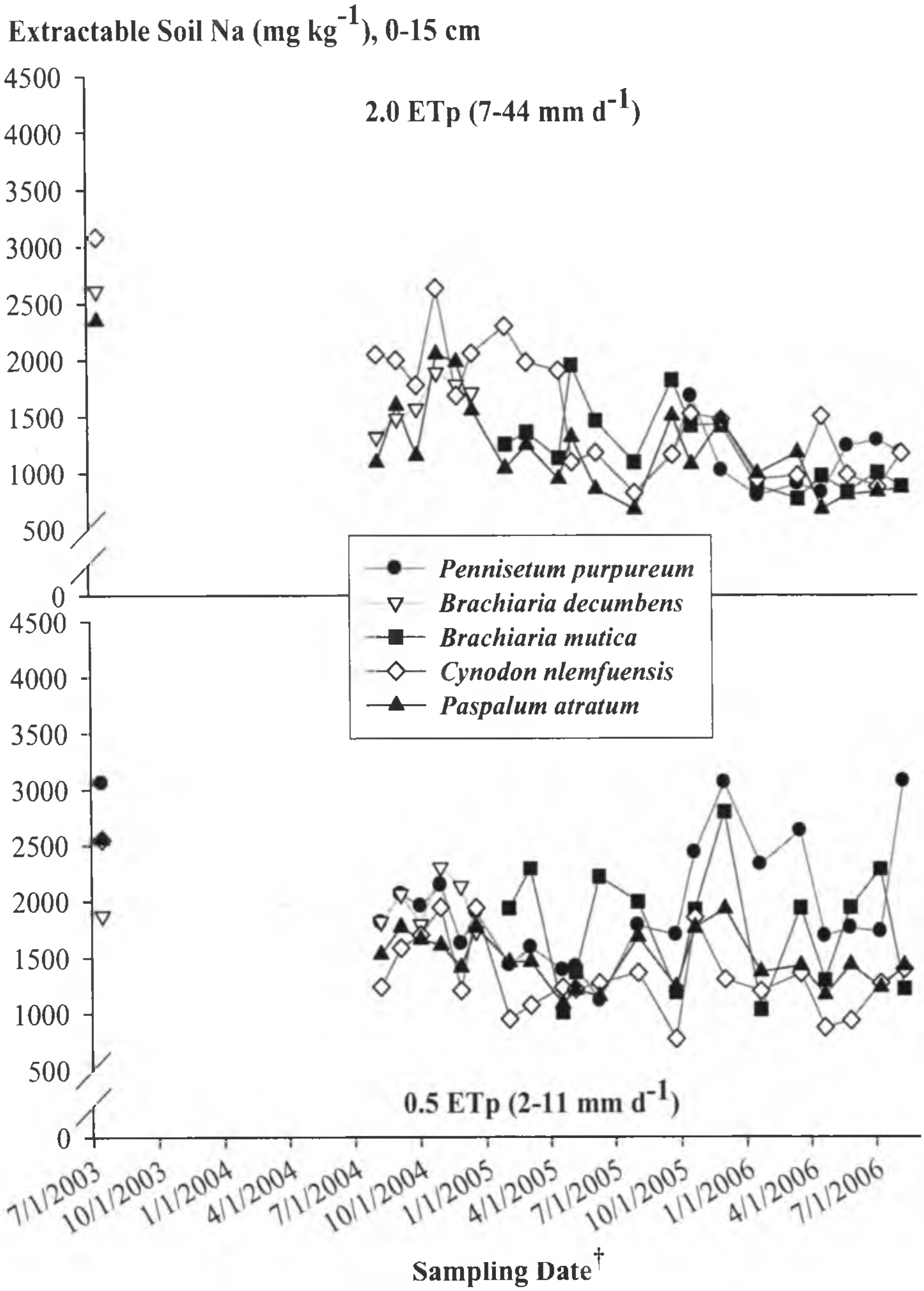

†Soil samples were collected before the irrigation was installed in July 2003; grass establishment period was from December 2003 to July 2004

**Figure A.27. Soil ESP at the 0-15 cm depth of effluent-irrigated soil (Mollisol) planted to tropical grasses, Waianae, O'ahu, Hawai'i.**



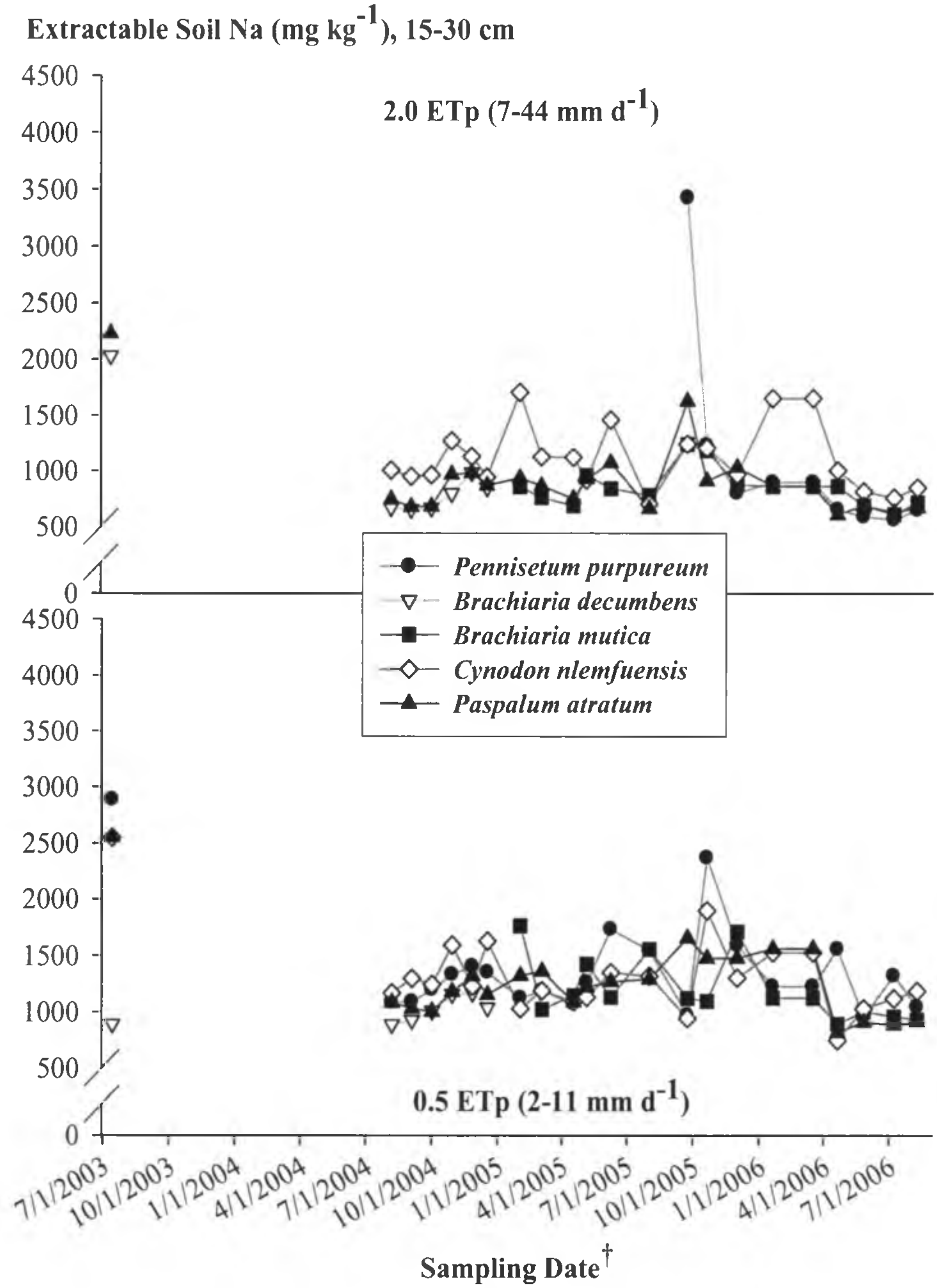

†Soil samples were collected before the irrigation was installed in July 2003; grass establishment period was from December 2003 to July 2004

**Figure A.28. Soil ESP at the 15-30 cm depth of effluent-irrigated soil (Mollisol) planted to tropical grasses, Waianae, O'ahu, Hawai'i.**




†Soil samples were collected before the irrigation was installed in July 2003; grass establishment period was from December 2003 to July 2004

**Figure A.29. Soil extractable Na at the 0-15 cm depth of effluent-irrigated soil at the 2.0 and 0.5 $ET_p$ rates, Waianae, O'ahu, Hawai'i.**




†Soil samples were collected before the irrigation was installed in July 2003; grass establishment period was from December 2003 to July 2004

**Figure A.30. Soil extractable Na at the 0-15 cm depth of effluent-irrigated soil at the 2.0 and 0.5 $ET_p$ rates, Waianae, O'ahu, Hawai'i.**

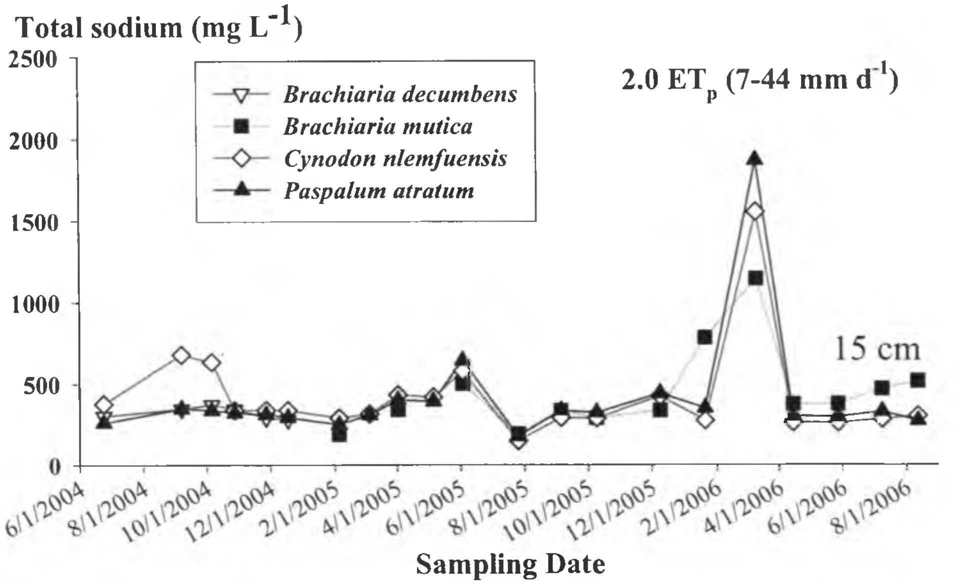



**Figure A.31. Soil solution total Na at the 15 cm depth of effluent-irrigated soil at the 2.0 $ET_p$ rate, Waianae, O'ahu, Hawai'i.**

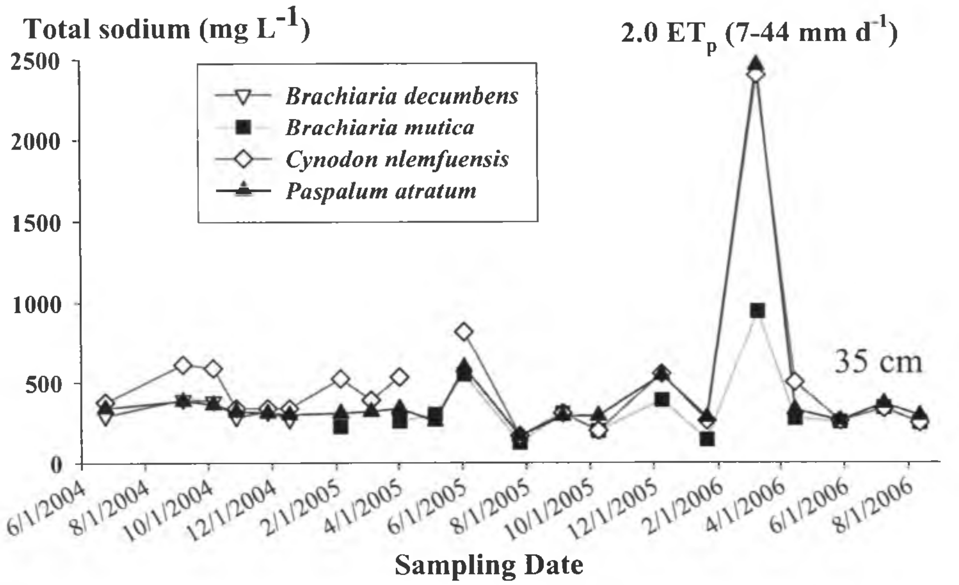



**Figure A.32. Soil solution total Na at the 35 cm depth of effluent-irrigated soil at the 2.0 $ET_p$ rate, Waianae, O'ahu, Hawai'i.**

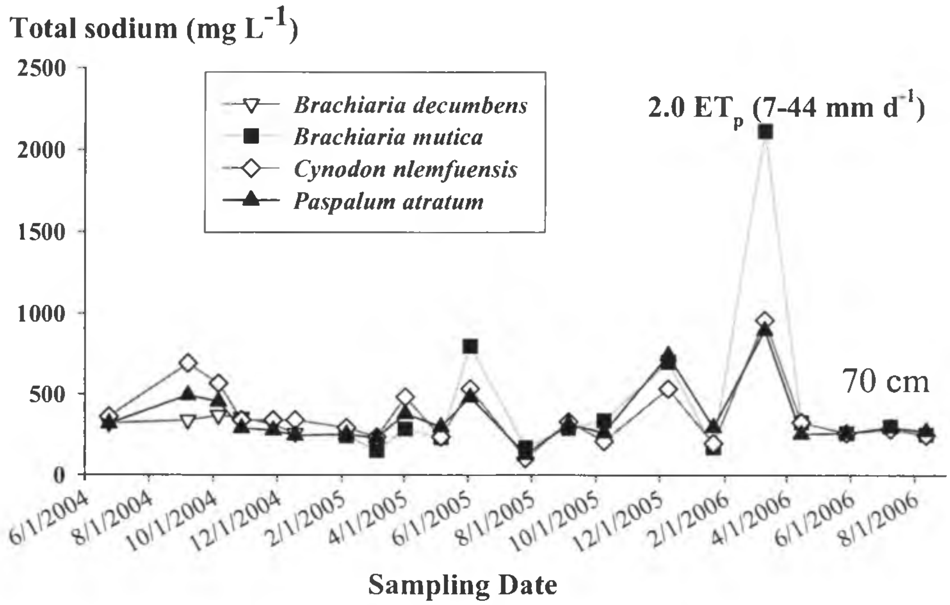



**Figure A.33. Soil solution total Na at the 70 cm depth of effluent-irrigated soil at the 2.0 ET$_p$ rate, Waianae, O'ahu, Hawai'i.**

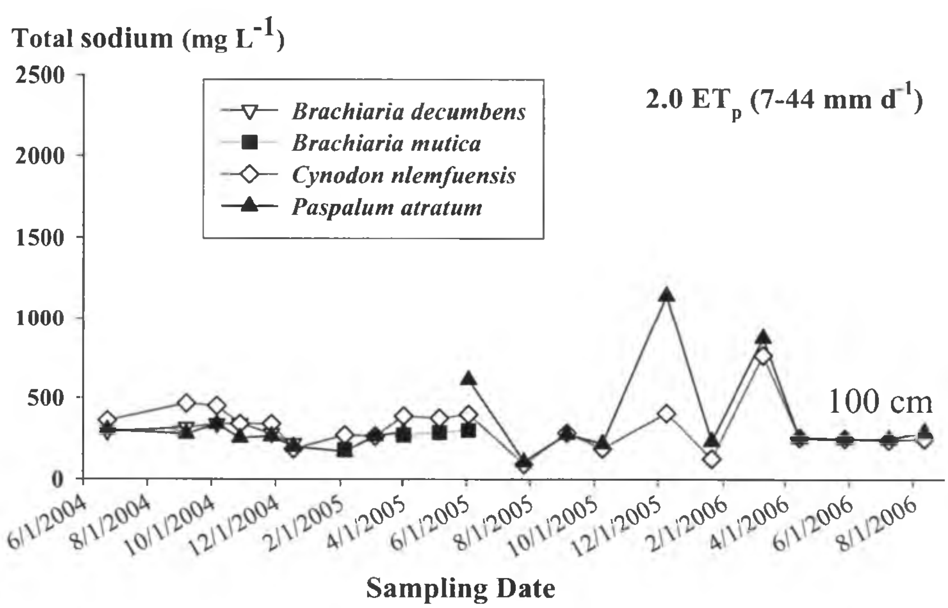



**Figure A.34. Soil solution total Na at the 100 cm depth of effluent-irrigated soil at the 2.0 ET$_p$ rate, Waianae, O'ahu, Hawai'i.**

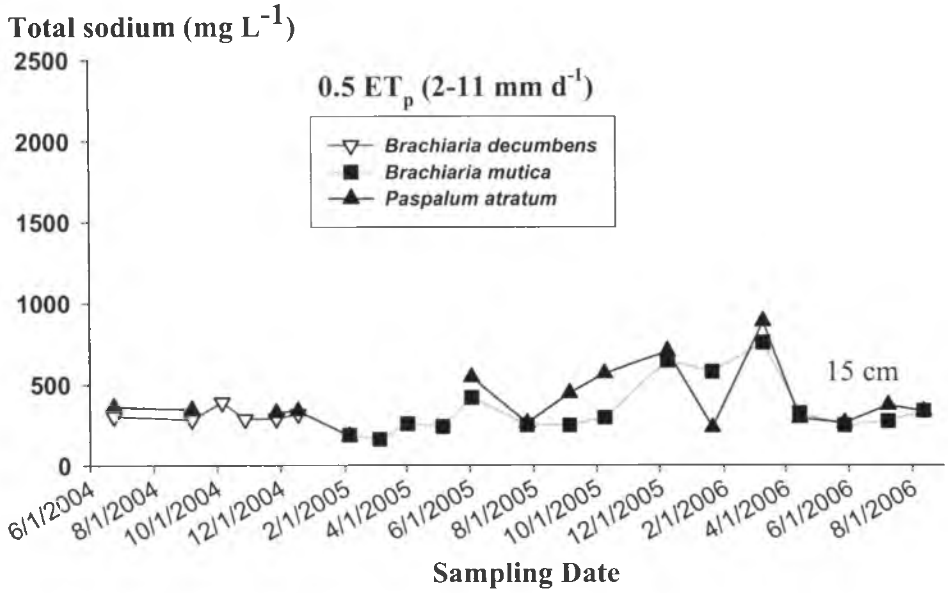



**Figure A.35. Soil solution total Na at the 15 cm depth of effluent-irrigated soil at the 0.5 ET$_p$ rate, Waianae, O'ahu, Hawai'i.**

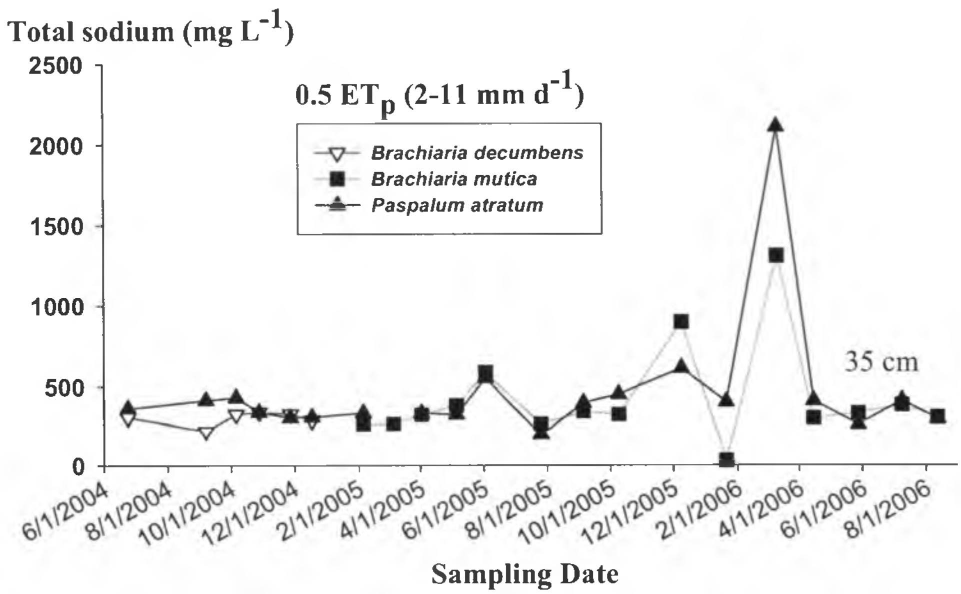



**Figure A.36. Soil solution total Na at the 35 cm depth of effluent-irrigated soil at the 0.5 ET$_p$ rate, Waianae, O'ahu, Hawai'i.**

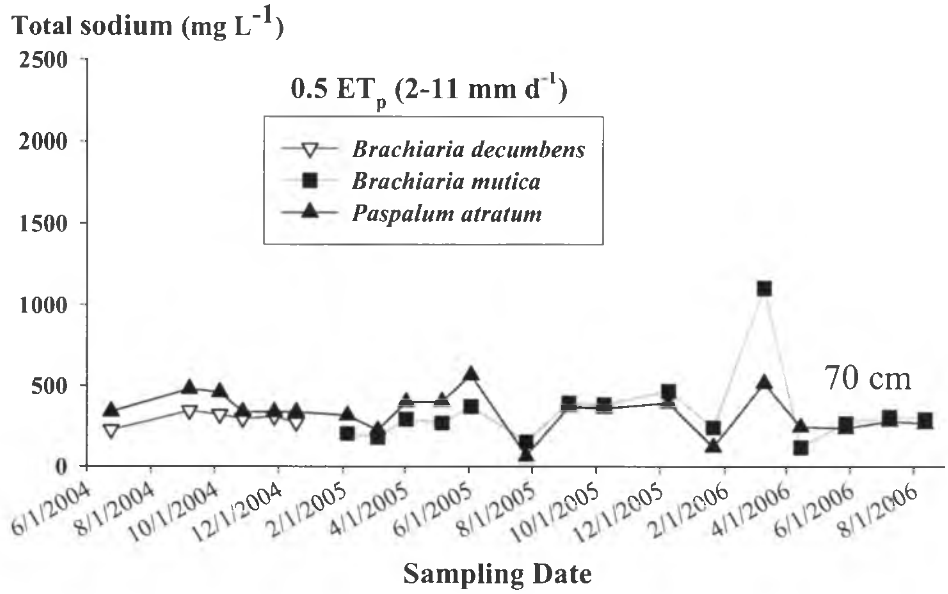



**Figure A.37. Soil solution total Na at the 35 cm depth of effluent-irrigated soil at the 0.5 ET$_p$ rate, Waianae, O'ahu, Hawai'i.**

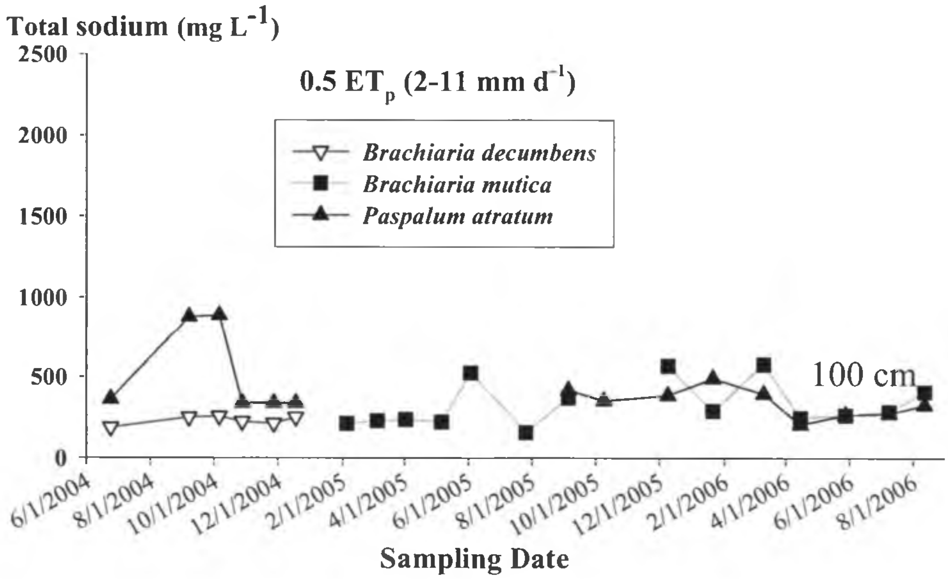
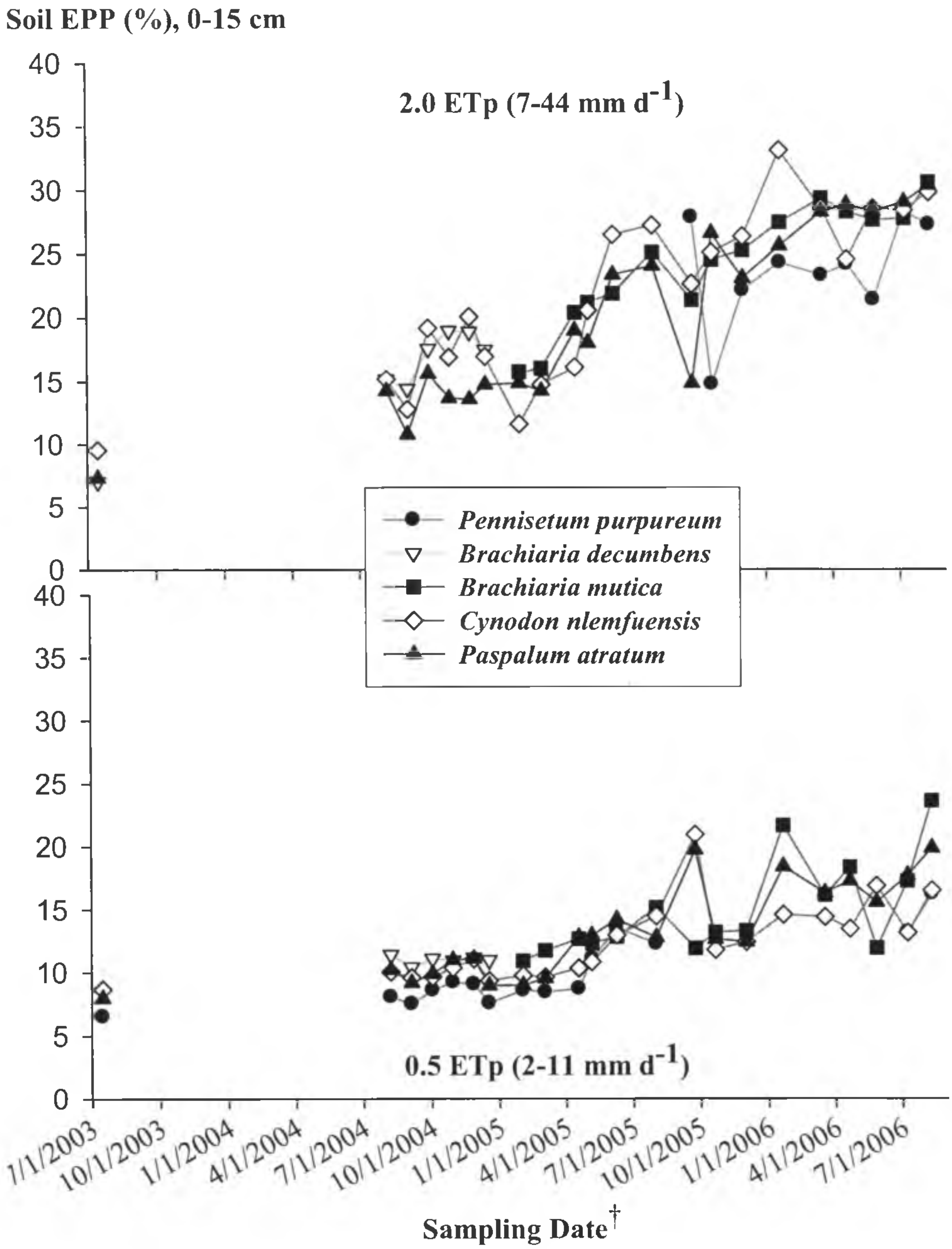
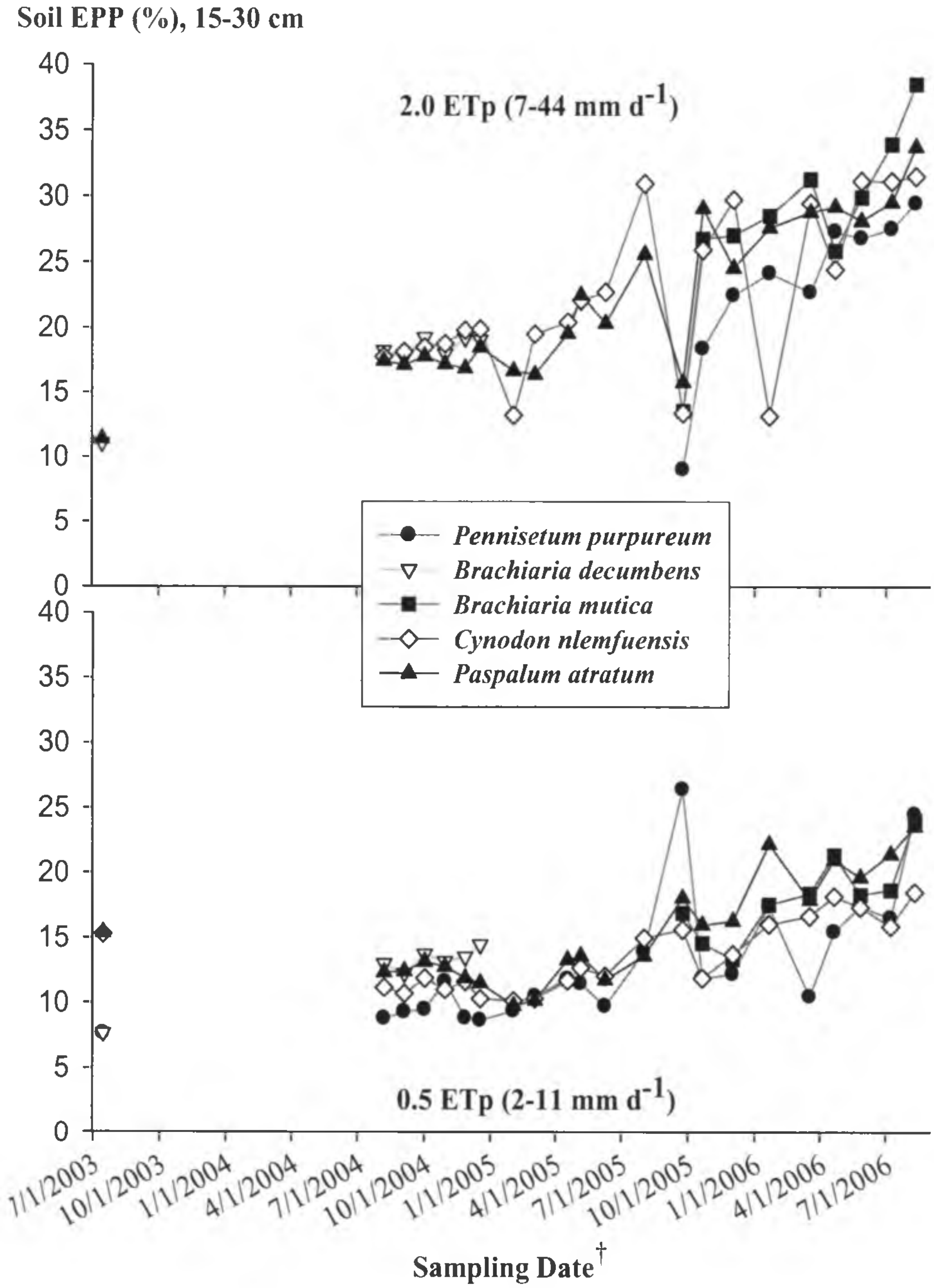



**Figure A.38. Soil solution total Na at the 35 cm depth of effluent-irrigated soil at the 0.5 ET$_p$ rate, Waianae, O'ahu, Hawai'i.**




†Soil samples were collected before the irrigation was installed in July 2003; grass establishment period was from December 2003 to July 2004

**Figure A.39. Soil EPP at the 0-15 cm depth of effluent-irrigated soil (Mollisol) planted to tropical grasses, Waianae, O'ahu, Hawai'i.**




†Soil samples were collected before the irrigation was installed in July 2003; grass establishment period was from December 2003 to July 2004

**Figure A.40. Soil EPP at the 15-30 cm depth of effluent-irrigated soil (Mollisol) planted to tropical grasses, Waianae, O'ahu, Hawai'i.**

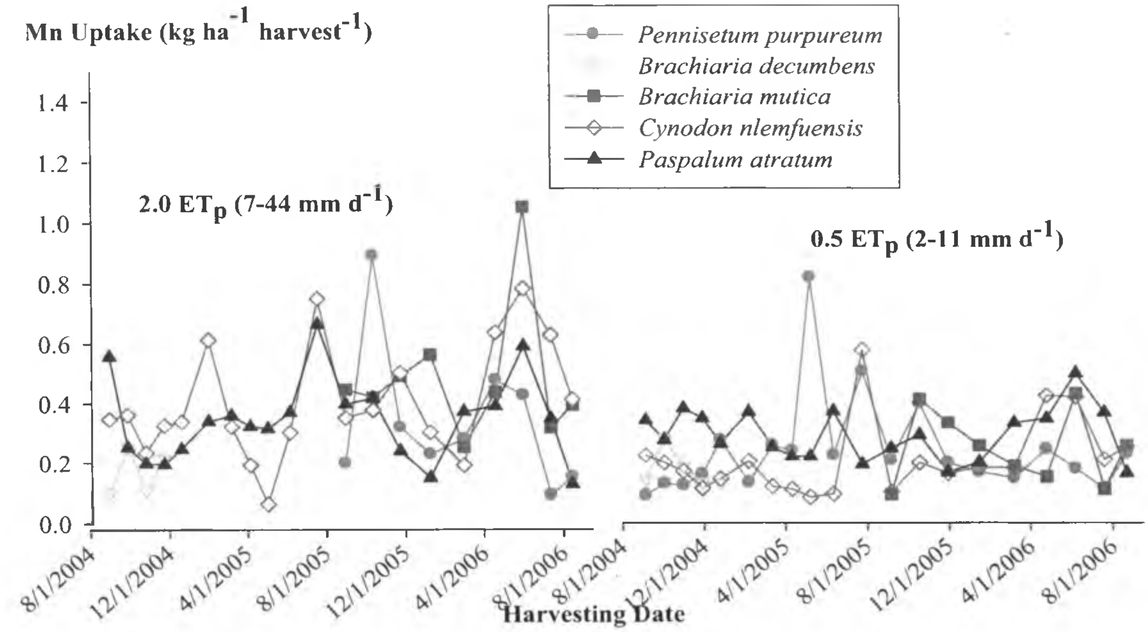



**Figure A.41. Manganese uptake of tropical grasses irrigated with dairy effluent, Waianae, Oahu, Hawai'i, Aug 2004-Aug 2006.**

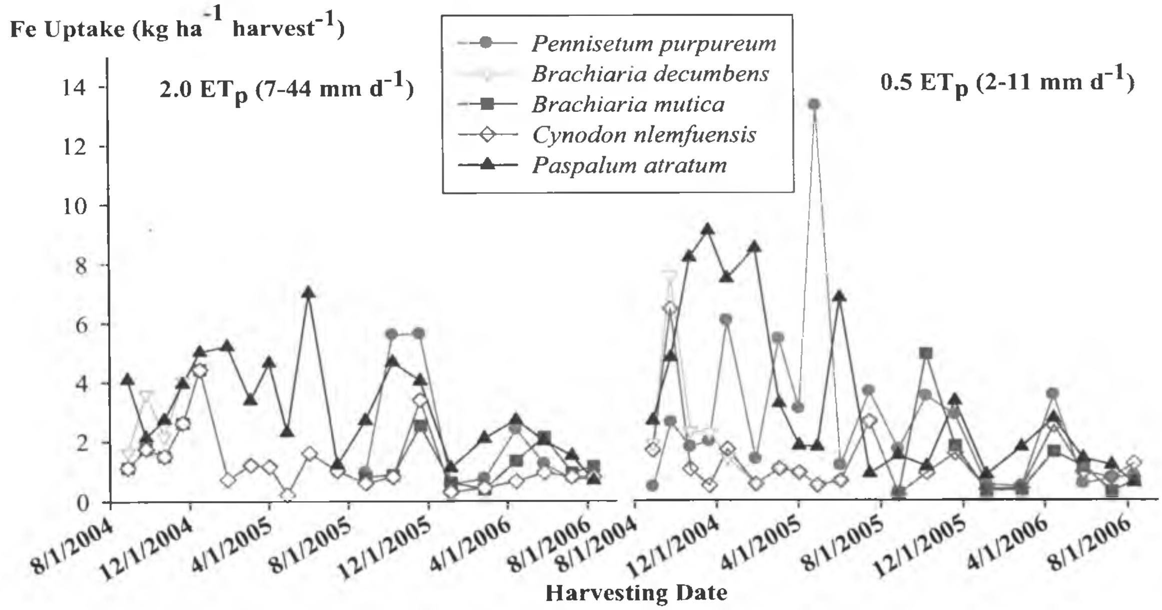



**Figure A.42. Iron uptake of tropical grasses irrigated with dairy effluent, Waianae, Oahu, Hawai'i, Aug 2004-Aug 2006.**

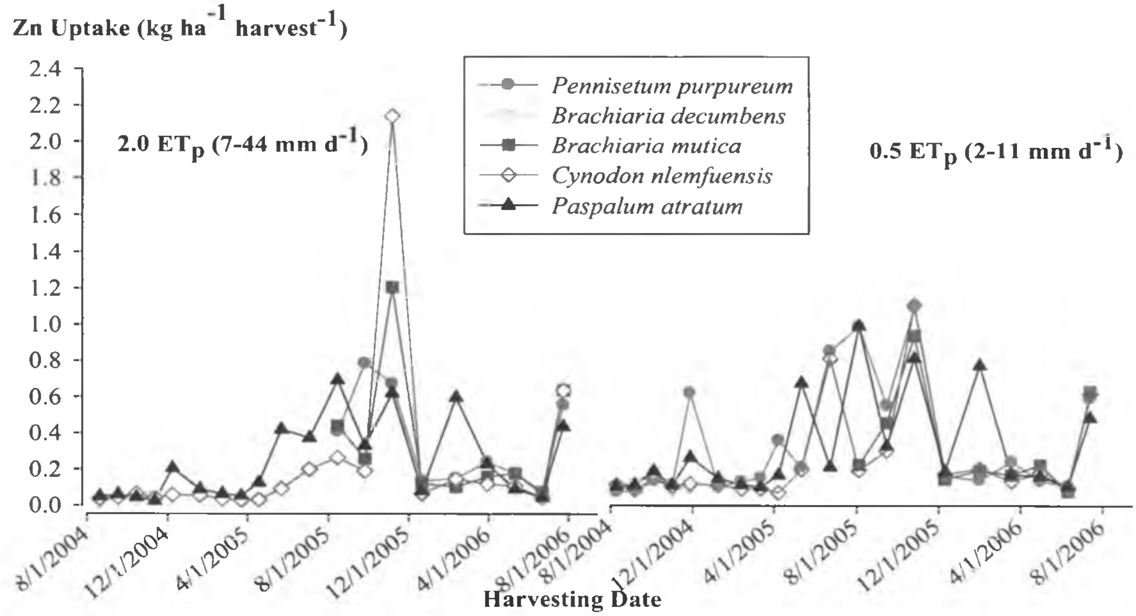



**Figure A.43. Zinc uptake of tropical grasses irrigated with dairy effluent, Waianae, Oahu, Hawai'i, Aug 2004-Aug 2006.**

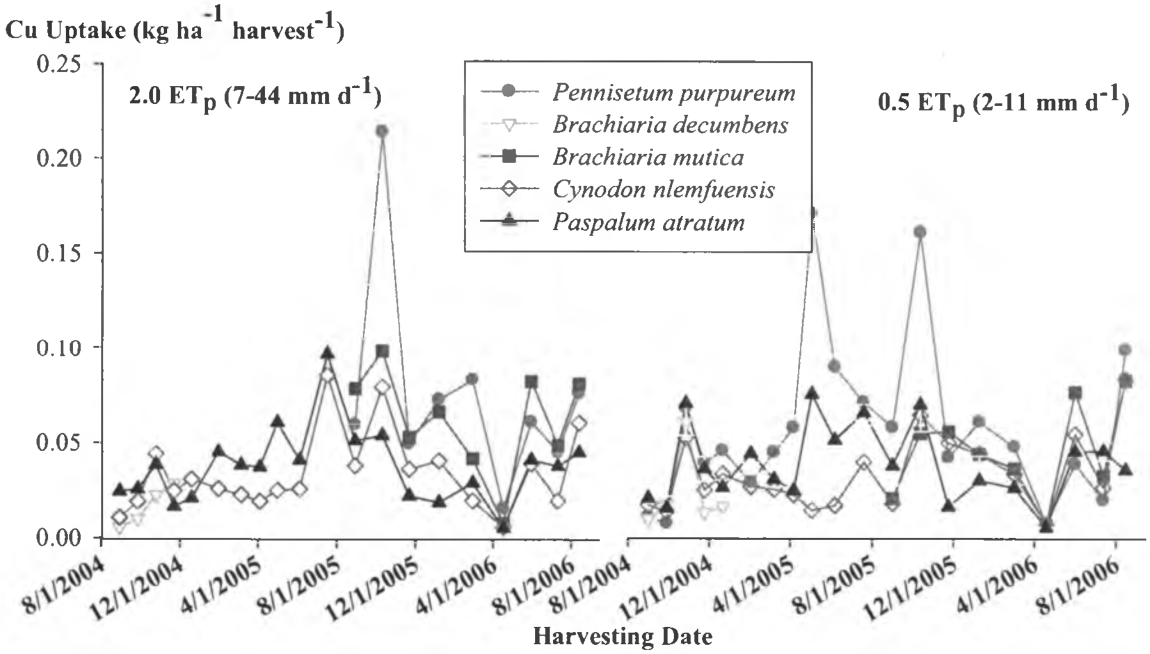



**Figure A.44. Copper uptake of tropical grasses irrigated with dairy effluent, Waianae, Oahu, Hawai'i, Aug 2004-Aug 2006.**

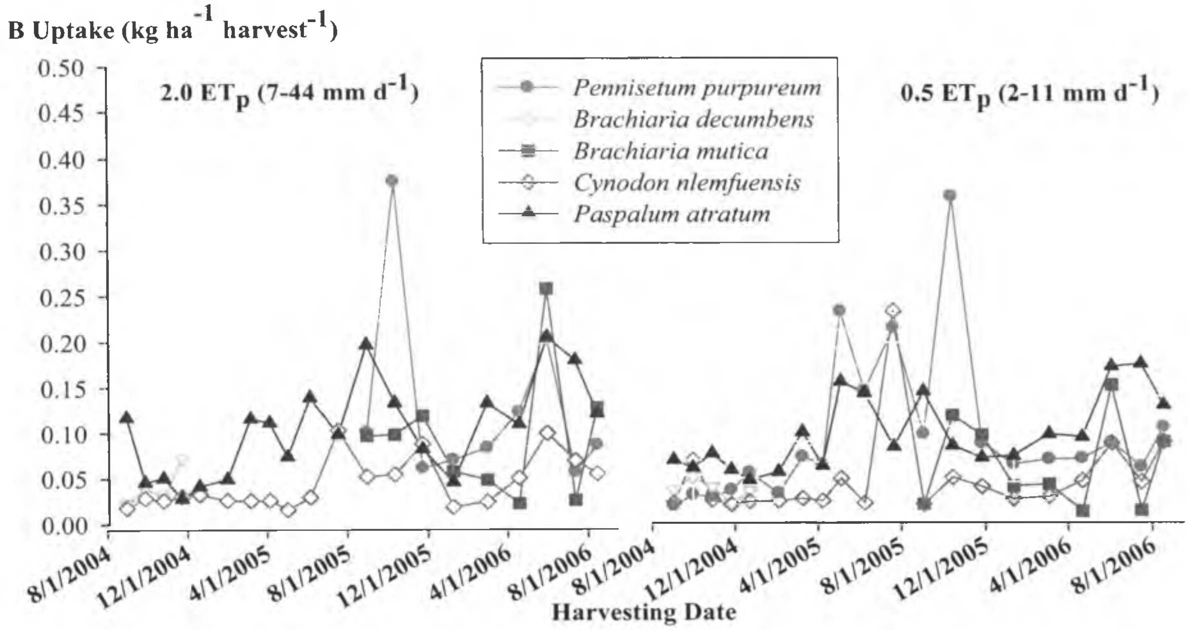



**Figure A.45. Boron uptake of tropical grasses irrigated with dairy effluent, Waianae, Oahu, Hawai'i, Aug 2004-Aug 2006.**

**Appendix 6.1. Prototype screens for effluent application decision-aid tool.**

WELCOME

to

EFFLUENT APPLICATION DECISION AID TOOL

**TPSS, University of Hawaii at Manoa**

***Location***

New Field **Add**

Search Database

| Location | Farm/ Climate/ Soil | Effluent Application | Crops & Milk Production | Summary | Recommendations |
|---|---|---|---|---|---|

## Farm Location

Field identifier: 0000001

Location (Town/County: Waianae | Oahu

Farm Name: Oahu Dairy

Farm identifier: 0000001

Owner/Manager: Mr. Dairyfarmer Ofhawaii

Address: 1010 Dairy St., Honolulu, HI 96800

E-mail: dairyfarmer@yahoo.com

SAVE | NEXT | BACK | Help

| Location | Farm/ Climate/ Soil | Effluent Application | Crops & Milk Production | Summary | Recommendations |
|---|---|---|---|---|---|

# Farm Information

Current crop(s) grown: Bana

Bana
Suerte
Others

Total land area: 1 ac 0 4047 ha

Field slope: 0 %

Field slope length: ft m

Distance to ocean/lake: ft m

river/stream: ft m

Elevation: 8 masl

SAVE NEXT BACK HELP

| Location | Farm/ Climate/ Soil | Effluent Application | Crops & Milk Production | Summary | Recommendations |
|---|---|---|---|---|---|

# Climatic Conditions

Rainfall:

May-Sept. [ ] in/mo [ ] mm/mo

Oct.-Apr. [ ] in/mo [ ] mm/mo

Potential evapotranspiration:

May-Sept. [ ] in/mo [ ] mm/mo

Oct.-Apr. [ ] in/mo [ ] mm/mo

Solar radiation:

May-Sept. [ ] in/mo [ ] mm/mo

Oct.-Apr. [ ] in/mo [ ] mm/mo

Temperature:

Min.: [ ] degC [ ] degF Ave.: [ ] degC [ ] degF

Max.: [ ] degC [ ] degC

SAVE NEXT BACK HELP

| Location | Farm/ Climate/ Soil | Effluent Application | Crops & Milk Production | Summary | Recommendations |
|---|---|---|---|---|---|

## Soil Information

Soil series: Pulehu

Soil type/mineralogy: Haplustoll

Range of phosphorus sorption capacity: 0-25 mg P $kg^{-1}$

Drainage class/hydraulic conductivity class: Poorly drained

Groundwater table depth: [ ] ft 40-80 m

Soil analysis: **Encode** **Import**

Soil texture:
Clay: 64 % Silt: 32 % Sand: 4 %

Bulk density: 1.3 g/$cm^3$ [ ] kg/$m^3$

Soil moisture content:

| | |
|---|---|
| 0-15 cm | |
| 15-30 cm | |

Fertilization/liming history: **Known** **Unknown**

SAVE NEXT BACK HELP

| Location | Farm/ Climate/ Soil | Effluent Application | Crops & Milk Production | Summary | Recommendations |
|---|---|---|---|---|---|

**Soil Data**

| Sampling date | pH | EC | OC (mg/kg) | N (mg/kg) | P (mg/kg) | K (mg/kg) | Ca (mg/kg) | Mg (mg/kg) | Na (mg/kg) | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Analysis method | 1 soil:1 water | 1 soil:2 water | Walkley-Black | Total Kjeldahl | Olsen | ICP | ICP | ICP | ICP | | |
| 1/1/06 | 8.2 | 3.4 | 1.20 | 0.25 | 150 | 3000 | 4000 | 2000 | 800 | | |
| | | | | | | | | | | | |
| | | | | | | | | | | | |
| | | | | | | | | | | | |
| | | | | | | | | | | | |

Note: shaded areas should be filled-in. The soil data worksheet to be imported should be in the same format at the built-in format in the model.

*(ESP and CEC will be calculated by the computer.)*

SAVE NEXT BACK HELP

| Location | Farm/ Climate/ Soil | Effluent Application | Crops & Milk Production | Summary | Recommendations |
|---|---|---|---|---|---|

**Fertilization/liming history:**

| Application date | Method | Fertilizer applied | Form applied | Amount applied (lb/ac) | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | N | P | K | Ca | Mg | Na | Fe | Mn | Zn | Cu | B |
| | | | | | | | | | | | | | | |
| | | | | | | | | | | | | | | |
| | | | | | | | | | | | | | | |
| | | | | | | | | | | | | | | |

*(this next table will be automatically filled-up by the computer)*

| Application date | Method | Fertilizer applied | Form applied | Amount applied (kg/ha) | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | N | P | K | Ca | Mg | Na | Fe | Mn | Zn | Cu | B |
| | | | | | | | | | | | | | | |
| | | | | | | | | | | | | | | |
| | | | | | | | | | | | | | | |
| | | | | | | | | | | | | | | |

SAVE NEXT BACK HELP

| Location | Farm/ Climate/ Soil | Effluent Application | Crops & Milk Production | Summary | Recommendations |
|---|---|---|---|---|---|

# Effluent Application

Area for application: 1 ac 0.4047 ha

Number of animals raised: 1500

Volume of effluent per animal per day: 40 gal L

Effluent properties: ***Import*** ***Encode***

Application rate: gal/d L/d

gal/ac/y L/ha/y

Frequency of application: 365 no./y 0.8 h/application/d

Application method: Subsurface drip

Application depth: 12 in m

Application time: evening

morning

noon

afternoon

evening

SAVE NEXT BACK HELP

| Location | Farm/ Climate/ Soil | Effluent Application | Crops & Milk Production | Summary | Recommendations |
| --- | --- | --- | --- | --- | --- |

**Effluent properties**

| Sampling date | pH | EC | N | P | K | Ca | Mg | Na | Fe | Mn | Zn | Cu | B | Mo | Al | SAR | TDS |
| --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- |
| Analysis method | | | | | | | | | | | | | | | | | |
| 1/1/01 | | | | | | | | | | | | | | | | | |
| 1/2/02 | | | | | | | | | | | | | | | | | |
| | | | | | | | | | | | | | | | | | |
| | | | | | | | | | | | | | | | | | |

SAVE NEXT BACK HELP

| Location | Farm/ Climate/ Soil | Effluent Application | Crops & Milk Production | Summary | Recommendations |
|---|---|---|---|---|---|

# Intended Crops

Species: Rana

Rana
California
Signal
Star
Suerte

Dry matter production: 50000 lb/ac/y kg/ha/y
*(autoconvert)*

Forage analysis: **Import** **Encode**

Feeding method: Cut & carry

Harvesting method: Mechanical

Harvesting interval: 30 d

30 d
45 d

Harvesting height: 6 in cm

3
6

SAVE NEXT BACK HELP

| Location | Farm/ Climate/ Soil | Effluent Application | Crops & Milk Production | Summary | Recommendations |
|---|---|---|---|---|---|

# Milk Production

No. of Animals

Volume of milk per animal gal/d L/d

Nutrient Analysis of Milk

N (%)

P (%)

K (%)

NRC nutrition requirements for dairy cattle Import

SAVE NEXT BACK HELP

| Location | Farm/ Climate/ Soil | Effluent Application | Crops & Milk Production | Summary Outputs | Recommendations |
| --- | --- | --- | --- | --- | --- |

# Summary Outputs

DM production and annual nutrient uptake

Water flux

Cumulative nutrient applied (per month)

Critical soil properties (P, EC, pH, ESP)

Nutrient balance (Input – Output)—excess or deficit

Carrying capacity

NEXT BACK PRINT

| Location | Farm/ Climate/ Soil | Effluent Application | Crops & Milk Production | Summary | Recommendations |
|---|---|---|---|---|---|

# Recommendations

Match the nutrient requirement of the crop

Match the water requirement of the crop

Increase/decrease the number of animals raised

Increase/decrease the area for application

Increase/decrease the rate of application

Change the type of grass grown into more robust-growing one.

Dilute the effluent with freshwater.

Avoid effluent application during the rainy days.

PRINT | NEW SIMULATION | EXIT

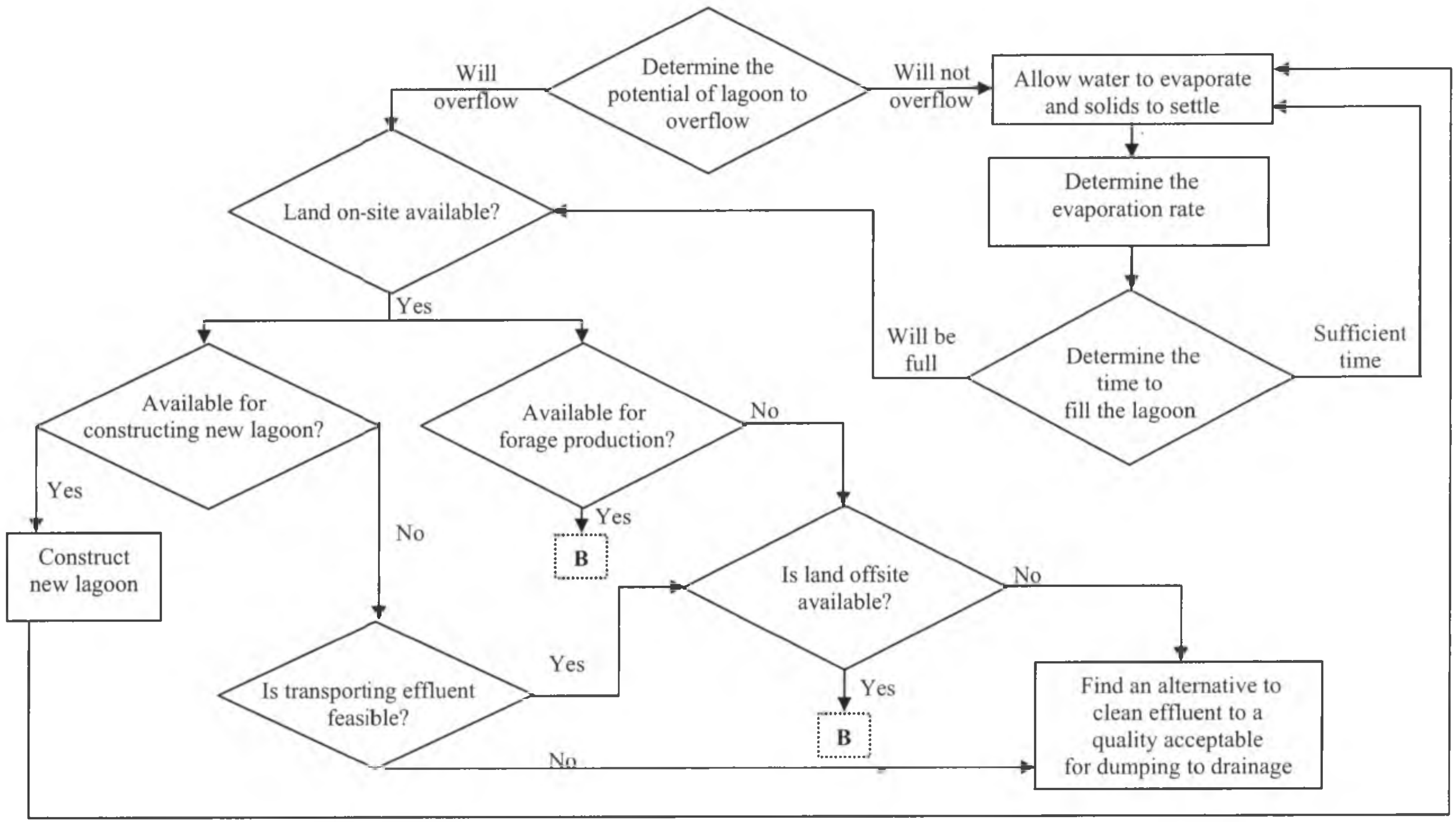



**Figure 6.1. Decision-making diagram for effluent management.**

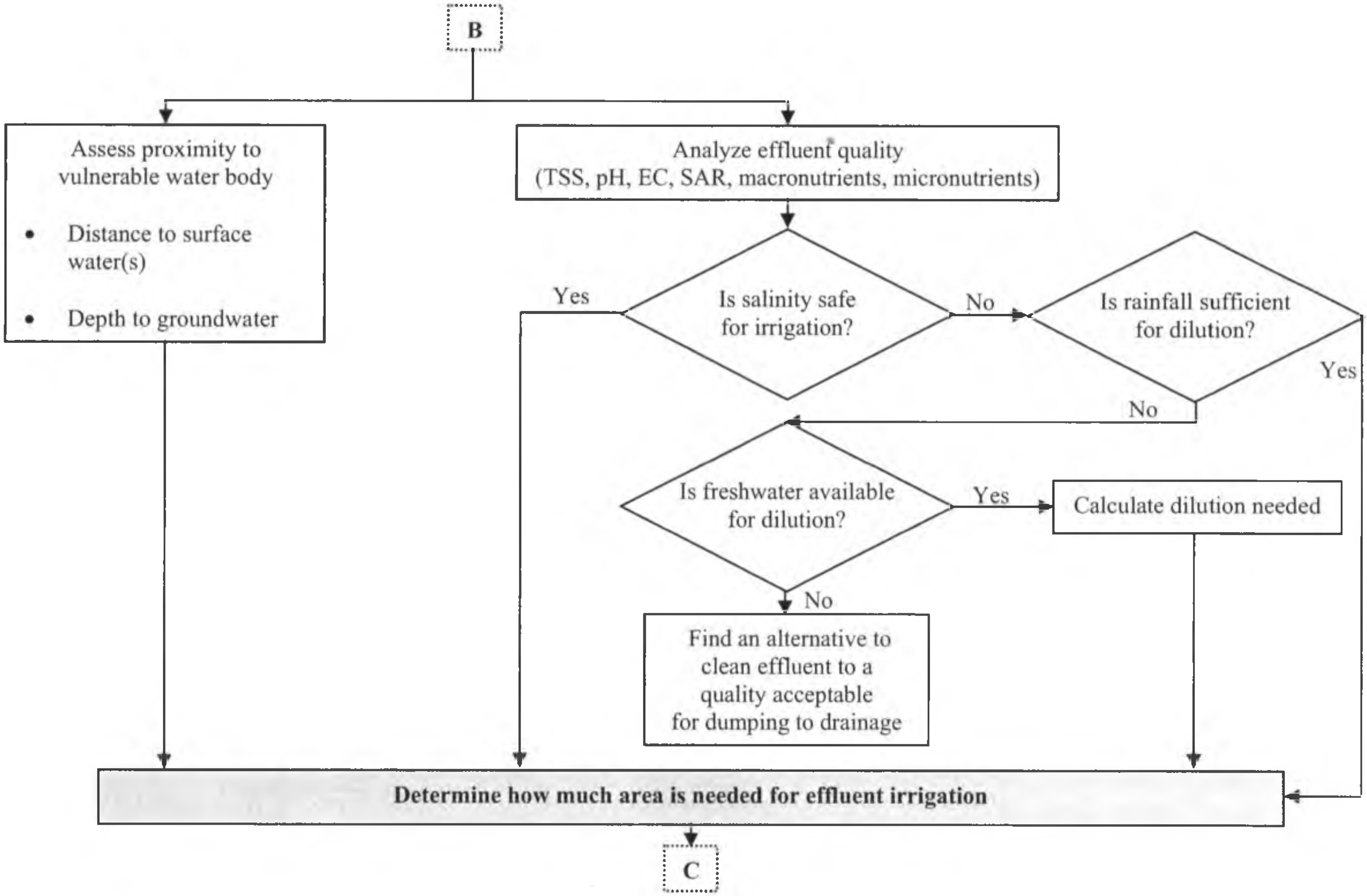



**Figure 6.1. Decision-making diagram for effluent management (cont.).**

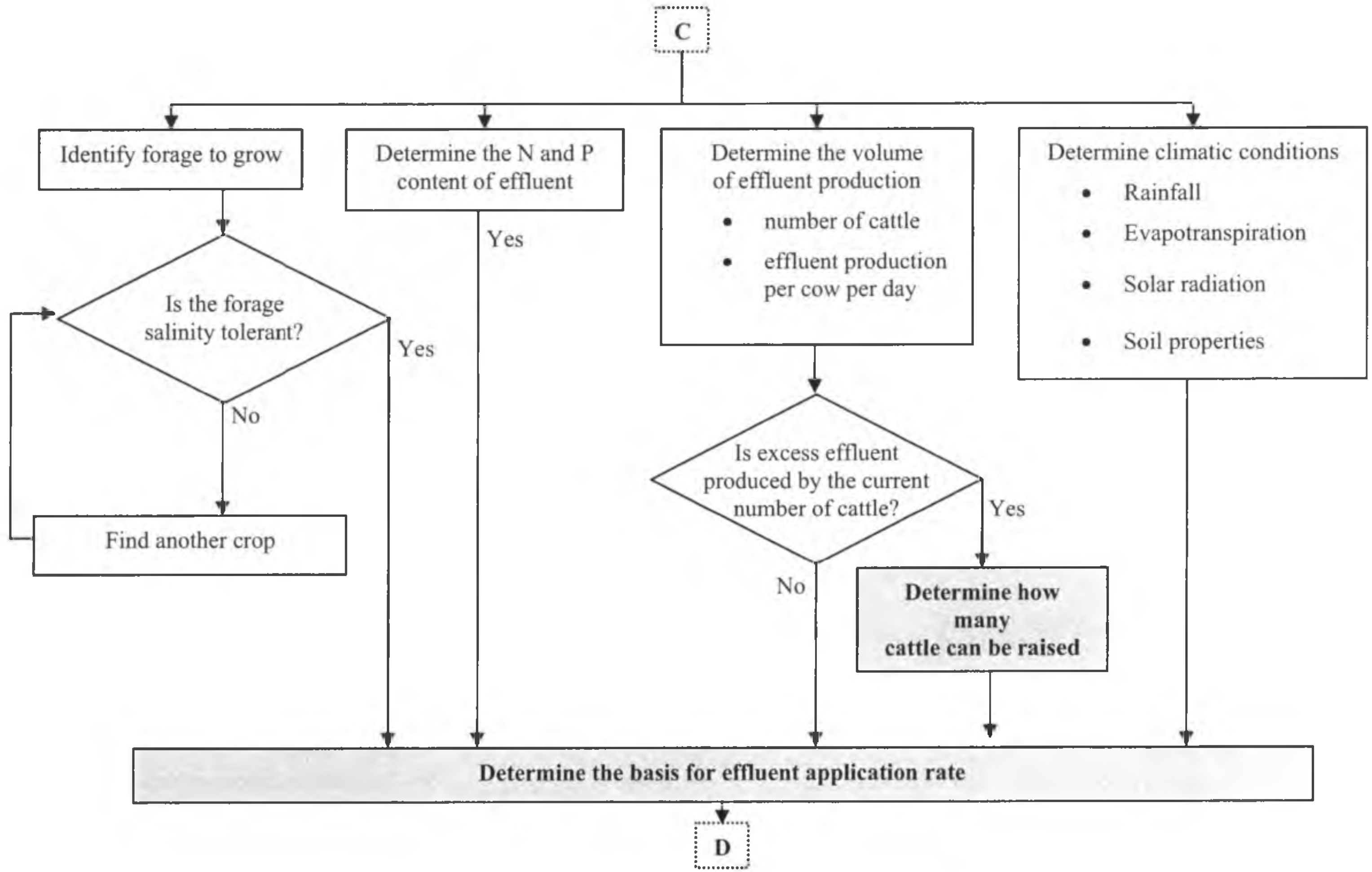



**Figure 6.1. Decision-making diagram for effluent management (cont.).**

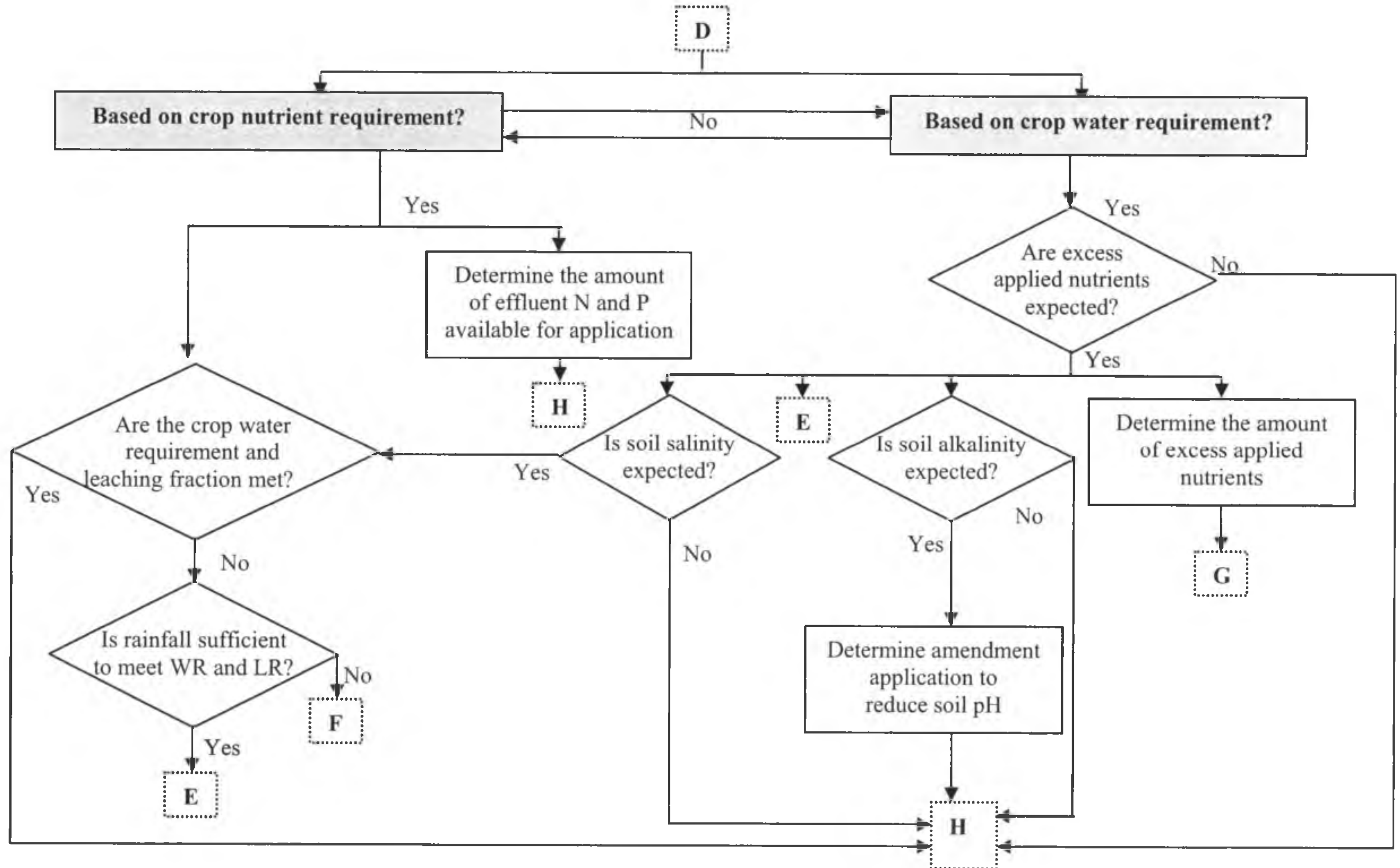



**Figure 6.1. Decision-making diagram for effluent management (cont.).**

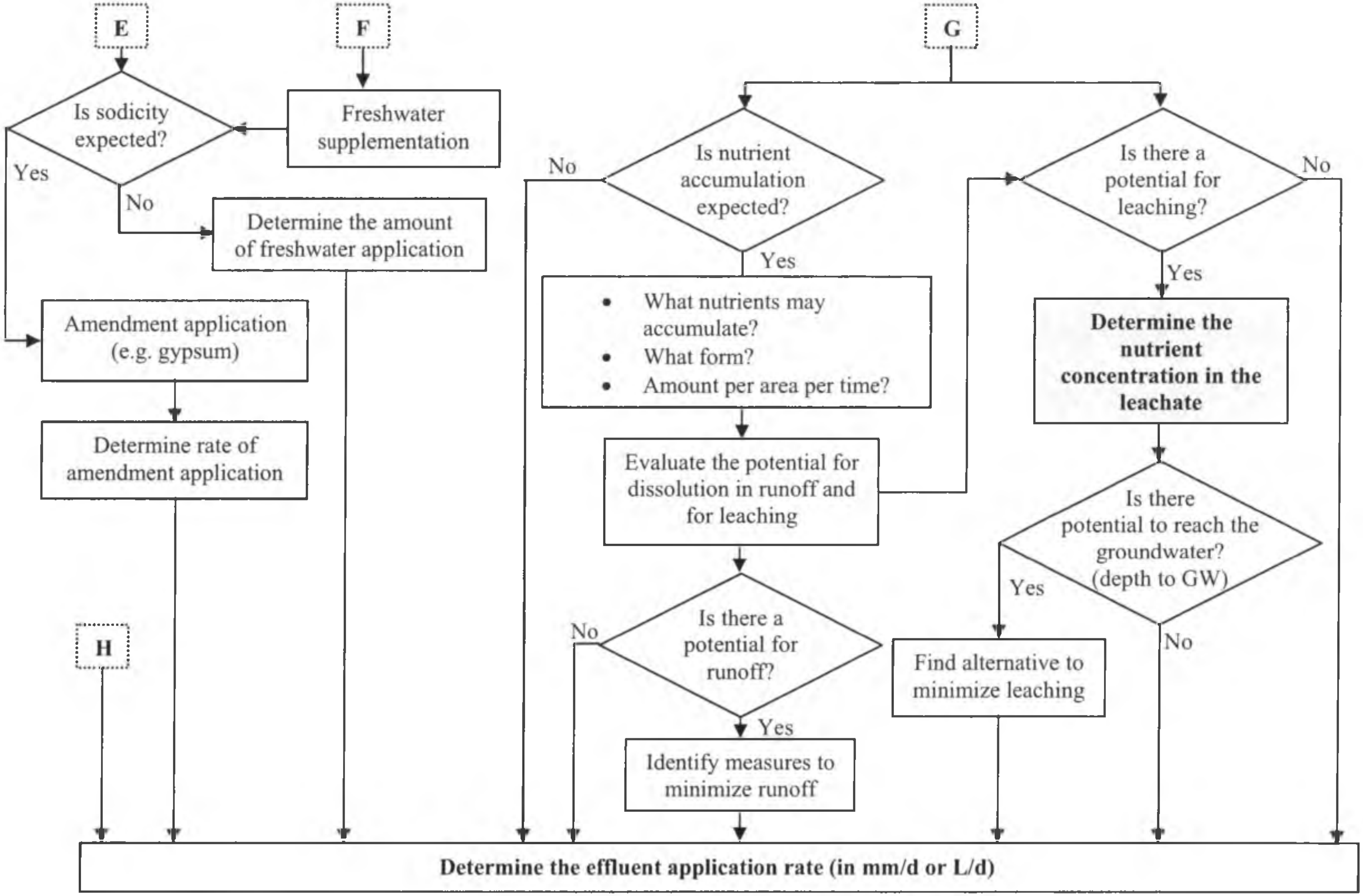



**Figure 6.1. Decision-making diagram for effluent management (cont.).**

## REFERENCES


Adeli, A., and J.J. Varco. 2001. Swine effluent as a source of nitrogen and phosphorus for summer forage grasses. Agron. J. 93:1174–1181.

Adeli, A., Sistani, K.R., Varco, J.J., Rowe, D.E. 2005. Effects of swine lagoon effluent relative to commercial fertilizer applications on warm-season forage nutritive value. Agron. J. 97:408-417.

Adjei, M.B., and J.E. Rechcigl. 2002. Bahiagrass production and nutritive value as affected by domestic wastewater residuals. Agron. J. 94:1400-1410.

Agbenin, J.O. and H. Tiessen, 1995. Phosphorus sorption at field capacity and soil ionic strength: kinetics and transformation. Soil Sci. Soc. Am. J. 59: 998-1005.

Akhtar, M., and S. M. Alam. 2001. Effect of incubation period on phosphate sorption from two P sources. Pakistan. J. Biol. Sci. 1(3):124-125.

Akhtar, M.S., B.K. Richards, P.A. Medrano, M. deGroot, and T.S. Steenhuis. 2003. Dissolved phosphorus from undisturbed soil cores: related to adsorption strength, flow rate, or soil structure? Soil Sci. Soc. Am. J.; 67: 458-470.

Aldrich, L. J., C.L. Munster, V.A. Haby, and J.M. Sweeten. 1997. Land application of poultry lagoon effluent. Trans. ASAE 40(6):1607-1615.

Allen, R.G. 2001. Reference Evapotranspiration calculation and software (Ref-ET v. 2.0) for FAO and ASCE standardized equations. University of Idaho, Moscow, ID.

Allen, G.H., and W.A.R. Cowdry. 1961. Yield from irrigated pasture in Burdekin. Queensland Agric. J. 87:207-213.

Allen, R.G., L.S. Pereira, D. Raes, and M. Smith. 2001. Crop evapotranspiration - Guidelines for computing crop water requirements. FAO Irrigation and drainage paper 56, Rome, Italy.

Alloway, B.J. 2004. Zinc in soils and crop Nutrition. [Online]. Available at http://www.iza.com/zwo_org/Publications/PDFs/ALLOWAY-all.pdf. (accessed 15 Aug. 2005, verified 31 Mar. 2006). International Zinc Association, Brussels, Belgium.

Alvarado, A. R., L.I. Arriojas, E. Chacón, S.C. Rodriguez, and F. Chacin. 1990. Estudios sobre henificacion del pasto barrera (Brachiaria decumbens Stapf) en condiciones de sabanas del Piedemonte barines I. Produccion y valor nutritivo de la material seca. (In Spanish, English Abstract). Zootecnia Trop. Vol. 8 (1 y 2):17-36.

Alvarez, R., Evans, L.A., Milham, P.J., and M.A. Wilson., 2004. Effects of humic material on the precipitation of calcium phosphate. Geoderma 118, 245 - 260.

Alvim, M.J., M.A. Botrel, R.S. Verneq, and J.A. Salvati. 1990. Aplicao de nitrogênio em acessos de Brachiaria. 1. Efeito sobre a produção de matéria seca. (In Portuguese, English Abstract). Past. Trop. 12:2- 6.

Anderson, G., E.G. Williams, and J.O. Moir. 1974. A comparison of the sorption of inorganic orthophosphate and inositol hexaphosphate by six acid soils. J. Soil Sci. 25:51–62.

Andrews, A.C. 1976. The productivity, herbage quality and physiology of bermuda grass (Cynodon dactylon) and kikuyu (Pennisetum clandestinum) in coastal New South Wales. Ph.D. thesis, Univ. of Sydney, Australia.

Antoniadis, V. and B.J. Alloway. 2002. Leaching of cadmium, nickel, and zinc down the profile of sewage sludge-treated soil. Commun. Soil Sci. Plant Anal. 33(1-2):273-286.

Appelt, H., N.T. Coleman, and P.F. Pratt. 1975. Interactions between organic compounds minerals, and ions in volcanic-ash derived soils: II. Effects of organic compounds on the adsorption of phosphate. Soil Science Society of America Proceedings, 39: 628-630.

Arvin, E. 1983. Observations supporting phosphate removal by biologically mediated chemical precipitation--a review. Water Sci. Technol. 15:43-63.

Assefa, B.A., J.J. Schoenau, and M.C.J. Grevers. 2004. Effects of four annual applications of manure on Black Chernozemic Soils. Can. Biosyst. Eng. 46:39-46.

Assouline, S. 2002. The effects of micro-drip and conventional drip irrigation on water distribution and uptake. Soil Sci. Soc. Am. J. 66(5):1630-1636.

Auerswald, K., M. Kainz, S. Angermüller, H. Steindl. 1996. Influence of exchangeable potassium on soil erodibility. Soil Use and Manage. 12(3):117–121.

Ayers R.S., and D.W. Westcot. 1985. Water quality for agriculture. FAO Irrigation and Drainage Paper No. 29 (Rev. 1). Rome, Italy.

Bahl, G.S.and G.S. Toor. 2002. Influence of poultry manure on phosphorus availability and the standard phosphate requirement of crop estimated from quantity-intensity relationships in different soils. Biores. Tech. 85:317-322.

Baker, J.L., and D.R. Timmons. 1994. Fertilizer management effects on leaching of labeled nitrogen from no-till corn in field lysimeters. J. Envi. Qual. 23:305-310.

Balks, M.R., W.J. Bond, and C.J. Smith. 1998. Effects of sodium accumulation on soil physical properties under an effluent-irrigated plantation. Aust. J. Soil Res. 36: 821-830.

Barber, S.A. 1995. Soil nutrient bioavailability: a mechanistic approach. 2nd ed. John Wiley and Sons, New York.

Bar-Ness and Chen 1991. Manure and peat-based iron-organo complexes. Plant Soil 130(1-2):45-50.

Barnett, G.M. 1994. Phosphorus forms in animal manure. Biores. Tech. 49:139-147.

Barrow, N.J. 1972. Influence of solution concentration of calcium on the adsorption of phosphate, sulphate, and molybdate by soils. Soil Sci. 113:175–180.

Barrow N.J. 1983. On the reversibility of phosphate sorption by soils. J. Soil Sci. 34:751–758.

Barrow, N.J. 1987. Return of nutrients by animals. p. 181–186. In R. W. Snaydon (ed.) Managed grasslands. Elsevier, Oxford, England.

Barrow, N.J., and T.C. Shaw. 1979. Effects of solution:soil ratio and vigour of shaking on the rate of phosphate adsorption by soil. J. Soil Sci. 30, 67-76.

Barton, L., L.A. Schipper, G. F. Barkle, M. McLeod, T.W. Speir, M.D. Taylor, A. C. McGill, A.P. van Schaik, N.B. Fitzgerald and S.P. Pandey. 2005. Land application of domestic effluent onto four soil types plant uptake and nutrient leaching. J. Environ. Qual. 34:635-643.

Bar-Yosef, B., U. Kafkafi, R. Rosenberg, and G. Sposito. 1988. Phosphorus adsorption by kaolinite and montmorillonite: I. Effect of time, ionic strength, and pH. Soil Sci. Soc. Am. J. 52:1585–1589.

Beck, M.A. and P.A. Sanchez. 1994. Soil phosphorus fraction dynamics during 18 years of cultivation on a typic paleudult. Soil Sci. Soc. Am. J. 58:1424-1431.

Benneth, W. F. (ed.). 1993. Nutrient deficiencies and toxicities in crop plants, APS Press, St. Paul, MN.

Bhat, K.K.S., and J.R. O'Callaghan. 1980. Behaviour in the soil of orthophosphate from pig slurry compared with that of KH2PO4. J. Agric. Sci. (Cambridge) 94:195–201.

Binh, L.H. 1998. Forage productivity of para grass in Vietnam. In A.C. de la Vina and F. A. Moog. Proceedings of the 6th Meeting of the Regional Working Group on Grazing and Feed Resources of Southeast Asia, held at Legazpi City, Philippines, 5-9 October 1998. Department of Livestock Development and FAO (Food and Agriculture Organization of the United Nations), Rome, Italy. pp. 167-171.

Bloom, P.R. 1981. Phosphorus adsorption by an aluminium-peat complex. Soil Sci. Soc. Am. J. 45:267-72.

Boddey R. M. and J. Dobereiner. 1988. Nitrogen fixation associated with grasses and cereals: recent results and perspectives for future research. Plant Soil 108: 53-65.

Boesch, D.F., R.B. Brinsfield, and R.E. Magnien. 2001. Chesapeake Bay eutrophication: Scientific understanding, ecosystem restoration, and challenges for agriculture. J. Environ. Qual. 30:303–320.

Bogdan, A.V. 1977. Tropical pasture and fodder plants. Longman, London.

Börling, K., E. Ottabong, and E. Barberis. 2001. Phosphorus sorption in relation to soil properties in some cultivated Swedish soils. Nutr. Cycling Agroecosyst. 59:39–46.

Borggaard, O.K., S.S. Jørgensen, J.P. Møberg, and B. Raben-Lange. 1990. Influence of organic matter on phosphate adsorption by aluminum and iron oxides in sandy soils. J. Soil Sci. 41:443–449.

Boruvka, L., and J. E. Rechcigl. 2003. Phosphorus retention by the Ap horizon of a spodosol as influenced by calcium amendments. Soil Sci. 168:699-706.

Brady, N.C., and R.R. Weil. 2002. The nature and properties of soils, 13th ed. Prentice Hall, New Jersey.

Brennan, R.F., M.D.A. Bolland, R.C. Jeffrey, and D.G. Allen. 1994. Phosphorus adsorption by a range of western Australian soils related to soil properties. Commun. Soil Sci. Plant Anal. 25:2785–2795.

Breeuwsma, A., and S. Silva. 1992. Phosphorus fertilization and environmental effects in the Netherlands and the Po region (Italy). Rep. 57. Agric. Res. Dep., The Winand Staring Centre for Integrated Land, Soil, and Water Res., Wageningen, the Netherlands.

Breeuwsma, A., J.G.A. Reijerink, and O.F. Schoumans. 1995. Impact of manure on accumulations and leaching of phosphate in areas of intensive livestock farming. p. 239–249. In K. Steele (ed.) Animal waste and the land-water interface. CRC Press, Boca Raton, FL.

Bremner, J.M., and C.S. Mulvaney. 1982. Nitrogen: Total. p. 565–624. In A.L. Page et al. (ed.) Methods of soil analyses. Part 2. Agron. Monogr. 9. 2nd ed. ASA and SSSA, Madison, WI.

Brink, G.E., D.E. Rowe, K.R. Sistani, and A. Adeli. 2003. Bermudagrass cultivar response to swine effluent application. Agron. J. 95:597–601.

Brink, G.E., K. R. Sistani, and D. E. Rowe. 2004. Nutrient uptake of hybrid and common bermudagrass fertilized with broiler litter. Agron. J. 96:1509-1515.

Bronswijk, J.J.B. 1989. Prediction of actual cracking and subsidence in clay soils. Soil Sci. 148:87–93.

Bumaya, A.H., and D.V. Naylor. 1988. Phosphorus sorption and extractability in Andic soil incubated with plant residues of variable P content. Plant Soil 112:77–81.

Burt, R., M.D. Mays, E.C. Benham, and M.A. Wilson. 2002. Phosphorus characterization and correlation with properties of selected benchmark soils of the United States. Commun. Soil Sci. Plant Anal. 33:117–141.

Busman, L., J. Lamb, G. Randall, G. Rehm, and M. Schmitt. 1998. Phosphorus in the agricultural environment: the nature of phosphorus in soils. [Online]. Available at http://www.extension.umn.edu/distribution/cropsystems/DC6795.html. (accessed Feb. 2005; verified Dec. 2006), University of Minnesota, MN.

Bussink, D.W. 1994. Relationships between ammonia volatilization and nitrogen fertilizer application rate, intake and excretion of herbage nitrogen by cattle on grazed swards. Fert. Res. 38:111–121.

Bustamante, J., M. Ibrahim, and J. Beer. 1998. Evaluación agronómica de ocho gramíneas mejoradas en un sistema silvopastoril con poró (Erythrina poeppigiana) en el trópico húmedo de Turrialba. (In Portuguese, with English Abstract). Agrofor. en las Amér. 5(19): 11-16.

Buxton, D.R., and S.L. Fales. 1994. Plant environment and quality. In Fahey, G.C., et al. (eds.) Forage quality, evaluation, and utilization. Proceedings of the National Conference on Forage Quality, Evaluation and Utilization held at the University of Nebraska, Lincoln on 13-15 Apr, ASA-CSSA-SSSA, Madison, WI.

Caballero, R., A. Bustos, and R. Roman Soil salinity under traditional and improved irrigation schedules in central Spain. Soil Sci. Soc. Am. J. 65(4):1210-1218.

Cameron, K. C., A. W. Rate, M. J. Noonan, S. Moore, N. S. Smith, L. E. Kerr. 1996. Lysimeter study of the fate of nutrients following subsurface injection and surface application of dairy pond sludge to pasture. Agric. Ecosyst. Envi. 58:187-197.

Cameron, K.C., H.J. Di, M.R. Anwar, J.M. Russell, and J.W. Barnett. 2003. The 'critical' ESP value: does it change with land application of dairy factory effluent? New Zealand J. Agric. Res. 46: 147-154.

Cao, X., W. Harris, M. Josan, and V. Nair, 2006. Inhibition of calcium phosphate precipitation under conditions typical of manure-amended soil solutions. ASA-CSSA-SSSA International Annual Meetings Abstracts, November 12-16, Indianapolis, IN.

Caro-Costas, R., F. Abruna, and J. Figerella. 1972. Effect of nitrogen rates, harvest interval and cutting heights on yield and composition of stargrass in Puerto Rico. J. Agric. Univ. PR 56:267–279.

Celi, L., S. Lamacchia, F.A.Marsan, and E. Barberis. 1999. Interaction of inositol hexaphosphate on clays: Adsorption and charging phenomena. Soil Sci. 164:574-585.

Council for Agricultural Science and Technology (CAST). 1996. Integrated animal waste management. Report Number 128. CAST, Ames, IA.

Cavallaro, N. and M.B. McBride. 1984. Zinc and copper status and fixation by an acid soil clay: Effect of selective dissolutions. Soil Sci. Soc. Am. J. 48:1050-1054.

Centro Internacional de Agricultural Tropical. 1979. Tropical Pastures Program Annual Report 1977. Centro Internacional de Agricultura Tropical (CIAT), Cali, Colombia.

Centro Internacional de Agricultural Tropical. 1984. Tropical Pastures Program Annual Report 1982. Centro Internacional de Agricultura Tropical (CIAT), Cali, Colombia. 362 p.

Chalmers, A.G., and P.J.A. Withers. 1998. Phosphorus leaching from soils enriched with phosphorus loadings from organic manures. p. 144–145. In R.H. Foy and R. Dils (eds.) Practical and innovative measures for the control of agricultural phosphorus losses to water. Organisation for Economic Co-Operation and Development, Paris.

Chang, C., T.G. Sommerfeldt, and T. Entz. 1991. Soil chemistry after eleven annual applications of cattle feedlot manure. J. Environ. Qual. 20: 475-480.

Chaparro, C. J. and L. E. Sollenberger (1997). Nutritive value of clipped 'Mott' elephantgrass herbage. Agron. J. 89(5): 789-793.

Chardon, W.J., O. Oenema, P. del Castilho, R. Vriesema, J. Japenga, and D. Blaauw. 1997. Organic phosphorus in solutions and leachates from soils treated with animal slurries. J. Environ. Qual. 26:372-378.

Chee, Y.K., and C.C. Wong. 1986. Forages in Malaysia. In G.J. Blair, D.A. Ivory, and T.R. Evans (eds.). Forages in Southeast Asian and South Pacific agriculture. ACIAR Proceedings Series No. 12. Canberra, Australia. p. 84-88.

Chin, F.Y. 1989. Justification, government policy, measures and strategy for the development of grazing reserves in Peninsular Malaysia. Proc. 1st Meet. of Regional FAO Working Group on Grazing and Feed Resources in S.E. Asia, Feb. 27-Mar. 3, Serdang, Malaysia, pp.77-82.

Chin, F.Y. 1995. Critical pasture management and utilization to reduce feed supplementation. Proc. 4th FAO Regional Forage Working Group S. E. Asia Workshop at Nha Trang, Vietnam, pp. 20-24.

Chin, F.Y. 1994. Managing and organizing communal grazing land for livestock production. Proc. FAO/DLVS workshop-cum-training on grassland management and forage production under different farming systems, May 10-13, Vientiane, Lao P.D.R.

Clark, D.R., C. J. Green, and J. A. Gordon. 2000. Laboratory exercises to demonstrate effects of ealts on plants and soils. J. Nat. Res. Life Sci. Edu. 29:41-45.

Clesceri, L. S., A. E. Greenberg and A. E. Eaton. 1998. Method 2320 (Alkalinity, Bicarbonate, Carbonate). In Franson, M. et al. (eds.). Standard Methods for the Examination of Water and Wastewater, 20th Edition, American Public Health Association, American Water Works Association, and Water Environment Federation, Washington, DC.

Cole, C.V. and S.R. Olsen and C.O. Scott. 1953. The nature of phosphate sorption by calcium carbonate. Soil Sci. Soc. Amer. Proc. 410:352-356.

Condron, L. M., B. L. Turner, and B. J. Cade-Menun. 2005. Chemistry and dynamics of soil organic phosphorus. p. 87-121. In Sims, J.T. and A.N. Sharpley (eds.) Phosphorus: agriculture and the environment. Agronomy Monograph 46. ASA, CSSA, and SSSA, Madison, WI.

Cook, B.G., B.C. Pengelly, S.D. Brown, J.L. Donnelly, D.A. Eagles, M.A. Franco, J. Hanson, B.F. Mullen, I.J. Partridge, M. Peters, and R. Schultze-Kraft. 2005. Tropical forages: an interactive selection tool. [CD-ROM], CSIRO, DPI&F(Qld), CIAT and ILRI, Brisbane, Australia.

Cooperband, L.R., and L.W. Good. 2002. Biogenic phosphate minerals in manure: Implications for phosphorus loss to surface waters. Environ. Sci. Technol. 36:5075-5082.

Currie, J.A. 1975. Para grass—viewpoint from two sides of the fence. Queensland Cane Growers Quart. Bull. 39: 28-29.

Dalal, R. 1977. Soil organic phosphorus. Adv. Agron. 29:83–117.

Daly K., D. Jeffrey, H. Tunney. 2001. The effect of soil type on phosphorus sorption capacity and desorption dynamics in Irish grassland soils. Soil Use Man. 17(1): 12-20.

Dantzman, C.L., M.F. Richter, and F.G. Martin. 1983. Chemical elements in soils under cattle pens. J. Environ. Qual. 12:164–168.

Davis, J.G., and B. Burgoa. 1995. Runoff and leaching of crop nutrients from soil in tilted beds as influenced by three rates of dairy lagoon effluent. J. Environ. Sci. Health 30(2):243-264 .

Deenik, J. 1997. Early warning indicators of groundwater contamination in soils with variable charge clays. M.Sc. Thesis, University of Hawaii at Manoa, Honolulu, HI.

Deifel, K.S., P.M. Kopittke, and N.W. Menzies. 2006. Growth response of various perennial grasses to increasing salinity. J. Plant Nut. 29(9):1573 – 1584.

Dendooven, L., E. Bonhomme, R. Merckx, and K. Vlassak. 1998. N dynamics and sources of N2O production following pig slurry application to a loamy soil. Biol. Fert. Soils 26:224–228.

Department of Health-Hawaii. 1996. The health state guidelines for livestock waste management. Wastewater Branch, DOH, Honolulu, HI.

Diez, J. A., A.I. de la Torre, M.C. Cartagena, M. Carballo, A. Vallejo, and M.J. Muñoz. 2001. Evaluation of the application of pig slurry to an experimental crop using agronomic and ecotoxicological approaches. J. Environ. Qual. 30:2165-2172.

Djodjic, F., K. Börling, and L. Bergström. 2004. Phosphorus leaching in relation to soil type and soil phosphorus content. J. Environ. Qual. 33:678–684.

Dobermann. A., and T. Fairhurst. 2000. Rice: nutrient disorders and nutrient management. Handbook series. Potash & Phosphate Institute (PPI), Potash & Phosphate Institute of Canada (PPIC) and International Rice Research Institute, pp. 191. Manila, Philippines.

Dodor, D.E., and K. Oya. 2000. Phosphate sorption characteristics of major soils in Okinawa, Japan. Commun. Soil Sci. Plant Anal. 31:277–288.

Duke, J.A. 1978. The quest for tolerant germplasm. p. 1–61. In ASA Special Symposium 32, Crop tolerance to suboptimal land conditions. Am. Soc. Agron. Madison, WI.

Duke, J.A. 1983. Handbook of Energy Crops [Online]. Available at http://www.hort.purdue.edu/newcrop/Indices/index_ab.html. (accessed 15 Aug. 2005, verified 31 Mar. 2006 ; updated 31 Jan. 2006). Department of Horticulture and Landscape Architecture, Purdue University, West Lafayette, IN.

Ebeling, A.M., L.G. Bundy, J.M. Powell, and T.W. Andraski. 2002. Dairy Diet Phosphorus Effects on Phosphorus Losses in Runoff from Land-Applied Manure Soil Sci. Soc. Am. J. 66:284-291.

Eghball, B. 2002. Soil properties as influenced by phosphorus- and nitrogen-based manure and compost applications. Agron. J. 94:128–135.

Eghball, B. 2003. Leaching of phosphorus fractions following manure or compost application. Commun. Soil Sci. Plant Anal. 34 (19-20):2803-2815.

Eghball, B., D.H. Sander and J. Skopp. 1995. Phosphorus rate affects phosphorus movement. Fluid J. Fall issue.

Eghball, B., G.D. Binford, and D.D. Baltensperger. 1996. Phosphorus movement and adsorption in a soil receiving long-term manure and fertilizer application. J. Environ. Qual. 25:1339–1343.

Eghball, B., and J. F. Power. 1999. Phosphorus- and nitrogen-based manure and compost applications: corn production and soil phosphorus. Soil Sci. Soc. Am. J. 63:895-901.

Elias-Azar, K., A.E. Laag, and P.F. Pratt. 1980. Bicarbonate-extractable phosphorus in fresh and composted dairy manures. Soil Science Society of America Journal, 44: 435-437.

Erich, M. S., Fitzgerald, C. B., and Porter, G. A. 2002. The effect of organic amendments on phosphorus chemistry in a potato cropping system. Agric. Ecosyst. Envi. 88:79-88.

Evans, S.D., P.R. Goodrich, R.C. Munter, and R.E. Smith. 1977. Effects of solid and liquid beef manure and liquid hog manure on soil characteristics and on growth, yield, and composition of corn. J. Environ. Qual. 6:361–368.

FAO Grassland Index. [Online]. Available at http://www.tropicalforages.info/key/Forages/Media/Html/index.htm. (accessed Feb 2005; verified Dec. 2006). Food and Agriculture Organization of the United Nations, Rome, Italy.

FAO. 1996. Irrigation scheduling: From theory to practice. Proceedings of the ICID/FAO Workshop on Irrigation Scheduling, Rome, Italy, 12-13 September 1995, FAO Water Reports 8, Food and Agriculture Organization of the United Nations, Rome, Italy.

Federer, W.T. 1956. Augmented (or hoonuiaku) designs. Hawai'i. Plant. Rec. 2:191–208.

Felhendler, R., Shainberg, I., and Frenkel, H. 1974. Dispersion and hydraulic conductivity of soils mixed solution. Trans 10th Int. Cong. Soil Sci., Moscow, vol. I. Nauka, Moscow, pp 103- 112.

Ferguson, R. B., J. A. Nienaber, R. A. Eigenberg and B. L. Woodbury. 2005. Long-term effects of sustained beef feedlot manure application on soil nutrients, corn silage yield, and nutrient uptake. J. Environ. Qual. 34:1672-1681.

Fernando, W.A.R.N., K. Xia, and C.W. Rice. 2005. Sorption and desorption of ammonium from liquid swine waste in soils. Soil Sci. Soc. Am. J. 69:1057-1065.

Field, J. A., R. B. Reneau, Jr., W. Kroontje, and J.S. Caldwell. 1986. Utilization of anaerobically digested poultry manure effluent nitrogen as fertilizer. Trans. ASAE 29(1):223-228.

Fine, P., A. Hass, R. Prost, and N. Atzmon. 2002. Organic carbon leaching from effluent irrigated lysimeters as affected by residence time. Soil Sci. Soc. Am. J. 66:1531-1539.

Fisher, M.J., and P.C. Kerridge. 1996. The agronomy and physiology of Brachiaria species. In CIAT and Embrapa. Brachiaria: Biology, agronomy and improvement. p. 43-52. CIAT, Columbia.

FLEPPC. 1998. List of Florida's invasive species. [Online]. Available at http://www.fleppc.org/ID_book/Pennisetum%20purpureum.pdf. (accessed May 2005; verified Dec. 2006). Florida Exotic Pest Plant Council, Miami, FL.

Foote, D.E., E.L. Hill, S. Nakamura, and F. Stephens. 1972. Soil Survey of the Islands of Kauai, Oahu, Maui, Molokai, and Lanai, State of Hawai'i, Soil Conservation Service, United States Department of Agriculture.

Fordham, A.W., and U. Schwertmann. 1977. Composition and reactions of liquid manure (Gulle), with particular reference to phosphate: II. Solid phase components. J. Environ. Qual. 6: 136–140.

Fox, R.L. 1969. Fertilization of volcanic ash soil in Hawaii. p. C6.1–6.13. In Panel on soil derived from volcanic ash in Latin America. Int. Inst. of Agric. Sci., Turrialba, Costa Rica.

Fox, R.L. 1981. External phosphorus requirements of crops. Hawaii J. Exp. Stn. J. series no. 2493. In R.H. Dowdy et al. (eds.) Chemistry in the soil environment. ASA Special Publication No. 40, ASA-SSSA, Madison, WI.

Fox, R.L. and E.J. Kamprath. 1970. Phosphate sorption isotherms for evaluating the phosphate requirements of soils. Soil Sci. Soc. Am. Proc. 34:902–906.

Fox, R.L. and P.G.E. Searle. 1978. Phosphate adsorption by soils of the tropics: The diversity of soils of the tropics. ASA Special Publication No. 34. American Society of Agronomy, Madison, WI.

Franzen, D. 2003. Managing saline soils in North Dakota. [Online]. Available at http://www.ext.nodak.edu/extpubs/plantsci/soilfert/sf1087-1.htm. (accessed 20 Jan. 2007). North Dakota State University Extension Service, Fargo, ND.

Freeman J.S., Rowell D.L. The adsorption and precipitation of phosphate onto calcite. J. Soil Sci. 1981;32:75-84.

Frenkel, H., J.O. Goertzen, J.D. Rhoades. 1978. Effect of clay type and content, ESP and electrolyte concentration on clay dispersion and soil hydraulic conductivity. Soil Sci. Soc. Am. J. 42:32-39.

Freund, R.J., and R.C. Littell. 1991. SAS System for Regression. 2nd ed. SAS Inst., Cary, NC.

Frossard, F., M. Brossard, M.J. Hedley, and A. Metherell. 1995. Reactions controlling the cycling of P in soils. p. 107–137. In H. Tiessen (ed.) Phosphorus in the global environment: Transfers, cycles and management. John Wiley & Sons, New York.

Frossard, E., J.W.B. Stewart, and R.J. St Arnaud, 1989. Distribution and mobility of phosphorus in grassland and forest soils of Saskatchewan. Can. J. Soil Sci. 69:401–416.

Fukumoto, G.K., and C.N. Lee. 2003. Stargrass for forage. Cooperative Extension Service, CTAHR, University of Hawai‘i at Manoa, Honolulu, HI.

Fukumoto, G.K., and C.N. Lee. 2002 (Rev. 2003). Signalgrass for forage. Cooperative Extension Service, CTAHR, University of Hawai‘i at Manoa.

Furoc, R.E., and E.Q. Javier. 1976. Integration of fodder production with intensive cropping involving rice. 1. Grass production with irrigated lowland rice field. 2. Herbage weeds during the juvenile stage of the rice crop. Phil. J. Crop Sci. 1: 146-148.

Gagnon, B., and R.R. Simard. 1999. Nitrogen and phosphorus release from on-farm and industrial composts. Can. J. Soil Sci. 79:481–489.

Garay, A.H., L. E. Sollenberger D. C. McDonald, G. J. Ruegsegger, R. S. Kalmbacher, and P. Mislevy. 2004. Nitrogen fertilization and stocking rate affect stargrass pasture and cattle performance. Crop Sci. 44:1348-1354.

Gentry, C.E., and R.B. Willis. 1988. Improved method for automated determination of ammonium in soil extracts. Commun. Soil Sci. Plant Anal. 19:721–737.

Geohring, L.D., O.V. McHugh, M.T. Walter, T.S. Steenhuis, M.S. Akhtar, and M.F. Walter. 2001. Phosphorus transport into subsurface drains by macropores after manure applications: implications for best manure management practices. Soil Sci. 12:896-909.

Gerritse, R.G. 1977. Phosphorus compounds in pig slurry and their retention in the soil. p. 257–266. In Proc. EEC Seminar on Utilization of Manure by Land Spreading. Publ. EUR 5672e. Commission of the European Communities, Brussels.

Gieskes, T., and D. Hackett. 2003. Hawai'i ethanol alternatives. Dept. of Business, Economic Development and Tourism. Strategic Industries Division. Honolulu: Dept. of Business, Economic Development and Tourism, State of Hawai'i, 47 p.

Golden, B.R., N.A. Slaton, R.J. Norman, E.E. Gbur, K.R. Brye, and R.E. DeLong. 2006. Recovery of nitrogen in fresh and pelletized poultry litter by rice. Soil Sci. Soc. Am. J. 70:1359-1369.

Gooddy, D.C., P. Shand, D.G. Kinniburgh, and W.H. Van-Riemsdijk. 1995. Field-based partition coefficients for trace elements in soil solutions. Eur. J. Soil Sci. 46:265–285.

Gough, J.D. and J.C. Ward. 1996. Environmental decision making and lake management. J. Env. Management 48(1):1-15.

Gross, C.M., J.S. Angle, and M.S. Welterlen. 1990. Nutrient and sediment losses from turfgrass. J. Environ. Qual. 19: 663-668.

Grossl, P.R., and W.P. Inskeep. 1991. Precipitation of dicalcium phosphate dihydrate in the presence of organic acids. Soil Sci. Soc. Am. J. 55:670–675.

Guo, F., R.S. Yost, N.V. Hue, C.I. Evensen and J.A. Silva. 2000. Changes in phosphorus fractions in soils under intensive plant growth. Soil Sci. Soc. Am. J. 64:1681-1689.

Habib, L. S. Hayfa, and J. Ryan. 1994. Phosphorus solubility changes with time in organically amended soil in a Mediterranean environment. Commun. soil sci. plant anal. 24(19-20): 3281-3290.

Habteselassie, M.Y., B.E. Miller, S.G. Thacker, J. M. Stark and J.M. Norton. 2006. Soil nitrogen and nutrient dynamics after repeated application of treated dairy-waste. Soil Sci. Soc. Am. J. 70:1328–1337

Halim, R.A., D.R. Buxton, J.J. Hattendorf, and R.E. Carlson. 1989. Water-stress effects on alfalfa quality after adjustment for maturity differences. Agron. J. 81:189-194.

Halim, R.A., and A.A. Hamid.1989. Flooding tolerance of four forage grasses. In Ahmed et al. (ed.). Modernization in tropical livestock and poultry production. 12th MSAP Annual Conference. p. 296–300.

Halim R.A. 1993. Feeding strategies for smallholder beef production. In C.P. Chen and C. Satjianon (eds.) Strategies for suitable forage-based livestock production in South East Asia. Proceedings of the Third Meeting of Regional Working Group on Grazing and Feed Resources of Southeast Asia, held at Khon Kaen, Thailand, 31 January-6 February 1993. Department of Livestock Development and Food and Agriculture Organization of the United Nations, Rome, Italy. pp. 221-230.

Hall, D. W. 1978. The grasses of Florida. Ph.D. Dissertation. The University of Florida, Tampa, FL.

Halliwell D.J., K.M. Barlow, and D.M. Nash. 2001. A review of the effects of wastewater sodium on soil properties and their implications for irrigation systems. Aust. J. Soil Res. 39:1259-1267.

Han, N., and M.L. Thompson. 1999. Copper-binding ability of dissolved organic matter derived from anaerobically digested biosolids. J. Environ. Qual. 28:939–944.

Hannam, R.J., and K. Ohki. 1988. Detection of manganese deficiency and toxicity in plants. p. 243–259. In R.D. Graham et al. (ed.) Manganese in soils and plants. Kluwer Acad. Publ., Dordrecht, the Netherlands.

Hannapel, R.J., Fuller, W.H., Fox, R.H., 1964a. Phosphorus movement in a calcareous soil: II. Soil microbial activity and organic phosphorus movement. Soil Sci. 97:421–427.

Hannapel, R.J., Fuller, W.H., Fox, R.H., Bosma, S., Bullock, J.S., 1964b. Phosphorus movement in a calcareous soil: I. Predominance of organic forms of phosphorus in phosphorus movement. Soil Sci. 97:350–357.

Handley, L.L., and P.C. Ekern. 1984. Effluent irrigation of para grass: Water, nitrogen, and biomass budgets. Water Resour. Bull. 20:669–677.

Handley, L. L., and P. C. Ekern. 1981. Irrigation of California grass with domestic sewage effluent: water and nitrogen budgets and crop productivity. WRRC Technical Report No. 141, Water Resources Research Center, University of Hawai'i at Manoa, Honolulu, HI.

Handreck KA. 1991. Interactions between iron and phosphorus in the nutrition of Banksia ericifolia L. f. var. ericifolia (Proteaceae) in soil-less potting media. Aust. J. Bot. 39: 373-384.

Hansen, N.C., T.C. Daniel, A.N. Sharpley, and J.L. Lemunyon. 2002. The fate and transport of phosphorus in agricultural systems. J. Soil Water Conserv. 57:408–417.

Hansen, J.C., and D.G. Strawn. 2003. Kinetics of phosphorus release from manure-amended alkaline soil. Soil Sci. 168:869–879.

Hansen, J.C., B.J. Cade-Menun, and D.G. Strawn. 2004. Phosphorus speciation in manure-amended alkaline soils. J. Environ. Qual. 33:1521–1527.

Hao, S. 2006. Higher milk prices sought. The Honolulu Advertiser. [Online]. Available at http://the.honoluluadvertiser.com/article/2006/Nov/28/bz/FP611280312.html. (Accessed Feb. 9, 2007), Honolulu, HI.

Harding, W. A. T. and B. Grof. 1978. Effect of fertilizer nitrogen on yield, nitrogen content and animal productivity of Brachiaria decumbens cv. 'Basilisk' on the wet tropical coast of North Queensland. Queensland J. Agric. Anim. Sci., 35: 11-22.

Hare, M.D., P. Booncharern, P. Tatsapong, K. Wongpichet, C. Kaewkunya, nd K. Thummasaeng. 1999a. Performance of para grass (Brachiaria mutica) and Ubon paspalum (Paspalum atratum) on seasonally wet soils in Thailand. Trop. Grasslds. 33: 75-81.

Hare, M.D., W. Suriyantratong, P. Tatsapong, C. Kaewkunya, K. Wongpichet, and K. Thummasaeng. 1999b. Effect of nitrogen on production of Paspalum atratum on seasonally wet soils in north-east Thailand. Trop. Grasslds. 33: 207-213.

Hare, M.D., K. Thummasaeng, W. Suriyajantratong, K. Wongpichet, M. Saengkham, P. Tatsapong, C. Kaekunya, and P. Booncharern. 1999c. Pasture grass and legume evaluation on seasonally waterlogged and seasonally dry soil in northeast Thailand. Trop. Grasslds. 33:65-74.

Hare, M.D., W. Suriyajantratong, K. Wongpichet, and K. Thummasaeng. 2001a. Paspalum atratum - from a wild native plant in Brazil to commercial forage seed production in Thailand in 10 years. International Herbage Seed Production Research Group Newsletter, 33: 5-8.

Hare, M.D., M. Saengkham, C. Kaewkunya, S. Tudsri, W. Suriyajantratong, K. Thummasaeng, and K. Wongpichet. 2001b. Effect of cutting on yield and quality of Paspalum atratum in Thailand. Trop. Grasslds. 35: 144-150.

Harlan, J.R. 1976. Tropical and subtropical grasses. In M.W. Simmonds (ed.). Evolution of crop plants. Longman Group, London. 19:142-144.

Harley, C.P., and R.C. Lindner. 1945. Observed response of apple and pear trees to some irrigation waters of north central Washington.Proc. Am. Soc. Hort. Sci. 46:35-44.

Harris, B. Jr., A. L. Adams, and H. H. Van Horn. 1994. Mineral needs of dairy cattle. Circular 468, Florida Cooperative Extension Service, Institute of Food and Agricultural Sciences, University of Florida, FL.

Hart, M., B. F. Quin, and M. L. Nguyen. 2004. Phosphorus runoff from agricultural land and direct fertilizer effects: a review J. Environ. Qual. 33: 1954-1972.

Havlin, J. L., and J. D. Beaton. 1998. Soil fertility and fertilizers, 6th ed. Prentice Hall, Upper Saddle River, NJ.

Haygarth, P.M., and A.N. Sharpley. 2000. Terminology for phosphorus transfer. J. Environ. Qual. 29:10–15.

Haynes, R.J., and P.H. Williams. 1993. Nutrient cycling and soil fertility in the grazed pasture ecosystem. Adv. Agron. 49:119–199.

Haynes, R.J., and M.S. Mokolobate. 2001. Amelioration of Al toxicity and P deficiency in acid soils by additions of organic residues: a critical review of the phenomenon and the mechanisms involved. Nutr. Cycling Agroecosyst. 59:47-63.

He, Z., and C.W. Honeycutt. 2001. Enzymatic characterization of organic phosphorus in animal manure. J. Environ. Qual. 30:1685–1692.

He, Z., T.S. Griffin, and C.W. Honeycutt. 2004. Phosphorus distribution in dairy manures. J. Environ. Qual. 33:1528–1534.

Heanes, D.L. 1984. Determination of total organic-C in soils by an improved chromic acid digestion and spectroscopic procedure. Commun. Soil Sci. Plant Anal. 15:1191–1213.

Heckrath, G., P.C. Brookes, P.R. Poulton, and K.W.T. Goulding. 1995. Phosphorus leaching from soils containing different phosphorus concentrations in the Broadbalk experiment. J. Environ. Qual. 24:904–910.

Hernandez-Daumas, S. 2000. Carbon and Nitrogen Cycling in a Tree-grass intercropping system in the humid tropics of Mexico. Ph. D. Dissertation, University of Edinburgh, Germany.

Henriksen A., and A. R. Selmer-Olsen. 1970. Automatic methods for determining nitrate and nitrite in water and soil extracts. Analyst 95:514-518.

Holford, I.C.R. 1997. Soil phosphorus: its measurement, and its uptake by plants. Aust. J. Soil Res. 35: 227–239.

Holford, I.C. and G.E.G. Mattingly. 1975. The high and low-energy phosphate absorbing surfaces in calcareous soils. The J. Soil Sci. 26: 407-417.

Holford, I.C.R., and W.H. Patrick, Jr. 1981. Effects of duration of anaerobiosis and reoxidation on phosphate sorption characteristics of an acid soil. Aust. J. Soil Res. 19:69–78.

Holford, I.C.R., C. Hird, and R. Lawrie. 1997. Effects of animal effluents on the phosphorus sorption characteristics of soils. Aust. J. Soil Res. 35:365–373.

Holm, L. G., D. L. Plucknett, J. V. Pancho, J. P. Herberger. 1977. The world's worst weeds: distribution and biology. University Press of Hawai'i, Honolulu, HI, 609 pp.

Hopkins, B., and J. Ellsworth. 2003. Phosphorus nutrition in potato production. Paper presented during the Idaho Potato Conference, Jan. 22-23, University of Idaho. [Online]. Available at http://www.ag.uidaho.edu/potato/research/files/Volume%2035/Vol%2035_75%20Phosphorus%20Nutrition.pdf. (accessed Mar. 2005; verified Dec. 2006), University of Idaho, ID.

Hountin, J. A., D. Couillard, and A. Karam. 1997. Soil carbon, nitrogen and phosphorus contents in maize plots after 14 years of pig slurry applications. The J. Agric. Sci. 129:187-191.

Huang, P. M., and A. Violante. 1986. Influence of organic acids on crystallization and surface properties of precipitation products of aluminum. In P. M. Huang and M. Schnitzer (eds.). Interaction of soil minerals with natural organics and microbes. Soil Sci. Soc. Amer. Spec. Publ. 17:549-592.

Hue, N.V. 1988. A possible mechanism for manganese phytotoxicity in Hawaii soils amended with a low-manganese sewage sludge. J. Environ. Qual. 17:473-479.

Hue, N. V. 1991. Effects of organic acids/anions on P sorption and phytoavailability in soils with different mineralogies. Soil Sci. 152:463-471.

Hue, N.V., H. Ikawa, and J.A. Silva. 1994. Increasing plant-available phosphorus in an Ultisol with a yard-waste compost. Commun. Soil Sci. Plant Anal. 25(19&20): 3291-3303.

Hue, N.V., R. Uchida, and M.C. Ho. 1997. Sampling and analysis of soils and plant tissues: how to make representative samples, how the samples are tested. In J.A. Silva and R. S. Uchida (eds.) Plant nutrient management in Hawaii's soils: approaches for tropical and subtropical agriculture. College of Tropical Agriculture and Human Resources, University of Hawai'i at Manoa, Honolulu, HI.

Humphreys, L.R., and I. Partridge. 1995. A guide to better pastures for the tropics and subtropics. 5th ed. [Online]. Excerpts available at http://www.tropicalgrasslands.asn.au/pastures/atra.htm. (revised 15 Jan. 2003; accessed 15 Feb. 2005; verified 31 Mar. 2006). Continuing Education, C.B. Alexander Agricultural College, New South Wales, Australia.

Humphrey, L.R. 1999. Forage utilization. In L. Falvey and C. Chantalakhana (eds.). Smallholder dairying in the tropics. International Livestock Research Institute (ILRI), Nairobi, Kenya. pp. 185-206.

Hutchison, K.J., and D. Hesterberg. 2004. Dissolution of phosphate in a phosphorus-enriched ultisol as affected by microbial reduction. J. Environ. Qual. 33:1793–1802.

Idris, A.B., and M. Najib. 2004. Role of silage from fodder, crop residues and agro-industrial by-products in commercial livestock production. In Proc. Regional Working Group on Grazing and Feed Resources for Southeast Asia, 8th, Kuala Lumpur, Malaysia. 22–28 Sept. 2003. Malaysian Agricultural Research and Development Institute, Kuala Lumpur, Malaysia.

Inskeep, W.P., and J.C. Silvertooth. 1988. Kinetics of hydroxyapatite precipitation at pH 7.4 and 8.4. Geochim. Cosmochim. Acta 52:1883-1893.

Iyamuremye, F., R.P. Dick, J. Baham. 1996. Organic amendments and phosphorus dynamics: I. phosphorus chemistry and sorption. Soil Sci. 161:426-435.

Jackman J.M., Jones R.C., Yost R.S., Babcock C.J. Rietveld estimates of mineral percentages to predict phosphorus sorption by selected Hawaiian soils. Soil Sci. Soc. Am. J. 1997;61:618-629.

Jensen, M.B., H.C.B. Hansen, N.E. Nielsen, and J. Magid. 1998. Phosphate mobilization and immobilization in two soils incubated under simulated reducing conditions. Acta Agric. Scand. Sect. B 48:11–17.

Johnson, A.F., D.M. Vietor, F.M. Rouquette, Jr., V.A. Haby, and M.L. Wolfe. 1995. Estimating probabilities of nitrogen and phosphorus loss from animal waste application. p. 411–418. In K. Steele (ed.) Animal waste and the land–water interface. Lewis Publ., New York.

Johnson, A.F., D.M. Vietor, F.M. Rouquette, Jr., and V.A. Haby. 2004. Fate of phosphorus in dairy wastewater and poultry litter applied on grassland. J. Environ. Qual. 33:735–739.

Jones, C.A., and J.S. Jacobsen. 2002. Effects of dust control coatings on P dissolution and uptake. In A.J. Schlegel (ed.) Proceedings of the Great Plains Soil Fertility Conference. Vol 9. Denver, Colorado. March 5-6.

Jones, J.W., G.Y. Tsuji, G. Hoogenboom, L.A. Hunt, P.K. Thornton, P.W. Wilkens,D.T. Imamure, W.T. Bowen and U. Singh. 1998. Decision support system for agrotechnology transfer: DSSAT v3. In G.Y. Tsuji et al. (eds.), Understanding options for agricultural production. Kluwer Academic Publishers, Boston. pp. 157-177.

Josan, M.S., V.D. Nair, W.G. Harris, and D. Herrera. 2005. Associated release of magnesium and phosphorus from active and abandoned dairy soils. J. Environ. Qual. 34:184–191.

Kalmbacher R.S., W.F. Brown, D.L. Colvin, L.S. Dunavin, A.E.Kretschmer, Jr., F.G., Martin J.J. Mullahey, and J.E. Rechcigl. 1997a. Suerte 'Atra' paspalum: its management and utilization. Florida Agric. Exp. Stn. Circular S-397, University of Florida, Gainsville, FL.

Kalmbacher R.S., F.G. Martin, and A.E. Kretschmer, Jr. 1997b. Performance of cattle grazing pastures based on Paspalum atratum cv. Suerte. Trop. Grasslds. 31:58-66.

Khiari, L., L.E. Parent, A. Pellerin, A.R.A. Alimi, C. Tremblay, R.R. Simard, and J. Fortin. 2000. An agri-environmental phosphorus saturation index for acid coarse-textured soils. J. Environ. Qual. 29:1561–1567.

King, L.D., P.W. Westerman, G.A. Cummings, M.R. Overcash, and J.C. Burns. 1985. Swine lagoon effluent applied to 'Coastal Bermudagrass: II. Effects on soil. J. Environ. Qual. 14(1):14-21.

Kingery, W.L., C.W. Wood, D.P. Delaney, J.C. Williams, and G.L. Mullins. 1994. Impact of long-term land application of broiler litter on environmentally related soil properties. J. Environ. Qual. 23:139–147.

Kinniburgh, D.G., J.K. Syers, and M.L. Jackson. 1975. Specific adsorption of trace amounts of calcium and strontium by hydrous oxides of iron and aluminum. Soil Sci. Soc. Am. Proc. 39:464–470.

Kleinman, P.J.A., R.B. Bryant, and W.S. Reid. 1999. Development of pedotransfer functions to quantify phosphorus saturations of agricultural soils. J. Environ. Qual. 28:2026–2030.

Kleinman, P.J.A., and A.N. Sharpley. 2002. Estimating phosphorus sorption saturation from Mehlich-3 data. Commun. Soil Sci. Plant Anal. 33:1825–1839.

Kohl, K. 2004. A valuable resource. [Online]. Available at http://www.extension.iastate.edu/pages/communications/epc/F97/valresource. html. (Accessed Dec. 2006; verified Jan. 2007), Iowa State University, Ames, IA.

Koster H.H., H. Meissner, R.J. Coertze. 1992. Variation in the production and quality of bana grass over the growing season using hand-clipped samples, South Africa. J. Anim. Sci. 22:31-34.

Kotuby-Amacher, J., R. Koenig, and B. Kitchen. 1997. Salinity and plant tolerance. [Online]. Available at http://extension.usu.edu/files/agpubs/salini.htm. (accessed 20 Feb. 2006; verified 31 Mar. 2006). AG-SO-03, Cooperative Extension Service, Utah State University, Logan, UT.

Kurunc, A., R. P. Flynn, and A. L. Ulery. 2004. Dairy lagoon effluent effects on soil chemical properties, corn yield and nutrient uptake. J. Agron. 3(3): 219-222.

Langeland, K. A. and Burks, K.C. (eds.) 1994. Identification and biology of bon-native plants in Florida's natural areas. [Online]. Available at http://www.fleppc.org/ID_book.htm. (accessed Feb. 2005; verified Dec. 2006). UF/IFAS Distribution, Gainesville, FL.

Laboski, C. A. M., and J. A. Lamb. 2004. Impact of manure application on soil phosphorus sorption characteristics and subsequent water quality implications. Soil Sci. 169(6):440-448.

Lado, M., M. Ben-Hur, and S. Assouline. 2005. Effects of effluent irrigation on seal formation, infiltration, and soil loss during rainfall. Soil Sci. Soc. Am. J. 69:1432–1439.

Lasat, M.M. 2000. Phytoextraction of metals from contaminated soil: a review of plant/soil/metal interaction and assessment of pertinent agronomic issues. J. Hazardous Substance Res. 5(1):10-15.

Latterell J.J., R.H. Dowdy, C.E. Clapp, W.E. Larson, D.R. Linden. 1982. Distribution of phosphorus in soils irrigated with municipal wastewater effluent: a five-year study. J. Environ. Qual. 11:124-28.

Lebron, L., D.L. Suares, and T. Yoshida. 2002. Gypsum effects on the aggregate size and geometry of three sodic soils under reclamation. Soil Sci. Soc. Am. J. 66:92–98.

Leytem, A.B., R.L. Mikkelsen, and J.W. Gilliam. 2002. Sorption of organic phosphorus compounds in Atlantic coastal plain soils. Soil Sci. 167:652–658.

Li, G. C., R. L. Mahler, and D. O Everson. 1990. Effects of plant residues and environmental factors on phosphorus availability in soils. Commun. Soil Sci. Plant Anal. 21:471-491.

Lindsay, W.L. 1979. Chemical equilibria in soils. John Wiley and Sons, Inc., New York.

Lindsay, W.L. 1984. Soil and plant relationships associated with iron deficiency with emphasis on nutrient interactions. J. Plant Nutr. 7:489–500.

Lindsay, W.L. 1991. Inorganic equilibria affecting micronutrients in soils. In J.J. Mortvedt et al. (ed.), Micronutrients in Agriculture, 2nd ed. p. 89–144. Soil Science Society of America, Madison, Wisconsin, USA.

Linquist, B.A., P.W. Singleton, R.S. Yost, and K.G. Cassman. 1997. Aggregate size effects on the sorption and release of phosphorus in an Ultisol. Soil Sci. Soc. Am. J. 61:160–166.

Littell R., W. Stroup, and R. Freund. 2004. SAS for Linear Models, 4th ed. SAS Publishing, Cary, N.C.

Loneragan, J.F., T.S. Grove, A.D. Robson, and K. Snowball. 1979. Phosphorus toxicity as a factor in zinc-phosphorus interactions in plants. Soil Sci. Soc. Am. J. 43:966-972.

Loneragan, J.F., and M.J. Webb. 1993. Interactions between zinc and other nutrients affecting the growth of plants. In A.D. Robson (ed.) Zinc in Soils and Plants, Ch. 9, Kluwer Academic Publishers, Dordrecht. pp. 119-134.

Maas, E.V. and Hoffman, G.J. 1977. Crop salt tolerance—current assessment. J. Irrig. and Drainage Div., ASCE 103 (IR2):115-134.

MacDonald, T. 2000. The effects of excessive liquid hog maaure applications on phosphorus concentrations in soil and surface runoff from corn and forage crops. M. Sc. Thesis, McGill University, Montreal, Canada.

Macoon, B., K.R. Woodard, L.E. Sollenberger, E.C. French, III, K.M. Portier, D.A. Graetz, G.M. Prine, and H.H. Van Horn, Jr. 2002. Dairy effluent effects on herbage yield and nutritive value of forage cropping systems. Agron. J. 94:1043–1049.

Maguire, R.O., J.T. Sims, and F.J. Coale. 2000. Phosphorus fractionation in biosolids-amended soils: Relationship to soluble and desorbable phosphorus. Soil Sci. Soc. Am. J. 64:2018–2024.

Mallarino, A.P. 1997. Interpretation of soil phosphorus tests for corn in soils with varying pH and calcium carbonate content. J. Prod. Agric. 10:163–167.

Mamedov, A.I., I. Shainberg, and G.J. Levy. 2000. Irrigation with effluent water: Effects of rainfall energy on soil infiltration. Soil Sci. Soc. Am. J. 64:732-737.

Marschner H. 1995 Mineral nutrition of higher plants, 2nd ed. Academic Press, New York, USA.

Mathers, A.C., and B.A. Stewart. 1974. Corn silage yield and soil chemical properties as affected by cattle feedlot manure. J. Environ. Qual. 3:143–147.

Mathews, B.W., J.R. Carpenter, L.E. Sollenberger and S. Tsang. 2005. Phosphorus in Hawaiian kikuyugrass pastures and potential phosphorus release to water. J Environ. Qual. 34:1214-1223.

Maule, A. 2000. Survival of verocytotoxigenic Escherichia coli in soil, water and on surfaces. J. Applied Microbiol. Sympo. Supp. 88:71S-78S.

McDowell, R., A. Sharpley, P. Brookes, and P. Poulton. 2001. Relationship between soil test phosphorus and phosphorus release to solution. Soil Sci. 166:137–149.

McDowell, R.W., and A.N. Sharpley. 2001. Approximating phosphorus release from soils to surface runoff and subsurface drainage. J. Environ. Qual. 30:508–520.

McIntyre, D.S. 1979. Exchangeable sodium, subplasticity and hydraulic conductivity of some Australian soils. Aust. J. Soil Res. 17:115–120.

Mclaughlin, M.R., Sistani, K.R., Fairbrother, T.E., Rowe, D.E. 2005. Overseeding common bermudagrass with cool-season annuals to increase hay yield and nitrogen and phosphorus uptake in a hay field fertilized with swine effluent. Agron. J. 97:487-493.

McLaughlin, M.R., T.E. Fairbrother and D.E. Rowe. 2004. Forage yield and nutrient uptake of warm-season annual grasses in a swine effluent spray field . Agron. J. 96(6):1516-1522.

McNeal, B.L., and N.T. Coleman. 1966. Effect of solution composition on soil hydraulic conductivity. Soil Sci. Soc. Am. Proc. 20:308-312.

McNeal, B.L., D.A. Layfield, W.A. Norvell and J.D. Rhoades. 1968. Factors influencing the hydraulic conductivity of soils in the presence of mixed salt solutions. Soil Sci. Soc. Am. J. 32:187-190.

Meiri, A., J. Kamburov, J. Shalhevet. Transpiration effects on leaching fractions. Agron. J. 69(5): 779-782.

Meléndez, N.F., M.J.A. Gonzalez, and J.P. Pérez. 1980. El pasto estrella Africana. (In Spanish, with English abstract) Bull. CA-7. Colegio Superior de Agricultura Tropical, Cárdenas, Tabasco, Mexico.

Miller, C.P. 1976. Growth and nitrogen uptake of irrigated grass-legume and nitrogen fertilized grass pasture in a Queensland tropical highland. J. Aust. Inst. Agr. Sci. 46:184.

Mills, H.A., and J. B. Jones Jr. 1996. Plant analysis handbook II: A practical sampling, preparation, analysis and interpretation guide. Micro Macro Publishing, Jefferson City, MO.

Mingelgrin, U., and J.W. Biggar. 1986. Copper species in aqueous sewage sludge extract. Water Air Soil Pollut. 28:351–359.

Minitab Inc., 2004. Minitab Release 14.13. College, PA, U.S.A.

Miranda, C.H.B., and R.M. Boddey. 1987. Estimation of biological nitrogen fixation associated with 11 ecotypes of Panicum maximum grown in nitrogen15-labeled soil. Agron. J. 79: 558-563.

Mislevy, P., and W.F. Brown. 1991. Management and utilization of complementary forages: Stargrass. 40th Florida Beef Cattle Short Course. Univ. of Fla., Gainesville.

Mislevy, P., W.F. Brown, R. Caro-Costas, J. Vicente-Chandler, L.S. Dunavin, D.W. Hall, R.S. Kalmbacher, A.J. Overman, O.C. Ruelke, R.M. Sonoda, A. Sotomayor-Rios, R.L. Stanley, Jr., and M.J. Williams. 1989. Florico Stargrass. Circular S-361, Florida Cooperative Extension Service, Institute of Food and Agricultural Sciences, University of Florida, Gainsville, FL.

Moraghan, J.T., and H.J. Mascagni. 1991. Environmental and soil factors affecting micronutrient deficiencies and toxicities. In J.J. Mortvedt et al. (eds.). Micronutrients in Agriculture, 2nd ed. SSSA Book Series No. 4, Soil Science Society of America, Madison, WI.

Mozaffari, M., and J.T. Sims. 1994. Phosphorus availability and sorption in an Atlantic Coastal Plain watershed dominated by animal-based agriculture. Soil Sci. 157:97–107.

Mugaas, R.J., M.L. Agnew, and N.E. Christians. 1997. Turfgrass management for protecting surface water quality. [Online]. Available at http://hubcap.clemson.edu/~blpprt/pasture/uptake.html, Clemson University, Clemson, SC.

Muir, J.P. 2001. Dairy compost, variety, and stand age effects on kenaf forage yield, nitrogen and phosphorus concentration, and uptake. Agron. J. 93:1169–1173.

Muller, Z.O. 1980. Feed from animal wastes: state of knowledge. FAO Animal Production and Health Paper No. 18, Food and Agriculture Organization of the United Nations, Rome, Italy.

Mullins, G., B. Joern and P. Moore. 2005. By-product phosphorus: sources, characteristics and management. In J.T. Sims and A.N. Sharpley (eds.) Phosphorus: Agriculture and the environment. Monograph no. 46, Amer. Society Agron., Crop Sci. Soc. Am., Soil Sci. Soc. Am., Madison, WI.

Muneer, M. and C.R. Lawrence. 2004. Phosphorus characteristics of soils treated with sewage effluent using land filtration at Werribee sewage treatment complex, Victoria [Online]. In B. Singh (ed.) SuperSoil 2004: Proc. Australian New Zealand Soils Conference, 3rd, 5–9 Dec. 2004. University of Sydney, Australia, Sydney, Australia. Available at www.regional.org.au/au/asssi/. (accessed 15 Aug. 2005, verified 31 Mar. 2006). The Regional Inst. Ltd., New South Wales, Australia.

Murphy, J., and J.P. Riley. 1962. A modified single solution method for the determination of phosphate in natural waters. Anal. Chim. Acta 27:31–36.

Nagarajah, S., A.M. Posner, and J.P. Quirk. 1970. Competitive adsorption of phosphate with polygalacturonate and other organic anions on kaolinite and oxide surfaces. Nature. 228:83-85.

Nahm, K. H. 2003. Bioavailability of phosphorus in poultry manure. Austr. J Exper. Agric. 44(8) 717–728.

Nair, V.D., D.A. Graetz, and K.R. Reddy. 1998. Dairy manure influences on phosphorus retention capacity of Spodosols. J. Environ. Qual. 27:522–527.

Nair, V.D., and D.A. Graetz. 2002. Phosphorus saturation in Spodosols impacted by manure. J. Environ. Qual. 31:1279–1285.

Nair, V.D., K.M. Portier, D.A. Graetz, and M.L. Walker. 2004. An environmental threshold for degree of phosphorus saturation in sandy soils. J. Environ. Qual. 33:107-113.

Nair V.D., D.A. Graetz, and K.M. Portier. 1995. Forms of phosphorus in soil profiles from dairies of South Florida. J. Environ. Qual. 59, 1244-1249.

Nair, V. D., D.A. Graetz and D. O. Dooley. 2003. Phosphorus release characteristics of manure and manure-impacted soils. Food, Agric. Environ. 1:217-223.

National Research Council. 2001. Nutrient requirements of dairy cattle. 7th Rev. ed. National Academy Press, Washington D.C.

National Academy of Press. 2001. Nutrient requirements of dairy cattle: 7th revised ed. Washington D.C.

Nelson, C.J., and L.E. Moser. 1994. Plant factors affecting forage quality. In Fahey, G.C., et al. (eds.). Forage quality, evaluation, and utilization. Proceedings of the National Conference on Forage Quality, Evaluation and Utilization held at the University of Nebraska, Lincoln on 13-15 Apr, ASA-CSSA-SSSA, Madison, WI.

Nelson, P.N., Baldock, J.A. Clarke, P., and Oades, J.M. 1999. Dispersed clay and organic matter in soil: their nature and associations. Aust. J. Soil Res. 37:289-315.

Nguyen, T.M., M.T. Vu, A.D. Chung, and T.L. Le. 2003. Improved grasses in commercial dairy production at peri-urban regions of 135 Hanoi, Hatay and Ho Chi Minh City. In Forages and Feed Resources in Commercial Livestock Production Systems, Proceedings of the 8th Meeting of The Regional Working Group on Grazing and Feed Resources for Southeast Asia, 22–28 September. Kuala Lumpur, Malaysia.

Nitis, I.M. 1999. Production of forage and fodder. In L. Falvey and C. Chantalakhana (eds.) Smallholder dairying in the tropics. International Livestock Research Institute (ILRI), Nairobi, Kenya. 462 pp.

Novak, J.M., and D.W. Watts. 2004. Increasing the phosphorus sorption capacity of southeastern coastal plain soils using water treatment residuals. Soil Sci. 169:206–214.

Novoa, L.G., and S. Rodriguez-Carrasquel. 1972. Estudio del comportamiento de los pastos para (Brachiaria mutica Stapf) y aleman (Echinochloa polystachya A.Bik. Hitch.). Agron. Trop. (Maracay) 22(6 Ser. 2):643–655.

Ohno, T. and B.S. Crannell. 1996. Green and animal manure-derived dissolved organic matter effects on phosphorus sorption. J. Environ. Qual. 25:1137-1143.

Ohno, T. and M.S. Erich. 1997. Inhibitory effects of crop residue-derived organic ligands on phosphate adsorption kinetics. J. Environ. Qual. 26:889-895.

Olsen, S.R., C.V. Cole, F.S. Watanabe, and L.A. Dean. 1954. Estimation of available phosphorus in soils by extraction with sodium bicarbonate. USDA Circ. 939. US Gov. Print. Office, Washington, D.C.

Olsen, S.R., and L.E. Sommers. 1982. Phosphorus. p. 403–427. In A.L. Page et al. (ed.) Methods of soil analysis. Part 2. 2nd ed. Agron. Monogr. No. 9. ASA and SSSA, Madison, WI.

Osmond, D.L., T.J. Smyth, R.S. Yost. D.L. Hoag, W.S. Reid, W. Branch, X. Wang and H. Li. 2002. Nutrient Management Support System (NuMaSS) v.2. Soil Management Collaborative research Support Program, Technical bulletin No. 2002-02, North Carolina State University, Raleigh, NC.

Oster, J.D., I. Shainberg, and J.D. Wood. 1980. Flocculation value and structure of Na/Ca montmorillonite and illite suspensions. Soil Sci. Soc. Am. J. 44:955-959.

Pavanasasivam V. 1973. Manganese studies in some soils with a high organic matter content. Plant Soil 38:245-257

Pant, H. K., P. Mislevy, and J. E. Rechcigl. 2004. Effects of phosphorus and potassium on forage nutritive value and quantity: environmental implications. Agron. J. 96:1299-1305.

Partridge I. 2003. Better pastures for the tropics and subtropics. [Online]. Available at http://www.tropicalgrasslands.asn.au/pastures/atra.htm. (accessed Feb., 2005; verified Dec. 2006), New South Wales, Australia.

Patrick, W.H. Jr., and A. Jugsujinda. 1992. Sequential reduction and oxidation of inorganic nitrogen, manganese, and iron in flooded soils. Soil Sci. Soc. Am. J. 56:1071–1073.

Pautler, M.C., and J.T. Sims. 2000. Relationships between soil test phosphorus, soluble phosphorus, and phosphorus saturation in Delaware soils. Soil Sci. Soc. Am. J. 64:765–773.

Pederson, G.A., G.E. Brink, and T.E. Fairbrother. 2002. Nutrient uptake in plant parts of sixteen forages fertilized with poultry litter: nitrogen, phosphorus, potassium, copper, and zinc. Agron. J. 94:895–904.

Peperzak, P., A.G. Caldwell, R.R. Hunziker, and C.A. Black. 1959. Phosphorus fractions in manures. Soil Sci. 87:293–302.

Phillips, I.R. 1998. Phosphorus availability and sorption under alternating waterlogged and drying conditions. Commun. Soil Sci. Plant Anal. 29:3045–3059.

Phillips, I.R. 2002. Phosphorus sorption and nitrogen transformation in two soils treated with piggery wastewater. Aust. J. Soil Sci 40(2):335–349.

Phoatong, S., and C. Phaikaew. 2001. Integration of forage crops in irrigated lowland rice in Thailand. In Proc. Regional Working Group on Grazing and Feed Resources for Southeast Asia, 7th, 2-7 July, 2001. Manado, North Sulawesi, Indonesia. FAO and University of Sam Rapulangi, North Sulawesi, Indonesia.

Pierzynski, G.M., and T.L. Logan. 1993. Crop, soil, and management effects on phosphorus soil test levels. J. Prod. Agric. 6:513–520.

Pierzynski, G.M., J.T. Sims, and G.F. Vance. 1994. Soils and environmental quality. Lewis Publ., Boca Raton, FL. 313 p.

Pierzynski, G.M., R.W. McDowell, and J.T. Sims. 2005. Chemistry, cycling, and potential movement of inorganic phosphorus in soils. p. 53-86. In J.T. Sims, and A.N. Sharpley (ed.) Phosphorus: Agriculture and the environment. ASA, CSSA, and SSSA, Madison, WI.

Pratt, P.F. 1984. Salinity, sodium, and potassium in an irrigated soil treated with bovine manure. Soil Sci. Soc. Am. J. 48:823-828.

Qian, P. and J. J. Schoenau. 2000. Fractionation of P in soil as influenced by a single addition of liquid swine manure. Can. J. of Soil Sci. 80: 561-566.

Rao, I. M., V. Borrero, J. Ricaurte, R. García, and M. A. Ayarza. 1996. Adaptive attributes of tropical forage species to acid soils. II. Differences in shoot and root growth responses to varying phosphorus supply and soil type. J. Plant Nutr. 19: 323-352.

Rasnake, M. 1996. Broiler litter production in Kentucky and potential use as a nutrient source. Publication AGR-168. Univ. Kentucky Coop. Extension Service, Lexington, KY.

Ravina, I. and Z. Markus. 1975. The effect of high exchangeable potassium percentage on soil properties and plant growth. Plant Soil 42(3): 661-672.

Redding, M.R. 2001. Pig effluent-P application can increase the risk of P transport: two case studies. Aust. J. Soil Res. 39, 169-174.

Reddy, K.R., M.R. Overcash, R. Khaleel and P.W. Westerman. 1980. Phosphorus adsorption-desorption characteristics of two soils utilized for disposal of animal wastes. J. Environ. Qual., 9(1): 86-92.

Reemtsma, T., A. Bredow, and M. Goehring. 1999. The nature and kinetics of organic matter release from soil by salt solutions. Eur. J. Soil Sci. 50:53–64.

Register of Australian Herbage Plant Cultivars. 1974. Brachiaria decumbens Stapf. (signalgrass) cv. Basilisk. J. Aust. Inst. Agric. Sci. 40 (1).

Register of Australian Herbage Plant Cultivars. 1990. 3rd ed. [Online]. Excerpts available at http://www.pi.csiro.au/ahpc/grasses/grasses.htm. (updated 21 Jan. 2004; accessed 15 Feb. 2005; verified 31 Mar. 2006). CSIRO Publ., Victoria, Australia.

Rengasamy, P., and K.A. Olsson. 1993. Irrigation and sodicity. Aust. J. Soil Res. 31: 821-837.

Rengasamy, P. 2002. Transient salinity and subsoil constraints to dryland farming in Australian sodic soils: an overview. Aust. J. Exp. Agric. 42(3) 351 – 361.

Reinhart, A. 2000. Assessment of and potential to reduce nitrate leaching in the Pearl Harbor watershed. Ph. D. Dissertation, University of Hawaii at Manoa, Honolulu, HI.

Reynolds, S.G. 1995. Pasture-cattle-coconut systems. FAO RAPA Publication 1995/7. FAO, Bangkok, Thailand, 668 pp.

Rhoades, J.D. 1974, Drainage for salinity control. In J. Van Schilfgaarde (ed) Drainage for agriculture. Amer. Soc. Agron. Monograph No. 17, pp 433–462.

Rhoades, J.D. and Merrill S.D. 1976, Assessing the suitability of water for irrigation: Theoretical and empirical approaches. In Prognosis of Salinity and Alkalinity. FAO Soils Bulletin 31. FAO, Rome. po. 69–110.

Rhoades, J.D., A. Kandiah, and A.M. Mashali. 1992. The use of saline waters for crop production, FAO irrigation and drainage paper 48, Food and Agriculture Organization of the United Nations, Rome, Italy.

Rietra, R. P. J. J., T. Hiemstra, and W. H. van Riemsdijk. 2001. Interaction between calcium and phosphate adsorption on goethite. Envi. Sci. Tech. 35, 3369-3374.

Roach C.G., R.D. Longhurst, and S.F. Ledgard. 2001. Land application of farm dairy effluent for sustainable dairy farming. Proceedings of the New Zealand Grasslands Association 63: 53-57.

Robert, F. J., and B.A. Carbon. 1969. Growth of tropical and temperate grasses and legumes under irrigation in southwest Australia. Trop. Grasslds. 3:103-116.

Roberts, O.T. 1970a. A review of pasture species in Fiji I. Grasses, Trop. Grasslds. 4:129-137.

Roberts, O.T. 1970b. Pasture improvement and research in Fiji. South Pac. Bull. 20:35-37.

Robinson, D.L. 1996. Fertilization and nutrient utilization in harvested forage systems–southern forage crops. p. 65–92. In R.E. Joost and C.A. Roberts (ed.) Nutrient cycling in forage systems. Proc. Symp., Columbia, MO. 7–8 Mar. 1996. Potash and Phosphate Inst., Manhattan, KS.

Robinson D.L., Eilers T.L. Phosphorus and potassium influences on annual ryegrass production. Louisiana Agric. 1996;39:10-11.

Robinson, J.S., and A.N. Sharpley. 1996. Reaction in soil of phosphorus released from poultry litter. Soil Sci. Soc. Am. J. 60:1583–1588.

Rowe, D.E., and T.E. Fairbrother. 2003. Harvesting winter forages to extract manure soil nutrients. Agron. J. 95:1209-1212.

Sah, R. N., and D. S. Mikkelsen. 1989. Phosphorus behavior in flooded-drained soils. I. Effects on phosphorus sorption. Soil Sci. Soc. Am. J. 53:1718-1722.

Sah, R.N., D.S. Mikkelsen, and A.A. Hafez. 1989. Phosphorus behavior in floodeddrained soils: II. Iron transformation and phosphorus sorption. Sci. Soc. Am. J. 53:1729-1732.

Sainty, G. R., and S. W. L. Jacobs. 1981. Water plants of New South Wales. Stanley Owens and Sons, Alexandria, New South Wales. 550 pp.

Salt, D.E., R.D. Smith, and I. Raskin. 1998. Phytoremediation. In L. Russel et al. (ed.) Annual review of plant physiology and plant molecular biology. p. 643–668. Annual Reviews, Palo Alto, CA.

Sample, E.C., R.J. Soper, and G.J.Racz.. 1980. Reactions of phosphate fertilizers in soils. In F.E. Khasawneh, E.C. Sample, E.J. Kamprath (eds.). The Role of Phosphorus in Agriculture. ASA, CSSA, and SSSA, Madison, WI.

Sanderson, M.A., R.M. Jones, M.J. McFarland, J. Stroup, R.L. Reed, and J.P. Muir. 2001. Nutrient movement and removal in a switchgrass biomass–filter strip system treated with dairy manure. J. Environ. Qual. 30:210-216.

Sanyal, S.K., S.K. De Datta, and P.Y. Chan. 1993. Phosphate sorption-desorption behavior of some acidic soils of south and Southeast Asia. Soil Sci. Soc. Am. J. 57:937–945.

Sarwar, M., J. L. Firkins, and M. L. Eastridge. 1992. Effects of varying forage or concentrate carbohydrates on nutrient digestibilities and milk production by dairy cows. J. Dairy Sci. 75:1533–1542.

SAS Institute. 2004. SAS/STAT user's guide. Version 9.1. SAS Inst., Cary, NC.

Satter, L. D., T. J. Kloppenstein, G. E. Erickson, and J. M. Powell., 2005, Phosphorus in dairy/beef nutrition. In J.T. Sims and A.N. Sharpley (eds.) Phosphorus: Agriculture and the environment. American Society of Agronomy Monograph Series No. 46. American Society of Agronomy, Madison, WI.

Schachtman, D.P., R.J. Reid, S. Ayling. 1998. Phosphorus uptake by plants: from soil to cell. Plant Physiol. 116:447-453.

Schroeder, J.W. 2004. Interpreting composition and determining market value: quality forage. North Dakota Extension Service Publication AS-1251, NDSU, Fargo, ND.

Shainberg I. 1984. The effect of electrolyte concentration on the hydraulic properties of sodic soils. In I. Shainberg and J. Shalhevet (eds.) Soil salinity under irrigation, Springer Verlag, New York. pp. 49-64.

Shainberg, I., R. Keren, N. Alperovitch, and D. Goldstein. 1987. Effect of exchangeable potassium on hydraulic conductivity of smectite-sand mixtures. Clays and Clay Min. 35(4):305-310.

Sharma,K., N. Lakshmi, K. Venugopalan, P. Mehta, A. Maheshwari and S. Bapna. 2003. X-ray diffraction: An approach to study interaction between cyanobacteria and dairy effluent. Curr. Sci. 85(9-10):1330-1334.

Sharpley, A.N., S.J. Smith, B.A. Stewart, and A.C. Mathers. 1984. Forms of phosphorus in soil receiving cattle feedlot waste. J. Environ. Qual. 13:211-215.

Sharpley, A.N.,and S.J.Smith. 1995. Nitrogen and phosphorus forms in soils receiving. manure. Soil Sci. 159(4):253-258.

Sharpley, A.N., S.J. Smith, and W.R. Bain. 1993. Nitrogen and phosphorus fate from long-term poultry litter applications to Oklahoma soils. Soil Sci. Soc. Am. J. 57:1131–1137.

Sharpley, A.N., and A.D. Halvorson. 1994. The management of soil phosphorus availability and its impact on surface water quality. p. 7–90. In R. Lai and B.A. Stewart (eds.) Soil processes and water quality. Lewis Publ., Boca Raton, FL.

Sharpley, A.N., and P.J.A. Withers. 1994. The environmentally sound management of agricultural phosphorus. Fert. Res. 39:133–146.

Sharpley A.N., S.C. Chapra, R. Wedepohl, J.T. Sims, T.C. Daniel, and K.R. Reddy. 1994. Managing agricultural phosphorus for protection of surface waters: issues and options. J. Environ. Qual. 23, 437-451.

Sharpley, A.N., and S. Rekolainen. 1997. Phosphorus in agriculture and its environmental implications. In H. Tunney, O.T. Carton, P.C. Brookes, A.E. Jonston, (eds.). Phosphorus loss from soil to water. CAB International, Wallingford, UK, pp. 1-53.

Sharpley, A.N. and I. Sisak. 1997. Differential availability of manure and inorganic sources of phosphorus in soil. Soil Sci. Soc. Am. J. 61:1503-1508.

Sharpley, A.N., and B. Moyer. 2000. Phosphorus forms in manure and compost and their release during simulated rainfall. J. Environ. Qual. 29:1462–1469.

Sharpley, A.N., R.W. McDowell, and P.J.A. Kleinman. 2004. Amounts, Forms, and Solubility of Phosphorus in Soils Receiving Manure Soil Sci. Soc. Am. J. 68: 2048-2057.

Siddique, M.T., J.S. Robinson, and B.J. Alloway. 2000. Phosphorus reactions and leaching potential in soils amended with sewage sludge. J. Environ. Qual. 29:1931–1938.

Siddique, M.T., and J.S. Robinson. 2004. Differences in phosphorus retention and release in soils amended with animal manures and sewage sludge. Soil Sci. Soc. Am. J. 68:1421–1428.

Simard R.R., D. Cluis, G. Gangbazo, and S. Beauchmin. 1995. Phosphorus status of forest and agricultural soils from a watershed of high animal density. J. Environ. Qual. 24, 1010-1017.

Simard, R.R., S. Beauchemin, and P.M. Haygarth. 2000. Potential for preferential pathways of phosphorus transport. J. Environ. Qual. 29:97–105.

Sims, J.T., R.R. Simard, and B.C. Joern. 1998. Phosphorus loss in agricultural drainage: Historical perspective and current research. J. Environ. Qual. 27:277–293.

Sims, J.T., and A.N. Sharpley (eds.) 2005. Phosphorus: Agriculture and the Environment. American Society of Agronomy Monograph Series No. 46. American Society of Agronomy, Madison, WI.

Singh, B.B., and M.A. Tabatabai. 1977. Effects of soil properties on phosphate sorption. Commun. Soil Sci. Plant Anal. 8:97–107.

Singh, B.B., and J.P. Jones. 1976. Phosphorus sorption and desorption characteristics of soil as affected by organic residues. Soil Sci. Soc. Am. J. 40:389-394.

Skerman, P.J., and F. Riveros. 1990. Tropical Grasses. FAO Plant Production and Protection Series No. 23. FAO, Rome, Italy, 832 pp.

Sleugh, B., B. Gilfillen, T. Willian, R. Stiles, and A. Canty. 2002. Uptake of nutrients by a cool season perennial and warm season annual grass. Amer. For. Grassld. Congress 77:193-197.

Smith, K.A., and S.W.F. Restall. 1971. The occurrence of ethylene in. anaerobic soil. J. Soil Sci., 22, 430-443.

So, H.B., and L.A.G. Aylmore. 1993. How do sodic soils behave? The effects of sodicity on soil behaviour. Aust. J. Soil Res. 31: 761-777.

Soil Survey Staff. 1972. Official Soil Series Description. [Online]. Available at http://soils.usda.gov/technical/classification/osd/. (accessed 15 Aug. 2005, verified 31 Mar. 2006). Natural Resources Conservation Service, United States Department of Agriculture.

Sotomayor Rios, A., A. Acosta, A. Matienzo, and J. Velez Fortuno. 1973. Evaluation of seven forage grasses at two cutting stages. J. Agr. U. Puerto Rico. 57(3):173–182.

Sotomayor-Rios A., J. Velez-Santiago, S. Torres-Rivera, and S. Silva.1976. Effect of three harvest intervals on yield and composition of nineteen forage grasses in the humid mountain region of Puerto Rico. J. Agric Univ. Puerto Rico. 60(3):294-309.

Sparrow, L.A., and N.C. Uren. 1987. Oxidation and reduction of Mn in acidic soils: Effect of temperature and soil pH. Soil Biol. Biochem. 19:143-148.

Stevenson, F.J., and A. Fitch. 1981. Reactions with organic matter. p. 69-96. In J.F. Loneragan et al. (ed.) Copper in soils and plants. Academic Press, Sydney, Australia.

Stevenson, F.J., and A. Fitch. 1981. Reactions with organic matter. p. 69–95. In J.F. Loneragan et al. (ed.) Copper in soils and plants. Academic Press, Sydney, Australia.

Stewart, J.W.B., and H. Tiessen. 1987. Dynamics of soil organic phosphorus. Biogeochem. 4:41–60.

Sumner, M.E. 1993. Sodic soils: new perspectives. Aust. J. Soil Res. 31: 683-750.

Swenson, R.M., C.V. Cole, and D.H. Sieling. 1949. Fixation of phosphate by iron and aluminum and replacement by organic and inorganic anions. Soil Sci. 67:3-22.

Swift, R.S. 1996. Organic matter characterization. In D.L. Sparks (ed.) Methods of soil analysis. Part 3. SSSA Book Ser. 5. p. 1011–1069. SSSA, Madison, WI.

Syers, J.K., and D. Curtin. 1989. Inorganic reactions controlling phosphorus cycling. In Tiessen, H. (ed.) Phosphorus cycles in terrestrial and aquatic ecosystems. Regional workshop 1: Europe. SCOPE/UNEP Proceedings, University of Saskatchewan, Saskatoon, Canada. pp. 17-29.

SYSTAT Software Inc. 2004. SigmaPlot Version 9.0. Point Richmond, CA, USA.

Tabbara, H. 2003. Phosphorus loss to runoff water twenty-four hours after application of liquid swine manure or fertilizer J. Environ. Qual. 32:1044-1052.

Taiz, L. and E. Zieger. 1998. Plant Physiology, 2nd Edition. Sinauer Associates, Inc, Massachusetts.

Takahashi, M., J.C. Moowaw, and J.C. Ripperton. 1966. Studies on napier grass. III. Grazing management. Hawai‘i Agric. Exp. Station Bull. 128.

Tamimi, Y. N., J.A. Silva, R.S. Yost and N.V. Hue. 1997. Adequate nutrient levels in soils and plants in Hawai'i (general guide). CTAHR publication AS-3, University of Hawai'i at Manoa, Honolulu, HI.

Tan, K.H. 1998. Principles of Soil Chemistry, 3rd ed. Switzerland: Marcel Dekker, Inc.

Tanji, K.K. 1990. Agricultural salinity assessment and management. ASCE Manual Reproductions of Engineering Practices 71, New York, NY.

Tanji, K.K. and N. C. Kielen. 2003. Agricultural drainage water management in arid and semi-arid areas. FAO Irrigation and Drainage Paper 61, Food and Agriculture Organization of the United Nations Rome, Italy.

Tanton, T.W., D.W. Rycroft, and F.M. Wilkinson. 1988. The leaching of salts from saline heavy clay soils: Factors affecting the leaching process. Soil Use Manage. 4(4).

Technicon, 1977. Technicon AutoAnalyzer II Industrial Method No 376-75. 1977. Digestion and sample preparation for the analysis of total Kjeldahl and/or total phosphorus in water samples using the Technicon BD-40 Block Digestor. Technicon Industrial Systems. Tarrytown, New York, USA.

The Honolulu Advertiser. 2004. [Online]. Available at http://the.honoluluadvertiser.com/article/2004/Oct/29/bz/bz04p.html. (accessed May 2005; verified Dec. 2006), Honolulu, HI.

Thomas, D., and B. Grof. 1986. Some pasture species for the tropical savannas of South America. III. *Andropogon gayanus* , *Brachiaria* species and *Panicum maximum*. Herbage Abs. 56:557–565.

Tiessen, H. 2005. Phosphorus dynamics in tropical soils. In J.T. Sims and A.N. Sharpley (eds.) Phosphorus: Agriculture and the environment. American Society of Agronomy Monograph Series No. 46. American Society of Agronomy, Madison, WI.

Tisdale, S., W.L. Nelson, and J.D. Beaton. 1985. Soil Fertility and Fertilizers 4th ed. NY: Macmillan Publishing Co.

Toor, G. S., L. M. Condron, H. J. Di, K. C. Cameron, B. J. Cade-Menun. 2003. Characterization of organic phosphorus in leachate from a grassland soil. Soil Biol. Biochem. 2003, 35, 1317-1323.

Toor, G.S., L.M. Condron, H.J. Di, K.C. Cameron, J.T. Sims. 2004. Impact of farm-dairy effluent application on the amounts and forms of phosphorus loss by leaching from irrigated grassland. New Zealand J. Agric. Res. 47: 479–490.

Toth, T., M. Kertesz, L.C. Guerra, J.L. Labrada, B. P. Machado, P.C. Fonseca, and M.N. Martínez. 1997. Plant composition of a pasture as a predictor of soil salinity. Revista de Biologia Tropical 45(4):1385-1393.

Toth, J.D., Z. Dou, J.D. Ferguson, D.T. Galligan, and C.F. Ramberg. 2006. Nitrogen- vs. phosphorus-based dairy manure applications to field crops. J. Environ. Qual. 35:2302-2312.

Toze, S. 2004. Reuse of effluent water–benefits and risks. Proc. Int. Crop Science Congress, New directions for a diverse planet. 4th. 26 Sep–1 Oct 2004, Brisbane, Australia.

Traina S.J., Sposito G., Hesterberg D., Kafkafi U. Effects of pH and organic acids on orthophosphate solubility in an acidic montmorillonitic soil. Soil Sci. Soc. Am. J. 1986;50:45-52.

Tunesi, S., V. Poggi, and C. Gessa. 1999. Phosphate adsorption and precipitation in calcareous soils: the role of calcium ions in solution and carbonate minerals. Nutr. Cycling Agroecosyst. 53(3):219-227.

Uehara, G. and G. Gillman. 1981. The mineralogy, chemistry, and physics of tropical soils with variable charge clays. Westview Press, Boulder, CO.

US Environmental Protection Agency. 1995. Guide manual on NPDES regulations for concentrated animal feeding operations. US-EPA Office of Water, Washington D.C.

US Environmental Protection Agency. 1998. Environmental impacts of animal feeding operations. US-EPA, Office of Water Standards and Applied Sciences Division.

US Salinity Laboratory. 1954. Diagnosis and improvement of saline and alkali soils. Agricultural Handbook No. 60, U.S. Department of Agriculture, Washington D.C.

Valle, C. B., C. Glienke, and G.O.C. Leguizamon. 1993. Breeding of apomictic Brachiaria through interspecific hybridization. In XVIII International Grassland Congress, New Zealand Grassland Association, Tropical Grasslands Society of Australia, New Zealand Society of Animal Production, Australian Society of Animal Production, Queensland Branch of New Zealand Institute of Agricultural Sciences, Palmerston North, New Zealand, Proceedings 1.

Vallejos, A. A.1988. Caracterización y evaluación agronómica y preliminar de accesiones de Brachiaria y Panicum en el trópico húmedo de Costa Rica. Tesis Mag.Sc. Turrialba, Costa Rica, CATIE, 126 pp.

van der Houwen J.A., and E. Vaisami-Jones. 2001. The application of calcium phosphate precipitation chemistry to phosphorus recovery: the influence of organic ligands. Envi. Tech. 22(11):1325-35.

Vanguard Software Corp. 1992-2002. DecisionPro for Windows V. 4.0.4. Cary, NC.

Velez-Santiago, J., and J.A. Arroyo-Aguilu. 1983. Nitrogen fertilization and cutting frequency, yield and chemical composition of five tropical grasses. J. Agric. Univ. PR 68:61–69.

Vicente-Chandler, J., F. Abruna, R. Caro-Costas, J. Figarella, S. Silva, and R.W. Pearson. 1974. Intensive grassland management in the humid tropics of Puerto Rico. Bull. 233. P. R. Agric. Exp. Stn., Rio Piedras, Puerto Rico.

Vicente-Chandler, J., and R.W. Pearson. 1960. Nitrogen fertilization of hot climate grasses. Soil Conserv. Mag. 25:269–272.

Vicente-Chandler J., Silva S., and Figarella J. 1959a. The effect of nitrogen fertilization and frequency of cutting on the yield and composition of three tropical grasses. Agron. J. 51:202-206.

Vicente-Chandler, J., S. Silva, and J. Figarella. 1959b. Effects of nitrogen fertilization and frequency of cutting on the yield and composition of napier grass in Puerto Rico. J. Agric. Univ. Puerto Rico 43(4):215-228.

Villapando, R.R. and D.A. Graetz. 2001. Phosphorus sorption and desorption properties of the Spodic horizon from selected Florida Spodosols. Soil Sci. Soc. Am. J. 65:331-339.

Voisin, A. 1963. Grass Tetany. [Online]. Available at http://www.soilandhealth.org/01aglibrary/010106voisin/010106gtintro.html#foreword. (accessed 15 Feb. 2005; verified 31 Mar. 2006). Bannerstone House, Springfield, IL and Crosby Lockwood and Son, Ltd., The Chaucer Press, Bungay, Suffolk. Great Britain.

Walmsley, D., V.A.L. Sargeant, and M. Dookeran. 1978. Effect of fertilizers on growth and composition of elephant grass (Pennisetum purpureum) in Tobago, West Indies. Trop. Agric. (Trinidad) 25: 329-334. [Online]. Available at http://www.fao.org/ag/AGP/AGPC/doc/Gbase/DATA/Pf000301.HTM, (accessed Jan. 2003; verified Dec. 2006), Food and Agriculture Organization of the United Nations, Rome, Italy.

Wagner, W.L., D.R. Herbst, and S.H. Sohmer. 1990. Manual of flowering plants of Hawai‘i. 2 vols. University of Hawai‘i Press and Bishop Museum Press, Honolulu, HI.

Wang, H.D., W.G. Harris, K.R. Reddy, and E.G. Flaig. 1995. Stability of phosphorus forms in dairy-impacted soils under simulated leaching. Ecol. Eng. 5:209–227.

Wang, D., J.A. Poss, T.J. Donovan, M.C. Shannon, and S.M. Lesch. 2002. Biophysical properties and biomass production of elephant grass under saline conditions. J. Arid Environ. 52(4):447–456.

Wang, H., G. N. Magesan, and N. S. Bolan. 2004. An overview of the environmental effects of land application of farm effluents. New Zealand J. Agric. Res. 47: 389–403.

Watanabe, F.S., and S.R. Olsen. 1965. Test of an ascorbic acid method for determining phosphorus in water and NaHCO3 extracts from soils. Soil Sci. Soc. Am. Proc. 29:677-678.

Weaver, D.M., and G.S.P. Ritchie. 1994. Phosphorus leaching in soils amended with piggery effluent or lime residues from effluent treatment. Environ. Pollut. 84:227–235.

Westerman, P. W., L. D. King, J. C. Burns, G. A. Cumming,s and M. R. Overcash. 1987. Swine manure and lagoon effluent applied to a temperate forage mixture: II. Rainfall runoff and soil chemical properties. J. Environ. Qual. 16(2):106-112.

Weir, C.C. and R.J. Soper. 1963. Interaction of phosphates with ferric organic complexes Canadian Journal of Soil Science, 43: 393-399.

Whalen, J.K., and C. Chang. 2002. Phosphorus sorption capacities of soils receiving annual feedlot manure amendments for 25 years. Commun. Soil Sci. Plant Anal. 33:1011–1026.

Whalen, J.K., C. Chang, G.W. Clayton, and J.P. Carefoot. 2000. Cattle manure amendments can increase the pH of acid soils. Soil Sci. Soc. Am. J. 64:962–966.

Williams, P. 1995. Animal production and European pollution problems. Animal Feed Sci. Tech. 53:135-144.

Willet, I.R. 1989. Causes and prediction of changes in extractable phosphorus during flooding. Aust. J. Soil Res. 27:45–54.

Wilkerson, V.A., D.R. Mertens, and D.P. Casper. 1997. Prediction of excretion of manure and nitrogen by Holstein dairy cattle. J. Dairy Sci. 80:3193–3204.

Wilkison, D.H., D.W.Blevins, and S.R.Silva. 2000. Use of isotopically labeled fertilizers to trace nitrogen fertilizer contributions to surface, soil, and ground water: J. Envi. Hydrol. 8(6):1-16.

Wolf, B. 1974. Improvements in the azomethine-H method for the determination of boron. Commun. Soil Sci. Plant Anal. 5:39-44.

Woodard, K.R., E.C. French, L.A. Sweat, D.A. Graetz, L.E. Sollenberger, B. Macoon, K.M. Portier, B.L. Wade, S.J. Rymph, G.M. Prine, and H.H. Van Horn. 2002. Nitrogen removal and nitrate leaching for forage systems receiving dairy effluent. J. Environ. Qual. 31:1980–1992.

Xavier, E.G., L.A. Petey, G. L. Cromwell, and M. D. Lindermann. 2004. Phosphorus excretion of pigs fed conventional or low-phytate corn-soybean meal diets without or with phytase. J. Anim. Sci. 82 (Suppl. 1):253.

Zemenchik, R.A., and K.A. Albrecht. 2002. Nitrogen use efficiency and apparent nitrogen recovery of Kentucky bluegrass, smooth bromegrass, and orchardgrass. Agron. J. 94:421–428.

Zhang, H., T.H. Dao, N.T. Basta, E.A. Dayton, and T.C. Daniel. 2002. Remediation techniques for manure nutrient loaded soils. National Center for Manure and Animal Waste Management, Raleigh, NC.

Zhang, H., J. L. Schroder, J. K. Fuhrman, N. T. Basta, D. E. Storm and M. E. Payton. 2005a. Path and multiple regression analyses of phosphorus sorption capacity. Soil Sci. Soc. Am. J. 69: 96-106.

Zhang, H., J.L. Schroder, J.J. Pittman, J.J. Wang,, and M. E. Payton. 2005b. Soil salinity using saturated paste and 1:1 soil to water extracts. Soil Sci. Soc. Am. J. 69:1146–1151.

Zhiping Q., I. M. Rao, J. Ricaurte, E. Amézquita, J. I. Sanz, and P. C. Kerridge. 2004. Root Distribution and Nutrient Uptake in Crop-Forage Systems on Andean Hillsides. J. Sust. Agric. 23(4).