CONTINUOUS SIMULATION OF GROUNDWATER USE AND EFFLUENT DISCHARGE IN CATFISH (*ICTALURUS PUNCTATUS*) PONDS AT FIVE LOCATIONS IN THE SOUTHEAST U.S.

By

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Long-term climatological data were used to evaluate the effectiveness of a drop/add management strategy to reduce groundwater use and effluent discharge in catfish ponds in the southeast U.S.. A drop/add approach is based on the creation of a storage volume in the pond for rainfall collection. The storage volume is created by allowing water level in the pond to decrease until some minimum level is reached. When the minimum level is reached, the pond is partly refilled, leaving the remaining volume available to capture incident precipitation. In this way, the role of precipitation in the water budget is increased. In the process, groundwater use and effluent release both become smaller.

The data consisted of 45 year precipitation and evaporation records from Fairhope, AL; Clemson, SC; Stoneville, MS, Stuttgart, AR; and Thomsons, TX. The data were used in a water balance levee pond model that included precipitation, evaporation, infiltration, overflow, groundwater pumping, and draining. The model appeared to indicate that the drop/add management scheme is an effective strategy to reduce groundwater use and effluent discharge.

The simulated results showed that variation of climate in the southeast U.S. was an important determinant of performance of the drop/add management scheme. At locations with positive P-0.8E, zero groundwater use could be achieved with low drop depths. At location with negative P-0.8E, zero groundwater use could be achieved for about 50% of the 45 simulated years. The model also indicated that effluent discharge cannot be avoided at most locations except at location with very low (negative) P-0.8E. The model also indicated that 65 to 100% of annual precipitation (depending on the P-0.8E's of the locations) can be captured and used in the ponds. Rainwater contribution to the total water budget ranged from 90 to 100%.

The sensitivity analysis showed that model sensitivity to pan coefficient and infiltration rate was affected by infiltration rate and pond water storage capacity (drop depth). The model was more sensitive to pan coefficient rather than to infiltration rate at lower infiltration rates and *vice-versa*. Both sensitivities of the model, however, increased when pond water deeper storage capacity was used.

Key words: aquaculture, pond, model, drop/add scheme

DEDICATION

I would like to dedicate this research to my parents Harinto Samad and Seruni, and my wife Sri hartatik, my son Afid Fitro Setiawan, and my daughter Vina Dwiayu Wardhani.

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CHAPTER I

INTRODUCTION

Aquaculture is defined as "the business of producing aquatic animals and plants in managed, unnatural aquatic ecosystems for profit" (Boyd and Schmittou, 1999). Aquaculture is the only viable aquatic food production alternative to meet food demand, in that fishing from natural resources has been exploited to, or beyond, the edge of maximum production. Capture fisheries were practically stagnating (87.7 million tons in 1998 and 90.3 million tons in 2003) but aquaculture production was expanding from 30.6 million tons in 1998 to 41.9 million tons in 2003 (FAO, 2004).

Commercial catfish (*Ictalurus punctatus*) production, the leading aquaculture industry in the United States, generates over 46 percent of the value of aquaculture production in the United States (Tucker et al., 2004). Important commercial catfish production is concentrated in southeastern states such as Mississippi, Arkansas, Alabama, and Louisiana. The combined acreage of these four states makes up 94 percent of all catfish production (Boyd and Queiroz, 2001; Tucker and Hargeaves, 2003b; Tucker et al., 2004). Among the primary producing states, Mississippi has the most acreage in catfish production and the greatest economic value, approximately $243 million in 2003 (Tucker et al., 2004). Some other catfish producing states with smaller acreage include Texas, Georgia, California, South Carolina, North Carolina, Missouri, Florida, and Kentucky.

Most commercial catfish ponds, particularly in Northwest Mississippi, are embankment-type/levee ponds (Steeby and Avery, 2002). The levee is made by filling with the soil dug from the area, which becomes the pond bottom. The pond bottom is sometimes lined with imported clay and compacted to reduce water losses through infiltration. Ponds are typically rectangular and have a 2:1 to 3:1 ratio of length to width. This type of pond is preferred because its rectangular shape makes management operations easier.

Groundwater is used as the primary water source in catfish ponds to make up water losses through evaporation and infiltration (Pote et al., 1988). Groundwater should not be overexploited beyond the recharge rates because this practice is not sustainable. Some areas in the US have suffered from groundwater overexploitation (USGS, 1995). Progressive irreversible drawdown of groundwater level and increasing abstraction cost are among the characteristics of groundwater being overexploited. Groundwater overexploitation could also result in other adverse impacts such as water-quality deterioration, land subsidence, and ecological damage (Custodio, 2002). As water shortages become more common in the future, it is predicted that competition for this resource in aquaculture will increase with other usages, particularly for drinking (Funge-Smith and Phillips, 2001).

Another problem in aquaculture is potential water quality deterioration of receiving waters due to nutrient release. Feed is put into the catfish ponds to maintain the fish growth and maximize the production (Gross et al., 2000), but only a small portion of the nutrients (about 30%) is recovered in the fish biomass. The rest is in uneaten feed and fish excretion products (Tucker and Hargeaves, 2003a). The contaminants are then physically and biochemically converted, decomposed, and assimilated into algal and bacterial biomass within the pond water. Some substances such nitrogen gas formed after nitrification-denitrification process volatile to the atmosphere but other substances may stay in the water. When the water is released (either intentionally or as a result of overflow after heavy rainfall) to receiving waters, the contaminants could potentially cause eutrophication and damage aquatic ecosystems down streams.

Inevitable pressure from environmentalist groups has led aquaculturists to look for environmentally responsible alternatives to current pond management techniques in order to reduce the environmental impact of effluent discharges from aquaculture ponds (Boyd and Queiroz, 2001). The United States Environmental Protection Agency (USEPA), has finalized effluent limitation guidelines (ELGs) for concentrated aquatic animal production (CAAP), or aquaculture facilities. The new regulation will apply to existing and new CAAP facilities which use flow-through, recirculating, or net pen systems; directly discharge wastewater; and produce at least 100,000 pounds of fish per year (USEPA, 2004).

Some farmers use flushing or water exchange, pumping groundwater and releasing portion of pond water at the same time, to improve pond water quality. Research has revealed that this practice has little effect on water quality improvement (Boyd, 1983). Catfish ponds are characteristically shallow, about 1 m, and have a large surface area. Many older ponds are more than 5 ha, but newly constructed ponds are typically 4-5 ha (Steeby and Avery, 2002). Based on typical inflow rate, hydraulic residence time is quite long. This lets physical, chemical, and biological processes occur in the pond water which convert and reduce the concentration of potentially harmful water quality constituents. If the pond is managed within its assimilation capacity (the capacity at which continuous conversion of the constituents will occur), the water constituents do not accumulate over time. This approach obviates the need for flushing or water exchange requirement to maintain pond water quality without affecting fish growth (Tucker et al., 1996; Zimba et al., 2003). Therefore, this effort could improve and minimize effluent discharge from the catfish ponds to the environment.

Catfish ponds have the potential to make use of rainwater, hence minimizing groundwater use. Maximizing rainwater use and reducing groundwater use in commercial catfish ponds (Cathcart et al., 1999; Parker et al., 1999; Pote and Wax, 1993) could maintain the sustainability of the precious natural resources and could reduce the production cost for groundwater pumping. Another advantage of this is that effluent discharge from catfish ponds can be reduced and environmental pollution can be minimized.

Research in pond water management related to groundwater use and effluent release have mostly been limited to water budget studies or studies related to feed composition and application rate. For example, research on the pond water budget was conducted for embankment fish ponds at Comayagua, Honduras (Green and Boyd, 1995). Similar researches on pond water budget model with small area of ponds at AIT, Thailand and El Carao, Honduras were also carried out (Nath and Bolte, 1998). At those researches, water source and sink were estimated.

The drop-add management strategy was first developed by Pote et al. (1988). This management strategy was explored for embankment catfish ponds at the Delta, Mississippi by using long-term climatological data. Following the drop-add management scheme, Cathcart et al. (1999) introduced continuous model for linked pond systems. These schemes maintained storage capacity to capture rainwater. Some reductions of groundwater use and effluent discharge were achieved in those modeling approaches. In the future, using storage of water in this type of water management will be increasing in aquaculture (Funge-Smith and Phillips, 2001). In broader scope, theoretical pond water budget and hydrology model for inland aquacultures to optimize water use and minimize effluent discharge was developed (Boyd and Gross, 2000). Recently published, symmetrical drop/add management scheme using 40-year climatological data was extended to the Southeast US (Cathcart et al., 2007).

In order to test the reliability of the drop-add management scheme, a long-term continuous simulation is very important to perform. This is because climate characteristics, particularly precipitation, tend to vary from year to year. Another reason is that the temporal and geographical distribution of rainfall has an effect on water accumulation in ponds. For example, an intensive short term research effort with on-site data collection may provide detailed information about what happened during the period of the research, but it will miss much of the intrinsic variability that occurs at that site over a longer period. Small intense storms may either strike or miss the research site, resulting in either over-estimation or under-estimation of precipitation. Long term studies with on-site climatological data collection are usually not practical, but it is only with the long term data that actual site characteristics can be determined. On the other hand, long term climatological records do exist in certain areas. The National Weather Service, in particular, supports many stations that have continuous climatological records in excess of 40 years. An alternative to intensive on-site collection uses these records to represent climatological performance in a region or sub-region of interest. Use of long term records allows continuous simulation of pond water balance which, in turn, may be used to mimic the dynamics of pond water level as an effect of the hydrological components (e.g. water addition or loss via overflow).

The drop-add management strategy needs to be applied across the southeast U.S. in order to promote sustainable aquaculture. Because this management scheme is highly dependent on regional climate, drop-add management strategy should be tested at various locations with different climatological characteristics. Once the management strategy has been tested at a specified location, the output can be used as a guideline by farmers or managers both for planning and modification of daily management operations.

The general objective of this research is to explore drop-add management strategies in catfish ponds by using long-term continuous simulations in order to minimize the groundwater use, maximize rainwater use, and minimize effluent discharge, based on regional climatological data. This study is accomplished by implementing a mathematical model of embankment-type/levee catfish ponds for locations in the states of Mississippi, Arkansas, Alabama, South Carolina, and Texas in order to assess the effect of the regional climate.

Specific objectives of this research include:

1. Comparison of climate patterns at Fairhope, AL; Clemson, SC; Stoneville, MS; Stuttgart, AR; and Thomsons, TX.

2. Discussions of the effects of the regional climates, infiltration rates, harvest discharge, drop depths, and fill depths on the drop-add management scheme.

3. Sensitivity analysis of the model.

4. Determination of optimum drop-add management schemes.

CHAPTER II

LITERATURE REVIEW

2.1 Problems of Groundwater Overexploitation

The amount of available fresh water in the world is very limited. UNEP (2002) estimated that the total volume of water on Earth is about 1.4 billion km^3. Of that amount, the volume of freshwater is about 35 million km^3 or only 2.5%. This total volume of fresh water is not all available because 24 million km^3 (68.9%) is in the form of ice and permanent snow cover in mountainous regions and the poles. Some 8 million km^3 or 30.8% is stored in the form of groundwater. Freshwater wetlands, lakes, and rivers contain an estimated 105,000 km^3 or 0.3% of the world freshwater (UNEP, 2002).

The fresh water resource is distributed unevenly among the regions or countries of earth. Table 1 shows that annual renewable water resources and availability per capita (annual renewable resources divided by population) varies among groups of countries (World Bank, 1992). Some areas still have high availability of water resources per capita, while other areas, such as Middle East and North Africa, have low availability of water resources. Other countries are in conditions where withdrawal is very high, even greater than the annual recharge rate (Pereira et al., 2002). There also are some people with less than 2000 m^3 per capita, which is considered to indicate that a region is water stressed since under these conditions populations face very large problems when a drought occurs (Pereira et al., 2002).

Table 1. World Availability of Water Resources *)

Country Group	Annual Internal Renewable Water Resources		Percentage of Population Living in Countries with Scarce Annual per Capita Resources	
	Total (1,000 km^3)	Per capita (1,000 m^3)	Less than 1,000 m^3	1,000-2,000 m^3
Sub-Saharan Africa	3,8	7,1	8	16
East Asia and Pacific	9.3	5.3	<1	6
South Asia	4.9	4.2	0	0
Eastern Europe and former USSR	4.7	11.4	3	19
Other Europe	2.0	4.6	6	15
Middle east and North Africa	0.3	1.0	53	18
Latin America and the Caribbean	10.6	23.9	<1	4
Canada and United States	5.4	19.4	0	0
world	40.9	7.7	4	8

*) Source: World Bank (1992)

The water crisis has been serious in some parts of the world and is getting more and more attention from world water societies (UNEP, 2003). This situation is likely to continue and become even worsen in the future because demand for fresh water is consistently increasing with population growth and lifestyle quality while the existing fresh water is only small portion of the total water in the world. During the last century total freshwater usage has risen six fold (UNEP-DEWA, 2003). In addition to low potential of renewable water resources (for certain areas), it was predicted that there would be two major causes contributing to water crises in the future: water quality deterioration and lack of investment (Biswas, 1999). As mentioned above, most available freshwater is stored underground, and it requires energy cost/investment to exploit the groundwater. In other parts of the world where people or industry can exploit groundwater easily, other problems (overexploitation) are arising (UNEP-DEWA, 2003). The problems associated with groundwater overexploitation include groundwater contamination, saltwater intrusion, progressive drawdown, increasing energy cost, ecological damage, and land subsidence.

In the United States, fresh water withdrawal is markedly increasing, from less than 200 billion gallons per day in 1950 to about 350 billions gallons per day in 2000 (USGS, 2000). The major consumers of the fresh water are thermoelectric power (for steam-driven turbine generators and cooling systems) and irrigation. Among the water resource regions, daily consumption varies relative to the renewable water supply (USGS, 1995). The renewable water supply is the sum of precipitation and imports of water, minus the water not available for use through natural evapotranspiration and exports (USGS, 1984). In eastern regions (of the US, Alaska, and The Pacific Northwest, the fresh water consumption is less than 10% of the renewable water supply. In California, the Great Basin, the Upper Colorado, Missouri, Arkansas, and the Texas-Gulf Region, the water consumption range is 10% to 40%. Greater consumption of fresh water occurs in the Rio Grande Region (~40-100%), while in the Lower Colorado, water consumption is more than 100% of the recharge rate (USGS, 1995).

Fresh water consumption is initially from surface water. When the surface water is not enough or the quality does not meet the requirement, demand of fresh water is satisfied by groundwater (Pereira et al., 2002). It was reported that groundwater contributed about 97% to fresh water consumptions (UNEP, 2002). When groundwater abstraction is close to the renewable water supply (recharge rate), it is said to be overdrawn or overexploited and generally it results in a decrease of the water level (Custodio, 2002). However, concern about groundwater overexploitation is not limited to the situation where withdrawal exceeds the renewable supply, resulting in the decrease of groundwater level. In practice, an aquifer is often considered as overexploited when some persistent negative results of aquifer development are felt or perceived, such as a continuous water-level drawdown, progressive water-quality deterioration, increase of abstraction cost, or ecological damage (Custodio, 2002).

Land subsidence is a major adverse effect of groundwater overexploitation. Land subsidence can be defined as the differential sinking of the ground surface with respect to surrounding terrain or sea level (Hua et al., 2004). Land subsidence can result from natural causes such as tectonic motion and sea level rise or man-induced causes such as the withdrawal of groundwater, oil and gas, the extraction of coal and ores and the underground excavation for tunnels. Groundwater withdrawal results in fluid pressure change in the layers. The pressure change in the layers induces both elastic and inelastic land compaction. The elastic compaction can be recovered if the water level rises again while inelastic compaction becomes permanent which can potentially cause land subsidence (Sun et al., 1999).

There are examples of land subsidence due to the decline of water level in many big cities and coastal areas in the world. Land subsidence, due to the decline of water level which has resulted from weak regulation of groundwater withdrawal, has caused some serious damage of buildings in Bangkok, Thailand (Phien-wej et al., 1998). The Thessaloniki coastal plain, Northern Greece, subsided at a rate of up to 10 cm/year during the last 40 years. As a consequence, the sea water invaded up to 2 km inland (Stiros, 2001). Land subsidence in the Indogangetic basin, India, caused by increasing rates of groundwater exploitation, has also been identified (Mishra et al., 1993).

In the United States, noticeable land subsidence occurred mostly in known regions where there is a high rate of groundwater exploitation. Damage of some infrastructures and flooding due to land subsidence has occurred in Long Beach and Santa Clara Valley, California and Houston-Galveston, Texas (Holzer, 1989). Observed vertical ground movement, relative to benchmarks outside the zone of pronounced subsidence in the vicinity of Pecos, Texas, was up to 200 mm between 1934 and 1956 (Rosepiler and Reilinger, 1977) . The decline in the water table was as much as 60 m during approximately the same period. Land subsidence due to water table decline in the form of sinks has also occurred on and near farmlands near Tucson, Pima County, Arizona, USA (Hoffmann et al., 1998).

2.2 Impacts of Eutrophication

Eutrophication is defined as the enrichment of surface water with plant nutrients and the subsequent (abundant) growth of plants within the waters (FAO, 1996; Pitois et al., 2001). The primary nutrients are nitrogen (N) (Bowen, 2005; Vitousek et al., 1997) and phosphorus (P) (Bennett et al., 2001) which are essential to all organisms. Abundance of the nutrients in aquatic ecology causes algal bloom. Because the life cycle of algae is short, they die very quickly. Upon decomposition of the algal biomass, microorganism consumption of dissolved oxygen (DO) can lead to foul odors and oxygen depletion. This can, in turn, lead to fish kills (Carpenter et al., 1998), change in the structure of zooplankton community (Pinto-Coelho, 1998), and disappearance of other aquatic species (Dolbeth et al., 2003; Riis and Sand-Jesen, 2001).

Public concern began to rise in the 1960s, when nutrient enrichment was rapidly making many bodies of water increasingly fertile (Pitois et al., 2001). Agriculture and urban activities are thought to be the major sources of phosphorus and nitrogen added to aquatic ecosystems (Carpenter et al., 1998). Nitrogen and phosphorus containing Fertilizers are intended to increase agricultural production to meet the demand for food. However, since only a portion of the nutrients is recovered in the agricultural products and very often more fertilizers are applied than required, there are some residues in the soil and plant litters (Bennett et al., 2001). When a raindrop hits the soil surface, the rainwater detaches the nutrient-containing soil particles. Runoff water washes outs theses residues in the soil and eventually the nutrients reach water bodies: lake, streams, estuaries, seashores, and oceans.

Excessive use of phosphorus in fertilizers, animal feeds, agricultural crops, and other products have been altering the global phosphorus cycle, increasing the levels of phosphorus in some of the world's soil, and elevating the potential phosphorus runoff to aquatic ecosystems. Bennett et al. (2001) reported that phosphorus change/accumulation in agricultural soils of developing countries increased from 1961 to 1996. A similar situation has been occurring for the nitrogen cycle. Agriculture, in addition to combustion of fossil fuel and other human activities, has been known to alter the nitrogen cycle substantially, generally increasing both the availability and the mobility of nitrogen over large regions of earth (Vitousek et al., 1997). Many of the mobile forms of nitrogen themselves have environmental consequences. It was reported that human activities have approximately doubled the rate of nitrogen input into the terrestrial nitrogen cycle, with these rates still increasing (Vitousek et al., 1997).

Detrimental impact of eutrophication on the environment is widespread in the world's waters. The hypoxia "dead zones" in the open oceans or coasts of the world are due to eutrophication (Dybas, 2005; Larsen, 2004). Hypoxia is defined in the broadest possible way as a condition of depressed dissolved oxygen (DO) concentration sufficient to cause an adverse ecological effect (Hagy-III, 2002). UNEP (2003b) reported that there are 146 coastal dead zones in the world, and that the number of dead zones has doubled with each passing decade. The dead zones range in size from small sections of coastal bays and estuaries to large seabeds spanning some 70,000 square kilometers. Most of these occur in temperate waters, concentrated off the U.S. coasts and in the seas of Europe. Some are in the waters off China, Japan, Brazil, Australia, and New Zealand. The largest dead zone is found in the Baltic Sea. The one in the Gulf of Mexico is the second largest. Of the world's known coastal dead zones, 43 occur in the US waters (UNEP, 2003).

The impacts of eutrophication around the Baltic Sea, where aquaculture is one of the sources (Gyllenhammar and Hakanson, 2005), have been reported in many papers. Ronnberg and Bonsdorff (2004) have identified that the amounts of nutrients in the brackish water of the Baltic Sea have increased several fold during the last century, causing severe ecological effects on the biota. During the last few decades it was reported that the perennial seaweed *Fucus vesiculosus* L. has rapidly declined in large parts of the Baltic Sea. Indirect effects of eutrophication, such as increased turbidity, sedimentation, grazing and occurrence of filamentous algae, have generally been suggested as major factors causing the decline (Berger et al., 2004). Lappalainen and Ponni (2000) reported that the Gulf of Finland is one of the most eutrophic areas in the Baltic Sea. It is the main fishing area for more than 150,000 recreational fishermen in Finland. In the eastern part of the Gulf of Finland eutrophication is also a serious environmental problem (Golubkov et al., 2003).

Many severe environmental impacts of eutropication in other parts of Europe are well documented. Riis and Sand-Jesen (2001) reported that European lowland streams have experienced increased perturbation and eutrophication during the past 100 years. Overall decline in richness and the directional change in dominance patterns among stream species were reported to be due to the loss of suitable habitats and the strong anthropogenic impacts, which have driven several European aquatic species close to extinction (Riis and Sand-Jesen, 2001). In Baldeggersee (Switzerland), as a result of progressive nutrient enrichment, changes occurred in the 1910s from a *Cyclotella* to a *Tabellaria fenestrata* dominated assemblage, and eventually in the 1950s to a *Stephanodiscus parvus* dominated diatom assemblage (Lotter, 1998). In the Netherlands, *Succisa pratensis* (a characteristic plant species of biodiverse slightly acidic grasslands (NARDETEA)) has been reported in decline over recent decades mainly attributed to the environmental stress due to anthropogenic acidification and eutrophication (Vergeer et al., 2003). Sicilian reservoirs, a vital resource for this densely populated island, have been found to be undergoing increased eutrophication processes, due to the lack of urban waste treatment plants and to intense agriculture (Naselli-Flores et al., 2003).

In South America, reports on the environmental impact of eutrophication are available. Moser et al. (2004) reported a growing degree of eutrophication in Praia Grande, Sao Sebastiao, Guaruja, and Santos Brazil. Change of the structure as well as seasonal patterns of the zooplankton community in Pampulha Lake (Brazil) due to the increasing eutrophication was reported by Pinto-Coelho (1998). In Lake Petén Itzá (Guatemala), geochemical records from the sediment core indicate increased phosphorus loading and organic matter accumulation due to the watershed deforestation and increased surface run-off since 1930. After 1965 the high nutrient concentrations have resulted in the dominance of eutrophic and hypereutrophic diatom species (Rosenmier et al., 2004).

The second largest zone of coastal hypoxia (oxygen-depleted waters) in the world is found on the northern Gulf of Mexico continental shelf adjacent to the outflows of the Mississippi and Atchafalaya Rivers (Rabalais et al., 2002). Paleoindicators in dated sediment cores indicate that hypoxic conditions likely began to appear around the turn of the last century and became more severe since the 1950s as the nitrate flux from the Mississippi River to the Gulf of Mexico tripled (Rabalais et al., 2002). Joyce (2000) found that since the 1970s, during the warm months, oxygen levels in the Mississippi plume region of the Gulf of Mexico fell from healthy concentrations to 2-3 mg/l. The Gulf of Mexico is usually teeming with fish, shrimp, and other sea creatures. In recent years, however, that abundance of marine life has virtually vanished in a huge area of water off the coast of Louisiana called the dead zone (Earth and Life Science, 1998). After the severe Mississippi River flood of 1993, the U.S. National Ocean Service reported that the oxygen-starved area in the Gulf of Mexico more than doubled in size to 18,000 square kilometers (Chafe, 2004). A seasonal dead zone in the northern Gulf of Mexico developed occasionally in the 1800s, but it's become more intense in the last few decades as farmers increased fertilizer use (Ferber, 2005).

The Chesapeake Bay is another example of the severe impact of eutrophication in the United States. Historical (1950-1999) dissolved oxygen (DO) data for Chesapeake Bay showed that moderate hypoxia (DO < 2.0 mg/l) increased about 3-fold, modulated by spring river flow, while severe hypoxia (DO < 0.7 mg/l) occurred only in high flow years during 1950-1960, but was present annually since 1968 (Hagy-III, 2002). A predictive mathematical model developed by Cerco (1995) also shows an increase in anoxic volume from the 1959-1968 decade to the 1979-1988 decade. The increase was associated with increasing nitrogen concentration in runoff from two major tributaries and with increased chlorophyll concentrations.

Bowen (2005) reported that N loading in the Waquoit Bay, Cape Cod, Massachusetts, more than doubled between 1938 and 1990, with wastewater N becoming the dominant N source during 1980s. These occurred even though the Waquoit Bay watershed retained 80% of N inputs, with only 20% entering the Bay. This increased N load altered the Bay, increasing biomass of phytoplankton and macroalgae (Bowen, 2005). The predominant source of nitrogen added to the bay changed from atmospheric deposition to wastewater disposal during the 1980s, reflecting the increasing urbanization of Cape Cod (Bowen and Valiela, 2001).

2.3 Water Quality of Embankment Catfish Ponds

Aquaculture is only a part of agricultural activities that is potentially the most water consuming activity per unit of area (Boyd and Gross, 2000). To promote fish production, feed and fertilizers are applied to fish ponds. Only about 30% of the nitrogen and phosphorus contained in the feed and fertilizers is recovered in fish at harvest (Gross et al., 2000). A major portion of the nutrient content is released in the form of fish excretion as metabolic products (Tucker and Hargeaves, 2003b). The metabolic products are then microbiologically synthesized and chemically decomposed into some water constituents such as microbial/algal biomass, carbon dioxide, and nitrogen gas. The water constituents will settle out to the pond bottom, escape to the atmosphere as a gas, or stay dissolved in the water. Most fish ponds are operated with some effluent discharges of water in the form of flow through, flushing, periodic drain, or overflow after heavy rain (Boyd, 2003). If aquaculture pond is not managed properly, the residues potentially make aquaculture one of polluting industries to the environment.

Catfish production is the leading aquaculture industry in the United States, with more than 94% located in southeastern states: Mississippi, Arkansas, Louisiana, and Alabama (Boyd and Queiroz, 2001; Tucker and Hargeaves, 2003b; Tucker et al., 2004). The production of channel catfish in ponds was over 270,000 metric tons of fish processed in 2000 (Tucker and Hargeaves, 2003b). Most production of commercial catfish is operated in multiple batch systems, with periodical fingerling stocking to maintain continuous harvest (USDA and APHIS, 1997). Typical embankment ponds are 4-10 ha in earthen levees and have depths of 1 – 1.5 m (Steeby et al., 2004). They use groundwater as the primary source of water supply.

The catfish pond system and the management practices used dictate water quality in the ponds and affect the quality of effluent discharge. Pond water contains primarily nutrients, organic matter, and occasionally soil particles (Tucker and Hargeaves, 2003b). Effluent contains a considerable amount of soil particles, particularly after heavy rainfall and intentional draining (when pond repair is performed). Soil particles from erosion of the pond levee is also caused by waves and is affected by aerator use (Steeby et al., 2004). The soil particles contained in the effluent threaten the sustainability of receiving waters because they cause sedimentation downstream. Nutrients from fish feeds and fertilizers in pond water consist primarily of nitrogen and phosphorus. Nutrients contained in the effluent cause eutrophication in receiving waters. Organic matter, mainly phytoplankton and zooplankton biomass, are also contained in the effluent of catfish ponds (Zimba et al., 2003). Plankton proliferate by deriving nutrients from feed residue and fish excretory products in pond water (Tucker and Hargeaves, 2003b). As described above, plankton will die, settle to the bottom of the water columns, and be decomposed by aerobic microorganisms leading to oxygen depletion in the effected waters.

Nitrogen dynamics in catfish ponds is well described by Gross et al.(2000). About 87.9% of the N input to ponds comes from fish feed. It is mainly in the form of ammonia (NH_3), ammonium (NH_4), and nitrate (NO_3). Nitrogen losses through fish harvest was 31.5%, denitrification was 17.4%, NH_3 volatilization accounted for 12.3%, and accumulation in bottom soils was 22.6% (Gross et al., 2000). Nitrification was about 70 mg N $m^{-2}d^{-1}$ and denitrification average 38 mg N $m^{-2}d^{-1}$. Phytoplankton removed NO_3-N at rate of 24 mg N $m^{-2}d^{-1}$ while mineralization of feed N to NH_3 averaged 59 N $m^{-2}d^{-1}$ (Gross et al., 2000).

Seasonal changes of water quality in catfish ponds were reported by Tucker et al. (1996). Concentrations of total nitrogen, total phosphorus, chemical oxygen demand, biochemical oxygen demand, and phytoplankton biomass in pond waters were highest in the summer. However, concentrations of dissolved inorganic nitrogen (ammonia, nitrite, and nitrate) were greatest in the winter because the assimilation rates are low during the period of slow phytoplankton growth (Tucker and Van der Ploeg, 1993; Tucker et al., 1996). It was also reported that concentrations of total suspended solids were low in winter because of low phytoplankton biomass. Long term temporal succession changes in nutrients, phytoplankton, and zooplanktons in catfish ponds were studied by Zimba et al. (2003). He reported that nutrients accumulated in ponds during the first 3 years post construction, and level off after that time. Algal composition became dominated by filamentous cyanobacteria after year 4, while zooplankton composition was dominated by larger copepods and cladocerans in older ponds (Zimba et al., 2003). Sediment oxygen demand, which is very important indicator in aquaculture ponds, is not affected by pond age (Berthelson et al., 1996). Similarly, sediment organic carbon concentrations do not increase with pond age (Steeby et al., 2004).

2.4 Drop-Add Management Strategy

The United States Environmental Protection Agency (USEPA), established Effluent Limitation Guidelines (ELGs) for concentrated aquatic animal production (CAAP), or aquaculture facilities on June 30 2004, and revised them on August 23 2004 (USEPA, 2004). The Regulation mandates reduced discharges of conventional pollutants (primarily total suspended solids), non-conventional pollutants such as nutrients, and also drugs that are used to manage fish health and chemicals, such as those used to clean fish tanks and nets. The new regulation applies to existing and new CAAP facilities which use flow-through, recirculating, or net pen systems; directly discharge wastewater; and produce at least 100,000 pounds of fish a year. In flow-through and recirculating systems the new regulation applies to the facilities that produce at least 100,000 pounds a year and discharge wastewater at least 30 days a year. When the rule is fully implemented, discharges of total suspended solids will be reduced by more than 500,000 pounds per year and biochemical oxygen demand and nutrients will be reduced by about 300,000 pounds per year (USEPA, 2004).

As expected by most aquacultural groups during the drafting, the regulation did not just set a numerical effluent standard, but required application of specific practices such as best management practices (BMPs) (Boyd, 2003; Silapajarn and Boyd, 2005). Implementation of wastewater treatment facilities such as settling basins, and retention structures is not applicable for aquaculture (Boyd and Queiroz, 2001; Tucker and Hargeaves, 2003b), because it would increase production cost and make the aquaculture industry not feasible (Engle and Valderrama, 2003; Kouka and Engle, 1994; Kouka and Engle, 1996).

It is generally known that catfish ponds cannot be operated without effluent discharge (Boyd, 2003). The effluent discharges from catfish ponds are typically due to flushing, intentional release, draining, and overflow after heavy rain (Boyd and Queiroz, 2001). Flushing exchanges the pond water, releasing water from the pond while simultaneously filling with groundwater. It was thought that this practice improved water quality. Most researchers do not recommend it because there is little evidence of improved water quality (Boyd and Gross, 2000; Burtle et al., 1996). Intentional release of some amount of water is typically done to facilitate harvest; however, in embankment ponds, such as in The Mississippi Delta, most catfish producers harvest the fish without partial release of water because the pond depth is relatively shallow and homogeneous, 1 – 1.5 m (Tucker and Hargeaves, 2003b).

Draining of catfish ponds is carried out when the earthen levees need renovation. Because of erosion by wave and rain, the pond levee is degraded slowly and pond depth becomes shallower. Farmers deepen the pond bottoms and fix the pond levees over a long period (between 10 to 15 years or more), making a settling basin impractical for storing drained effluent (Boyd and Queiroz, 2001; Ozbay and Boyd, 2004). Moreover, by using long ditches, solids will settle within 15 minutes and also other water quality concentrations (phosphorus and nitrogen) will be lower than initial concentrations after traveling 150-200 m long (Hargreaves et al., 2005a; Hargreaves et al., 2005b).

Although effluent discharge during pond renovation cannot be avoided, this is not the case for day-to-day operation. While the pond is in use, minimization or prevention of environmental pollution by catfish ponds can be accomplished by minimizing the volume of discharge (either overflow or intentional draining).

A drop-add management strategy known as the “6/3 Management Scheme” was initially proposed by Pote et al. (1988). This management scheme was intended to maximize the use of rainwater while minimizing groundwater pumping for use in catfish ponds. In this scheme, water level was allowed to drop by 15 cm (6 inches) below the outfall before water was added. The amount of water then added raised the water level by 7.5 cm (3 inches), leaving a 7.5 cm water storage capacity to capture potential rainwater. By using this management scheme the use of groundwater was reduced about 57% (Pote and Wax, 1993; Pote et al., 1988) relative to the industry practice of complete filling.

Another mathematical model of add-drop management was developed to test various management scenarios of linked ponds with respect to the “6/3 management scheme” (Cathcart et al., 1999). In the study, instead of deepening all the production ponds, it was proposed to select a pond to deepen as a production and storage pond. This pond was linked to conventional production ponds. Therefore, only the production/storage pond is deeper than the production pond, making more practical and efficient work during the construction. The model predicted that effluent discharge would be reduced by 40-90% and groundwater usage was reduced by 40-75% following this practice for ponds deepened by 30-60 cm.

As a follow up of the “6/3 management scheme”, water quality was tested (Tucker et al., 1996). The results showed that managing pond water levels by 7.5 cm below the overflow structure reduced average discharge of nitrogen, phosphorus, and organic matter by about 70% through degradation and overflow reduction. The model also showed that harvesting fish without draining ponds would reduce average annual nutrient and organic matter discharge by over 60%.

Most catfish ponds, particularly those in The Mississippi Delta, are embankment ponds. Groundwater is used as the primary source of water to fill the ponds with rainwater as a supplemental source of water. Water conservation can provide a financial benefit to producers (Davis et al., 2002). The 6/3 water management scheme has been included in a pilot catfish verification program, initiated by the Arkansas Cooperative Extension System to provide production support to the catfish industry since May of 1993 (University of Arkansas, 2004). There may be some variations of the 6/3 scheme, or even seasonal flexibility for other locations depending on rainfall patterns. For example, In Alabama storage of 3-4 inches is recommended (Auburn University and USDA/NRCS, 2002). In Oklahoma water drop level of 12-18 inches is recommended for watershed ponds before pumping (Williams, 2000).

Maintaining water storage capacity of the ponds and not draining ponds when harvesting appear to be an effective tool not only to maximize groundwater and rainwater uses but also to reduce pollutant discharge (Boyd and Gross, 2000; Cathcart et al., 1999; Pote et al., 1988; Tucker et al., 1996).

2.5 Simulation Modeling in Aquaculture Ponds

Simulation modeling has been developing extensively in the complex systems of aquaculture ponds. Most simulation models are used to describe pond dynamics and management options with the purposes of optimizing production, minimizing environmental impact, and increasing economic profit. Computers have had great effects on model development. Complicated calculations can be solved by using computer aids in a short time period. By using computers, one can easily deal with numerous sets of data that would be impractical for manual calculation.

Piedrahita (1998) grouped aquaculture pond models into empirical and mechanistic models. Empirical models treat every pond as a big "black box", where correlation between input and output is explained by using statistical tests. Mechanistic models, on the other hand, require more detail than empirical models. In a mechanistic model, a pond is divided into several smaller compartments, each of which contains relationships among state variables. A mechanistic model tries to explain every relationship among the state variables by employing mathematical equations. However, as natural processes are too complex to always be able to be simulated mathematically, the use of empirical models remains a useful solution. In actual practices, pond simulations may always involve both empirical and mechanistic models (Piedrahita, 1998).

There are several benefits of using simulation models as a primary analysis tool in the complex systems of aquaculture ponds. One has an opportunity for knowledge synthesis, whereby a large body of knowledge about aquaculture ponds can be integrated into a comprehensive representation of the system that can be used to explore the effect of different management scenarios. Simulation models are valuable tools for predicting system responses to conditions that are either too complex or expensive to explore experimentally. Simulation models also provide an opportunity to explore a much larger set of operating conditions, environments, management strategies, and constraints compared to physical experiments (Nath, 1996).

In order to enhance production, simulation models have been constructed to examine fish carrying capacity of the ponds (Duarte et al., 2003), fish growth (Tai et al., 1994), fish diseases (Lotz et al., 2003), and to get optimal harvesting strategy (Yu and Leung, 2005). By using simulation models, pond managers can test management options and observe optimal results before implementation of field action.

Simulation models have also been developed to evaluate pond water quality (Buonomo et al., 2005; Schawartz, 2004), such as temperature, nutrient dynamics, and sediment characteristics. A computer model of temperature distribution in a freshwater pond was initially developed by Cathcart and Wheaton (1987). Other models of temperature distribution in aquaculture ponds are in Losordo and Piedrahita (1991) and Culberson and Piedrahita (1996). Temperature gradient/stratification can be critical in aquaculture ponds because it can cause stress, influences resistance to diseases, and prevents vertical mixing. Because of the density gradient, contaminants such as metabolites accumulate on the pond bottom, resulting in low dissolved oxygen (Berthelson et al., 1996). Other models of temperature effects on fish growth were studied by Jackson and Wang (1998) and Lamourex et al. (2005), Nutrient (particularly nitrogen as well as phosphorous) accumulation degrades the pond water system. Interactions between various *N*-species are complex and difficult to integrate. Modeling can improve our ability to evaluate this complex system. Jamenez-Montealegre (2002) integrated existing knowledge about nitrogen transformations in fishponds into a model that calculated the amount of various *N*-compounds in the water column and in the sediment. The model was also used to gain insight into the relative importance of transformation processes between the various *N*-compounds. This better understanding of the biochemical processes in ponds facilitated management. Montoya et al. (2002) developed another simulation model of nitrogen dynamics, in shrimp culture systems. Modeling of nitrogen and phosphorus exchanges at the sediment–water interface of an intensive fishpond system was also done by Lefebvre et al. (2001).

A simulation model to analyze the water flow and sediment transport in aquaculture raceways was developed by Huggin et al. (2004 and 2005) using a computational fluid dynamics (CFD) software package. The simulation was used to evaluate the efficiency of solids settling in the quiescent zone of existing trout raceways. This efficiency was based on the percentage of solids removed, which corresponded to the percentage of solids introduced into the raceway. (with settling taking place primarily in the quiescent zone). In commercial channel catfish ponds, a model of sediment oxygen demand was developed by Steeby et al. (2004).

Models were also used to optimize allocation of resources in the activity of aquaculture ponds by using multiple criteria decision making (MCDM) (El-Gayar and

Leung, 2001), and operational research (OR) (Bjorndal et al., 2004). These models sought a desirable allocation of resources and activity levels that would strike an acceptable balance among the various development goals under consideration subject to resource constraints, market constraints, and pollution constraints. Leung and Rowland (1989) developed another financial model in aquaculture.

An integrated model intended for aquaculture pond planning and management as Decision Support Systems (DSS) was developed by Bolte et al. (2000), Nath (1996), and Ernst et al. (2000). Decision Support Systems integrate knowledge in the form of mathematical models, rule-based ("expert") systems, and/or databases into user-friendly software systems focused on developing, analyzing and optimizing management strategies. Decision support systems are potentially valuable tools for assessing the economic and ecological impacts of alternative decisions on aquaculture production. This approach was then developed into a Decision-Support Software, called AquaFarm, which provided: (1) simulation of physical, chemical, and biological unit processes; (2) simulation of facility and fish culture management; (3) compilation of facility resource and enterprise budgets; and (4) a graphical user interface and data management capabilities (Ernst et al., 2000).

Among the available aquaculture pond simulation models, water management modeling has gotten little attention in spite of its potentially important role in groundwater conservation, enhancing rainwater harvesting, and minimizing environmental pollution. There has been research on pond water budget and management but it has not been incorporated into simulation models (Boyd and Gross, 2000; Green and Boyd, 1995). Reducing groundwater use and increasing rainwater usage in ponds could significantly reduce pumping cost. In addition, reducing groundwater pumping would prevent adverse effect of groundwater overexploitation, thus maintaining the sustainability of the groundwater resource. Moreover, water management modeling and effective management can be used to evaluate effluent discharge minimization from fishponds, which in turn reduces environmental pollution to receiving waters. Since some researchers feel that treating pond effluent is not feasible (Boyd and Queiroz, 2001; Tucker et al., 1996), the effort to minimize effluent discharge seems a promising solution.

Nath and Bolte (1998) developed a short-term simulation model of water budget in a watershed-type aquaculture pond. This descriptive model was verified by using available data from existing aquaculture ponds. Pote et al. (1988) and Pote and Wax (1993), simulated drop-add water management strategies in embankment-type catfish ponds, by using long-term climatologic data. As a long-term simulation, verification was not performed. Tucker et al. (1996) integrated the model developed by Pote et al. (1988), Pote and Wax (1993) with constituent concentration to simulate mass discharges from catfish ponds. A mathematical model was also used by Cathcart et al. (1999) to simulate the drop-add water management strategies in linked embankment pond systems. Another model, in spreadsheet form, to develop water resources for aquaculture ponds in developing counties was developed by Tollner et al. (2004).

CHAPTER III

METHODOLOGY

3.1 Data Processing and Site Selection

Forty five year (1961-2005) data sets of precipitation and pan evaporation from the National Weather Service were used in this research. Pan evaporation refers to evaporation measurement by using the National Weather Service Class A pan. The 5 selected locations in the southeastern US, (the major production areas for catfish) were Fairhope, AL; Clemson, SC; Stoneville, MS; Stuttgart, AR; and Thomsons, TX (Figure 1). These locations were selected to represent variability within the southeastern subtropical humid climate, which is affected by the continent middle latitude climate, the maritime tropical climate, and the subtropical dry-summer climate. The subtropical humid climate in the southeast US is much affected by the cool middle latitude in winter (from the north) and by warm tropical air in summer (from the south). The precipitation gradient is pronounced, being high in the south and low in the north. Some southern coastal regions show maximum summer rainfall, while inland regions show maximum winter rainfall. The southeastern region also shows an east-west precipitation gradient which has high precipitation in the east and low precipitation in the west due to the subtropical dry-summer climate found in Texas (Trewartha, 1968).

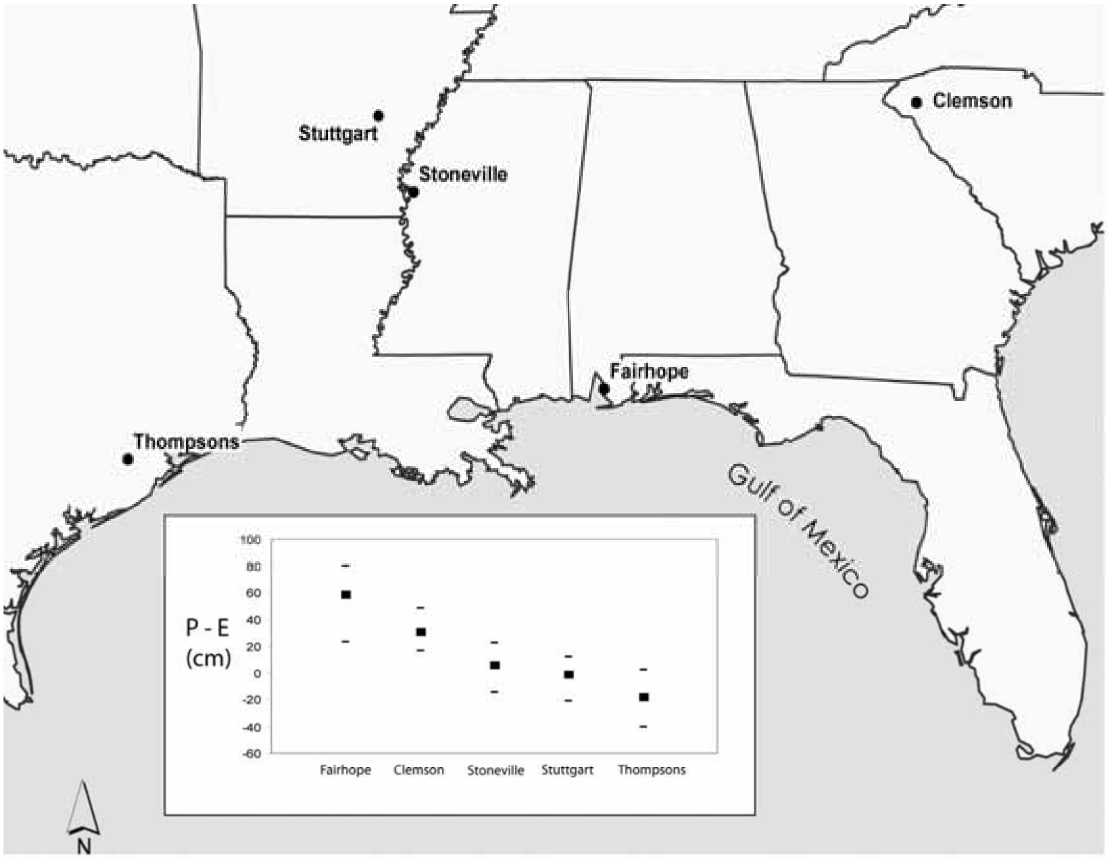


Figure 1. Locations and P-0.8E values for the locations used in this study.

The rainfall data are complete from the 5 chosen locations. The evaporation data have missing records, errors of transcription, and measurement errors (the latter two identified by their unrealistic magnitude). Missing data were estimated using the method described in Cathcart et al (2007). The missing data were filled using linear interpolation for one or two blank spaces or by using average of all previous records for more blank spaces. For blank spaces followed by combined records, the combined record was distributed evenly to fill the blanks. A combined record was identified as an extremely high value following blank spaces. For very long blank records, missing data was substituted using data from next nearest location with the same time periods. For example for Stuttgart, AR; long missing evaporation data was substituted by using evaporation data from Keiser, AR. For Thomsons, TX; a long list of missing evaporation data was substituted using evaporation data from Beaumont, TX. By doing this correction, the precipitation and evaporation data sets from the five locations were completed and used for simulation.

3.2 Pond Water Balance Model

A mathematical model of mass balance used in this research is similar to those used by Cathcart et al. (1999), and Pote and Wax (1993). The model is used to simulate sources and sinks of water for embankment-type catfish ponds in the five selected locations; Fairhope, AL; Clemson, SC; Stoneville, MS; Stuttgart, AR; and Thomsons, TX.

The following model of pond water depth approximates mass balance with assumptions of constant density and constant area of pond surface:

$$\frac{\Delta H}{\Delta t} = PR + GW - E - I - D \tag{1}$$

Where;

$\Delta H/\Delta t$: change in pond water depth per time step (cm/day)
PR : precipitation (cm/day)
GW : groundwater Use (cm/day)
E : evaporation (cm/day)
I : infiltration/seepage (cm/day)
D : overflow or intentional discharge (cm/day)

Pond water depth increases by precipitation and groundwater pumping. Groundwater is pumped if pond water depth drops below the minimum allowable level. Pond water depth decreases due to evaporation, infiltration, and overflow or intentional discharge. Overflow occurs when rainwater depth exceeds the pond overflow structure or pond storage capacity. Intentional discharge may occur when harvesting the fish crop and during pond levee repair. Discharge for repair is not incorporated in this simulation. When harvesting pond water depth is emptied to a depth of 1 m.

Rainfall depth can be plugged directly to the Equation 1, while pan evaporation is corrected by using a pan coefficient of 0.8 in order to estimate evaporation of pond water. This constant is a common average used for fish pond evaporation (Boyd, 1985). The pan coefficient is not really constant, but varies seasonally. Boyd (1985) found that the pan coefficient varied from 0.71 to 0.90 for experimental catfish ponds at Auburn, AL. In this research, a pan coefficient of 0.8 was used to estimate pond water evaporation. This is consistent with the work of Pote at al, (1988), Cathcart et al. (1999), and Cathcart et al. (2007).

Infiltration rate or seepage is normally measured by measuring change of pond water depth and correcting for pond evaporation loss on rainless days. Infiltration from pond bottoms is affected by seasons, soil types, and construction process (Davis et al., 2002). It was reported that daily infiltration rates of 1000 m^2 experimental fish ponds in El Carao (Honduras) ranged from 0.11 to 0.43 cm in rainy season and 0.06 to 0.60 cm in dry season (Green and Boyd, 1995). Pan coefficient of 0.81 was used for the evaporation loss correction, to estimate the infiltration rates. In other research from the same location (El Carao, Honduras) and 375 m^2 ponds at AIT Bangkok Thailand, infiltration rates were estimated 0.58 and 0.44 cm/day respectively with reduction of 25% during rainfall events (Nath and Bolte, 1998). Following FAO (1977) guideline these Authors used a pan coefficient of 0.75 in their research. In Gualaca (Panama), infiltration rates of new constructed ponds ranged from 1.9 to 5.8 cm/day and averaged 3.1 cm/day (Teichert-Coddington et al., 1988). Pond bottoms were lined with imported 30% clay soil by 10-15 cm. The imported soil proved to be permeable because soil particles were coated with iron and aluminum oxides resulting in a granular soil structure (Teichert-Coddington et al., 1988). Then infiltration reduction was performed by using chicken litter. After the chicken litter application infiltration ranged from 0.8 to1.7 cm/day or reduction of 54-76% was achieved (Teichert-Coddington et al., 1989). Pan coefficient of 0.83 was used for pond evaporation correction in the research.

Properly constructed ponds seldom have infiltration rates above 0.5 or 0.6 cm/day (Boyd and Gross, 2000). High infiltration rates are usually associated with sandy soils or failure to install seepage reduction measures in ponds during construction. Boyd (1986) categorized infiltration rates in ponds as: 1. low (0-0.5 cm/day), 2. moderate (0.5-1.0 cm/day), 3. high (1.0-1.5 cm/day), and 4. extreme (>1.5 cm/day). Zero infiltration can be achieved by using proper bottom liner (Boyd, 1985). In other research, Parker et al. (1999) found that infiltration rates from unlined beef cattle feedlot runoff storage ponds ranged from 1.11 cm/day (no sludge accumulation) to 0.5 cm/day (22 years of sludge accumulation), with 49 to 73% of total infiltration through the sidewalls. This fact indicates that pond bottoms tend to self seal over time.

In the above research, variations of pond infiltration were affected by soil type, liner used (engineering control), seasons, and age. All the cited research was conducted at small ponds ($\leq$ 1000 ^{2}m) so that the effect of side inflow from levees was seasonally pronounced. For catfish ponds, the effect of side inflow from levees can be expected to be small given that catfish ponds have a very large surface area as compared to the perimeters. Hence, assuming constant infiltration rates of catfish ponds for long period is reasonable. Pote and Wax (1993) used constant infiltration rates of 0 and 0.2 cm/day, while Cathcart et al. (1999) used infiltration rates 0 and 0.1 cm/day. These low infiltration rates were reasonable given that most catfish ponds in the Mississippi Delta have been used for years and that many of the ponds were constructed over thick clay deposits. In other research performed by Tucker et al. (1996), infiltration rate of catfish pond bottoms was assumed to be 0.04 cm/day, the same as values reported from local rice fields.

Considering that infiltration rate is related to engineering controls and pond age, this research elaborates a wider range of infiltration rates: from zero to 0.5 cm/day. Self sealing in the long term causes older ponds to approach zero infiltration. Moreover, Boyd (1985) had nearly zero infiltration when he investigated pond evaporation rates in his research. On the other hand, high infiltration rate of 0.5 cm/day is explored because of the fact that newly constructed ponds tend to have higher infiltration rates and that higher rates are characteristic in other regions of the US southeast. Rates higher than 0.5 cm/day are not investigated because such ponds are lined improperly (Boyd and Gross, 2000), and are not feasible for industry.

3.3 Simulation

The simulation calculates daily change of pond water depth in Equation 1 and pond water depth using Euler's forward numerical integration technique as the following;

$$H_{i+1} = H_i + \frac{\Delta H_i}{\Delta t_i} \tag{2}$$

Where: H_{i+1} and H_i correspond to pond water depth at day i+1 and day i.

The simulation is run by inputting forcing variables; precipitation and evaporation, while infiltration rates are held constant. The simulation mimics pond water dynamics, up and down. When precipitation exceeds the pond storage capacity, overflow occurs. When pond water below the minimum level, groundwater is pumped.

Assumptions used in this simulation include:

1. Pond shape is rectangular (constant surface area), in order to simplify the calculation of water depth changes
2. Pond surface area is 1 ha.
3. At initial condition (time "0"), ponds are full of water. This is quite realistic for January 1.
4. When rainfall exceeds the maximum depth on day i, overflow occurs on day i+1.
5. When pond water drops below the minimum level on day i, groundwater is pumped on day i+1.
6. Pond bottoms are above water table, in order to get negative infiltration.
7. Water quality is homogeneous for the entire pond.

The model includes appropriate safeguards, such as a mass conservation, to detect a miscalculation. Simulation is similar to that in Cathcart et al. (1999), written in VisSim 4.0 package program and spreadsheet for some calculations.

Various scenarios with wide ranges of drop depths are used to test various management schemes in this research. This is to investigate the possibility of using extensive drop depths in drop/add management schemes as adjusted to local climates. Amounts of annual groundwater use, effluent discharge, and percentage of rainwater harvested are recorded and then averaged annually. Total numbers of groundwater pumping and effluent discharge events (in days) are also recorded and split seasonally. Most of the simulations reported here deal with unlinked pond systems. Some simulations of linked pond systems are reported as well.

3.4 Scenarios and Analysis

3.4.1 Climate Characteristics

This analysis is intended to characterize the location climates relative to each other. Precipitation and evaporation data set from each location were compared in order to describe variability among the locations. Annual, seasonal, and monthly accumulations were determined. In order to identify the possibility of long-term climate change, 3 blocks of a 15-year data set from each 45 year data set were built and compared. Spatial variability of the climate was determined from east to west locations (across South Carolina-Mississippi-Texas) and from south to north locations (across

Alabama-Mississippi-Arkansas). Analyses were quantified using ANOVA followed by LSD multiple comparisons, as necessary, in SAS.

3.4.2 Pond performance

Pond performance was analyzed based on the records of annual groundwater use, effluent discharge, and percentages of rainwater stored for each drop/add management scheme. Percentage of rainwater stored was calculated using the difference between amount of annual rainfall and annual effluent discharge. Pond performances with various drop/add management schemes and infiltration rates at each of locations were compared. Add or fill depths of 7.5 cm (3 inches) were used in all simulations. This value was recommended by (Pote and Wax, 1993). Their recommendation was based on the following: if farmers use a pump having a capacity of 1000 GPM (Steeby and Avery, 2002), it takes about one day to add 7.5 cm of water to a pond having an area of 5 ha.

Extensive drop depths were used in the simulations to explore the range of pond performances with various management schemes. When drop depth is increased, groundwater use and effluent discharge decrease (rainwater storage increases). The improvement in performance approaches a limit asymptotically and then flattens when the limit is reached. This limit represents the best performance (minimum groundwater use and effluent discharge) possible for the simulation at that geographic location. Drop depths were varied from 15 cm (6 inches), which was initially recommended by (Pote and Wax, 1993), to the drop depth at which performance no longer improved.

3.4.3 Sensitivity Analysis

Sensitivity analysis is the general term used to describe analyses conducted to determine variables to which a model is sensitive. Sensitivity analysis techniques can be grouped into two categories: deterministic or single-variable, and stochastic (Jesiek and Wolfe, 2005). Various techniques have been developed to quantify sensitivity of the model using the deterministic approach. The most basic method is differential analysis, or the direct method, determined from simple partial derivatives (Jesiek and Wolfe, 2005). When solving the partial differential is not possible, relative sensitivity can also be calculated for any point. In the deterministic or single-variable approach, only one value is given for each input parameter and the analysis is for one scenario (Ma et al., 1998).

In most sensitivity analyses, a base set of inputs is selected which is used to generate a corresponding base set of outputs. The inputs are individually varied over some reasonable range and the effect on the output is presented numerically or graphically. The following index representing the ratio of relative ranges of output and input (Baffaut et al., 1997; Walker et al., 2000) is used in this research,

$$S = \frac{\left(\dfrac{O_2 - O_1}{O_{avg}}\right)}{\left(\dfrac{I_2 - I_1}{I_{avg}}\right)} \tag{3}$$

Where,

S	= the relative sensitivity index
I_1, I_2	= smallest and greatest input values tested for a given parameter
I_{avg}	= average of I_1 and I_2
O_1, O_2	= model output values corresponding to I_1 and I_2
O_{avg}	= average of O_1 and O_2, corresponding to I_{avg}

An index of 1 indicates that the output ranges about the average output to the same degree as the tested input ranges about the average input. A negative value indicates that input and output are inversely related. The greater the absolute value of the index, the greater the impact an input parameter has on a particular output. Because it is dimensionless, S provides a basis for comparison with other input variables. A drawback is that the index is undefined if average output is zero.

3.4.4 Effect of Intentional Harvest Discharge

Portion of water is discharged intentionally from catfish ponds when harvesting the fish crop to lower pond water depth to about 1 m, so that the harvesting work can be done easier. It is because the limited depth of the nets/seines that farmers use. Intentional harvest discharge may significantly increase groundwater use, increase effluent discharge, and lower rainwater harvested with the same volume.

This scenario is to determine the increases of annual groundwater use and effluent discharge and decrease of annual rainwater stored when a single harvest date is scheduled once per year. The harvest time is scheduled once per year and varied at each end of months throughout one year, so that there are 12 possibilities of harvest dates. In the 12 scenarios, the increases of annual groundwater use and effluent discharge, also increase of rainwater stored, are compared. The increases of annual groundwater use and effluent discharge, and decrease of rainwater stored are calculated by running the simulation with and without incorporation of the single harvest per year for 45 years.

Another technique to save the harvest discharge is to transfer the water to an adjacent pond, so at least there is a pair of two ponds. When harvesting the first pond, the pond is emptied to 1 m deep and the water is transferred to the second pond. Similarly when harvesting the second pond, the second pond is emptied to about 1 m deep and the water is transferred to the first pond. This scenario is investigated if there is some reductions of groundwater use and effluent discharge compared to the harvest discharge without transfer.

3.4.5 Mass Discharge of Water Constituents

Mass discharge of water constituents is estimated by multiplying concentrations of selected water quality variables and volume of effluent discharge. Simulation is run for 45 years, and annual mass discharge is recorded and averaged. Two management schemes are compared in this scenario. One is 15/7.5 (6/3 inch) as recommended in previous research (Pote and Wax, 1993; Pote et al., 1988), for catfish ponds at the Delta Mississippi. The other scheme is one selected in the scheme selection section. Data for selected water quality variables from previous research at the Delta Mississippi (Tucker and Hargeaves, 2003a; Tucker et al., 1996) are used for the comparison. It is assumed that the water quality data is applicable for all locations and over a long time period since other research revealed that water quality of catfish pond does not change appreciably over time (Zimba et al., 2003). From other locations, research at Auburn, AL (Seo and Boyd, 2001) also showed comparable concentrations of catfish pond water quality.

3.4.6 Linked Pond System

The linked pond system was developed in previous research for catfish ponds at the Delta Mississippi (Cathcart et al., 1999). This system consists of production ponds (shallower) and production/storage ponds (deeper). The relative numbers of each pond type depends on the configuration used. A 1:1 configuration refers to one production pond linked to one production/storage pond. The use of a production/storage pond allows additional water storage with minimal pond modification. The idea behind the use of linked pond system is that there is no need to modify all ponds when trying to increase water storage capacity if this system is implemented to existing ponds. Performances of linked pond system with 1:1 configuration are simulated for 45 years over each of the locations. Groundwater use, effluent discharge, and percentage of rainwater harvested are recorded annually and then averaged. In this scenario, many management schemes over ranges of infiltration rates (0-0.5 cm/day) are also compared. Potential problems associated with implementing linked pond systems when infiltration rates are high are also discussed.

CHAPTER IV

RESULTS AND DISCUSSION

4.1 Climate Characteristics

4.1.1 Annual Records

Fairhope, AL; Clemson, SC; Stoneville, MS; Stuttgart, AR; and Thomson, TX were selected in this research to describe regional variability of climates (based on precipitation and evaporation) in the southeastern states that have catfish production. The climate difference is tested by using one-way classification analysis of variance (ANOVA) followed by multiple comparisons using least significant different (LSD) at $\alpha = 0.05$. SAS with the GLM procedure is used in the statistical analysis, and the outputs are put in Appendix A.

Table 2. Annual Precipitation

Location	Num. of Obs.	Mean (cm)*	Std. Dev.	Maximum		Minimum	
				cm	Year	cm	Year
Fairhope	45	168.96^{a}	33.04	238.9	1978	103.3	1968
Clemson	45	136.52^{b}	27.34	181.7	1975	91.3	1981
Stoneville	45	133.14^{bc}	27.34	182.2	1961	91.7	1981
Stuttgart	45	123.08^{cd}	22.48	172.2	1990	88.4	1988
Thomsons	45	117.75^{d}	27.14	175.4	1997	60.3	1988

*) Means with different letters are significantly different at α=0.05.

Table 2 contains annual average precipitation, standard deviation, maximum, and minimum of precipitation at each location for each 45 year record. Mean in Table 2 is average precipitation per year. The annual precipitation among the locations is significantly different at level $\alpha=0.05$. P value < 0.0001 (Appendix A) indicated that the significance is very strong. Fairhope has the highest precipitation, but some other locations have no significant difference. For example, annual precipitations between Clemson and Stoneville, and Stoneville and Stuttgart are not significantly different. Stuttgart and Thomsons are not significantly different either with respect to the annual precipitation.

Although some pairs of locations were not significantly different, precipitation gradient was still visible, high in the south and the east (Fairhope and Clemson); and low in the north and the west (Stuttgart and Thomsons). This is a typical subtropical climate in a continent. Tropical wind from the south brings moisture to the continent; result in moist climate in the southern parts on a continent. The subtropical climate in a continent is also characterized by humid in the eastern parts and dry in the western parts (Trewartha, 1968).

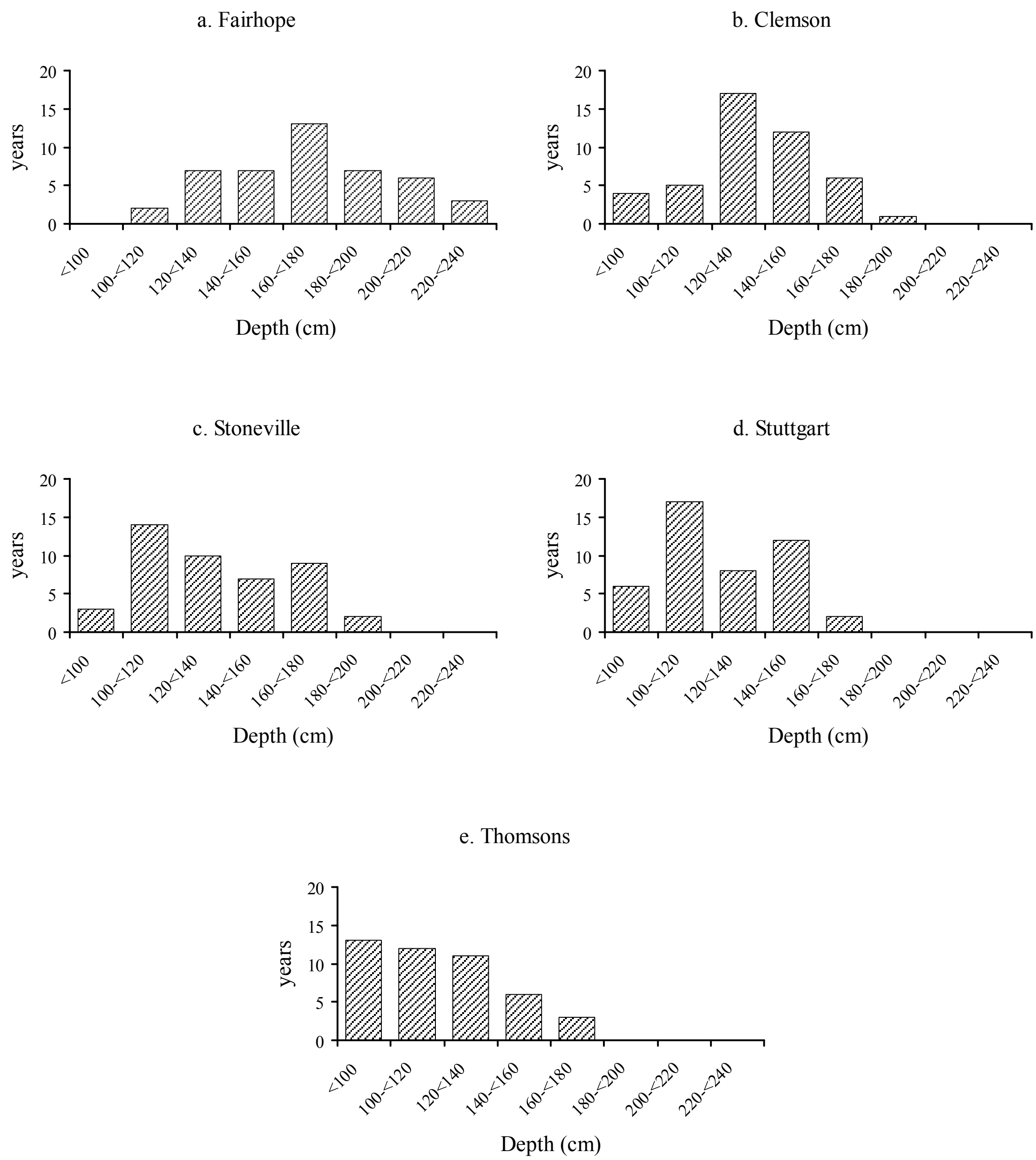


Figure 2. Histogram of Annual Precipitation at Each Location

Figure 2 shows histograms of the annual precipitation at each location. From Figure 2 we can see that histograms of Fairhope and Clemson are relatively close to normal, while those of the other three locations have slightly longer tails to the right as represented by the standard deviations and skew coefficients. More variability of annual precipitation with 7 intervals at Fairhope is also visible as compared to other locations which have fewer intervals (5 or 6). There is a peak bar of 13 years at the 160-180 cm interval and relatively homogeneous heights for other intervals at Fairhope. Clemson has various heights and a peak of 17 years at 120-140 cm interval, while other locations (Stoneville, Stuttgart, and Thomsons) have various heights and a peak at lower depth intervals.

Table 3 shows annual P-0.8E as well as the wettest and driest values for the 45 year period. The mean is the average of cumulative P-0.8E per year. The P-0.8E is calculated as precipitation minus 0.8 of pan evaporation. The pan coefficient of 0.8 is used because, as discussed earlier, it is the typical average for pond evaporation as compared to pan evaporation. SAS outputs of means, one-way ANOVA, and LSD comparisons are put in Appendix B.

Table 3. Annual Accumulation of P-0.8E

Location	Num. of Obs.	Mean (cm)*	Std. Dev.	Wettest		Driest	
				cm	Year	cm	Year
Fairhope	45	58.5[a]	34.8	133.5	1978	-15.5	2000
Clemson	45	28.4[b]	26.7	82.8	1975	-30.4	2000
Stoneville	45	8.1[c]	29.2	58.5	1974	-31.0	1965
Stuttgart	45	-0.9[c]	24.9	49.7	1990	-40.7	1986
Thomsons	45	-15.3[d]	31.9	52.2	1973	-86.2	1988

*) Means with different letters are significantly different at $\alpha=0.05$

Among the locations, means of the annual P-0.8E significantly differ at level $\alpha=0.05$, with $p < 0.0001$ (Appendix B), except for Stoneville and Stuttgart. Fairhope has the wettest P-0.8E while Thomsons has the driest P-0.8E. The east to west gradient represented by Clemson-Stoneville-Thomsons is clear, changing from wet at Clemson to dry at Thomsons. This gradient reflects change of the subtropical humid climate in the east to the subtropical summer dry climate in the west. The south to north gradient may reflect the effect of wet maritime air in the south and dry continent air in the north. The south to north gradient represented by Fairhope-Stoneville-Stuttgart is also visible, even between Stoneville and Stuttgart is not significant. It could be because Stoneville and Stuttgart are relatively close to each other, as compared to other locations. Regardless of this lack of significance, using P-0.8E as the variable to describe the spatial variability of climate is more appropriate than using precipitation alone.

Figure 3 shows frequency histograms of annual P-0.8E at each location. Just as in Figure 2, Fairhope has the highest variation of P-0.8E among the locations. In the bar graph, this shows up as a greater number of intervals and relatively homogeneous bar heights for each depth interval. It is also apparent that the graph for Fairhope is shifted to the right relative to the other locations. Clemson, which is the second wettest location, has a peak bar at 20-40 cm depth interval and more variations of bar heights. Other drier locations (Stoneville, Stuttgart, and Thomsons) have peaks years at lower depth intervals. Thomson which has the second highest variance (square of standard deviation) of P-0.8E is represented by 8 intervals of depths. However, statistically the variances may not be significantly different.

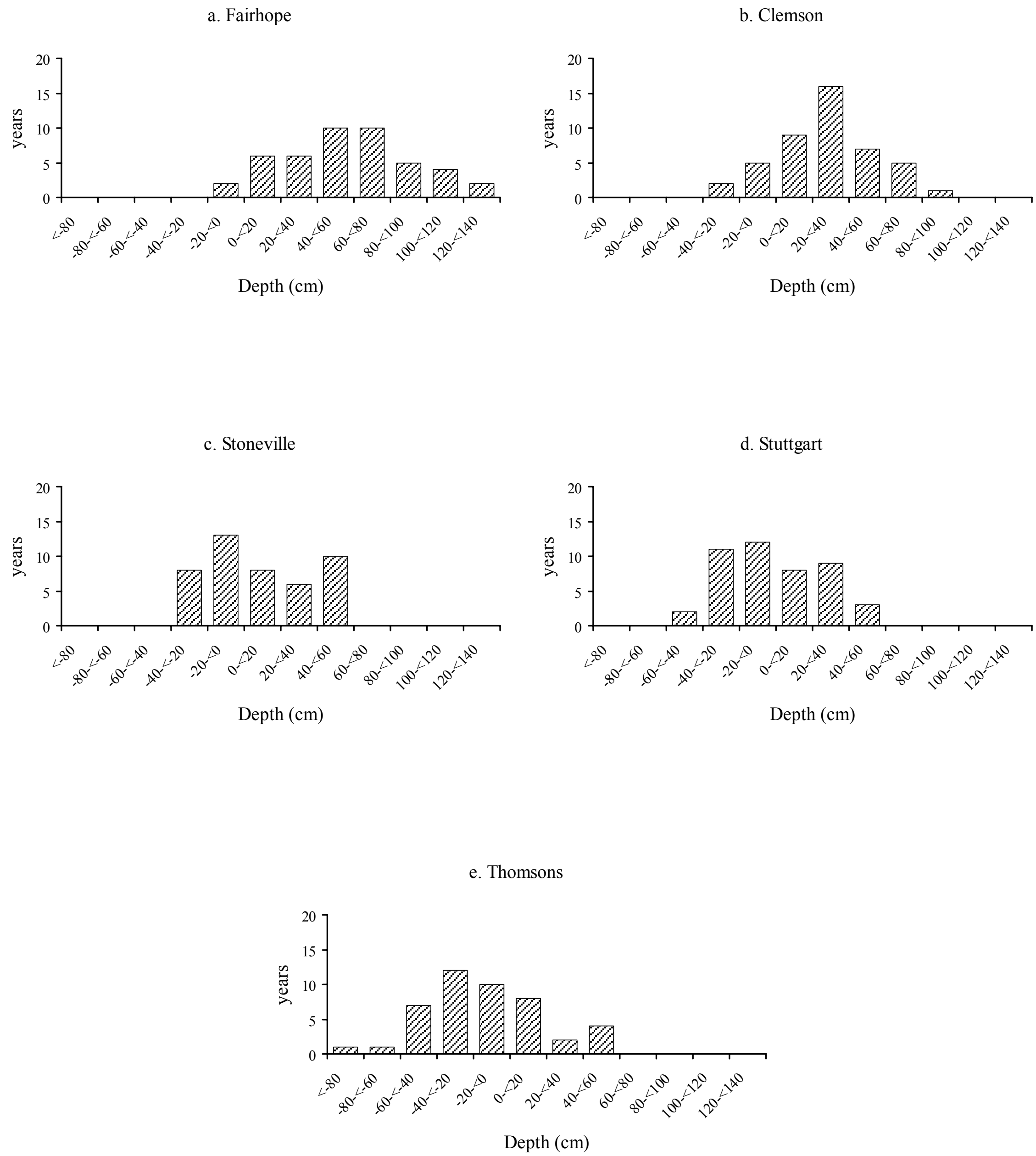


Figure 3. Histogram of Annual P-0.8E at Each Location

With regard to pond water management, the above climatologic data suggest that it will be easier to reduce groundwater use and more difficult to reduce effluent discharge at Fairhope than at Thomsons. Conversely, it will be easier to reduce effluent discharge and more difficult to reduce groundwater use at Thomsons than at Fairhope.

4.1.2 Seasonal Records

Table 4. Seasonal Average of 45 Year Precipitations

Location	Num. of Obs.	Mean (cm) [1,2,3,4,5)]			
		Spring	Summer	Fall	Winter
Fairhope	45	40.1^{bA} (18.4)	53.5^{aA} (15.4)	33.4^{cA} (14.1)	41.9^{bA} (14.2)
Clemson	45	33.4^{bB} (11.6)	33.0^{bB} (12.2)	31.6^{bA} (10.7)	38.6^{aA} (10.3)
Stoneville	45	35.7^{aAB} (12.5)	25.1^{bC} (9.7)	33.7^{aA} (12.4)	38.6^{aA} (12.2)
Stuttgart	45	35.9^{aAB} (11.0)	23.5^{bC} (8.6)	32.4^{aA} (12.5)	31.3^{aB} (10.3)
Thomsons	45	30.6^{aC} (13.3)	34.1^{aB} (16.2)	29.5^{aA} (13.3)	23.6^{bC} (10.6)

1) Means with different letters are significantly different at $\alpha=0.05$
2) Lowercases represent horizontal comparisons among seasons in each location
3) Uppercases represent vertical comparisons among locations in each season
4) Values in parentheses represent standard deviation
5) Location and season significantly interact with p value < 0.0001

Table 4 shows seasonal averages for the 45 year precipitation record for each location. Spring is from day 80 to 171, summer from day 172 to 263, fall from day 264 to 354, and winter from day 1 to 79 and day 355 to 365. SAS outputs of two-way classification analysis of variance and LSD multiple comparisons using GLM procedures are put on Appendix C. Location and season significantly interact with respect to precipitation at level $\alpha=0.05$. The significance is very strong as indicated by P value <0.0001 (Appendix C). Complete LSD multiple comparisons of the interaction effects on Appendix C are selected for comparisons among seasons within each location (horizontal) and among locations within each season (vertical); and presented on Table 4. Each location shows specific seasonal pattern of precipitation, but some show similarity. For example, Fairhope and Thomsons which are in coastal region have high precipitations in summer. Clemson, Stoneville, and Stuttgart which are far from the coast have high precipitation in winter. Both Stoneville and Stuttgart have low summer precipitations which are typical for in land region.

Vertical comparison is among locations within each season with respect to the seasonally precipitation. Each season also show some different patterns among the locations. In spring, Fairhope is the highest and Thomsons is the lowest with respect to precipitation. In summer, even Thomsons is dry location, precipitation is not significantly different from Clemson. In fall, all locations are not significantly different. In winter precipitation is high in Fairhope, Clemson, and Stoneville; and low in Thomsons.

Table 5. Seasonal Average of 45 Year P-0.8E

Location	Num. of Obs.	Mean[1,2,3,4,5] (cm)			
		Spring	Summer	Fall	Winter
Fairhope	45	3.1^{cA} (19.1)	16.3^{bA} (16.4)	13.2^{bA} (15.3)	25.9^{aA} (14.7)
Clemson	45	-2.3^{cAB} (13.2)	-6.3^{cB} (14.5)	13.9^{bA} (11.5)	23.1^{aA} (10.2)
Stoneville	45	-8.5^{cC} (14.0)	-20.3^{dC} (10.8)	12.9^{bA} (12.8)	24.0^{aA} (12.1)
Stuttgart	45	-5.5^{bBC} (13.0)	-22.1^{cC} (10.3)	11.6^{aA} (13.1)	15.1^{aB} (10.6)
Thomsons	45	-10.3^{bC} (15.2)	-11.1^{bB} (18.3)	3.6^{aB} (15.0)	2.6^{aC} (11.6)

1) Means with different letters are significantly different at α=0.05
2) Lowercases represent horizontal comparisons among seasons in each location
3) Uppercases represent vertical comparisons among locations in each season
4) Values in parenthesis represent standard deviation
5) Location and season significantly interact with p value < 0.0001

Table 5 presents seasonal averages for the 45 year P-0.8E record from each location. Location and seasonal significantly interact with respect to P-0.8E, with p value <0.0001. Two-way classification analysis of variance and complete LSD multiple comparisons are in Appendix D. The LSD multiple comparisons of interaction effects are selected for among seasons within location (horizontal) and among locations within season (vertical), and presented on Table 5. In general, all locations have significantly the high P-0.8E's in winter and low in summer. Except for Fairhope, all locations have negative P-0.8E's (deficit) in spring and summer. Fall P-0.8E's are significantly higher than spring P-0.8E's for all locations.

Among the locations, within each season (vertical comparisons), P-0.8E's shows some different patterns. Fairhope has consistently the highest or high P-0.8E's in all seasons, while Thomsons has consistent the lowest P-0.8E's in fall and winter. In spring, P-0.8E's of Stoneville, Stuttgart, and Thomsons are not significantly different. In summer, among the locations, Stoneville and Stuttgart have significantly the lowest P-0.8E's. Figure 4 shows histogram of seasonal P-0.8E at each location.

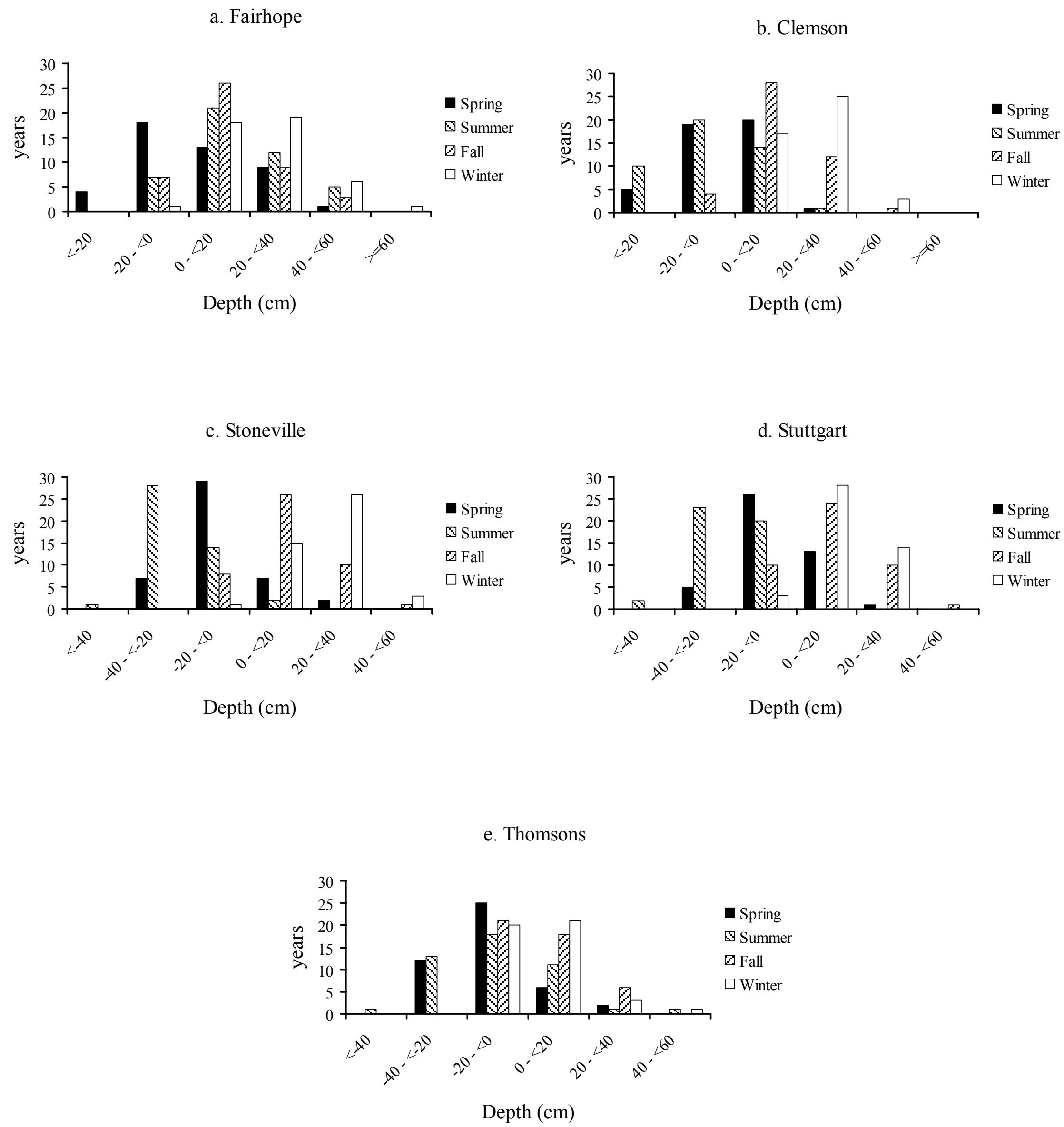


Figure 4. Histogram of Seasonal P-0.8E at Each Location

An initial look at the mean seasonal P-0.8E records suggests that the drop/add approach may have limited effect on effluent reduction at Fairhope, since P-0.8E is positive for the majority of years at all seasons. There is not a reliable P-0.8E "deficit" within which to store surplus from other seasons. Similarly, Thomsons initially appears not be a good candidate for groundwater use reduction, since the average precipitation surplus is not large at any season. Conversely, it would appear that properly designed ponds at Fairhope should rarely require groundwater and properly designed ponds at Thomsons should rarely release effluent.

At Clemson, spring and summer show an average P-0.8E deficit of 8.6 cm and a fall-winter surplus of 27 cm. This suggests that, based on seasonal averages, it will be relatively easy to design a pond for low groundwater use and more difficult to limit effluent release.

Stoneville and Stuttgart have spring-summer deficits that are very close in magnitude to their fall-winter surpluses. This suggests that both of these sites are good candidates for both reduced groundwater use and effluent release.

4.1.3 Monthly Records

Figure 5 shows mean monthly precipitation and pond evaporation (0.8E) at each location for the 45 year record. Monthly precipitation or evaporation is the total of cumulative daily precipitation or evaporation within each one month period. To get a single value for every month, monthly values are averaged. Thus the monthly precipitation and evaporation refer to averages of 45 observations.

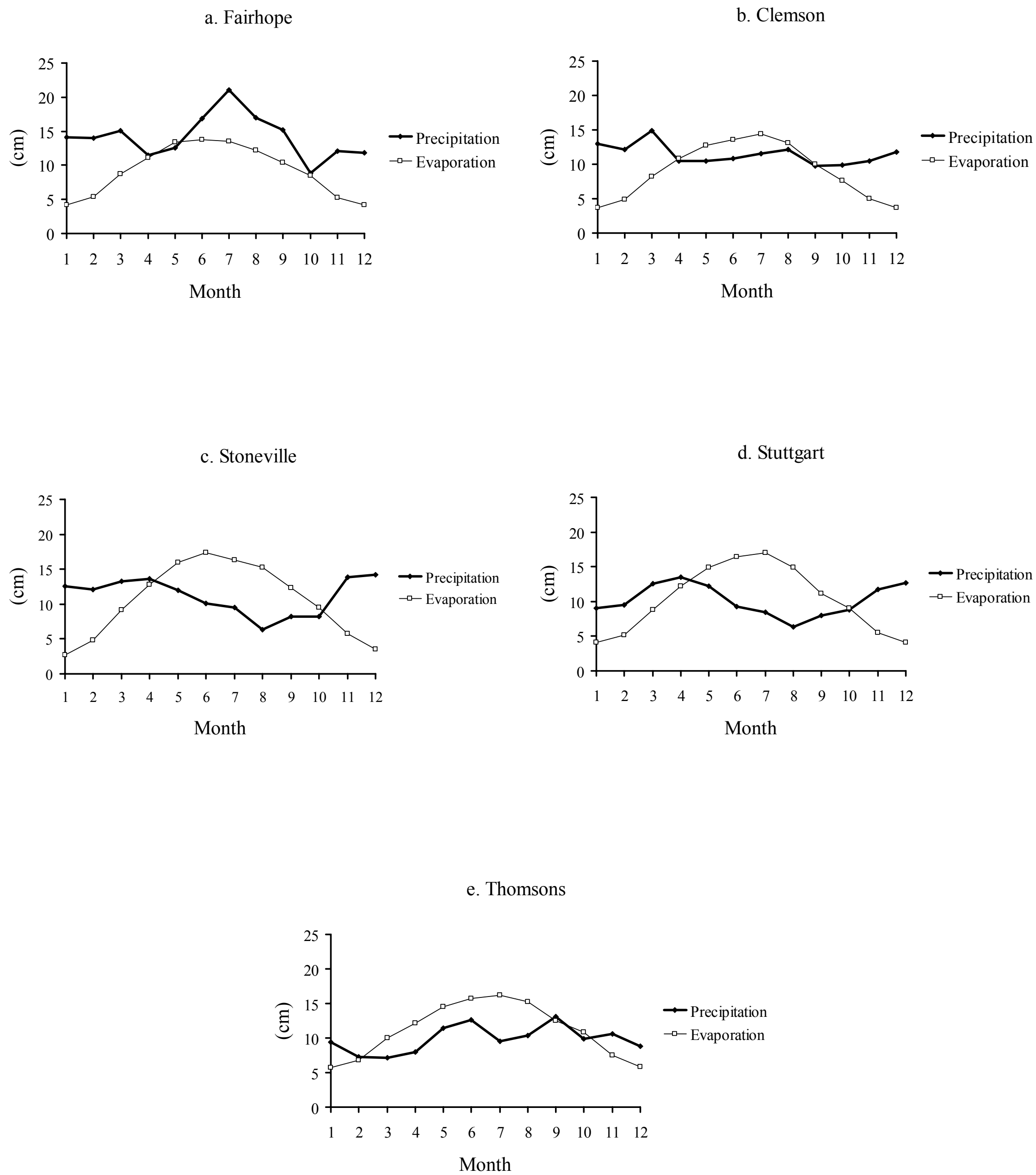


Figure 5. Monthly Precipitation and Pond Evaporation at Each Location

Precipitation of each location shows a specific pattern. At Fairhope (Figure 5a), precipitation spikes in summer; while at Clemson (Figure 5b) precipitation distribute almost evenly with a little peak in winter. Stoneville and Stuttgart (Figures 5c and 5d) show almost similar patterns of precipitation, high in winter and low in summer. Thomsons (Figure 5e) shows more homogeneous precipitation with little low in winter. Evaporation, however, looks similar across all locations, low in beginning and end of year and high in summer

Comparing monthly P to 0.8E provides a partial insight into the problem of reducing groundwater use and effluent release at these locations. The areas of the graphs between the P and 0.8E curves represent either the precipitation surplus to be stored (if $P > 0.8E$) or the deficit to be filled (if $P < 0.8E$). At Stoneville and Stuttgart, these areas are very nearly equal. Hence, if the problem was merely to design for average conditions, groundwater use and effluent release could be reduced commensurately. At Clemson, it is clear that the best that could be accomplished would be to design for zero groundwater use. The best discharge reduction possible would be that amount used to supply water needs during dry periods. At Thomsons, it appears unlikely that ponds could be designed for zero groundwater use. The best that could be accomplished would be to use all of the monthly surplus to meet part of the water needs. Based on the averages, it would appear that Fairhope should never need to use groundwater and a drop/add scheme would serve no purpose.

As mentioned, looking at averages provides only a partial view of the problem. Both additional complexity and additional opportunities occur because of the inherent variability of P-0.8E in the pond system. Seasons and months can be wetter or dryer than normal. So designing for average conditions may provide too little storage capacity during wet years and too much storage capacity during dry years. Additionally, precipitation is not evenly distributed throughout the month or season. Much of the precipitation for a given time period may occur over just a few days. If storage capacity is not adequate to retain precipitation from peak events, then much of the water resource may be lost.

4.1.4 Evaluation of P-0.8E in 15-Year Blocks

The 45 year records of P-0.8E's were divided into 15 year blocks (1961-1975, 1976-1990, 1991-2005) to see whether observable changes in P-0.8E occurred over time. Annual P-0.8E is then broken down into four seasons. Each of the seasonal P-0.8E's per year are summed and annually averaged. Three-way cross classification analysis of variance using SAS with the GLM procedure is used and the result is presented in Appendix E.

At significant level $\alpha=0.05$, the test revealed that there is no significant interaction among location, year block, and season with respect to P-0.8E's (p value=0.8575). Location and year block did not significantly interact either (p value=.04395). It means that the 15-year blocks within each season of each location are homogeneous. The 15-year blocks within each location are homogeneous as well. Seasonal P-0.8E's among 15-year blocks at each location are summarized on Tables 6 - 10, while annual P-0.8E's among 15-year blocks at each location are in Table 11.

Table 6. Seasonal Accumulation of P-0.8E's among 15-Year Blocks at Fairhope

Block	Num. Of Obs.	Mean (cm)	Stdev.	Block	Num. Of Obs.	Mean (cm)	Stdev.
Spring:				Fall:			
61-75	15	1.7	14.0	61-75	15	9.4	9.6
76-90	15	5.4	20.7	76-90	15	10.7	12.8
91-05	15	2.1	22.8	91-05	15	19.6	20.3
Summer:				Winter:			
61-75	15	16.9	14.9	61-75	15	24.4	11.8
76-90	15	11.8	19.0	76-90	15	27.5	16.3
91-05	15	20.2	14.9	91-05	15	24.4	15.4

Table 7. Seasonal Accumulation of P-0.8E's among 15-Year Blocks at Clemson

Block	Num. Of Obs.	Mean (cm)	Stdev.	Block	Num. Of Obs.	Mean (cm)	Stdev.
Spring:				Fall:			
61-75	15	1.5	10.1	61-75	15	12.6	8.0
76-90	15	-4.0	13.9	76-90	15	15.1	12.7
91-05	15	-4.2	14.7	91-05	15	13.6	13.2
Summer:				Winter:			
61-75	15	-6.9	11.8	61-75	15	26.4	10.2
76-90	15	-6.1	13.8	76-90	15	21.3	10.1
91-05	15	-5.9	18.1	91-05	15	21.7	10.2

Table 8. Seasonal Accumulation of P-0.8E's among 15-Year Blocks at Stoneville

Block	Num. Of Obs.	Mean (cm)	Stdev.	Block	Num. Of Obs.	Mean (cm)	Stdev.
Spring				Fall			
61-75	15	-7.5	14.2	61-75	15	10.1	13.4
76-90	15	-8.7	15.7	76-90	15	13.4	12.7
91-05	15	-8.0	12.9	91-05	15	13.1	13.2
Summer							
61-75	15	-18.6	9.7	61-75	15	24.6	13.9
76-90	15	-22.3	12.6	76-90	15	21.9	12.1
91-05	15	-20.0	10.4	91-05	15	25.5	10.7

Table 9. Seasonal Accumulation of P-0.8E’s among 15-Year Blocks at Stuttgart

Block	Num. Of Obs.	Mean (cm)	Stdev.	Block	Num. Of Obs.	Mean (cm)	Stdev.
Spring:				Fall:			
61-75	15	-7,3	13.2	61-75	15	7.8	14.9
76-90	15	-3.4	14.0	76-90	15	14.0	11.2
91-05	15	-6.0	11.2	91-05	15	13.1	13.0
Summer:*)				Winter:			
61-75	15	-16.5[a]	6.9	61-75	15	14.3	7.4
76-90	15	-24.7 [b]	12.4	76-90	15	15.0	13.9
91-05	15	-25.2 [b]	9.0	91-05	15	16.2	10.1

Table 10. Seasonal Accumulation of P-0.8E’s among 15-Year Blocks at Thomsons

Block	Num. Of Obs.	Mean (cm)	Stdev.	Block	Num. Of Obs.	Mean (cm)	Stdev.
Spring:				Fall:*)			
61-75	15	-8.1	17.8	61-75	15	-1.3 [b]	10.3
76-90	15	-16.1	8.5	76-90	15	0.8[ab]	17.1
91-05	15	-6.6	16.8	91-05	15	11.2[a]–	14.4
Summer:				Winter:			
61-75	15	-9.4	15.8	61-75	15	-0.6	8.9
76-90	15	-10.6	22.7	76-90	15	0.0	7.8
91-05	15	-13.2	16.7	91-05	15	8.3	15.2

Table 11. Annual P-0.8E’s among 15-Year Blocks at Each Location

Location	P-0.8E* (cm)		
	1961-1975	1976-1990	1991-2005
Fairhope	54.3 (36.9)	56.0 (32.9)	67.3 (34.6)
Clemson	34.7 (24.8)	25.9 (26.6)	24.6 (29.2)
Stoneville	10.0 (34.1)	4.5 (27.9)	11.1 (26.4)
Stuttgart	-1.7 (23.6)	0.9 (29.9)	-1.9 (22.3)
Thomsons	-19.5 (31)	-25.9 (30.8)	-0.4 (30.9)

*) values in parenthesis represent standard deviation

Figure 6 shows annual averages of cumulative P-0.8E's for each of 15-year blocks at each location. Although the year blocks did not significantly differ, from the curves we can see some consistent deviations of plots. For example, Fairhope (Figure 6a) shows that block 91-05 is consistently higher than the other two blocks, and block 61-75 is consistently below the other two blocks from beginning to the end of year. Similarly, Clemson (Figure 6b) shows that block 61-75 is consistently above the other two blocks.

Stoneville and Stuttgart (Figures 6c and 6d) show almost similar patterns of plots, but of the three blocks Stuttgart has more homogeneous. Block 76-90 of Stoneville looks consistently below blocks 61-75 and 91-05. Thomsons (Figure 6e) appears to show more variability of P-0.8E's among the three blocks with block 91-05 consistently at the top and block 76-90 at the bottom.

If we look at Table 11, generally the variances (square of standard deviation) are quite high. The variability seems to cause the statistical insignificancy among the 25-year blocks at each of locations. Consistent differences visible on the graphs, however, should not be overlooked in that they may cause different pond performances. It means that using 45 year climatological data in the pond model is safer than using any 15 year climatological data. Because a pond model using 45 year climatological data can handle climate variability in any 15 year blocks within the 45 year period.

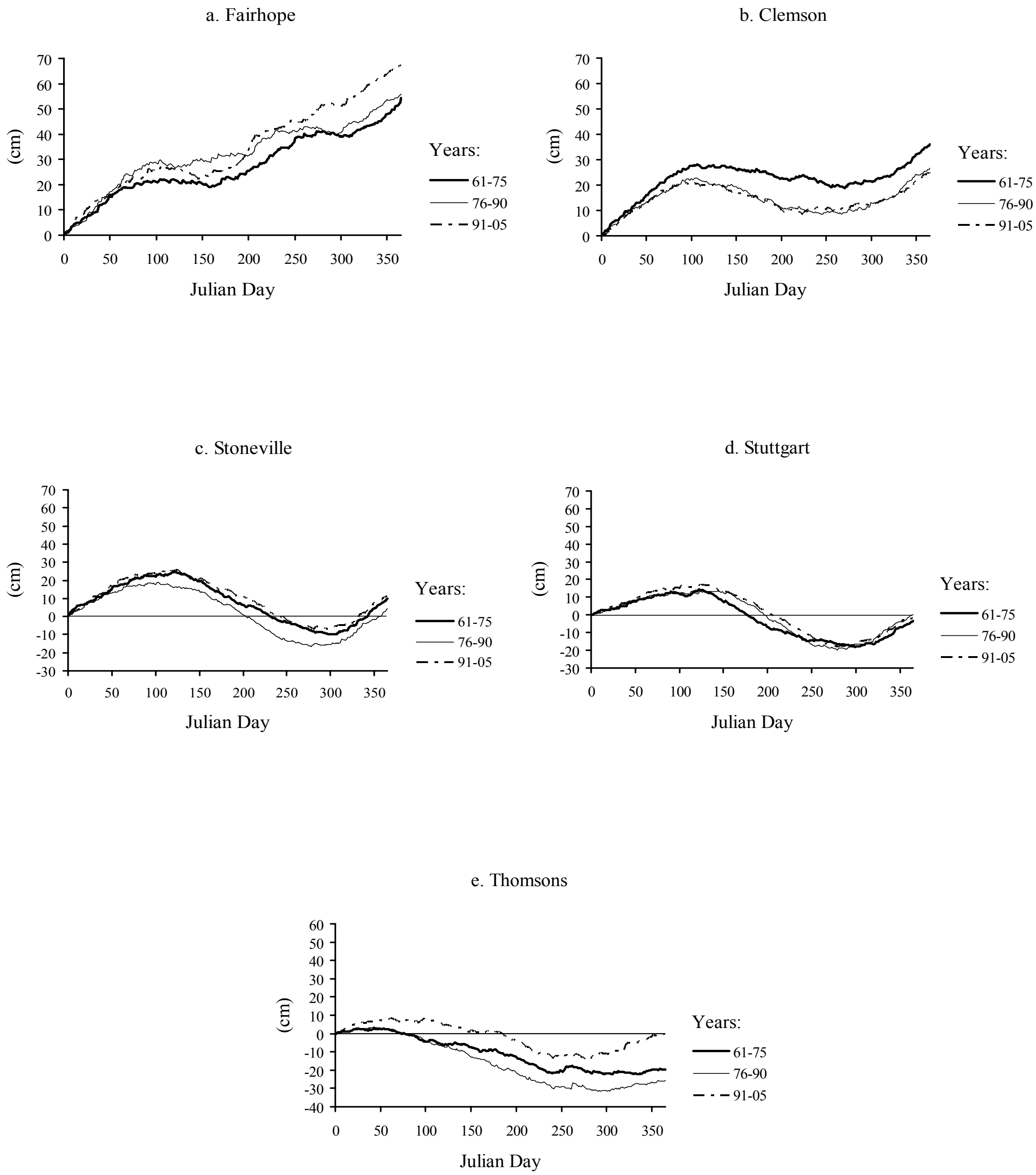


Figure 6. Cumulative P-0.8E's in One Year Cycle for Each of 15-Year Blocks

4.2. Pond Performance

4.2.1 Effect of Climate on Pond Performance

Figure 7 shows simulated pond performance results for schemes of 7.5 cm add depth and various drop depths from 7.5 cm to 210 cm at each location. Add or fill depth is the depth of groundwater added and drop depth is the height of the outflow structure measured from the minimum pond water level allowed. The simulation was run from drop depths of 7.5 cm until the lines became level. An add depth of 7.5 cm is used because farmers typically use pumping rates of 3.785 m^3 (1000 gallon) per minute. With this pumping capacity farmers can add 7.5 cm groundwater to a 4-ha pond in a long work day. Zero infiltration is used to assess climatological effects on drop/add management schemes.

In Figure 7a, we can see that groundwater use drops sharply when drop depth is increased from 7.5 cm to 30 or 45 cm. At Fairhope (the wettest location), groundwater use can be reduced easily with a fairly low drop depth. Groundwater use is practically zero when drop depth is 30 cm. Conversely, at Thomsons (the driest location), groundwater use cannot be zeroed. Groundwater use at Thomsons is almost 60 ha-cm/year at a drop depth of 7.5 cm. It drops to 20 ha-cm/year at a drop depth of 45 cm, and levels off at about 15 ha-cm/year at deeper drop depths. Other locations have zero groundwater at deeper drop depths. Groundwater use at Clemson is zero at a drop depth of 120 cm but at a drop depth of 45 cm groundwater use is also close to zero (2.43 ha-cm/year). Stoneville and Stuttgart require more extended drop depths to get zero groundwater use. Groundwater uses at the two locations become zero at a drop depth of

180 cm. If we look at Table 5, Stoneville and Stuttgart have the lowest P-0.8E's in summer and biggest differences between P-0.8E's in summer (driest) and winter (wettest) among the other locations. It seems that these two locations use more groundwater in summer (dry period) and discharge more effluent in winter. This fact has caused Stoneville and Stuttgart to require deeper drop depths to get zero groundwater. At Thomsons, although it has the lowest annual P-0.8E, seasonal variations of P-0.8E is not as high as at Stoneville and Stuttgart. For this reason, groundwater use leveled off with a shallower drop depth (groundwater use never becomes zero at Thomsons).

Figure 7b shows the number of years that predicted groundwater use was zero as drop depth was adjusted. When drop depth is 7.5 or 15 cm, relatively few years require zero groundwater. When drop depth is increased to 30, 45 or 60 cm, the number of zero groundwater use years increases dramatically. For Fairhope, zero groundwater can be increased to more than 40 years with a drop depth of 30 cm, and no groundwater at all when drop depth is 45 cm. For Clemson, zero groundwater use is predicted for 38 years if a drop depth of 45 cm is used. This shows that most make-up water need can be derived from rainwater and groundwater is added only in very dry years. For Thomsons, zero groundwater use can be increased to 20 years when a drop depth of 75 cm is used. After that, further increases are not effective to increase zero groundwater use years. For Stoneville and Stuttgart, large drop depths are required to appreciably increase the number of zero groundwater years. Drop depths of 120 or 135 cm are required to increase the number of zero groundwater use to about 40 years, after which the curve is essentially flat.

Figure 7c shows simulated effluent discharges at various drop depths. The graph show, as expected, that Fairhope has the highest discharge rate and increased drop depth has only a little effect on effluent release. Effluent discharge drops from 87 ha-cm/year at a drop depth of 7.5 cm to 60 ha-cm/year at drop depth of 30 cm. At higher drop depths, effluent discharge cannot be reduced at all. Relatively high effluent discharge also occurs at Clemson. Effluent discharge decreases from 61 ha-cm/year at a drop depth of 7.5 cm to 31 ha-cm/year at drop depth of 45 cm, with no improvement at greater drop depths. Stoneville and Stuttgart have nearly identical predicted effluent discharges, with Stoneville a little higher than Stuttgart. Effluent discharge at Stoneville plateaus at 9 ha-cm/year when drop depth is 180 cm. Predicted effluent discharge at Stuttgart levels off at about 1 ha-cm/year at drop depth of 180 cm. Thomsons is the only location where effluent discharge reached zero and, even there, it required a fairly deep drop depth. At every drop depth, Thomsons has the lowest effluent discharge among all the locations.

Figure 7d shows the number of predicted zero effluent discharge years for each site in the study. From this graph, we can see that there were no zero discharge years predicted for Fairhope. Clemson had a maximum of 9 years with no discharge. Unlike the previous graphs, this graph of zero effluent year showed quite different shapes between Stoneville and Stuttgart regardless of the fact that the P-0.8E's of both location were not significantly different (Table 3). Stoneville had a maximum of 29 zero effluent years, while the maximum at Stuttgart was 42 years. This was likely caused by the high P-0.8E in winter at Stoneville. The seasonal P-0.8E's of the two locations are not significantly different except for winter, which was significantly higher for Stoneville

(Table 5). This illustrates that seasonal variations in P-0.8E can cause differing performances.

Implementation of a drop/add strategy is really the introduction of rainwater harvesting to pond aquaculture. Rainwater harvesting systems are judged both on their capture efficiency (percentage of incident precipitation stored for use) and on the amount of need that they serve (contribution to the water needs of the system). Figure 7e shows the predicted percentage of rainwater that can be stored, relative to the total of annual precipitation. From this graph, we can see that for dryer locations (Stoneville, Stuttgart, and Thomsons) 80% to almost 100% of rainwater can be captured and stored with drop depths of 45 cm or higher. This indicates that the drop/add approach at these locations can capture and use most of the incident rainfall. For wetter locations, (Fairhope and Clemson) capture efficiencies were nearly 65% and 80%, respectively.

Figure 7f shows the percentage that rainwater contributed to the total water budget. In general, at a drop depth of 15 cm, rainwater contribution is more than 70% of total water used. All location but Thomsons can have rainwater contributions that approach 100% of water requirements if drop depths are great enough. Maximum rainwater contribution for Thomsons is about 89%.

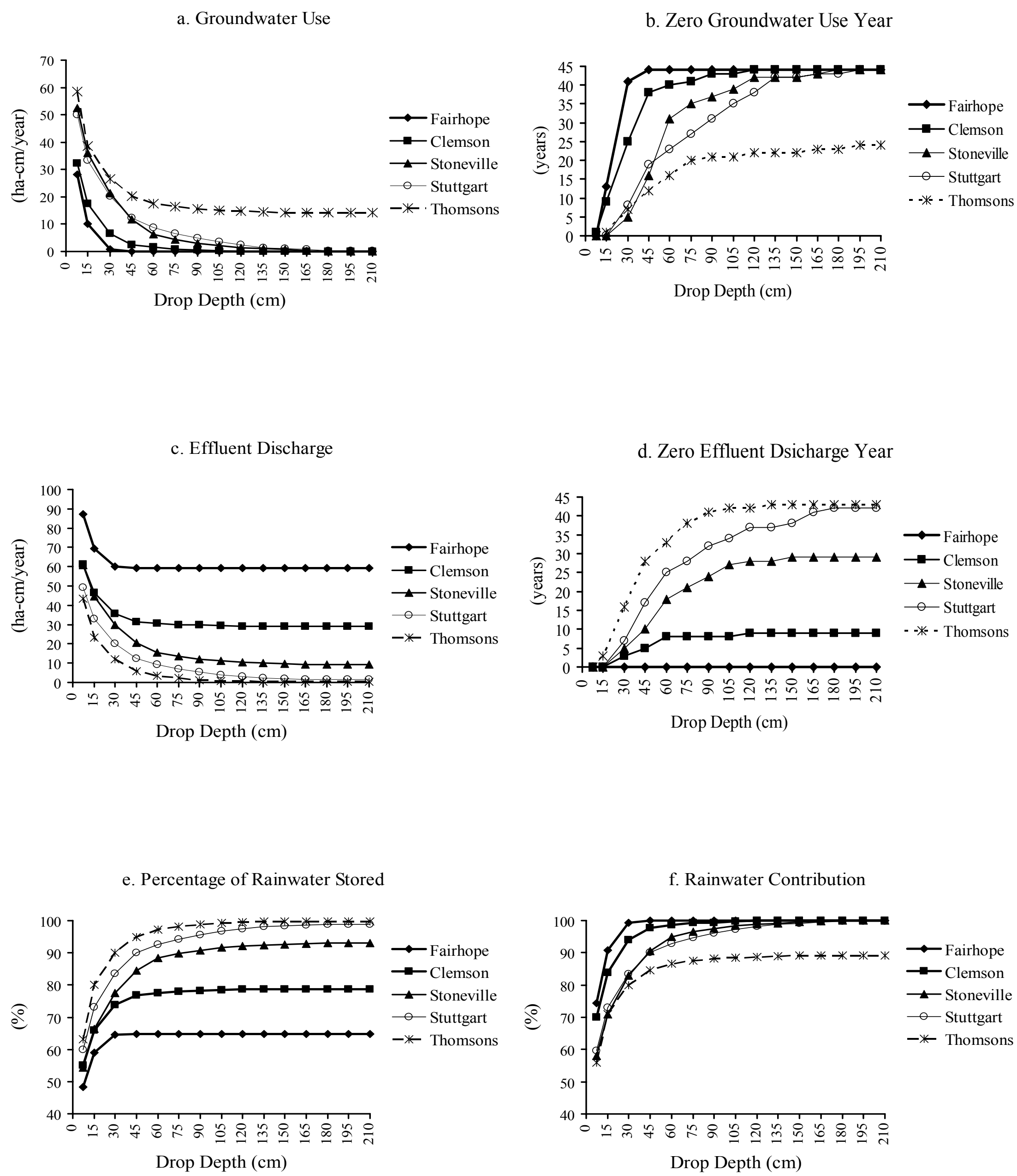


Figure 7. Pond Performance of Schemes with Various Drop Depths and 7.5 cm Fill Depth at Zero Infiltration

Extreme drop depths were used in the preceding simulations to find the climatological limits of the drop/add approach. It is interesting to note, in Figure 7, that much or, in some cases, nearly all of the advantage of the greatest drop depths was found at drop depths of 45 to 75 cm. Work at the Mississippi State Delta Research Center to evaluate this approach (SRAC, 2002) showed that drop depths to 30 to 60 cm presented little operational difficulty. Although 75 cm was not tested, it is likely that this drop depth would be practicable as well.

4.2.2 Seasonal Evaluation

Figure 8 shows seasonal groundwater use of various drop depths and 7.5 cm fill depth at zero infiltration. At Fairhope, high groundwater use in spring was consistent with the lowest P-0.8E in this season (Figure 8a). At Clemson, the highest groundwater use was in summer, followed by spring and fall for drop depths of 30 cm and less (Figure 8b).

For Stoneville and Stuttgart (Figure 8c and 8d), groundwater uses look similar. This is not surprising because the P-0.8E's of the two locations are not different. On the average, groundwater use in spring is higher than in fall for drop depths of 7.5 and 15 cm. This is also consistent with P-0.8E which is higher in fall than in spring (Table 5). But, for drop depths higher than 15 cm, groundwater use in fall is higher than in spring (while groundwater is being used). This is because winter storage is used first in the following spring. During the fall, following an extended period of low P-0.8E, stored precipitation may be minimal, leading to increased groundwater use in the fall.

For Thomsons, P-0.8E's of spring and summer are not significantly different; and P-0.8E's of fall and winter are not significant either (Table 5). But if we look at the graph (Figure 8e), the groundwater use in spring and summer look quite different. Similarly, groundwater uses in fall and in winter also look different. This is also because of the differences of pond water levels. Even pond water level is decreasing in spring (due to negative P-0.8E) but it is still higher than in summer that more groundwater is used in summer than in spring. Similarly, even pond water level is increasing in fall (due to Positive P-0.8E) it is still lower than in winter that groundwater use in fall is higher than in winter.

Figure 9 shows seasonal effluent discharge from each location. In general, effluent discharge in winter is high due to high P-0.8E in winter except for Thomsons, where effluent discharge is low all the time. Higher effluent discharge in spring than in fall is also visible at all locations particularly for some deeper drop depths, except for Thomsons, in spite of higher P-0.8E's in fall than in spring.

Figure 10 shows average of daily pond water level for a 45/7.5 management scheme at zero infiltration at each location. Pond water level is high in winter, decreases in spring, is low in summer, and then increases in fall. These graphs are representations of rainwater accumulations in pond. Pond water level is high in winter because of positive P-0.8E accumulations in fall and winter. Pond water level decreases in spring because of negative P-0.8E accumulation. Pond water level is low in summer because of negative accumulations in spring and summer. Finally, pond water level increases again in fall because of positive P-0.8E accumulation in fall.

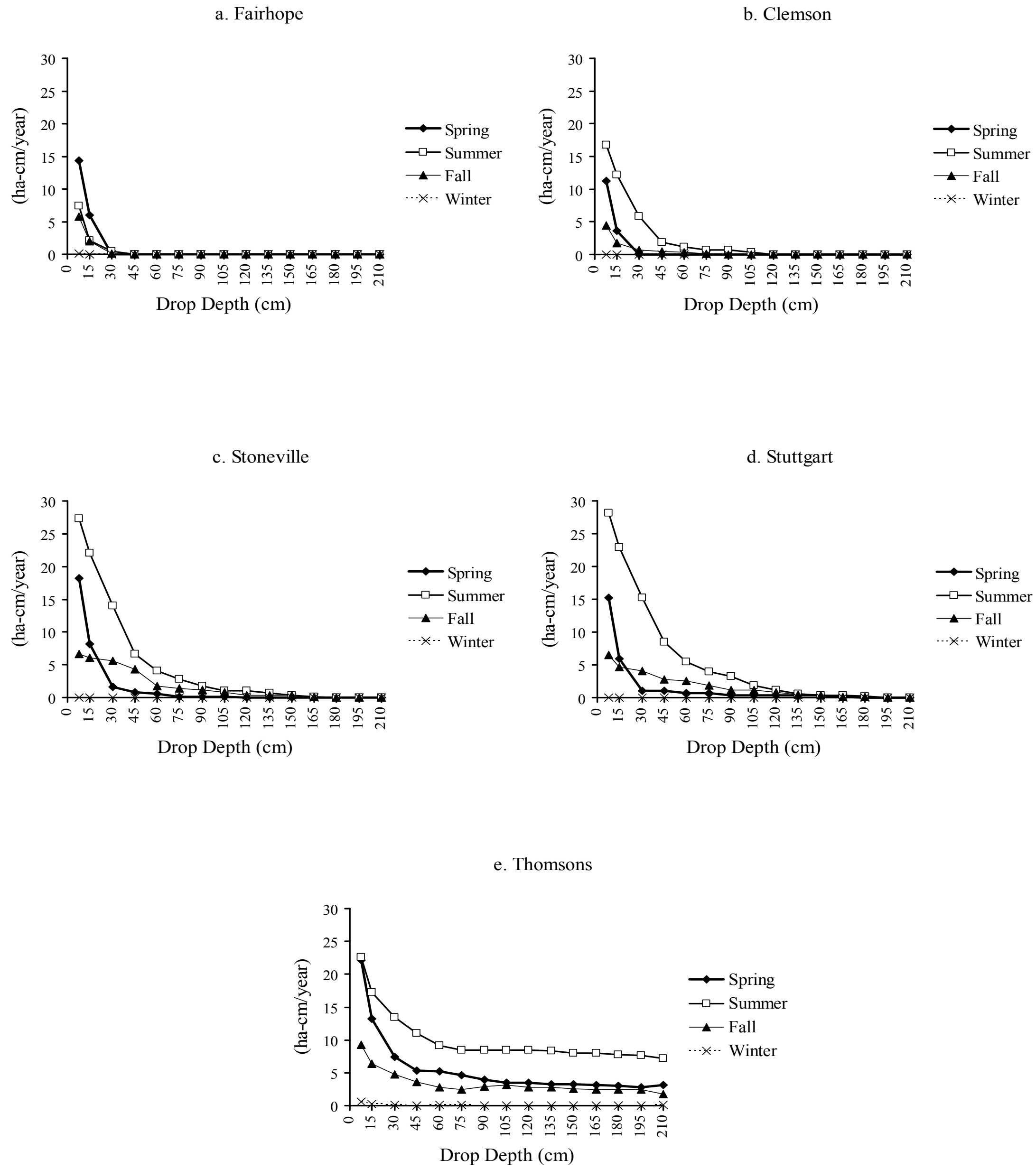


Figure 8. Seasonal Groundwater Use of Schemes with Various Drop Depths and 7.5 cm Fill Depth at Zero Infiltration

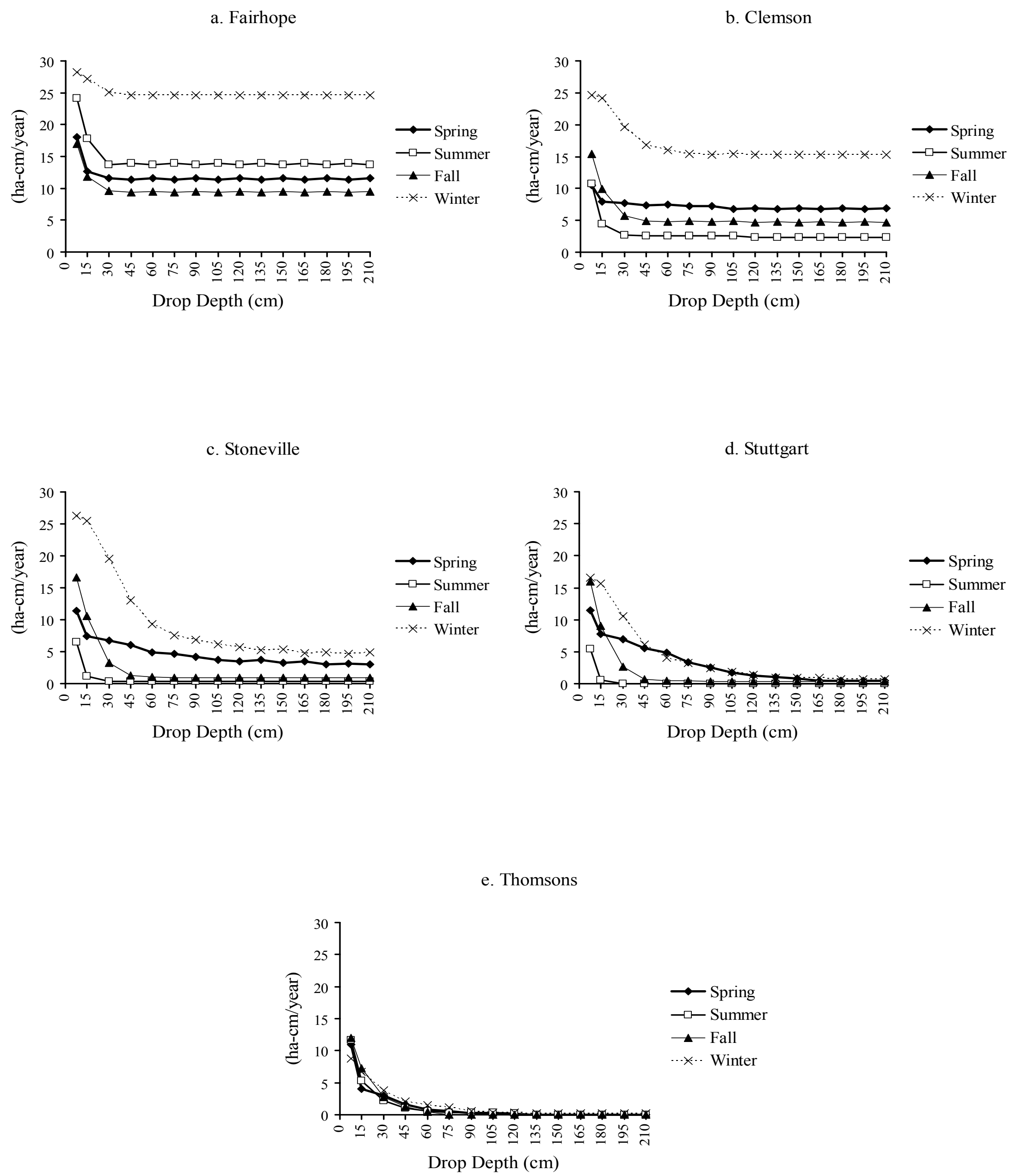


Figure 9. Seasonal Effluent Discharge of Schemes with Various Drop Depths and 7.5 cm Fill Depth at Zero Infiltration

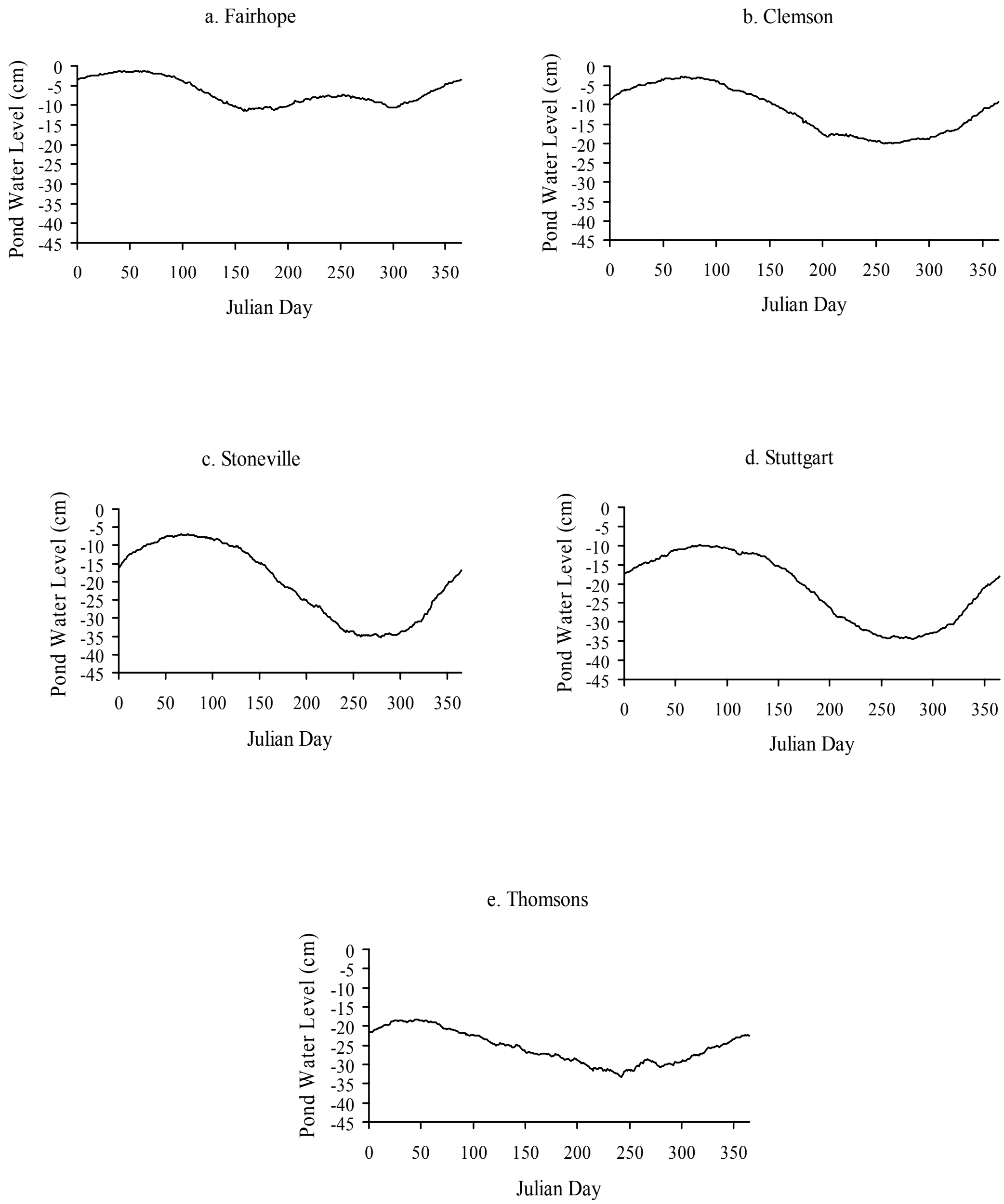


Figure 10. Average of Pond Water Level for a 45/7.5 Management Scheme at Zero Infiltration

4.2.3 Effect of Infiltration Rates

The general effect of increasing infiltration is to increase groundwater requirements and decrease effluent discharge to receiving waters. Although a drop in effluent release decreases release of nutrients to surface waters, this is not the advantage that it appears to be. Shallow subsurface effluent release can contribute to surface eutrophication. This has been illustrated by experiences with failed septic tank waste treatment systems. Also, under some circumstances, infiltration of nutrient rich water can adversely affect shallow aquifers. The following section should be viewed as emphasizing the importance of well sealed aquacultural ponds to the long-term sustainability of the pond aquaculture industry.

Figure 11 shows pond performance at Fairhope for infiltration rates of 0 to 0.5 cm/day and drop depths of 7.5 to 210 cm. Zero predicted groundwater use can be achieved for infiltration rates up to 0.1 cm/day (Figure 11a). For an infiltration rate of 0.2 cm/day, minimum groundwater use remains low (13 ha-cm/year), with approximately 50% of the years requiring no groundwater (Figure 11b). When infiltration rates are higher (0.3 to 0.5 cm/day), predicted groundwater use becomes large (about 50 ha-cm/year for infiltration rate of 0.3 cm/day and 125 ha-cm/year for infiltration rate of 0.5 cm/day). At these infiltration rates, zero groundwater use years are uncommon.

At Figure 11c, we can see that effluent discharge drops sharply from about 60 ha-cm/year (at zero infiltration) to 23 ha-cm/year at an infiltration rate of 0.1 cm/day. At an infiltration rate of 0.1 cm/day, there are 11 zero effluent discharge years (Figure 11d).

When infiltration rate is 0.2 cm/day, surface effluent discharge decreases to nearly 0 ha-cm/day.

Figure 11e shows percentage of rainwater that can be stored (relative to the average of annual precipitation). Maximum percentage of rainwater stored increases from 65% (for infiltration rate of 0 cm/day) to about 86% (for infiltration rate of 0.1 cm/day). For higher infiltration rates, maximum rainwater stored can be maximized to 100% with fairly low drop depths. Figure 10f shows percentage of rainwater contribution to total water budget (groundwater plus rainwater). We can see that the total water budget can be satisfied by rainwater at all the time for infiltration rates of 0 and 0.1 cm/day. When infiltration rate increases to 0.2 cm/day, rainwater contribution is still about 90%. For higher infiltration rate (0.5 cm/day), rainwater contribution decreases to about 55%.

From the above simulation results, we know that if infiltration rate is low (close to 0 cm/day), the groundwater is low or can be zero, and rainwater contribution is high. However, high effluent discharge at low or zero infiltration rates cannot be avoided at Fairhope as the wettest location. At the same time, high infiltration rate is avoided because of high groundwater use and potential to cause groundwater pollution.

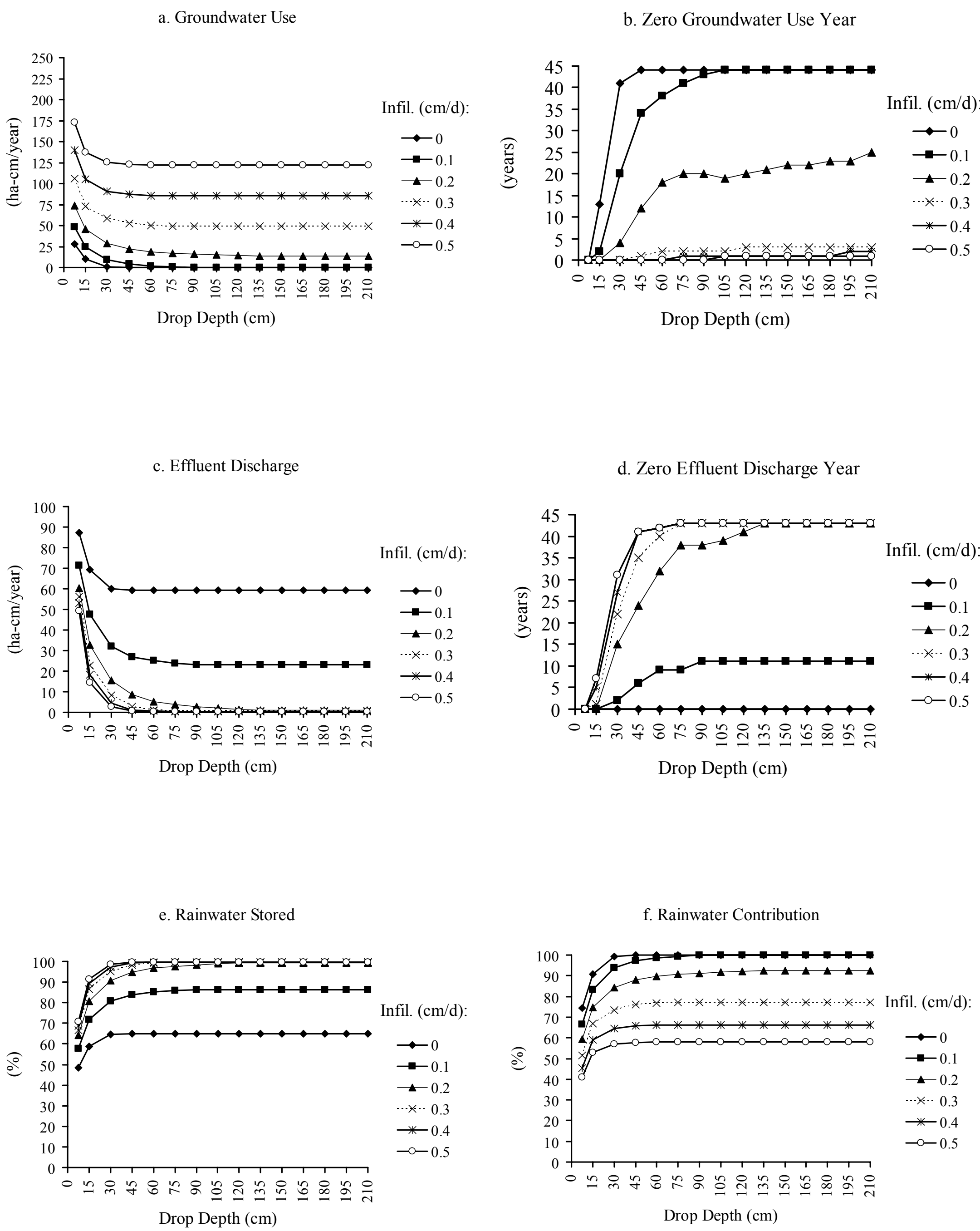


Figure 11. Effect of Infiltration Rates on Pond Performance of Schemes with Various Drop Depths and 7.5 cm Fill Depth at Fairhope

If we look at the shapes of the curves, drop depths of 30 or 45 cm can be selected as the optimum options depending on the infiltration rates. For example, drop depth of 30 cm is enough to be used for infiltration rate of 0 cm/day or higher infiltration rates (0.4 and 0.5 cm/day) because the curves essentially flatten at higher drop depths than 30 cm. For infiltration rates of 0.1 to 0.3 cm/day, drop depths of 45 cm can be enough to choose.

Figure 12 shows pond performance at Clemson, for infiltration rates of 0 to 0.5 cm/day and drop depths of 7.5 to 210 cm. As at Fairhope, annual and seasonal P-0.8E is relatively large. As a result, it is still possible to achieve zero groundwater use at modest infiltration rates (up to 0.1 cm/day). At greater infiltration rates, zero groundwater use years become uncommon. Not surprisingly, the amount of rainwater used to meet water needs increases as infiltration increases, but it's overall percent contribution decreases as groundwater requirements increase with increasing infiltration.

Figures 13 and 14 show pond performance at Stoneville and Stuttgart, respectively. The figures are very similar, which is not surprising given the similarity of their P-0.8E records. Lacking the high P-0.8 values of Clemson and Fairhope, both sites show steadily increasing groundwater use requirements as infiltration increases (Figures 13a, 14a), although, in both cases, it is possible to achieve zero groundwater use years at low infiltration rates (Figures 13b, 14b). Effluent discharge, of course, plummets as pond water is "discharged" instead through the bottom (Figures 13c, 14c). At both sites, nearly all of the precipitation contributes to meeting the water budget (Figures 13e, 14e) but the importance of precipitation to the water budget steadily decreases as infiltration increases (Figures 13f, 14f).

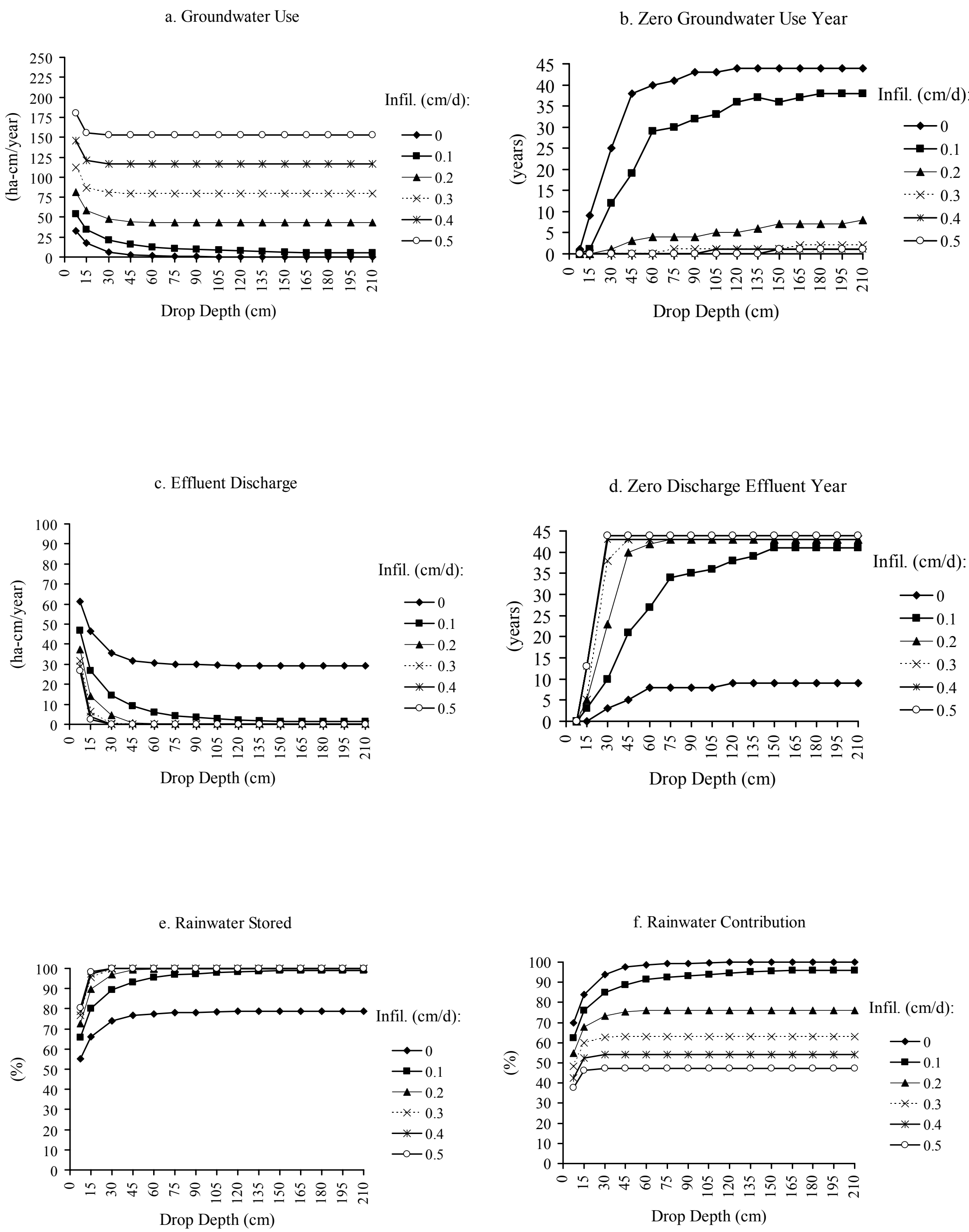


Figure 12. Effect of Infiltration Rates on Pond Performance of Schemes with Various Drop Depths and 7.5 cm Fill Depth at Clemson

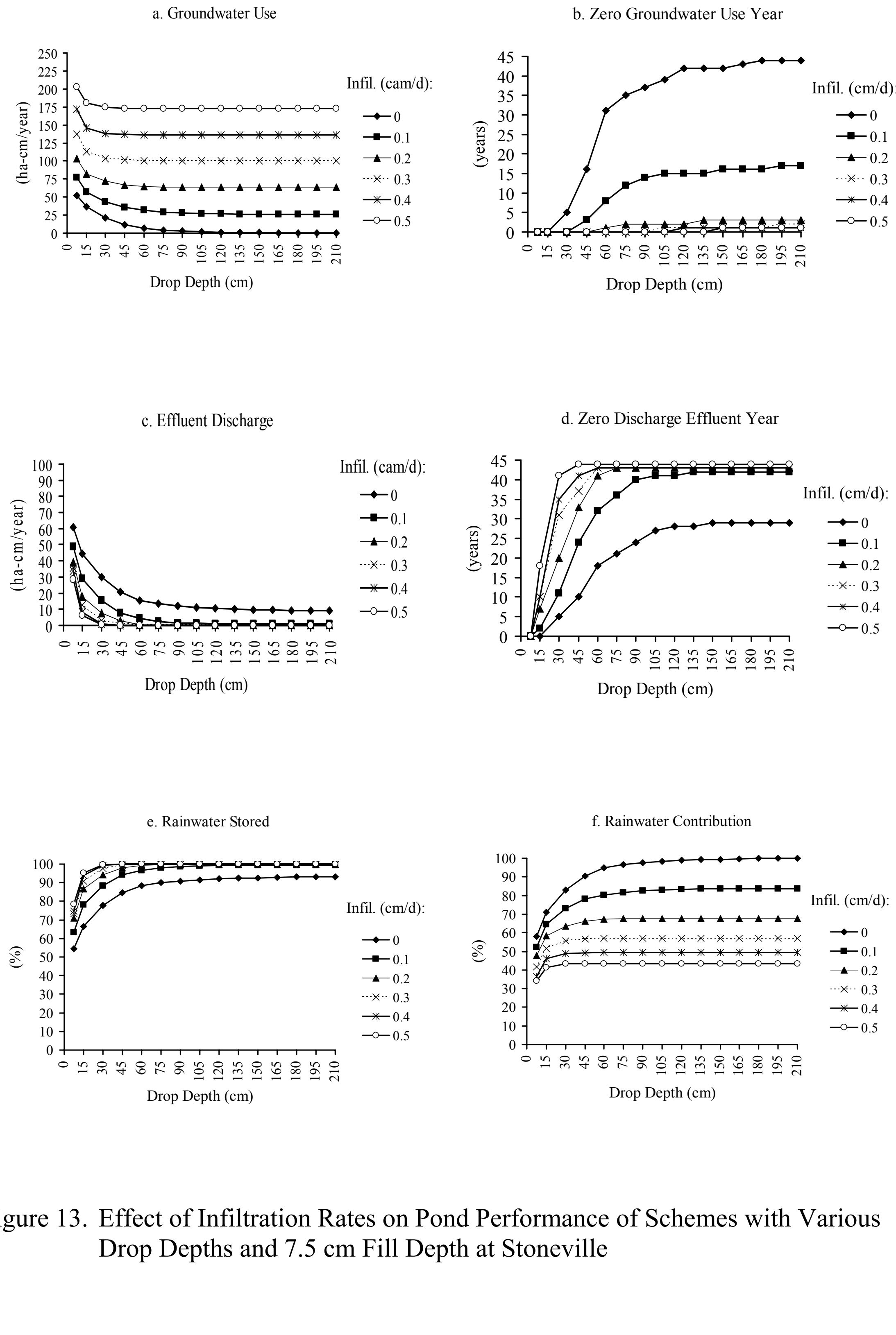


Figure 13. Effect of Infiltration Rates on Pond Performance of Schemes with Various Drop Depths and 7.5 cm Fill Depth at Stoneville

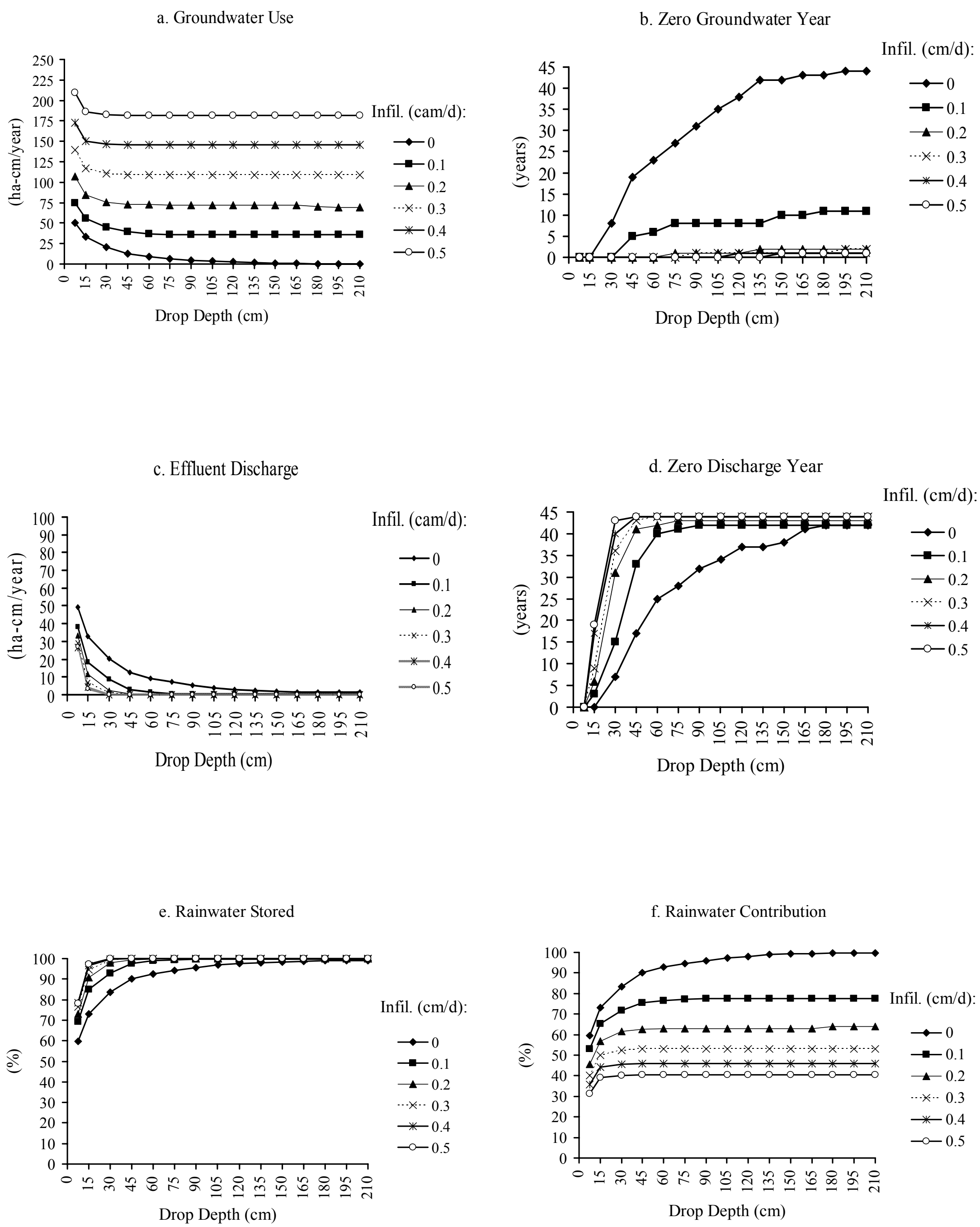


Figure 14. Effect of Infiltration Rates on Pond Performance of Schemes with Various Drop Depths and 7.5 cm Fill Depth at Stuttgart

Figure 15 shows pond performance at Thomsons for infiltration rates of 0 to 0.5 cm/day and drop depths of 7.5 to 210 cm. Annual P-0.8E at Thomsons is the lowest among the locations (-15.3 cm) and elimination of all groundwater use requirements does not appear to be possible. For zero infiltration, minimum groundwater use is about 14 ha-cm/year on the average, but the number of zero groundwater is still 24 years (more than 50% of the years). Minimum groundwater use increases to 50 ha-cm/day (Figure 15a) or the number of zero groundwater use drops to 4 years (Figure 15b) when infiltration rate increases from 0 to 0.1 cm/day. When infiltration rates increase to 0.2 cm/day and higher, groundwater uses are very high, at least more than 80 ha-cm/year while years of zero groundwater uses are practically zero.

Effluent discharges to surface waters (Figure 15c) are low at Thomsons for all infiltration rates even at low drop depths. All effluent discharges can be zero or the number of zero effluent discharge is at maximum of 45 years (Figure 15d). When infiltration rate increases to 0.1 cm/day or higher, zero effluent discharges can be achieved at relatively low drop depths. At infiltration rate of 0.5 cm/day, zero effluent discharge can be achieved at all time with drop depth of 30 cm.

Figure 15e shows percentage of rainwater that can be applied to the water budget at Thomsons. Rainwater storage is more than 60% for infiltration rate of 0 cm/day and at drop depth of 7.5 cm, and close to 100% for higher infiltration rates at drop depths of 15 to 30 cm. Figure 15f shows the percentage of the water budget supplied by precipitation. As at the other sites, the percent contribution decreases as infiltration increases.

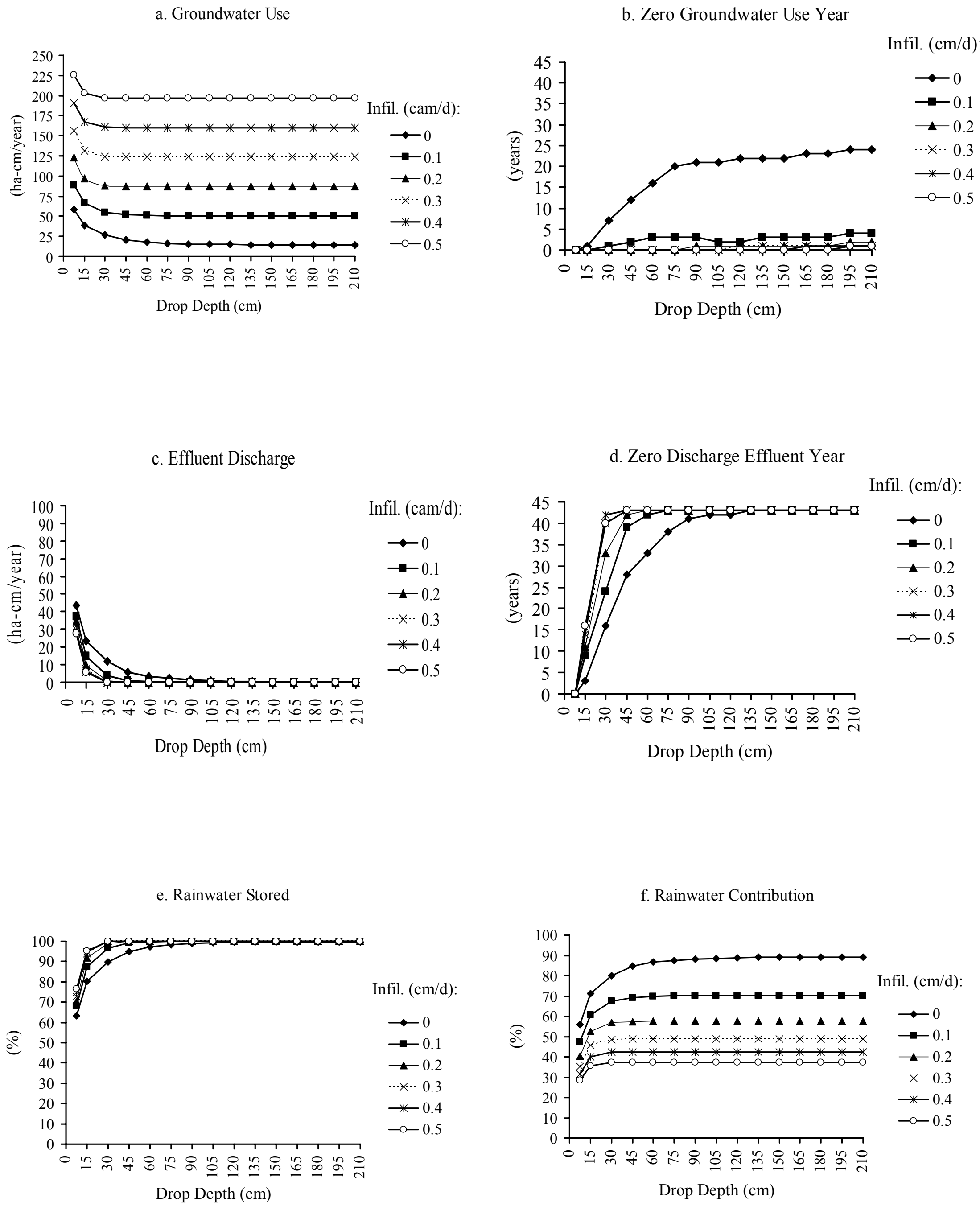


Figure 15. Effect of Infiltration Rates on Pond Performance of Schemes with Various Drop Depths and 7.5 cm Fill Depth Thomsons

4.2.4 Summary

Low infiltration rates result in low groundwater use and a high contribution of rainwater to the water budget. When infiltration rates increase, groundwater and rainwater that can be stored increase but effluent discharge decreases. As described above, the reduction of effluent discharge cannot be viewed as an advantage.

Predicted groundwater use and effluent discharge suggest that, for ponds with low infiltration rates (0 to 0.1 cm/day), a management scheme 45/7.5 will give generally good performance. It is probably reasonable to assume that many ponds will have some infiltration but that, for well designed ponds, this value will be 0.2 cm/day or less.

For Clemson, if infiltration rate can be reduced to around 0.1 cm/day, groundwater use and effluent discharge can be maintained low while rainwater can be stored to more than 90% by using management scheme 45/7.5. At Fairhope, groundwater use can be kept low at infiltration rates of 0.2 cm/day or less. For the other three locations, a well sealed pond (with nearly 0 infiltration) is preferable from a water conservation standpoint. At Stuttgart and Stoneville, low infiltration (0.1 cm/day or less) can result in acceptable groundwater use reductions. At infiltration rates greater than 0.1 cm/day, performance deteriorates rapidly.

Figure 16 illustrates pond water level at each location using a 45/7.5 management scheme. This figure shows that effluent discharge (above the overflow depth) occurs only during heavy rainstorms or very wet years.

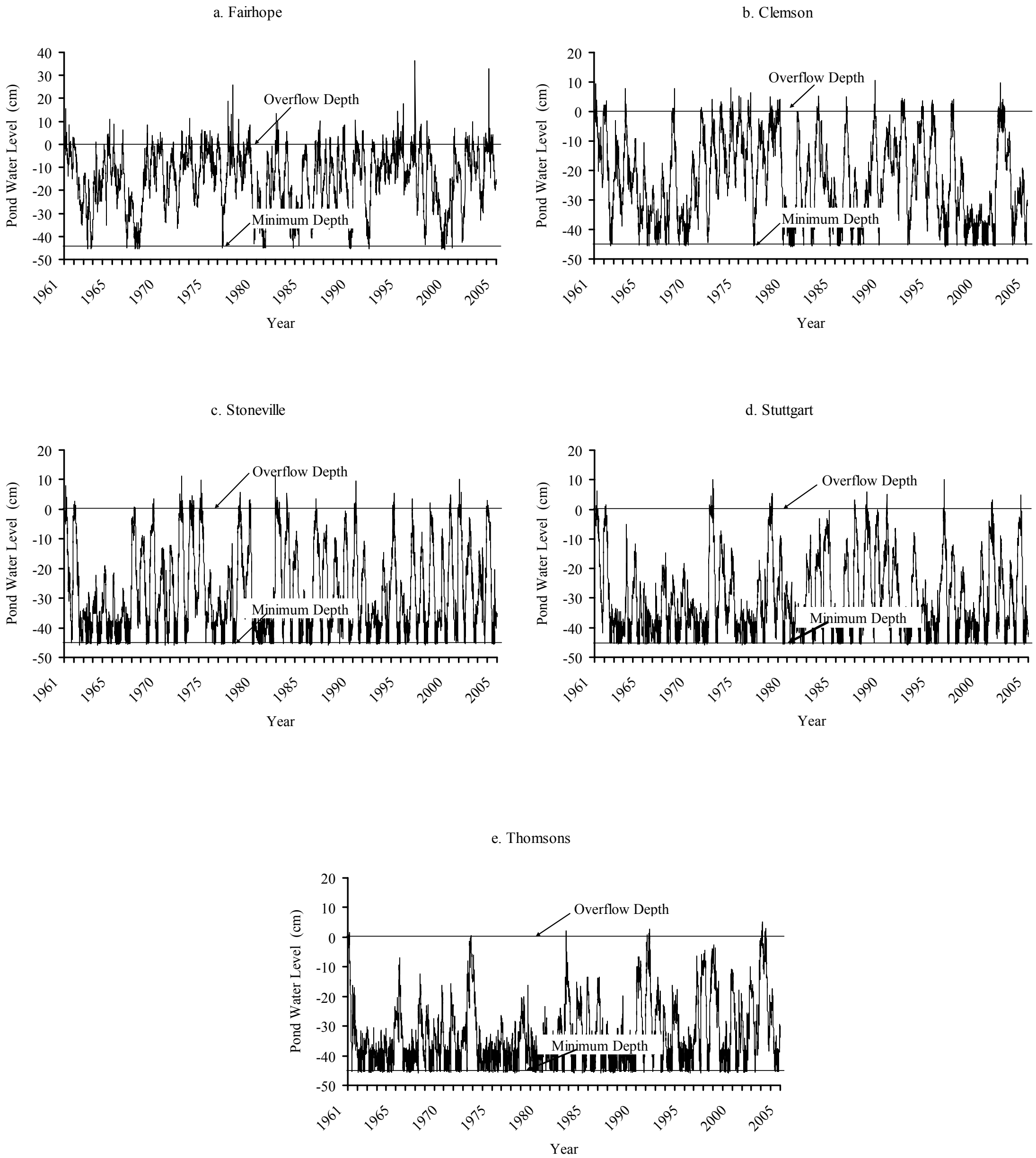


Figure 16. Pond Water Level for a 45/7.5 Scheme and 0.1 cm/day Infiltration Rate

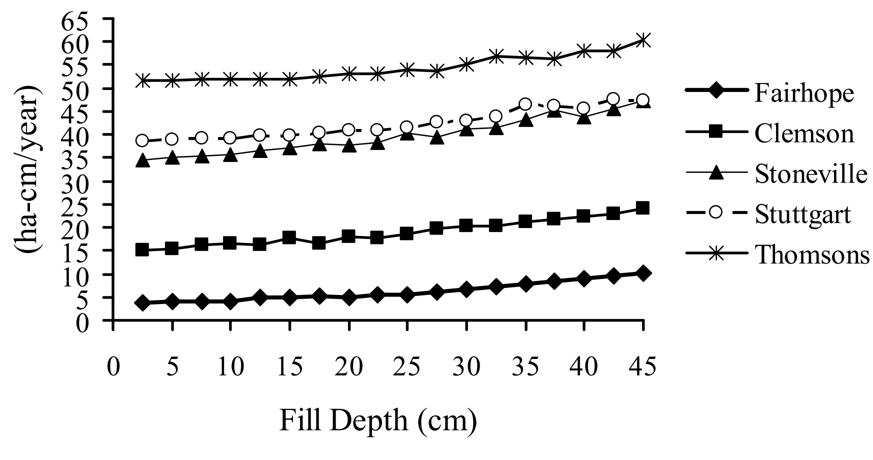
4.3 Effect of Deeper Fill Depths

The 45/7.5 scheme appears to perform well for low infiltration rates at most locations. A fill depth of 7.5 cm is used because it is appropriate for a 1000 GPM pump that is typical for catfish ponds. If a lower fill depth is used, groundwater use and effluent discharge are lower because there is more storage capacity, but filling will be more frequent during dry periods. Conversely, if a deeper fill depth is used, groundwater use and effluent discharge will increase while pumping frequency will decrease. This scenario is used to evaluate how much groundwater use and effluent discharge respond to the fill depths. A drop depth of 45 cm and an infiltration rate of 0.1 cm/day are used for this analysis.

Figure 17 shows groundwater use (Figure 17a) and effluent discharge (Figure 17b) as fill depths vary from 2.5 to 45 cm at each location. The slopes of these curves are small, indicating that increasing fill depth does has a small affect on groundwater use and effluent discharge. For fill depths of 2.5 to 15 cm, groundwater use and effluent release increases by about 2 to 3 ha-cm/year. When fill depth is increased to 25 cm, groundwater use and effluent discharge increase by 2 to 6 ha-cm/year. For a fill depth of 45 cm, groundwater use and effluent discharge increase around 6 to 9 ha-cm/year.

It means that farmers have plenty flexibilities to use fill depths without worrying much about increases of groundwater use and effluent discharge. Various management Schemes from 45/2.5 to 45/25 may be used dependent on pump capacity they own and their convenience.

a. Groundwater Use



b. Effluent Discharge

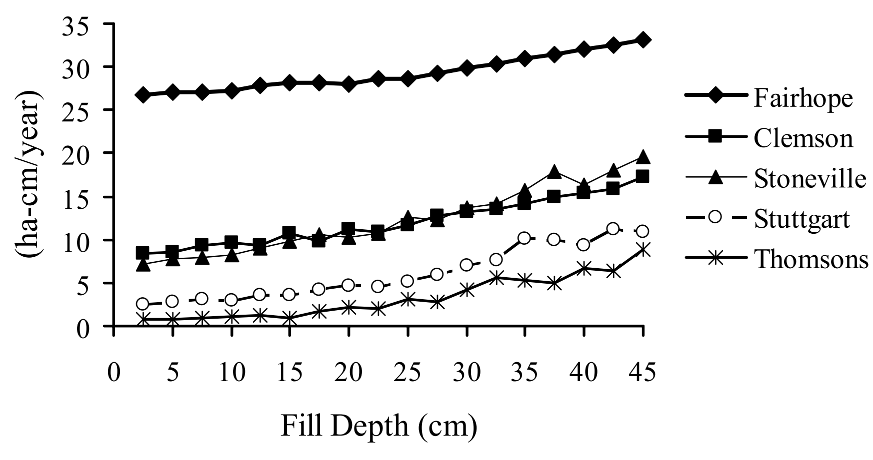


Figure 17. Groundwater Use and Effluent Discharge of Schemes with a 45 cm Drop Depth and Various Fill Depths from 2.5 to 45 cm at Each Location

4.4 Sensitivity Analysis

Table 12 summarizes groundwater use and effluent discharge using 15/7.5 and 45/7.5 management schemes, a pan coefficient of 0.8, and infiltration rates of 0.1 and 0.4 cm/day. This data set is used as the base set for the sensitivity analysis of the model to pan coefficient and infiltration rate (as the input parameters). Groundwater and effluent discharge are the model outputs. Linear sensitivity is not the same for every drop depth and every infiltration rate since the relationship between model inputs and outputs is not always linear. For example at wet locations such as Fairhope, groundwater use for deeper drop depths and at low infiltration rates (0 and 0.1 cm/day) are always zero. Similarly for dry locations, effluent discharges for deeper drop depths at higher infiltration rates are zero no matter what the drop depth is (as shown previously).

Therefore, this linear sensitivity analysis is performed using four base sets: two management schemes (15/7.5 and 45/7.5) and two infiltration rates (0.1 and 0.4 cm/day). These four base sets are used to evaluate model sensitivity in low and high pond water storage capacities and at low and high infiltration rates. The sensitivity analysis is not performed on management schemes having higher drop depths since effluent discharge will be always zero particularly at high infiltration rates. Linear sensitivity is calculated using Equation 3.

Table 12. Groundwater Use and Effluent Discharge of 15/7.5 and 45/7.5 Management Schemes Using Pan Coefficient=0.8 at 0.1 cm/day and 0.4 cm/day Infiltration Rates, as Base Sets for Sensitivity Analysis

Location	Groundwater Use (ha-cm/year)		Effluent Discharge (ha-cm/year)	
	at Infiltration Rate =		at Infiltration Rate =	
	0.1 cm/day	0.4 cm/day	0.1 cm/day	0.4 cm/day
Scheme 15/7.5:				
Fairhope	24.6	104.7	47.6	18.2
Clemson	34.4	120.6	26.9	3.8
Stoneville	57.0	145.7	29.0	8.3
Stuttgart	55.4	150.5	18.5	4.1
Thomson	66.3	166.9	14.9	6.0
Scheme 45/7.5:				
Fairhope	3.9	87.0	27.0	1.2
Clemson	16.1	116.2	9.3	0.1
Stoneville	35.3	136.9	7.9	0.2
Stuttgart	39.3	145.6	3.1	0.0
Thomson	51.9	160.2	1.0	0.0

4.4.1 Relative Sensitivity to Pan Coefficient

Table 13 contains sensitivity of the model (using groundwater use as the model output) to pan coefficient at each of locations. Pan coefficient is varied from 0.7 to 0.9 with average of 0.8 as the base on Table 16. Among the locations, relative sensitivity of groundwater use is consistently high at the wet location (Fairhope) and low at the dry location (Thomsons) for both management schemes (15/7.5 and 45/7.5) and the two infiltration rates (0.1 and 0.4 cm/day). This means that changing pan coefficient will have a greater effect on the model output (groundwater use) in wet locations than in dryer locations. The positive index of relative sensitivity reflects increasing pan coefficient will be followed by increasing groundwater use.

Comparing two infiltration rates (0.1 and 0.4 cm/day), the relative sensitivity of groundwater use to pan coefficient is also consistently higher at 0.1 cm infiltration and lower at higher at 0.4 cm/day for both management schemes (15/7.5 and 45/7.5). It has been understood that increasing infiltration rate is followed by increasing groundwater use. But this analysis reveals that groundwater use is more responsive to pan coefficient at low infiltration rates than at high infiltration rates. Change of groundwater use due to change of pan coefficient decreases when infiltration rate increases.

The relative sensitivities of groundwater use to pan coefficient for the two management schemes also shows another consistent difference. It is higher for the 45/7.5 management scheme than for the 15/7.5 scheme. Despite the fact that higher pond water storage capacity reduces groundwater use, groundwater is more responsive to change of pan coefficient for higher pond water storage capacity (scheme 45/7.5) than for lower pond water storage (scheme 15/7.5) at both low and high infiltration rates (0.1 and 0.4 cm/day).

If effluent discharge is used as the model output, all indices of relative sensitivity are negative (Table 14), meaning that increasing pan coefficient will lead to decreasing effluent discharge. However, among the locations, the sensitivity is not consistent with the wetness of the locations. For a negative index, the lower index of relative sensitivity (further from zero) is more responsive than higher index.

Between the two infiltration rates, except for Stoneville, effluent discharge is generally more responsive to pan coefficient at lower infiltration (0.1 cm/day) than at higher infiltration (0.4 cm/day), for both management schemes (15/7.5 and 45/7.5). This is similar to what was seen with groundwater use, which is also more responsive to the change of pan coefficient at low infiltration rates than at higher infiltration rates.

Between the two management schemes, relative sensitivity of effluent discharge to pan coefficient is also consistent with groundwater use sensitivity. In general, effluent discharge is more sensitive to change of pan coefficient at a 45/7.5 management scheme (higher pond water storage capacity) than at a 15/7.5 management scheme (lower pond water storage capacity) regardless of lower discharge for management scheme 45/7.5.

The explanation for this sensitivity behavior can be explained by looking at Equation 4, which was used to calculate sensitivity ($S = (\Delta O/O_{avg})/(\Delta I/I_{avg})$, where O and I are the output and input variables). The denominator of Equation 4 is 0.2/0.8 = 0.25, meaning that sensitivity will be the numerator multiplied by 4. Looking at groundwater use, at a pan coefficient of 0.7, both Fairhope and Clemson use relatively little groundwater. The dryer locations use much more groundwater. When pan coefficient increases to 0.9, the volume change in groundwater use is greater in the dryer locations but the ratio of the change to the mean value is greater in the wetter locations because mean groundwater use is much smaller in the wetter locations. This effect is more marked at the 45/15 scheme since mean groundwater use is negligible in the wetter locations when drop depth is large. The same reasoning can be applied to effluent discharge. In this case, the sensitivity is greatest at the dry locations because mean discharge is smaller there than at the wetter locations.

Pan coefficient (the proportionality constant that relates pond evaporation to class A pan evaporation) is one of the “weaker” model inputs in that it is not based upon site specific measurements but on averages from aquacultural literature. The literature values also indicate that pan coefficient may vary with time of the year. Ideally, models should have low sensitivity to independent variables of uncertain accuracy. In this case, the greatest sensitivity occurs when the model has been adjusted to give good performance. The increased sensitivity of the model to small adjustments in pan coefficient as model output approaches optimum may be the greatest weakness of this model.

Table 13. Relative Sensitivity of Groundwater Use to Pan Coefficient

Location	Groundwater Use (ha-cm/year) at 0.1 cm/day Infiltration		Relative Sensitivity	Groundwater Use (ha-cm/year) at 0.4 cm/day Infiltration		Relative Sensitivity
	Pan Coefficient =			Pan Coefficient =		
	0.7	0.9		0.7	0.9	
Scheme 15/7.5:						
Fairhope	16.69	34.23	2.85	92.48	116.94	0.93
Clemson	25.59	44.46	2.19	107.50	133.70	0.87
Stoneville	45.34	69.58	1.70	131.38	160.73	0.81
Stuttgart	43.76	68.92	1.82	136.01	166.64	0.81
Thomson	51.71	80.23	1.72	151.09	183.40	0.77
Scheme 45/7.5:						
Fairhope	1.02	9.49	8.61	73.53	100.51	1.24
Clemson	8.43	25.47	4.23	102.71	129.58	0.92
Stoneville	22.47	49.28	3.04	121.28	152.49	0.91
Stuttgart	25.75	53.61	2.84	130.10	161.16	0.85
Thomson	35.94	67.80	2.46	143.59	176.88	0.83

Table 14. Relative Sensitivity of Effluent Discharge to Pan Coefficient

Location	Effluent Discharge (ha-cm/year) at 0.1 cm/day Infiltration		Relative Sensitivity	Effluent Discharge (ha-cm/year) at 0.4 cm/day Infiltration		Relative Sensitivity
	Pan Coefficient =			Pan Coefficient =		
	0.7	0.9		0.7	0.9	
Scheme 15/7.5:						
Fairhope	53.33	43.26	-0.85	19.71	16.58	-0.69
Clemson	31.63	23.51	-1.21	4.18	3.45	-0.76
Stoneville	32.95	25.94	-0.97	9.62	7.53	-1.01
Stuttgart	22.35	16.66	-1.23	5.21	4.84	-0.35
Thomson	16.79	12.12	-1.26	6.79	5.83	-0.64
Scheme 45/7.5:						
Fairhope	37.82	18.87	-2.81	1.43	0.79	-2.18
Clemson	14.82	5.19	-4.16	0.08	0.05	-1.81
Stoneville	10.78	6.20	-2.31	0.21	0.09	-2.93
Stuttgart	5.01	2.04	-3.80	0.02	0.02	-0.10
Thomson	1.70	0.41	-5.33	0.02	0.01	-1.13

4.4.2 Relative Sensitivity to Infiltration Rate

Tables 15 and 16 contain relative sensitivities of groundwater use and effluent discharge to changes in infiltration rates at each location. Infiltration rates range from 0.0-0.2 cm/day (low rate) to 0.3-0.5 cm/day (high rate). A pan coefficient average of 0.8 is used in this analysis. Groundwater use using a pan coefficient average of 0.8 at infiltration rates of 0.1 and 0.4 cm/day was as presented on Table 11. In general, among the locations, relative sensitivity of groundwater use to change of infiltration is higher at wet location (Fairhope) and low at dry location (Thomsons) for both management schemes (15/7.5 and 45/7.5) and both infiltration rates (low and high rates). As with pan coefficient, the model appears to be more sensitive to changes in infiltration when dependent variables (groundwater use or effluent discharge) are small. Although less pronounced, there is a similar effect when looking at effluent discharge (Table 16).

For management scheme 45/7.5, at wet locations (Fairhope and Clemson), groundwater use is more sensitive to infiltration at lower infiltration, but at dry locations (Stoneville, Stuttgart, and Thomsons) groundwater use is more sensitive to infiltration at higher infiltration. At wet locations, minimum groundwater use is predicted when infiltration is low. At dryer locations, discharge of effluent to surface waters is minimal when infiltration is large. Again, the sensitivity is greatest when the dependent variable(s) are minimized.

The greatest sensitivities appear to occur in the predictions of effluent discharge when infiltration is large (0.3 to 0.5 cm/day) and the management scheme is 45/15. The explanation of this is similar to the explanations presented previously: sensitivity is greatest when dependent variable values are smallest.

Sensitivity of the model to infiltration is less troubling than sensitivity to pan coefficient. As mentioned, use of the pan coefficient is based on the assumption that ponds in specific locations behave similarly to research ponds located elsewhere, and that the pan coefficient remains constant throughout the year. Infiltration, by contrast, is inherently site specific and subject to some reasonable degree of local measurement. The model does exhibit variable sensitivity to infiltration but the independent variable is at least measurable locally.

Table 15. Relative Sensitivity of Groundwater Use to Infiltration

Location	Groundwater Use (ha-cm/year) Infiltration Rate = 0.0	0.2	Relative Sensitivity	Groundwater Use (ha-cm/year) Infiltration Rate = 0.3	0.5	Relative Sensitivity
Scheme 15/7.5:						
Fairhope	10.1	46.1	0.73	72.9	137.4	1.23
Clemson	17.5	57.9	0.59	86.7	155.8	1.15
Stoneville	36.2	82.2	0.40	113.3	180.1	0.92
Stuttgart	33.4	84.8	0.46	116.5	186.3	0.93
Thomson	38.5	97.4	0.44	131.7	203.0	0.85
Scheme 45/7.5:						
Fairhope	0.0	21.8	2.77	52.2	123.1	1.63
Clemson	2.4	44.2	1.30	79.9	152.7	1.25
Stoneville	11.8	66.7	0.78	101.3	173.3	1.05
Stuttgart	12.3	73.3	0.78	109.2	182.3	1.00
Thomson	20.3	87.3	0.64	123.8	196.7	0.91

Table 16. Relative Sensitivity of Effluent Discharge to Infiltration

Location	Effluent Discharge (ha-cm/year) Infiltration Rate = 0.0	0.2	Relative Sensitivity	Effluent Discharge (ha-cm/year) Infiltration Rate = 0.3	0.5	Relative Sensitivity
Scheme 15/7.5:						
Fairhope	69.45	32.59	-0.39	22.79	14.35	-0.93
Clemson	46.49	13.98	-0.60	6.30	2.56	-1.96
Stoneville	44.65	17.73	-0.46	12.36	6.26	-1.48
Stuttgart	32.98	11.48	-0.58	6.69	3.46	-1.56
Thomson	23.40	9.46	-0.47	7.36	5.53	-0.61
Scheme 45/7.5:						
Fairhope	59.32	8.71	-0.94	2.86	0.74	-3.61
Clemson	31.63	0.85	-1.66	0.18	0.05	-4.81
Stoneville	20.73	2.92	-1.12	1.00	0.01	-12.00
Stuttgart	12.42	0.71	-1.87	0.06	0.02	-3.88
Thomson	5.93	0.08	-3.00	0.03	0.00	-4.00

4.5 Effect of Intentional Harvest Discharge

Water is sometimes discharged intentionally from catfish pond when fish are harvested in order to reach a depth to about 1 m, so that the harvesting work can be done easier. This is because the limited depth of the nets that farmers use. Intentional harvest discharge may increase totals of annual effluent discharge and groundwater use.

In this scenario, simulation is run for 45 years and annual groundwater use and effluent discharge are recorded and then averaged. The increase of annual groundwater use and effluent discharge are calculated by using the difference between the simulation outputs with and without incorporation of a single harvest per year, with harvest date at the end of each month. For each location, 45/7.5 cm management scheme at infiltration rate of 0.1 cm/day is used.

Figure 18 shows the effect of harvest discharge on the increase of annual effluent discharge. All locations show about the same trend, increase of effluent discharge rises in late fall to about late winter, and decrease from spring to about mid fall. This corresponds to the precipitation minus evaporation (P-0.8E) and pond water depth at each of locations. High precipitation, low evaporation, and high pond water depth have caused high effluent discharge when harvest is performed in winter. In spring even P-0.8E is negative, the increases of effluent discharge due to harvest discharge are high because pond water level is still high. In summer, effect of harvest discharge is low because low precipitation, high evaporation, and low pond water depth. In fall even when P-0.8E is positive, pond water level is still low and increase of effluent discharge due to the harvest discharge is low.

If we compare the peak of the increases among the locations, Fairhope and Thomsons have lower peaks, while Stoneville have higher peak in late winter or early spring. This is because the differences between maximum and minimum P-0.8E's at Fairhope and Thomsons are not as high as at Stoneville and Stuttgart. Clemson and Stuttgart have moderate differences between the peak and the lowest increases of the discharge.

Based on all the graphs, we determine that best times for harvest is between late summer to fall in order to avoid high increase of effluent discharge. This is just a single harvest per one year. For two time harvests per year, it will need a strategy to schedule the second best time for the other harvest. End of January (day 30) may be a good time for the second harvest because soon after water is released pond water will be made up by abundant rainwater in winter.

Figure 19 shows examples of combinations of two time harvests per year. The total increase of the effluent discharge from any combination of two harvest dates per year is more or less summation of discharges from each of the harvest dates. For example, increase of discharge from two harvest dates per year on days 30 and 270 at Stoneville is 26 ha-cm/year as on Figure 22a, equal to 21 ha-cm/year (discharge increase of harvest on day 30) plus 5 ha-cm/year (discharge increase of harvest on day 270) as on Figure 19c. Combinations of two harvest dates on days 30 and 270, and on days 180 and 360 provide low increases of effluent discharges for most locations.

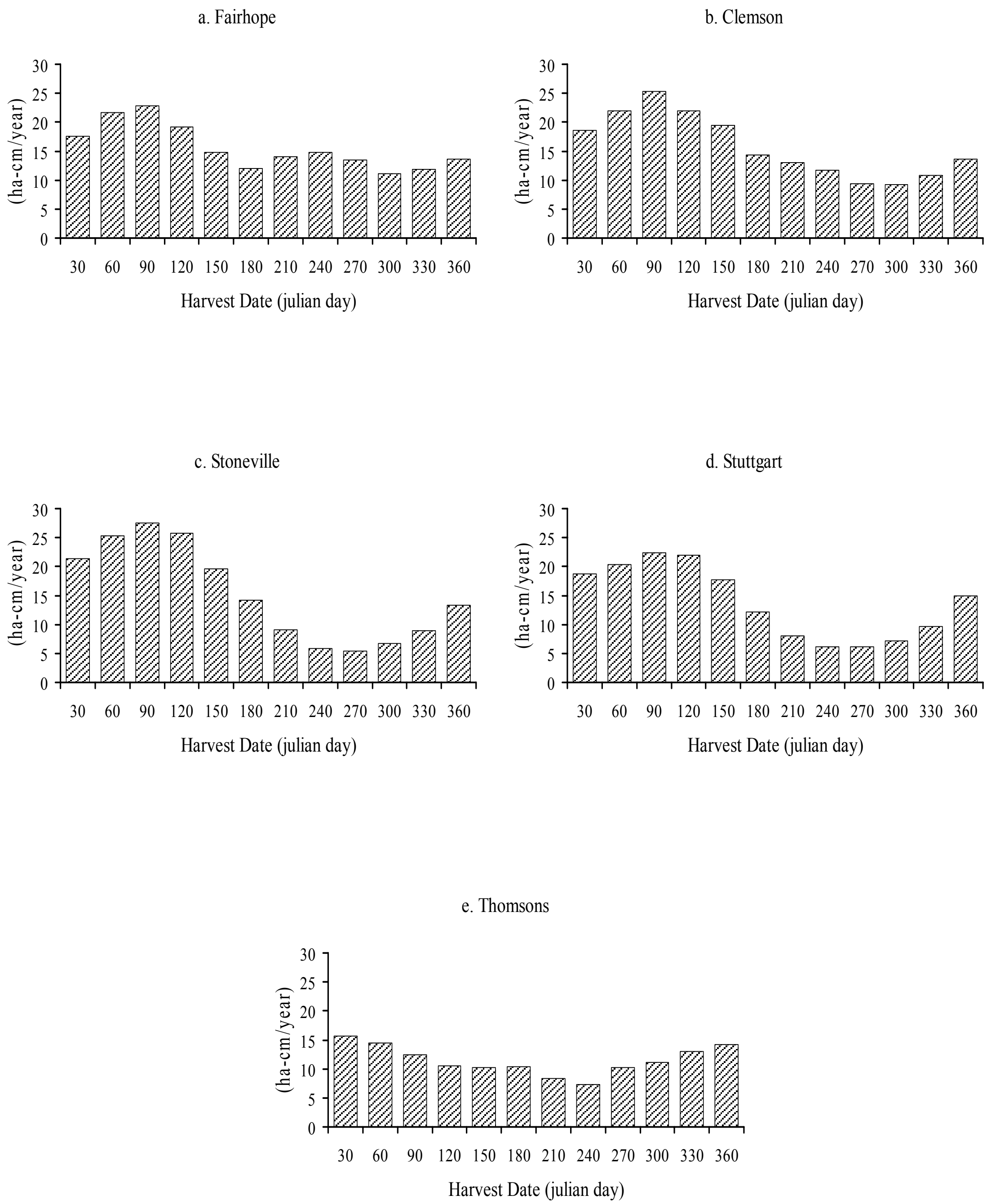


Figure 18. Increase of Effluent Discharge from One Time Harvest per Year of a 45/7.5 Management Scheme at 0.1 cm/day Infiltration

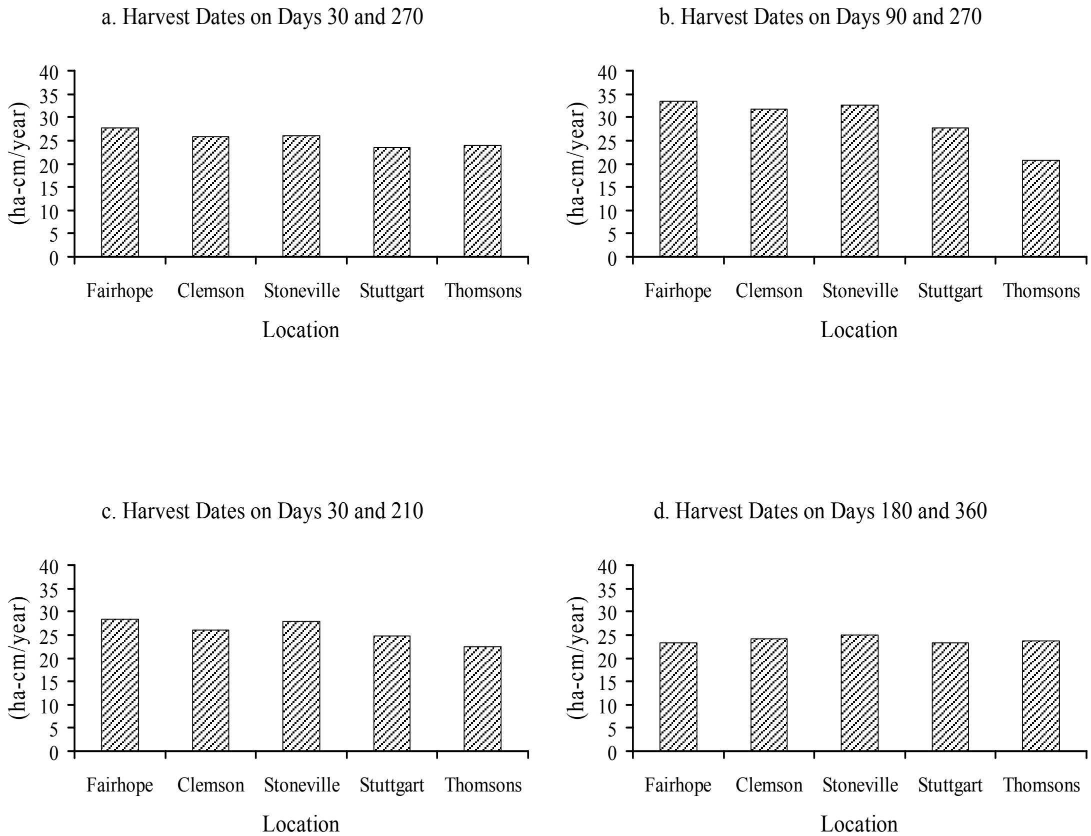


Figure 19. Increase of Effluent Discharge from Two Time Harvests per Year of a 45/7.5 Management Scheme at 0.1 cm/day Infiltration

Instead of discharging water to environment, farmers can transfer the water to next closest pond that is not full yet. If all ponds are full when harvesting times, the water cannot be saved to other ponds. This scenario is to estimate the amount of water that can be saved by using the technique. A pond (first pond) is emptied to the depth of -45 cm (the minimum water level) when harvesting, and the water is transferred to a next closest pond (second pond). So, when water level of the ponds is less than half of the drop depth (-22.5 cm), all the water from the first pond can be transferred to the second pond since the storage capacity is enough. However, if water level of both ponds is above half, the storage capacity of the second is not enough to store all the transferred water. Some portion of the water has to be discharged to the environment. When harvesting the second pond, similar technique is performed, the second pond is drained to -45 cm, and the water in stored in the first pond.

Hence, after harvesting the two ponds, they have different water level (different storage capacity). To simulate this technique, it is assumed that a portion of the water is transferred back from the first to the second pond so that the water of the two ponds is always at the same level. This method is equivalent to discharging pond water to half depth (-22.5 cm) instead of minimum depth (-45 cm) when harvesting. The pond is partially drained only when water level is above -22.5 cm. This is the portion of water (above -22.5 cm) that cannot be stored in the second pond using the transfer technique as mentioned above.

Figure 20 shows increase of annual effluent discharge at each of the locations if water is transferred to next closest pond or pond is emptied to -22.5 cm depth when the pond is harvested. We can see that increase of effluent discharge can be greatly reduced. Increase of effluent discharge at all locations is less than 10 ha-cm/year. Most locations still show peak increases at winter but for dry locations there are minor or no increases at all in summer and fall. Stoneville and Stuttgart show no increase of annual effluent discharge when harvest is carried out in late summer and fall. Thomsons shows just minor increase of annual effluent discharge at all time schedules of harvest.

Transfer technique or partial drain of water to half of drop depth appears to be an effective method in reducing effluent discharge when harvesting. This method is very helpful in reducing environmental pollution when discharge cannot be avoided when fish harvest is conducted. As a typical set of ponds consists of four ponds, transferring water from harvested pond to next closest pond should not be a problem. In the above scenario, the partially drained water of first pond is transferred to second pond only. If the drained water is transferred to the other three ponds (besides the four ponds), the there should be more water that can be saved and may be no harvest discharge at all time.

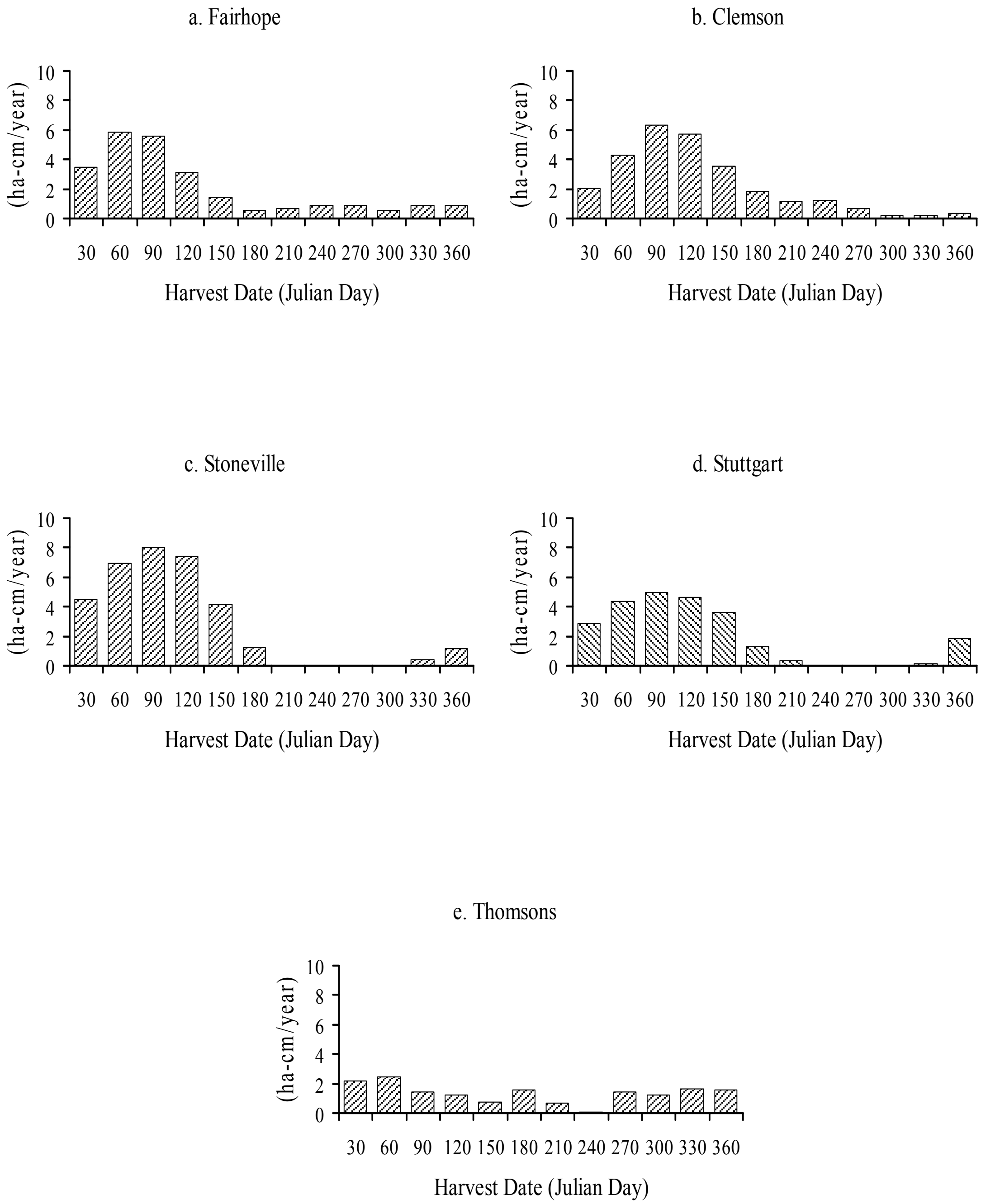


Figure 20. Increases of Effluent Discharge from One Time Harvest per Year of a 45/7.5 Management Scheme at 0.1 cm/day Infiltration if Water is Transferred to Adjacent Pond

4.6 Mass Discharge of Water Constituents

This scenario is to estimate average of mass of selected water quality variables discharged annually at each location. Mass discharges from two management schemes (15/7.5 and 45/7.5) at an infiltration rate of 0.1 cm/day are compared. Scheme 15/7.5 (6/3 inch) has been recommended in previous research for catfish ponds in the Delta Mississippi (Pote and Wax, 1993; Pote et al., 1988). Scheme 45/7.5 appears to provide good performance at an infiltration rate of 0.1 cm/day for most locations.

Table 17. Concentrations of Selected Water Quality Variables (Means and Ranges) in Potential Overflow Effluents from 20 Commercial Channel Catfish Ponds in Northwest Mississippi Sampled over 2 years. TSS = Total Suspended Solids; TN = Total Nitrogen; TP = Total Phosphorus; BOD_5 = 5 Day Biochemical Oxygen Demand.

Variables	Concentration (mg/l)			
	Spring	Summer	Fall	Winter
TSS	129 (46-289)	122 (40-225)	87 (22-175)	101 (39-194)
TN	4.9 (1.5-7.9)	6.6 (2.6-14.1)	6.5 (2.9-10.8)	5.3 (0.6-8.8)
TP	0.34 (0.15-0.58)	0.53 (0.23-1.24)	0.3 (0.14-0.62)	0.34 (0.13-0.62)
BOD_5	14.9 (8.2-27.1)	23.6 (10.5-41.2)	11.0 (1.9-34.0)	12.8 (4.8-29.7)

Source: (Tucker and Hargeaves, 2003a), (Tucker et al., 1996)

Selected water quality variables (Table 17) are adopted from previous research at the Delta Research and Extension Center (Tucker and Hargeaves, 2003a; Tucker et al., 1996). The Catfish pond water quality data were collected over 2 years in 1991 to 1993 (Tucker et al., 1996). Since this scenario is for comparison purposes only, between the two management schemes at each of locations, it is assumed that the concentrations of the selected water quality variables are applicable to each of the locations. In fact, this water quality data were comparable to data from other research on experimental catfish ponds at Auburn, AL (Seo and Boyd, 2001). Based on available literature, water quality of catfish ponds tends to be stable over time regardless of pond ages (Zimba et al., 2003). So, seasonal water quality is also assumed constant over time in this scenario.

To calculate mass discharge (kg/ha/year), concentration (mg/l) is multiplied by volume of effluent discharge (l/ha/year). The simulation was run for 45 years, and then annual mass discharge was recorded and averaged. The mass discharge was from overflow only, while intentional drain and harvest discharge were not included in this scenario. Outputs of the simulation using the two management schemes are presented on Tables 18 to 22.

Table 18. Mass Discharge (kg/ha/year) of Selected Water Quality Variables Using Schemes 15/7.5 and 45/7.5 at Infiltration Rate of 0.1 cm/day at Fairhope*)

Variables	Scheme 15/7.5					Scheme 45/7.5				
	Sp	Sm	Fl	Wn	Total	Sp	Sm	Fl	Wn	Total
TSS	120.1	156.2	62.9	183.0	522.2	88.3	84.1	28.1	99.5	299.9
TN	4.6	8.4	4.7	9.5	27.2	3.4	4.5	2.1	5.2	15.2
TP	0.3	0.7	0.2	0.6	1.8	0.2	0.4	0.1	0.3	1.0
BOD_5	14.1	30.0	7.9	22.9	74.9	10.2	16.3	3.6	12.6	42.6

*) Sp: spring, Sm: summer, Fl: fall, and Wn: winter

At Fairhope (Table 18), the wettest location, predicted mass discharge was higher compared to other locations. Using scheme 45/7.5, predicted mass discharge of the constituents was reduced about 40% as compared to that of scheme 15/7.5. In an earlier section, it was shown that effluent discharge of Fairhope at infiltration rate of 0.1 cm/day cannot be reduced effectively by implementing deeper drop depth. Concentrations are generally high in summer (Table 17), but mass discharges are not much different between summer and winter because volume of effluent discharge is higher in winter than in summer. Mass discharges in fall are generally the lowest because of low concentration and low volume of effluent discharge at one time.

Table 19. Mass Discharge of Selected Water Quality Variables Using Schemes 15/7.5 and 45/7.5 at Infiltration Rate of 0.1 cm/day at Clemson*)

Variables	15/7.5 Scheme					45/7.5 Scheme				
	Sp	Sm	Fl	Wn	Total	Sp	Sm	Fl	Wn	Total
TSS	59.8	29.6	46.5	146.2	282.0	48.4	2.2	1.7	51.8	104.1
TN	2.3	1.6	3.4	7.7	15.0	1.8	0.1	0.1	2.7	4.8
TP	0.2	0.1	0.2	0.5	0.9	0.1	0.0	0.0	0.2	0.3
BOD_5	7.1	5.7	5.8	18.5	37.1	5.6	0.4	0.2	6.6	12.8

*) Sp: spring, Sm: summer, Fl: fall, and Wn: winter

Mass discharge at Clemson (Table 19) is much lower that that at Fairhope for both management schemes. Mass discharge at Clemson can be reduced about 60% when 45/7.5 scheme is used. Mass discharges in winter are generally much higher than those in summer even summer concentrations are high. It is clearly due to high volume of effluent discharge in winter. Summer and fall mass discharge are lower than in spring and winter due to low volume of effluent discharge.

Table 20. Mass Discharge of Selected Water Quality Variables Using Schemes 15/7.5 and 45/7.5 at Infiltration Rate of 0.1 cm/day at Stoneville

Variables	15/7.5 Scheme					45/7.5 Scheme				
	Sp	Sm	Fl	Wn	Total	Sp	Sm	Fl	Wn	Total
TSS	58.3	11.3	59.3	168.1	297.0	38.8	0.0	1.0	48.7	88.5
TN	2.1	0.6	4.4	8.9	16.1	1.5	0.0	0.1	2.6	4.1
TP	0.2	0.0	0.2	0.6	1.0	0.1	0.0	0.0	0.2	0.3
BOD_5	6.5	2.2	7.5	21.5	37.6	4.5	0.0	0.1	6.2	10.8

*) Sp: spring, Sm: summer, Fl: fall, and Wn: winter

Total mass discharge at Stoneville (Table 20) was not much different from that at Clemson. The difference was not consistent between the two management schemes. Stoneville has slightly higher total mass discharge for scheme 15/7.5 and slightly lower for scheme 45/7.5 when compared to Clemson. It seems that higher total discharge of Stoneville for scheme 15/7.5 resulted from higher discharges in fall and winter. But Clemson and Stoneville are not significantly different with respect to fall and winter P-0.8E's (Table 5). It seems that Stoneville has higher frequency of rainfall in fall and winter so that the 15/7.5 scheme cannot perform as well as in Clemson. But 45/7.5 scheme can handle the high frequency of rainfall in fall and winter better than the 15/7.5 scheme, making Stoneville's total mass discharge lower than that of Clemson. This is consistent with the order of P-0.8E's of the two locations. Total mass discharge at Stoneville can be reduced about 70% when scheme 45/7.5 is used as compared to scheme 15/7.5.

Table 21. Mass Discharge of Selected Water Quality Variables Using Schemes 15/7.5 and 45/7.5 at Infiltration Rate of 0.1 cm/day at Stuttgart*)

Variables	15/7.5 Scheme					45/7.5 Scheme				
	Sp	Sm	Fl	Wn	Total	Sp	Sm	Fl	Wn	Total
TSS	58.1	3.8	49.8	79.9	191.7	26.2	0.0	0.3	10.8	37.3
TN	2.2	0.2	3.7	4.2	10.3	1.0	0.0	0.0	0.6	1.6
TP	0.2	0.0	0.2	0.3	0.6	0.1	0.0	0.0	0.0	0.1
BOD_5	6.8	0.7	6.3	10.1	23.9	3.0	0.0	0.0	1.4	4.4

*) Sp: spring, Sm: summer, Fl: fall, and Wn: winter

At Stuttgart (Table 21), reduction of mass discharge was about 80% when the 45/7.5 scheme was used as compared to the 15/7.5 scheme. Compared to Stoneville, total mass discharge of Stuttgart is much lower that that of Stoneville although P-0.8E's of the two locations are not significantly different (Table 3). It seems that the big difference is caused by the difference of summer and winter discharges. Stoneville discharges higher mass in summer and winter than Stuttgart does, particularly for the 15/7.5 scheme. But, winter rainfalls and P-0.8E's between the two locations are the only significant differences (Table 4 and 5). Higher discharge of Stoneville in summer is likely caused by higher rainfall frequencies in Stoneville than in Stuttgart.

Table 22. Mass Discharge of Selected Water Quality Variables Using 15/7.5 and 45/7.5 Schemes at Infiltration Rate of 0.1 cm/day at Thomsons

Variables	15/7.5 Scheme					45/7.5 Scheme				
	Sp	Sm	Fl	Wn	Total	Sp	Sm	Fl	Wn	Total
TSS	33.8	54.5	41.9	29.8	160.0	1.9	3.1	0.5	5.2	10.7
TN	1.3	3.2	2.9	1.5	8.9	0.1	0.2	0.0	0.3	0.6
TP	0.1	0.2	0.1	0.1	0.6	0.0	0.0	0.0	0.0	0.0
BOD_5	3.9	11.3	5.0	3.7	23.9	0.2	0.7	0.0	0.7	1.6

*) Sp: spring, Sm: summer, Fl: fall, and Wn: winter

It is apparent that the improvement of mass discharge was affected by the P-0.8E's of the locations, being the most efficient at Thomsons where mass reduction was about 90% when the 45/7.5 scheme was used (Table 22). As mentioned earlier, effluent discharge is easily controlled by using deeper drop depths for drier locations. It should be noted that, particularly for drier locations, the average of annual mass discharges consist of many zero years (as discussed earlier). Discharge may occur in heavy storm and wet years only.

In this section, it has been demonstrated that mass discharge can be greatly reduced by using management schemes with deeper pond water storage capacity. This appears to be very effective for most locations. The constituent mass discharge, addition to concentration basis, is a critical criterion particularly when computing Total Maximum Daily Load (TMDL). Some states, such as Mississippi, use environmental impact in rivers as the water quality standard instead of effluent quality. The dissolved oxygen of rivers is normally used to measure the environmental impact in rivers. The Mississippi standard is that oxygen concentration in rivers should be $\geq$ 5 mg/l for non toxic materials. Based on this standard, TMDL over all dischargers to a certain river segment is computed to find the contribution of each discharger. So, each discharger may be different in terms of effluent quality, while the quantity of the effluent they release will be determined based on the TMDL.

While implementing waste water treatment to aquaculture is difficult in term of economical feasibility, drop/add management schemes using adequate pond water storage capacity to minimize effluent discharge is a promising alternative. The drop/add management scheme also enhances retention time, promoting in-pond treatment of nutrients and thus improving water quality.

4.7 Linked Pond System

Linking ponds may be more appropriate if applied to existing catfish ponds because it needs less pond modification. A group of adjacent ponds can be modified into a linked pond system regardless of the number and layout of the individual ponds relative to each other. This system consists of production ponds and a production/storage pond (Cathcart et al., 1999). The Production ponds are shallow, about 1 m deep while production/storage pond is deeper, for example 1.5 – 2 m deep. Thus the depth of production ponds do not need to be increased. The production/storage pond does need to be deepened. The function of the production/storage pond is simultaneously for fish culture and rainwater storage. Overflow may occur in the production pond(s) and this water is stored in the production/storage pond. When the production/storage pond is full, overflow will occur and the water may be discharged to the environment. Conversely, water is pumped from the production/storage pond to fill the production pond(s) if water level in the production pond drops below the minimum level. Groundwater is only pumped to the production pond(s) when water in the production/storage pond drops down to the minimum level.

Drop/add management scheme of 15/7.5 cm or 6/3 inches (Pote and Wax, 1993) can be applied to the production pond, so each pond has supplementary storage capacity. Deeper schemes such as 90/7.5 cm can be used for production/storage pond in order to get additional storage. The following scenario tested a linked pond system consisting of 15/7.5 cm for the production pond(s) and various drop depths (with a fill depth of 7.5 cm) for the production/storage pond. The system was tested at 6 infiltration rates for each of the 5 locations. A pond configurations of 1:1 (i.e., 1 production pond and 1 production/storage pond per one linked pond system), was tested in this scenario (Cathcart et al., 1999).

4.7.1 Groundwater Use

Figure 21 shows groundwater use in the linked pond system at each location. Each locations had zero predicted groundwater use at zero infiltration except for Thomsons. Fairhope had zero groundwater use at infiltration rates of 0 and 0.1 cm/day at relatively low drop depths (about 60 and 165 cm). Drier locations had zero groundwater use at zero infiltration with deeper drop depths. Clemson had practically zero groundwater use at drop depth of 105 cm. Stoneville and Stuttgart had zero groundwater use at deeper drop depths (210 and 255 cm each); while Thomsons did not have zero groundwater use even at zero infiltration. For an infiltration rate of 0.1 cm/day, the lowest groundwater use at Clemson was about 12 ha-cm/day. Drier locations (Stoneville, Stuttgart, and Thomsons) had higher lowest groundwater use: 52, 71, and 100 ha-cm/year, respectively.

For high infiltration rate, 0.5 cm/day, the lowest groundwater use at Fairhope was about 245 ha-cm/year at a drop depth of 120 cm. At a drop depth of 45 cm, the curve was nearly flat. At other locations, the curves of groundwater use were about flat from the beginning (drop depth of 15 cm) or by 30 cm for infiltration rates of 0.4 and 0.5 cm/day. The highest groundwater use was at Thomsons; about 400 ha-cm/year for an infiltration rate of 0.5 cm/day and 325 ha-cm/year for an infiltration rate of 0.4 cm/day.

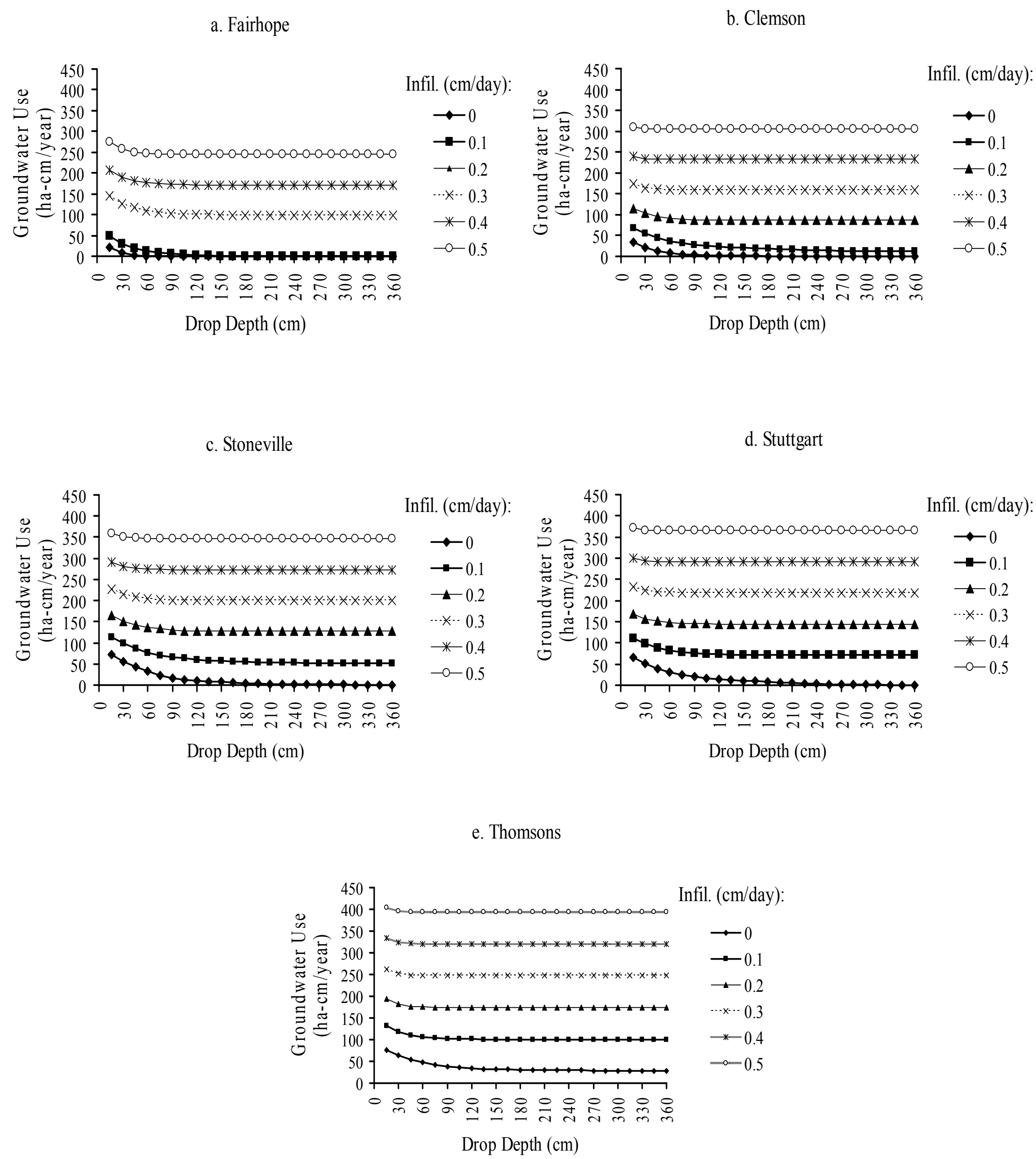


Figure 21. Effect of Infiltration Rate on Groundwater Use of Production/Storage Ponds with Various Drop Depths and a 7.5 cm Fill Depth

4.7.2 Effluent Discharge

Figure 22 shows effluent discharge of the 1:1 linked pond system at each location. At Fairhope, zero effluent discharge cannot be achieved for infiltration rates of 0 and 0.1 cm/day (Figure 22a). The lowest effluent discharge is about 119 ha-cm/year (at drop depth of 60 cm) for infiltration rate of 0 cm/day and 46 ha-cm/year (at drop depth of 165 cm) for infiltration rate of 0.1 cm. At Clemson (Figure 22b), effluent discharge flattened to 59 ha-cm/year at drop depth of 165 cm for an infiltration rate of 0 cm/day. Stoneville and Stuttgart had the lowest effluent discharges with a very deep drop depth for zero infiltration (Figures 22c and 22d). Thomsons is the only location that can have zero effluent discharge at zero infiltration (Figure 22e).

For higher infiltration rates, practical zero effluent discharges can be achieved. Fairhope has nearly zero discharge at drop depths of 240 cm for an infiltration rate of 0.2 cm/day. Clemson, Stoneville, and Stuttgart have nearly zero discharge at lower drop depths for infiltration rates of 0.2 cm/day. Thomsons has zero discharge at even lower drop depth (60 cm) for an infiltration rate of 0.2 cm/day.

At an infiltration rate of 0.5 cm/day, zero effluent discharge can be achieved with lower drop depths. Zero effluent discharge is achieved at Fairhope with drop depth of 120 cm. For other drier locations, very zero effluent discharge is achieved with very low drop depths (15 or 30 cm). At drop depths of 30, 45, or 60 cm effluent discharges are generally zero already. As previously mentioned, zero release refers solely to discharge to receiving surface waters. This does not truly make the system zero-discharge, since transport through the pond bottom can degrade both surface and groundwaters.

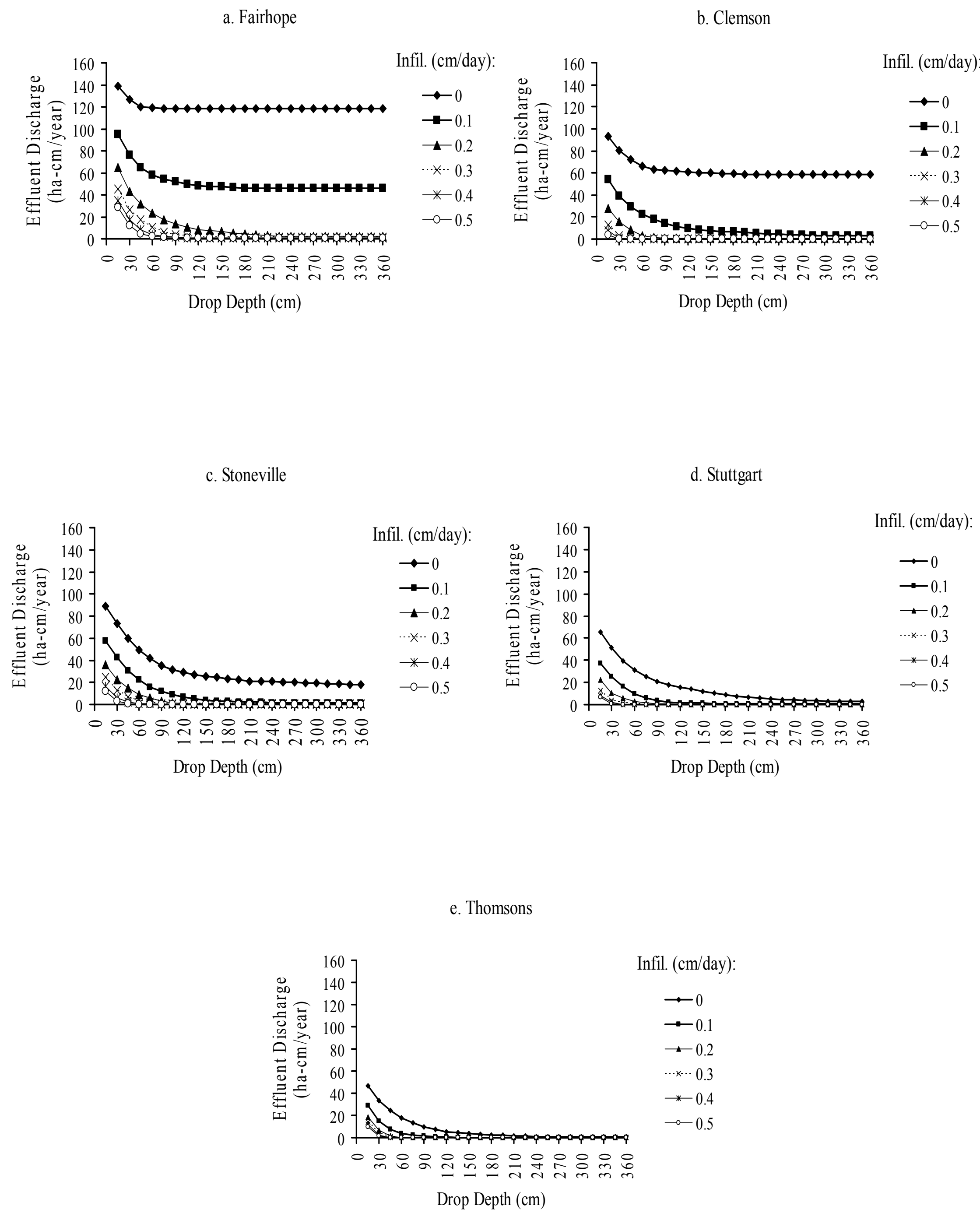


Figure 22. Effect of Infiltration Rate on Effluent Discharge of Production/Storage Ponds with Various Drop Depths and a 7.5 cm Fill Depth

4.7.3 Rainwater Stored

Figure 23 presents predicted percentages of rainwater that were applied to the pond water budget at each location. The percentages of rainwater stored varied from location to location and depended on infiltration rates. At Fairhope, 30% of the precipitation was used to meet water needs for an infiltration rate of 0 cm/day. For an infiltration rate of 0.1 cm/day, 73% was used for pond needs. Percentages of about 80 to 100% were stored for higher infiltration rates.

Clemson had the highest percentage of rainwater stored (about 57%) for an infiltration rate of 0 cm/day. At higher infiltration rates, percentages of rainwater that can be stored ranged from 60 to 100%. At infiltration rates of 0.3 to 0.5 cm/day, predicted percentages rainwater stored exceeded 90% with a drop depth only 15 cm.

Stoneville and Stuttgart had percentages of rainwater stored that exceeded 80% for an infiltration rate of 0 cm/day at maximum drop depths. At infiltration rates of 0.1 and 0.2 cm/day, 60% of precipitation can be applied to the water budget at drop depths of 30 and 45 cm, respectively. For Stoneville, the more than 80% of incident precipitation can be stored with drop depth of 15 cm at infiltration rates of 0.3 cm/day or higher. At Stuttgart, more than 90% can be stored with a drop depth of 15 cm at the same infiltration rates.

At Thomsons, the driest location, the percentage of stored rainwater exceeds 60% at an infiltration rate of 0 cm/day. At an infiltration rate of 0.1 cm/day, the percentage is more than 70% when drop depth is 15 cm. At higher infiltration rates, rates in excess of 80% can be stored with a drop depth of $\geq$ 15 cm.

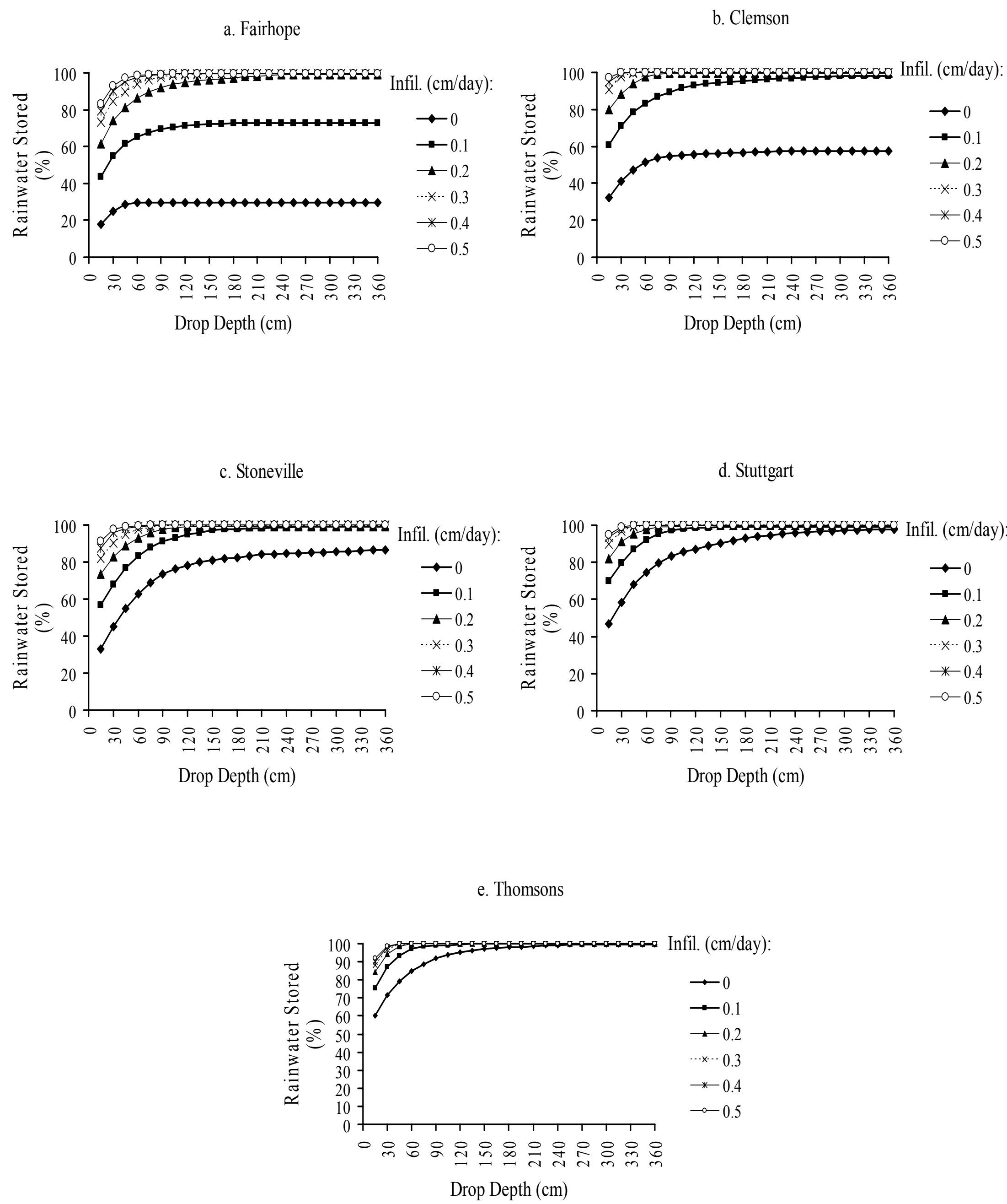


Figure 23. Effect of Infiltration Rate on Rainwater Stored of Production/Storage Ponds with Various Drop Depths and a 7.5 cm Fill Depth

4.7.4 Summary

The linked pond system is geometrically identical to the same number of unlinked ponds, each having a common drop depth calculated by averaging the drop depths used in the system. For example, a 1:1 system consisting of 15/7.5 pond and a 75/7.5 pond would be mathematically equivalent to two unlinked 45/7.5 ponds. The principle advantage of the linked system, as mentioned, is that it requires that fewer ponds be modified. This both reduces initial cost and increases the ease of implementing an aggressive drop/add system.

Figure 24 shows projected pond performance at drop/add depths that appear to provide good performance. The best ponds are those that do not leak. In much of the SE United States, though, some degree of infiltration has been reported. Figure 28 shows performance for a pond at 0.1 cm/day infiltration is. In all of these scenarios, pond performance provides the advantages cited above for aggressive applications of the drop/add approach.

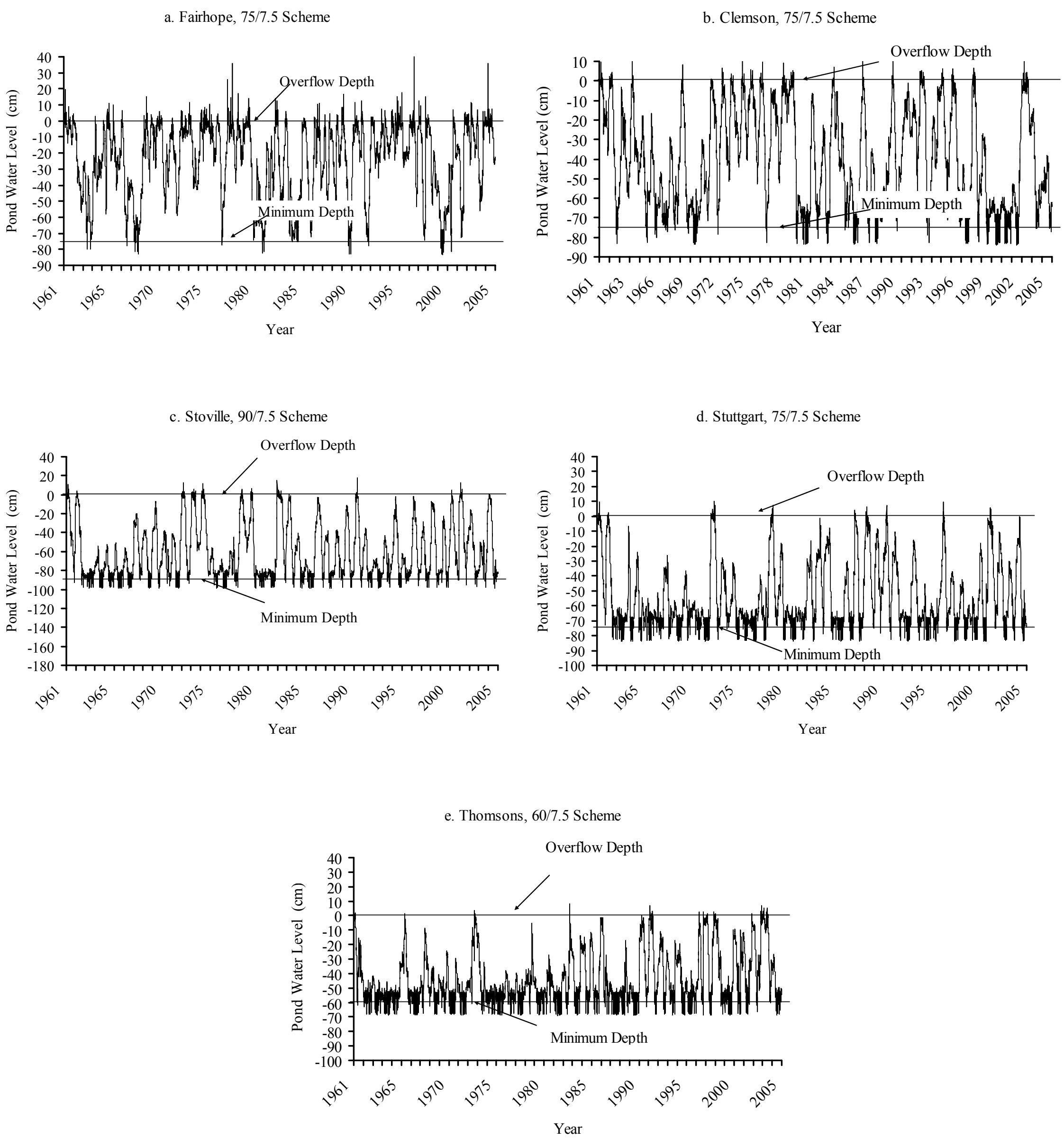


Figure 24. Pond Water Level of Production/Storage Pond at 0.1 cm/day Infiltration Rate at Each Location

CHAPTER V

CONCLUSION

Precipitation minus pond evaporation (P-0.8E) among the five locations in the Southeastern US was found; 58.5 cm, 28.4 cm, 8.1 cm, -0.9 cm, and -15.3 cm respectively for Fairhope, AL; Clemson, SC; Stoneville, MS; Stuttgart, AR; and Thomsons, TX. The south-to-north climate gradient across Fairhope, Stoneville, and Stuttgart was observed. Fairhope was the wettest but P-0.8E's of Stoneville and Stuttgart were not significantly different when statistical tests were applied. Similarly, the east-to-west climate gradient across Clemson, Stoneville, and Thomsons was also clear. Clemson was the wettest location and Thomsons was driest. Most locations had a precipitation-evaporation deficit (negative P-0.8E) in spring and summer except for. Fairhope had water positive P-0.8E at all year around.

At all locations, climates did not show significant differences among three consecutive blocks of 15 years (1961-1975, 1976-1990, 1991-2005) with respect to annual and seasonal P-0.8E's. The lack of significance seemed to be caused by high variability of the climate. Therefore, it is better to use 45 year climatological data in the model than to use any 15 year data sets, in order to handle the variability over 45 years.

The drop/add management scheme appeared to be an effective strategy to reduce groundwater use and effluent discharge of catfish ponds. The differences of climates among the locations in The Southeast US affected the performance of drop/add management schemes. Most locations could have zero groundwater use at zero infiltration, except for Thomsons. Groundwater use at Thomsons could not be eliminated (minimum of 15 ha-cm/year) but the number of zero groundwater use years was still about 50% of the simulation period. At this ultimate point (zero infiltration), rainwater contribution to the total water budget at Thomsons could be almost 90%.

At all infiltration rates, effluent discharge to receiving surface waters could be easily eliminated but groundwater use increased appreciably. A management scheme of 45/7.5 could be used with good results at most locations, depending on infiltration rates. At a wet location, such as Fairhope, well sealed ponds worked well with regard to reducing use of groundwater but it was difficult to effectively reduce effluent discharge. For drier locations, a management scheme 45/7.5 performed well at low infiltration rates, (≤ 0.1 cm/day). At these infiltration rates, groundwater use was generally still low and effluent discharge was close to zero for most locations. The contribution of rainwater to the water budget was 90% at Clemson, 70% at the Thomsons, and between these limits at the other locations for low infiltration rates. Deeper fill depths have little effect on the increases of groundwater use and effluent discharge. Schemes such as 45/15 or 45/30 could be used at infiltrate rate of 0.1 cm/day at most locations as well.

The sensitivity analysis showed that model sensitivity to pan coefficient and infiltration rate was affected by infiltration rate and pond water storage capacity (drop depth). The model was more sensitive to pan coefficient rather than to infiltration rate at lower infiltration rates and *vice-versa*. Both sensitivities of the model, however, increased when pond water deeper storage capacity was used.

Draining pond water to the depth of -45 cm (from zero/overflow level) for harvesting increased total annual groundwater use and effluent discharge for all locations. The increases were the highest if harvests were scheduled at late winter to early spring for most locations. Minimum increases were about 5 to 15 ha-cm/year if harvests were scheduled on late summer to late fall. The increases of groundwater use and effluent discharge could be reduced greatly if the water was transferred to an adjacent pond. The increases of groundwater use and effluent discharge were all less than 10 ha-cm/year for all time schedules for each of location, being minimum or no discharge at all for late summer to late fall schedules.

Mass discharge of some water constituents was reduced about 40% at Fairhope to 90% at Thomsons when 45/7.5 scheme was used instead of a 15/7.5 scheme. As concentration of some water quality constituents is high even in winter, reduction of mass discharge from aquaculture ponds using a drop/add management scheme becomes very important especially when computing TMDL.

Linked pond systems are an interesting alternative in that it needs less pond modification when drop/add management scheme is applied to existing ponds. It was shown that linked pond system worked very well in reducing groundwater use and effluent discharge, and increasing rainwater capture at most infiltration rates.

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APPENDIX A

SAS OUTPUTS OF ONE WAY CLASSIFICATION ANALYSIS OF VARIANCE AND LSD MULTIPLE COMPARISONS FOR ANNUAL PRECIPITATION AMONG LOCATIONS

The SAS System

The MEANS Procedure

Analysis Variable : Annual Precipitation

LOCATION	N Obs	Mean	Std Dev
Clemson	45	136.26578	22.39110
Fairhope	45	168.95667	33.03972
Stonevil	45	133.13867	27.33977
Stuttgar	40	122.00975	22.17130
Thomsons	40	115.82200	27.40881

The SAS System

The GLM Procedure

Class Level Information

Class	Levels	Values
LOCATION	5	Clemson Fairhope Stonevil Stuttgar Thomsons

Number of Observations Read	215
Number of Observations Used	215

The SAS System

The GLM Procedure

Dependent Variable: Annual Precipitation

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	4	73363.1386	18340.7846	25.43	<.0001
Error	210	151449.2324	721.1868		
Corrected Total	214	224812.3709			

R-Square	Coeff Var	Root MSE	Annual Mean
0.326331	19.74659	26.85492	135.9978

Source	DF	Type I SS	Mean Square	F Value	Pr > F
LOCATION	4	73363.13857	18340.78464	25.43	<.0001

Source	DF	Type III SS	Mean Square	F Value	Pr > F
LOCATION	4	73363.13857	18340.78464	25.43	<.0001

The SAS System

The GLM Procedure

t Tests (LSD) for Annual Precipitation

NOTE: This test controls the Type I comparisonwise error rate, not the experimentwise error rate.

Alpha	0.05
Error Degrees of Freedom	210
Error Mean Square	721.1868
Critical Value of t	1.97132
Least Significant Difference	11.436
Harmonic Mean of Cell Sizes	42.85714

NOTE: Cell sizes are not equal.

Means with the same letter are not significantly different.

t Grouping		Mean	N	LOCATION
	A	168.957	45	Fairhope
	B	136.266	45	Clemson
	B			
C	B	133.139	45	Stonevil
C				
C	D	122.010	40	Stuttgar
	D			
	D	115.822	40	Thomsons

APPENDIX B

SAS OUTPUTS OF ONE WAY CLASSIFICATION ANALYSIS OF VARIANCE

AND LSD MULTIPLE COMPARISONS FOR ANNUAL P-0.8E

AMONG LOCATIONS

The SAS System

The MEANS Procedure

Analysis Variable: Annual P-0.8E

LOCATION	N Obs	Mean	Std Dev
Clemson	45	28.89178	26.91596
Fairhope	45	58.54200	34.80909
Stonevil	45	8.10267	29.20526
Stuttgar	40	-1.57975	25.04008
Thomsons	40	-18.05325	31.35234

The SAS System

The GLM Procedure

Class Level Information

Class	Levels	Values
LOCATION	5	Clemson Fairhope Stonevil Stuttgar Thomsons

Number of Observations Read	215
Number of Observations Used	215

The SAS System

The GLM Procedure

Dependent Variable: Annual P-0.8E

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	4	150448.9526	37612.2381	42.58	<.0001
Error	210	185508.9472	883.3759		
Corrected Total	214	335957.8998			

R-Square	Coeff Var	Root MSE	Annual Mean
0.447821	181.8577	29.72164	16.34335

Source	DF	Type I SS	Mean Square	F Value	Pr > F
LOCATION	4	150448.9526	37612.2381	42.58	<.0001

Source	DF	Type III SS	Mean Square	F Value	Pr > F
LOCATION	4	150448.9526	37612.2381	42.58	<.0001

The SAS System

The GLM Procedure

t Tests (LSD) for Annual P-0.8E

NOTE: This test controls the Type I comparisonwise error rate, not the experimentwise error rate.

Alpha	0.05
Error Degrees of Freedom	210
Error Mean Square	883.3759
Critical Value of t	1.97132
Least Significant Difference	12.657
Harmonic Mean of Cell Sizes	42.85714

NOTE: Cell sizes are not equal.

Means with the same letter are not significantly different.

t Grouping	Mean	N	LOCATION
A	58.542	45	Fairhope
B	28.892	45	Clemson
C	8.103	45	Stonevil
C			
C	-1.580	40	Stuttgar
D	-18.053	40	Thomsons

APPENDIX C

SAS OUTPUTS OF TWO-WAY CLASSIFICATION ANALYSIS OF VARIANCE AND LSD MULTIPLE COMPARISONS FOR SEASONAL PRECIPITATION

The SAS System

The MEANS Procedure

Analysis Variable : Seasonal Precipitation

LOCATION	SEASON	N Obs	Mean	Variance	Std Dev
Clemson	fall	45	31.599	113.608	10.659
	spring	45	33.366	133.781	11.566
	summer	45	32.988	148.366	12.181
	winter	45	38.565	107.091	10.348
Fairhope	fall	45	33.426	200.019	14.143
	spring	45	40.129	337.996	18.385
	summer	45	53.462	236.563	15.381
	winter	45	41.940	199.478	14.124
Stonevil	fall	45	33.699	153.089	12.373
	spring	45	35.699	155.140	12.456
	summer	45	25.116	93.697	9.680
	winter	45	38.625	149.996	12.247
Stuttgar	fall	45	32.435	156.371	12.505
	spring	45	35.890	121.982	11.045
	summer	45	23.471	74.614	8.638
	winter	45	31.283	106.718	10.330
Thomsons	fall	45	29.453	178.071	13.344
	spring	45	30.598	177.577	13.326
	summer	45	34.066	261.880	16.183
	winter	45	23.628	112.609	10.612

The SAS System

The GLM Procedure

Class Level Information

Class	Levels	Values
LOCATION	5	Clemson Fairhope Stonevil Stuttgar Thomsons
SEASON	4	fall spring summer winter

Number of Observations Read	900
Number of Observations Used	900

The SAS System

The GLM Procedure

Dependent Variable: Seasonal Precipitation

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	19	39380.1451	2072.6392	12.88	<.0001
Error	880	141620.4212	160.9323		
Corrected Total	899	181000.5663			

R-Square	Coeff Var	Root MSE	P_E Mean
0.217569	37.34219	12.68591	33.97206

Source	DF	Type I SS	Mean Square	F Value	Pr > F
LOCATION	4	17941.29430	4485.32357	27.87	<.0001
SEASON	3	1237.26459	412.42153	2.56	0.0536
LOCATION*SEASON	12	20201.58620	1683.46552	10.46	<.0001

Source	DF	Type III SS	Mean Square	F Value	Pr > F
LOCATION	4	17941.29430	4485.32357	27.87	<.0001
SEASON	3	1237.26459	412.42153	2.56	0.0536
LOCATION*SEASON	12	20201.58620	1683.46552	10.46	<.0001

The SAS System

The GLM Procedure

Class Level Information

Class	Levels	Values
LOC_SEA	20	C FALL C SP C SU C W F FALL F SP F SU F W STON FALL STON SP STON SU STON W STUT FALL STUT SP STUT SU STUT W T FALL T SP T SU T W

Number of Observations Read	900
Number of Observations Used	900

The SAS System

The GLM Procedure

Dependent Variable: Seasonal Precipitation

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	19	39380.1451	2072.6392	12.88	<.0001
Error	880	141620.4212	160.9323		
Corrected Total	899	181000.5663			

R-Square	Coeff Var	Root MSE	P_E Mean
0.217569	37.34219	12.68591	33.97206

Source	DF	Type I SS	Mean Square	F Value	Pr > F
LOC_SEA	19	39380.14508	2072.63921	12.88	<.0001

Source	DF	Type III SS	Mean Square	F Value	Pr > F
LOC_SEA	19	39380.14508	2072.63921	12.88	<.0001

```
                              The SAS System

                             The GLM Procedure
                 t Tests (LSD) for Seasonal Precipitation

      NOTE: This test controls the Type I comparisonwise error rate, not the
                         experimentwise error rate.

                  Alpha                               0.05
                  Critical Value of t              1.96266
                  Least Significant Difference       5.249

         Means with the same letter are not significantly different.

              t Grouping                Mean      N    LOC_SEA*)
                        A             53.462     45    F SU
                        B             41.940     45    F W
                        B
              C         B             40.129     45    F SP
              C         B
              C         B    D        38.625     45    STON W
              C         B    D
              C    E    B    D        38.565     45    C W
              C    E         D
              C    E    F    D        35.890     45    STUT SP
              C    E    F    D
         G    C    E    F    D        35.699     45    STON SP
         G         E    F    D
         G    H    E    F    D        34.066     45    T SU
         G    H    E    F    D
         G    H    E    F    D        33.699     45    STON FALL
         G    H    E    F    D
         G    H    E    F    D        33.426     45    F FALL
         G    H    E    F
         G    H    E    F            33.366     45    C SP
         G    H         F
         G    H         F             32.988     45    C SU
         G    H         F
         G    H         F             32.435     45    STUT FALL
         G    H         F
         G    H         F             31.599     45    C FALL
         G    H         F
         G    H         F             31.283     45    STUT W
         G    H
         G    H                       30.598     45    T SP
              H         I             29.453     45    T FALL
                        I
              J         I             25.116     45    STON SU
              J                       23.628     45    T W
              J                       23.471     45    STUT SU
```

*) F: Fairhope, C: Clemson, STON: Stoneville, STUT: Stuttgart, T: Thomsons, SP: Spring, SU: Summer, FALL: Fall, W: Winter.

APPENDIX D

SAS OUTPUTS OF TWO-WAY CLASSIFICATION ANALYSIS OF VARIANCE

AND LSD MULTIPLE COMPARISONS FOR SEASONAL P-0.8E

The SAS System

The MEANS Procedure

Analysis Variable: Seasonal P-0.8E

LOCATION	SEASON	N Obs	Mean	Variance	Std Dev
Clemson	Fall	45	13.880	132.289	11.502
	Summer	45	-6.339	209.586	14.477
	spring	45	-2.294	175.267	13.239
	winter	45	23.126	104.068	10.201
Fairhope	Fall	45	13.208	233.402	15.277
	Summer	45	16.343	269.501	16.416
	spring	45	3.081	366.514	19.145
	winter	45	25.911	216.071	14.699
Stonevil	Fall	45	12.860	164.820	12.838
	Summer	45	-20.258	117.539	10.842
	spring	45	-8.501	195.775	13.992
	winter	45	24.002	146.865	12.119
Stuttgar	Fall	45	11.624	171.993	13.115
	Summer	45	-22.134	105.671	10.280
	spring	45	-5.545	159.977	12.648
	winter	45	15.148	111.676	10.568
Thomsons	Fall	45	3.555	223.783	14.959
	Summer	45	-11.085	333.960	18.275
	spring	45	-10.297	231.390	15.212
	winter	45	2.572	134.689	11.606

The SAS System

The GLM Procedure

Class Level Information

Class	Levels	Values
LOCATION	5	Clemson Fairhope Stonevil Stuttgar Thomsons
SEASON	4	Fall Summer spring winter

Number of Observations Read	900
Number of Observations Used	900

The SAS System

The GLM Procedure

Dependent Variable: Seasonal P-0.8E

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	19	177389.3235	9336.2802	49.08	<.0001
Error	880	167412.7475	190.2418		
Corrected Total	899	344802.0710			

R-Square	Coeff Var	Root MSE	P_E Mean
0.514467	349.8169	13.79282	3.942867

Source	DF	Type I SS	Mean Square	F Value	Pr > F
LOCATION	4	36988.5945	9247.1486	48.61	<.0001
SEASON	3	109498.3429	36499.4476	191.86	<.0001
LOCATION*SEASON	12	30902.3862	2575.1988	13.54	<.0001

Source	DF	Type III SS	Mean Square	F Value	Pr > F
LOCATION	4	36988.5945	9247.1486	48.61	<.0001
SEASON	3	109498.3429	36499.4476	191.86	<.0001
LOCATION*SEASON	12	30902.3862	2575.1988	13.54	<.0001

The SAS System

The GLM Procedure

Class Level Information

Class	Levels	Values
LOC_SEA	20	C FALL C SP C SU C W F FALL F SP F SU F W STON FALL STON SP STON SU STON W STUT FALL STUT SP STUT SU STUT W T FALL T SP T SU T W

Number of Observations Read	900
Number of Observations Used	900

The SAS System

The GLM Procedure

Dependent Variable: Seasonal P-0.8E

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	19	177389.3235	9336.2802	49.08	<.0001
Error	880	167412.7475	190.2418		
Corrected Total	899	344802.0710			

R-Square	Coeff Var	Root MSE	P_E Mean
0.514467	349.8169	13.79282	3.942867

Source	DF	Type I SS	Mean Square	F Value	Pr > F
LOC_SEA	19	177389.3235	9336.2802	49.08	<.0001

Source	DF	Type III SS	Mean Square	F Value	Pr > F
LOC_SEA	19	177389.3235	9336.2802	49.08	<.0001

The SAS System

The GLM Procedure

t Tests (LSD) for Seasonal P-0.8E

NOTE: This test controls the Type I comparisonwise error rate, not the experimentwise error rate.

Alpha	0.05
Critical Value of t	1.96266
Least Significant Difference	5.707

Means with the same letter are not significantly different.

t Grouping		Mean	N	LOC_SEA*)
	A	25.911	45	F W
	A	24.002	45	STON W
	A	23.126	45	C W
	B	16.343	45	F SU
	B	15.148	45	STUT W
	B			
	B	13.880	45	C FALL
	B			
	B	13.208	45	F FALL
	B			
	B	12.860	45	STON FALL
	B			
	B	11.624	45	STUT FALL
	C	3.555	45	T FALL
	C			
D	C	3.081	45	F SP
D	C			
D	C	2.572	45	T W
D				
D	E	-2.294	45	C SP
	E			
F	E	-5.545	45	STUT SP
F	E			
F	E	-6.339	45	C SU
F				
F		-8.501	45	STON SP
F				
F		-10.297	45	T SP
F				
F		-11.085	45	T SU
	G	-20.258	45	STON SU
	G			
	G	-22.134	45	STUT SU

*) F: Fairhope, C: Clemson, STON: Stoneville, STUT: Stuttgart, T: Thomsons, SP: Spring, SU: Summer, FALL: Fall, W: Winter.

APPENDIX E

SAS OUTPUT OF THREE-WAY CLASSIFICATION ANALYSIS OF VARIANCE

FOR P-0.8E'S OF 15-YEAR BLOCKS

The SAS System

The MEANS Procedure
Analysis Variable: P-0.8E

LOCATION	SEASON	YEAR_BLK	N Obs	Mean	Variance	Std Dev
Clemson	fall	1	15	13.229	69.362	8.328
		2	15	15.282	158.566	12.592
		3	15	13.130	184.674	13.589
	spring	1	15	2.019	105.280	10.261
		2	15	-4.581	192.587	13.878
		3	15	-4.319	223.042	14.935
	summer	1	15	-6.946	138.716	11.778
		2	15	-6.137	190.313	13.795
		3	15	-5.933	329.055	18.140
	winter	1	15	26.369	104.027	10.199
		2	15	21.308	101.824	10.091
		3	15	21.701	104.233	10.209
Fairhope	fall	1	15	9.447	92.343	9.610
		2	15	10.621	162.635	12.753
		3	15	19.557	413.055	20.324
	spring	1	15	1.720	194.644	13.951
		2	15	5.383	429.831	20.732
		3	15	2.139	518.814	22.777
	summer	1	15	17.107	219.849	14.827
		2	15	11.714	360.836	18.996
		3	15	20.209	226.721	15.057
	winter	1	15	24.239	124.500	11.158
		2	15	27.935	323.461	17.985
		3	15	25.559	223.604	14.953
Stonevil	fall	1	15	11.269	189.457	13.764
		2	15	13.761	160.778	12.680
		3	15	13.549	163.680	12.794
	spring	1	15	-7.827	194.334	13.940
		2	15	-9.500	243.788	15.614
		3	15	-8.175	175.500	13.248
	summer	1	15	-18.557	94.808	9.737
		2	15	-22.261	159.122	12.614
		3	15	-19.957	107.981	10.391
	winter	1	15	24.644	193.166	13.898
		2	15	21.892	146.701	12.112
		3	15	25.470	114.190	10.686
Stuttgar	fall	1	15	7.783	222.174	14.905
		2	15	13.976	126.293	11.238
		3	15	13.113	167.970	12.960
	spring	1	15	-7.260	173.499	13.172
		2	15	-3.360	196.315	14.011
		3	15	-6.016	124.467	11.156
	summer	1	15	-16.531	47.141	6.866
		2	15	-24.716	154.105	12.414
		3	15	-25.156	80.294	8.961
	winter	1	15	14.281	54.997	7.416
		2	15	15.007	192.853	13.887
		3	15	16.156	101.215	10.061

Thomsons	fall	1	15	-1.332	106.663	10.328
		2	15	0.814	293.342	17.127
		3	15	11.184	207.317	14.399
	spring	1	15	-8.132	318.414	17.844
		2	15	-16.124	72.025	8.487
		3	15	-6.635	281.021	16.764
	summer	1	15	-9.439	248.495	15.764
		2	15	-10.623	515.203	22.698
		3	15	-13.193	277.998	16.673
	winter	1	15	-0.613	78.721	8.872
		2	15	0.039	61.280	7.828
		3	15	8.290	230.539	15.183

The SAS System

The GLM Procedure

Class Level Information

Class	Levels	Values
LOCATION	5	Clemson Fairhope Stonevil Stuttgar Thomsons
SEASON	4	fall spring summer winter
YEAR_BLK	3	1 2 3

Number of Observations Read	900
Number of Observations Used	900

The SAS System

The GLM Procedure

Dependent Variable: P-0.8E

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	59	184252.6682	3122.9266	16.34	<.0001
Error	840	160549.4028	191.1302		
Corrected Total	899	344802.0710			

R-Square	Coeff Var	Root MSE	RAINFALL Mean
0.534372	350.6329	13.82499	3.942867

Source	DF	Type I SS	Mean Square	F Value	Pr > F
LOCATION	4	36988.5945	9247.1486	48.38	<.0001
SEASON	3	109498.3429	36499.4476	190.97	<.0001
LOCATION*SEASON	12	30902.3862	2575.1988	13.47	<.0001
YEAR_BLK	2	620.1902	310.0951	1.62	0.1980
LOCATION*YEAR_BLK	8	1519.0556	189.8819	0.99	0.4395
SEASON*YEAR_BLK	6	1524.7858	254.1310	1.33	0.2411
LOCATI*SEASON*YEAR_B	24	3199.3131	133.3047	0.70	0.8575

Source	DF	Type III SS	Mean Square	F Value	Pr > F
LOCATION	4	36988.5945	9247.1486	48.38	<.0001
SEASON	3	109498.3429	36499.4476	190.97	<.0001
LOCATION*SEASON	12	30902.3862	2575.1988	13.47	<.0001
YEAR_BLK	2	620.1902	310.0951	1.62	0.1980
LOCATION*YEAR_BLK	8	1519.0556	189.8819	0.99	0.4395
SEASON*YEAR_BLK	6	1524.7858	254.1310	1.33	0.2411
LOCATI*SEASON*YEAR_B	24	3199.3131	133.3047	0.70	0.8575