

Final Report

To

The South Florida Water Management District

For the project titled:

South Dade Stormwater Treatment Area Evaluation

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INTRODUCTION

Project Summary

This research explored how to best ecologically engineer watersheds to minimize the effects of land use change caused by rapid population growth. Specifically, various urbanization condition and spatial wetland network designs were tested for their relationships to stormwater hydrology and water quality. The area studied was the C-102 and C-103 sub-basins in the rapidly growing region to the south of Miami, Florida.

Since the project watershed was in a heterogeneous landscape with unique hydrologic characteristics, and because spatial evaluations of wetlands are not well supported by existing models, a new type of spatial model was developed. A cell-based hydrologic model, using local environmental and management variables, was developed using Geographic Information Systems software. The spatial model then employed an iterative technique to simulate the temporal dynamics characteristic of the hydrology found in both low-relief landscapes and in wetlands.

Model development involved calibration, validation using alternate urbanization and rainfall conditions, and sensitivity analyses of four important model processes. Calibration was primarily challenged by instabilities related to the spatial lateral runoff algorithms. The calibrated model did not simulate as accurately during extremes of high or low rainfall conditions. Simulation accuracy was not found to be considerably different between the simulation results of two discrete sub-basins. The development steps revealed that the influences of input data, mathematical equations, and parametric settings on simulation results were somewhat scale-dependent.

Model results suggested that urbanization has increased the annual watershed discharge of constituents into Biscayne Bay by approximately one order of magnitude. In addition, simulation results indicated that the conversion of wetlands into farms, residences, and canals has fundamentally changed the primary deposition pathway of the phosphorus cycle from biological to chemical.

Simulation results of land use scenarios where 10% of the urbanized landscape was configured into nine different spatial arrangements of stormwater treatment wetlands suggested that although all wetland networks treated stormwater pollutants, the treatment achieved was variable. Reductions in annual discharge of constituents to the ocean achieved by the treatment wetland networks was between 14.8% and 41.3% for Total Phosphorus, and between 34.2% and 63.9% for Total Suspended Solids. Comparisons of wetland network simulation results suggested that location of wetlands in areas where the landscape naturally converges water and constituents reduced constituent discharges into the ocean more than did wetlands located either randomly or only in close proximity to pollutant sources.

Plan of Study

A temporally-dynamic, spatial model of water quantity and water quality was developed using Geographic Information Systems (GIS) software. Water quality constituents modeled were total phosphorus and total suspended solids. Model inputs included topography, soils, land use, climate, pollutant loads, and water management operations. The model was calibrated to observed data for an average rainfall year, and then validated by simulating higher and lower rainfall years, and using a different study basin. A sensitivity analysis was then conducted to identify key model variables.

The model was then used to simulate various land use scenarios. The scenarios reflected pre-development and projected future land use. Several scenarios were then simulated with different spatial arrangements of wetlands. Distributions were created both at random and by matching stormwater wetland location with landscape attributes theoretically conducive for treatment effectiveness.

Background & Review of Literature

Characterization of Urban Stormwater Flows & Effects

Urbanization modifies stormwater quantity and quality as a direct result of alterations to the physical system governing hydrodynamics, changes in sources and forms of materials on the land surface, and due to secondary effects in downstream ecological systems. Conversion of natural landscapes into urban impervious surfaces, such as parking lots, roofs, and roads, increases the magnitude of surface runoff volume due to decreased infiltration rates (US Soil Conservation Service, 1986, Viessman, Jr. et al., 1989). In addition, the speed at which the runoff moves across the land surface increases as resistance to overland flow drops (Viessman, Jr. et al., 1989, Lin & Perkins, 1989, Brooks et al., 1991). The rapid concentration of water at lower elevations in the watershed increases frequency and severity of downstream flooding, exacerbating regional stormwater problems (US Soil Conservation Service, 1986, Yeh & Labadie, 1997, Behera et al., 1999). The increased rate at which surface water drains off and the reduced infiltration volume combine to reduce average soil water and groundwater tables, which in turn lowers average base flow between storm events (Viessman, Jr. et al., 1989, Brooks et al., 1991). Figure 1 indicates simplified hydrologic response and groundwater

a)



b)



Figure 1. Hydrologic response to storm events and groundwater table differences between: a) undeveloped, and b) urbanized landscape. Arrows indicate direction of water flows, and thickness of arrows indicates relative magnitude. Groundwater tables are indicated by dashed lines.

level differences between undeveloped and urbanized landscapes. The increased peak flows and reduced base flows translate into flashier hydropattern fluctuations for downstream areas (Viessman, Jr. et al., 1989, Mitsch & Gosselink, 1993). Secondary effects occur as vegetation is cleared and as remaining vegetation responds to the altered water budgets. Vegetative biomass reduction modifies evapotranspiration rates, eliminates interception losses, and reduces infiltration as soil pores are compacted (Viessman, Jr. et al., 1989, Carr & Rushton, 1995).

Sources of Pollutants in Stormwater Runoff

Stormwater runoff has several constituents that impact receiving waterbodies. In fact, non-point source emissions have been identified as the primary source of pollution to most waterbodies (Henigar & Ray, 1992, Kehoe et al., 1994). Increased constituent loads are caused by atmospheric deposition, septic tanks, substrate weathering, and application of chemicals on the landscape. Atmospheric deposition rates increase as combustion in power plants and automobiles increases (Reigner, 1997). Septic tanks tend to contribute phosphorus, nitrogen, BOD, and pathogens into groundwaters that can eventually end up in surface waters (USEPA, 1991, Valiela et al., 1992). Weathering of substrates can contribute constituents through several pathways. Transportation highways generate pollutants as car parts wear, lubricants leak, and as the road surface deteriorates (Adkins, 1995). Roofs of buildings weather, which causes increased metal loads (Rushton, 1997). A significant amount of stormwater pollution, especially suspended sediment, is due to erosion caused by construction activities (Wooten et al., 1992, USEPA, 1992). Construction activities, which often involve clearing vegetation,

altering topography, and stockpiling soil, frequently lead to nearly 100-fold increases in suspended solids levels in runoff (Brooks et al., 1991, USEPA, 1992, Henigar & Ray, 1992). Increased surface flow rates, coupled with removal of the vegetative cover, may lead to increased erosion of urban soils and scour in waterways such as canals and rivers (Brooks et al., 1991). Since many constituents are attached to the sediment particles, suspended solids levels are closely correlated with levels of several pollutants (Henigar & Ray, 1992, Carr & Rushton, 1995, Wooten et al., 1992). Pollution due to direct applications includes constituent loads originating in runoff from gas stations that includes sediments, diverse hydrocarbons, and metals (USEPA, 1991). Lawn fertilizer is also considered to be a major source of non-point source phosphorus load (USEPA, 1991). Some of the other applied materials include landscaping pesticides and herbicides, animal waste, soaps, oils, cleaning agents, salts, and tar (USEPA, 1991, Alleman, 1995). An abundance of literature exists that has discussed and documented the correlation of increased pollutant loads and watershed urbanization (see for example Schropp et al., 1990, Correll et al., 1992, Valiela et al., 1992, Campbell et al., 1995, Parker, 1998). Pollutant and sediment concentrations are highest in the "first flush" of stormwater, with 80-95% of loading occurring as the first half inch of precipitation falls (Kehoe et al., 1994).

Interactions of Constituents & the Landscape

As constituents travel, the landscape transforms and sequesters them through a variety of mechanisms. The general processes that occur are substrate interactions, biological transformations and uptake, downstream transport, and sedimentation. Soil properties influence constituent dynamics in many different ways. For example, while

apatite rock and sandy soils have been shown to leach nutrients, calcite precipitation and coprecipitation of phosphorus as calcium phosphate occurs in hard waters, such as areas with abundant limestone (Gilliland, 1973, Haan, 1995, House et al., 1998). Soil redox potential can also affect phosphorus dynamics, in particular where reduced soils exist with an abundance of iron, aluminum, or sulfides (Mitsch & Gosselink, 1993, Tam & Wong, 1999). Microbes and plants ingest and utilize many of these compounds and elements as part of their metabolism, transform them, and then either release them directly as by-products, sequester them in their tissues, or drop them during biotic senescence. Since metabolisms are strongly related to hydrologic patterns, nutrient concentrations, and soil redox conditions, all of these forces come together and feedback to regulate biological interactions with constituents (Busnardo et al., 1992). Although topography and drainage control hydrologic constituent transport, key landscape attributes that govern transport mechanisms are substituted from uplands to lowlands (Campbell et al., 1995, Flaig & Reddy, 1995). In upland areas, transport tends to be governed by relief and depths to the spodic soil horizon, whereas sediment chemistry, biological uptake, and sediment accumulation and erosion processes control transport in wetlands and streams (Flaig & Reddy, 1995). All of these transport processes are occurring in heterogeneous spatial and temporal dimensions. The processes not only change the magnitudes of pollutants as they travel downstream, but they undergo transformations in oxidation state, solubility, and organic or inorganic state (Correll et al., 1992).

Effects of Stormwater Runoff on Ecosystems

The consequences of non-point source pollution, now considered the primary cause of impairment in US surface waters, are documented extensively throughout literature (Baker, 1992). Lakes and estuaries undergo significant physical, chemical, and ecological change as a result of poorer water quality and altered hydrology. Aquatic systems frequently are evolved to a diet of lower levels of nutrients, and one or more of these nutrients will limit productivity (Pennock & Sharp, 1994). Increasing nutrient levels tends to encourage primary production, and phytoplankton are often the quickest to respond (LaPointe & Clark, 1992). Phytoplankton blooms, along with increases in suspended solids and dissolved organic matter, reduce water clarity and change aquatic oxygen cycles (Lathrop & Lillesand, 1989, McPherson & Miller, 1994). In extreme or unfortunate cases, fish kills, red tides, and increased disease rates among wading birds will occur (Okaichi, 1987, Lowe, 1991, Spalding et al., 1994). More commonly, reduction of light has been demonstrated to cause species shifts and reduce biomass and productivity of benthic flora, which in turn adversely impacts the associated benthic animals (McPherson & Miller, 1994, Czerny & Dunton, 1995, Buzzelli et al., 1998). Another potential stress to benthic communities is from altered salinity and temporal salinity flux where streams and canals discharge into estuaries, especially for marine communities not adapted to small freshwater increases (Montague & Ley, 1993). Urban stormwater can impact coastal wetlands due to changes in sediment quality and quantity, which are important factors for mangrove seedling productivity (Duarte et al., 1998). Disrupted sediment deliveries have resulted in extensive coastal wetland fragmentation in the Mississippi River delta, as wetland marshes near canals and spoil banks change into small ponds, which then become larger ponds (Turner & Rao, 1990). Additionally,

elevated concentrations of heavy metals and organic compounds pose a health hazard to human drinking water and commercial seafood consumption (Valiela et al., 1992, Alleman, 1995).

Urban Stormwater Management & the Role of Wetlands

Urban stormwater is not well suited for intensive treatment facilities, such as those for drinking water and wastewater, due to the stochastic nature of stormwater flows, which makes sizing and operation infeasible. Traditional stormwater management was engineered to remove rainfall as quickly as possible through constructed conveyance systems of ditches, pipes, gates, and pumps. Often, these systems shifted the problem from upstream to downstream flooding, altered hydrologic flows, and severely decreased water quality. To control stormwater problems, several best management practices (BMPs) have been developed. Grassy swales (essentially vegetated street gutters) and stormwater ponds that either retain or detain water serve to provide flood protection (Whalen & Cullum, 1988, Rushton et al., 1997). Other BMPs, such as sediment barricades, specially designed box culverts, and permeable asphalt, offer a more high-tech approach when used in addition to other BMPs or existing sewer networks (Wooten et al., 1992, England, 1997, Rushton, 1997). BMPs, however, have persistent maintenance expenses associated with their use (USEPA, 1992, Kehoe et al., 1994, England, 1997). Another option for solving water and sediment quality problems is through source abatement, although applicability and cost-effectiveness are variable and limited (Elliott, 1998). Source abatement includes street sweeping, to reduce the supply of materials waiting to be washed into waterbodies, stringent controls on automobile

emissions, to decrease atmospheric inputs, and elimination of certain products from manufacturing processes (Carr & Rushton, 1995, Elliott, 1998).

Characteristics of Stormwater Wetland Functions

Wetlands are increasingly constructed as part of stormwater management, although most sites are still young (Kehoe et al., 1994, Carr & Rushton, 1995, Singleton & Williams, 1995). Stormwater wetlands function hydrologically much as detention systems do, where flood protection is provided by storing water for a long enough duration to reduce downstream peakflows (Winston, 1996). However, the shallower, intermediate hydroperiod zones of wetlands also create habitat for vascular plants, which provide the additional function of enhanced pollution treatment (Reinelt & Horner, 1995). Pollution abatement mechanisms include the reduction in velocity as a discharge enters a wetland, which causes sediments to settle, a variety of anaerobic and aerobic processes that remove certain chemicals from water, mineral uptake by vegetation and burial in sediment, and microbial activity in the sediment (Carr & Rushton, 1995). Stormwater wetland constituent removal rates have been calculated to remove between 76 to 99 percent of total suspended solids loading, and from 52 to 99 percent of total phosphorus loading (Hey et al., 1994). Nutrient removal has been shown to be highly dependent on vegetation (Carr & Rushton, 1995, Kent et al., 1997). For example, a study comparing plant mesocosms and mesocosms with only water column plankton found mean phosphorus removal improved from 21 to 62 percent with plants (Johengen & LaRock, 1993). The presence of plants increases resistance to flow, slowing down the water, and fostering sedimentation, which is the dominant pathway for pollutant removal (Brueske & Barrett, 1994, Jadhav & Buchberger, 1995, Reinelt & Horner, 1995, Kent et

al., 1997). Direct metabolic plant uptake treats about 8% of annual phosphorus loading in stormwater wetlands, and wetland release of phosphorus can occur during winter months (Hey et al., 1994, Reinelt & Horner, 1995). Another possible sink for phosphorus in stormwater wetlands may result from calcium carbonate chemistry, as observed recently in a wastewater wetland with a limestone substrate (Nelson, 1998).

Spatial Interactions of Wetland Network Patterns, Hydrology & Water Quality

Temporal hydrologic patterns vary in different parts of a watershed, including the transitions from terrestrial to aquatic, upstream to downstream, and urban to undeveloped areas. In the upper reaches of a watershed, flows tend to be generated by precipitation inputs, and runoff and base flows become the dominant sources in the lower parts of the watershed (Brinson, 1993, Mitsch & Gosselink, 1993). As a result, upper watershed hydropatterns fluctuate more rapidly than they do in the lower watershed, as shown in Figure 2. In addition, hydrodynamic heterogeneity results from spatially-variable distributions of the soils, urban lands, and in large watersheds, climate features.

Spatial Water Quality Patterns in Watersheds

Since hydrology regulates nutrient cycling and varies in space, water quality exhibits spatial heterogeneity as well. The length and landscape characteristics of flow paths between source areas, wetlands, and watershed discharges are spatial properties that influence the nutrient retention of the basin wetland network. The distance from a sediment source is negatively correlated with sediment load, which also affects loading rates for constituents that occur in particulate form, such as phosphorus and metals (Brooks et al., 1991, Brueske & Barrett, 1994). For regions with considerable groundwater flow, location of and distance to sources is particularly important due to the

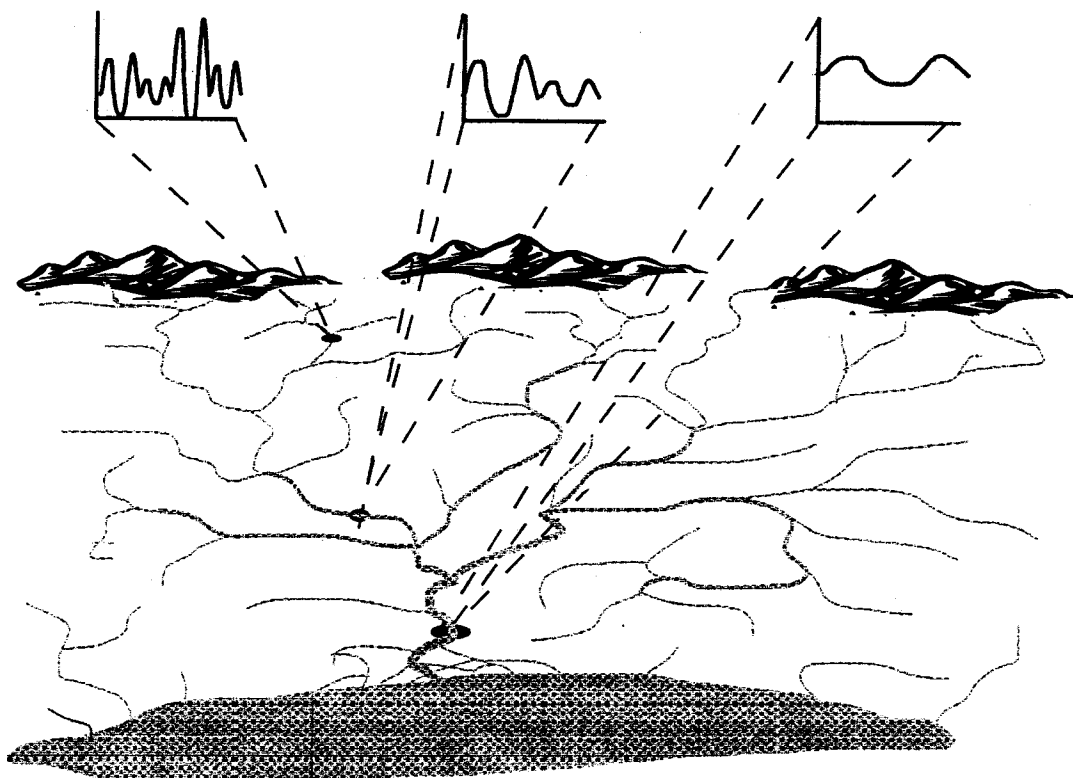


Figure 2. The variation in hydropattern throughout a watershed. Upper watershed hydrology oscillates quickly between high and low flows, whereas the baseflow-dominated systems downstream have smoother hydropatterns.

longer travel times involved (Valiela et al., 1992). As flows and constituents travel the distances across a landscape, they interact with heterogeneous watershed attributes. Landscape characteristics that can affect water quality along flow paths include land use/land cover, soils, and topography. The spatial configuration of land cover and land use has been shown to affect loads discharged from watersheds, modify aquatic N:P ratios, and shift species of nitrogen and phosphorus compounds (Correll et al., 1992, Gumbricht et al., 1996). The location of soils with high hydraulic conductivity affected hydrology and plant communities in a small wetland designed to control highway runoff (Winston, 1996).

Effects of Wetland Location

Wetland systems self-design around the magnitude and timing of the distribution of water they receive (Mitsch & Gosselink, 1993, Bedford, 1996, Kadlec & Knight, 1996). Since hydrology, including parameters such as flow rate, storage volume, and hydropattern, varies depending on watershed location, the site a wetland occupies in a landscape will influence wetland type and functions (Baker, 1992, Brinson, 1993). Furthermore, the location of wetlands modifies the storage characteristics of the watershed, which in turn affects downstream hydrology. Consequently, the spatial design of a wetland network should be considered when planning watershed-scale projects. For example, a National Wetland Inventory estimate of hydroperiod throughout the landscape was the primary criteria used to design a feasible wetland buffer strip in Dade county, Florida (Gong, 1994).

The wetland distribution may affect water quality and constituent loads discharged from a watershed into a receiving waterbody. To test this theory, a recent

study was conducted in which wetland landscape position was compared to total phosphorus retention and sediment accumulation rates, and no significant correlation was found (Greiner & Hershner, 1998). However, all sites were located in the coastal zone of a North Carolina watershed, which the researchers suggested may have been too similar at a landscape scale to detect significant differences. Models created to study the effect of wetland distributions, and some field data, suggest that upper watershed wetlands uptake a higher percentage of input constituent loads than do wetlands located in the lower parts of a watershed (Mitsch, 1992, Brinson, 1993, Greiner & Hershner, 1998, Tilley & Brown, 1998). On the other hand, a different model used to compare urban stormwater wetland landscape configurations showed that a large wetland at the base of the watershed treated nitrogen and phosphorus required the least land area (Tillis, 1996).

Importance of Wetland Size

The size of individual wetlands is cited as an important consideration for watershed-scale water quality management. A controversy exists in the ecological engineering literature about what the real effects of wetland size are, in part because of a lack of actual field studies specific to the question. Models have suggested that large wetlands function more effectively in buffering pulses of water quantity during floods (Ogawa & Male, 1983, Tilley & Brown, 1998). However, it has been argued that building large numbers of smaller wetlands is a better and more economical strategy for wetlands to survive extreme events (Loucks, 1989). Small wetlands may be more effective than large downstream wetlands at trapping nutrients and sediments, based on interpretations of data used to study other phenomenon (Van der Valk & Jolly, 1992, Brinson, 1993, Raisin et al., 1997, Greiner & Hershner, 1998). Alternatively, one model

predicts that a large wetland offers the most effective treatment of urban stormwater runoff on a land area basis (Tillis, 1996).

Wetlands self-organize to a size that is related to the flow of available hydrologic energy, which in turn is dependent upon upstream area and slope. Since the energy varies throughout a watershed, a wetland size class distribution develops, and the distribution can be used to characterize wetland network structure. In natural landscapes, the wetland network is hierarchical, with abundant small wetlands located in the upper watershed converging water into wetlands that increase in size and decrease in number toward the basin terminus. Accordingly, the size class distribution and spatial configuration are closely related wetland network properties. One study concluded that the different components of the hierarchy perform separate stormwater management functions: small neighborhood wetlands are best suited for treating phosphorus, medium sub-basin wetlands trap suspended sediment, and large downstream wetlands attenuate floods (Tilley & Brown, 1998).

Hydrologic Modeling of Non-point Source Flow

Traditional Models

Models of non-point source flows are tools that mathematically simulate the interaction of driving forces, landscape attributes, and management alternatives to predict water quantity and quality. Different applications of hydrologic models include, but are certainly not limited to, reservoir management, sewer design, soil loss estimation, stormwater modeling, climate scenario simulation, studying pollutant transport, effects of wells on water tables, and flood-impact assessment. Since there are many stakeholders involved in understanding hydrology for diverse situations, an abundant inventory of

hydrologic models exist that range from the simple to the complicated. Simple models are easy to develop, understand, and apply to rapidly generate results. Where detailed numerical estimates are demanded, mathematical complexity is employed in the hopes of predicting the finer resolution dynamics of reality. As a result, models range in central assumptions, breadth of utility, and in their extent and resolution in space-time, all of which determine their validity under the circumstances. Hydrologic models partition processes into separately handled components where the algorithms are grouped into surface water, groundwater, or water quality computational blocks. Traditional models are generally tailored to simulate one type of hydrologic or ecological system, such as a stream, and are linked to models that handle other systems, for example, reservoirs, aquifers, or estuarine fish populations, which results in a unit process representation of landscapes.

Surface Water Models

Surface water quantity and quality has attracted most of the attention of stormwater modelers. Simple surface water models simulate flows and loads at an endpoint of a discrete hydrologic unit (such as a watershed), where the hydrologic unit is a homogenous system. Although the mathematics vary, the more complex models are generally organized to function as link-node networks, where each node functions as the endpoint described above for the simpler models, and spatial connectivity is achieved through use of links. The additional complexity allows the hydrologic unit to be subdivided, generating spatial heterogeneity. Calculations are performed for each node using local input parameters, such as rainfall, impervious surface, and slope, and then a routing procedure transmits overland flow between appropriate nodes, in the analogous

manner that a sewer network distributes flow from sewer junctions through pipes to the next junction. Some surface models divide the analysis extent into a grid of cells, usually triangles, squares, or hexagons. Data is collected for each cell, and this is used as input for the calculations. The cells are linked together in the same manner as the link-node network based above, where flow routing is performed by defining flow directions (usually based on elevations) for each cell. Models that simulate surface water quality utilize algorithms basing constituent loads or concentrations on source quantities and runoff volumes, and then simulating transport dynamics and landscape uptake. Source quantity is determined using several methods, including direct application rates (such as fertilizer use), empirical constituent buildup-washoff dynamics, the Universal Soil Loss Equation, or Event-Mean Concentrations (EMCs), which are empirically determined runoff concentrations from discrete land use types for average storms (Donigan & Huber, 1991, Harper, 1994). Hydrographs and pollutographs at nodes and outfalls are generated. Input parameters can be adjusted to determine storm response and pollutant load under various climate, management, and design alternatives.

Donigan and Huber (1991) have written a thorough comparative study of the most widely used surface water models, notably SWMM, HSPF, CREAMS, HEC, STORM, and AGNPS, for simulating non-point source flows. SWMM and the HEC models are primarily used to model urban hydrology, while the others have found more use for agricultural and watershed-wide studies (Donigan & Huber, 1991). The US Army Corps of Engineers HEC models are simple models utilized for planning purposes where effects of numerous design alternatives on reservoir storage and treatment need to be simulated (Morris, 1996). SWMM, a complex model with the capability to simulate hydrology in

detailed urban stormwater systems, is perhaps the most popular stormwater model in use today, and is used to design sewers (Huber & Dickinson, 1988). Calibration of SWMM output has been found to be particularly sensitive to adjusting parameters dependent on percentage impervious surface (Warwick & Tadepalli, 1991). In one study, calibration was achieved by reducing percentage impervious surface from the actual amount of 30% to 25% (Warwick & Tadepalli, 1991). It follows that SWMM may not be an appropriate model for studying changes in watershed hydrology as a result of urbanization. Models such as HSPF and AGNPS are preferable for conducting non-urbanized watershed studies, since they represent rural hydrology more accurately and their water quality algorithms are more appropriate (Summer et al., 1990, Donigan & Huber, 1991). However, HSPF is data and labor intensive, especially for planning-level land use change studies, and this has resulted in models, such as WEPP and CREAMS, that can accurately simulate individual constituents with reduced input requirements (Donigan & Huber, 1991, Wilcox et al., 1992, Negahban et al., 1995). HSPF, with mathematical complexity that attempts to simulate many small-scale processes, may yield invalid results, since it has been reported to have different model parameter sets (calibration values) for different rainfall years (Jacomino & Fields, 1997).

Groundwater Models

Traditional groundwater models function somewhat differently, using finite-element or finite-difference mathematical methods, instead of the link-node concept, to simulate continuous values of water tables (Wang & Anderson, 1982). Groundwater models typically require extensive data about several soil properties for many different depths or discrete soil horizons. The two equations usually used to simulate groundwater

movement are Darcy's Law, which incorporates data about hydraulic head gradients, cross sectional area, and soil hydraulic conductivity or transmissivity (depending on the form of the equation used), and the Continuity Equation, which regulates flows according to the conservation of mass principle (Wang & Anderson, 1982, Viessman et al., 1989).

Coupled Models

For simulation of hydrology in areas where the phreatic aquifer freely connects surface and shallow ground water, such as in low-relief or karst landscapes, linking a surface model with a groundwater model is desirable. A US Geological Survey (USGS) groundwater model, MODFLOW, was linked to a surface water model to simulate the complex hydrologic system in Dade county, Florida, and results were more accurate by comparison with results from a canal hydrology model (Swain et al., 1996). The space-time dynamics for the coupled relationship of surface and shallow ground waters are characterized by the Variable Source Area (VSA) concept, illustrated as Figure 3, which describes changes in a landscape unit's surface runoff patterns in response to changing water tables. Variable source areas are lands that contribute to infiltration flows when dry, and as water tables rise and soils become saturated, infiltration decreases, surface runoff increases, and they eventually become surface storage hydrologic units (Brooks et al., 1991). By definition, wetlands, which are lands of intermediate hydroperiods, are VSAs, and their relationship in the surface water runoff network is inconsistent. Traditional non-point source models do not have VSA routines and have a flow routing network fixed at the beginning of the simulation period, and as a result cannot simulate spatially dynamic runoff zones or represent wetlands correctly (Konyha, 1995, Endreny & Wood, 1999).

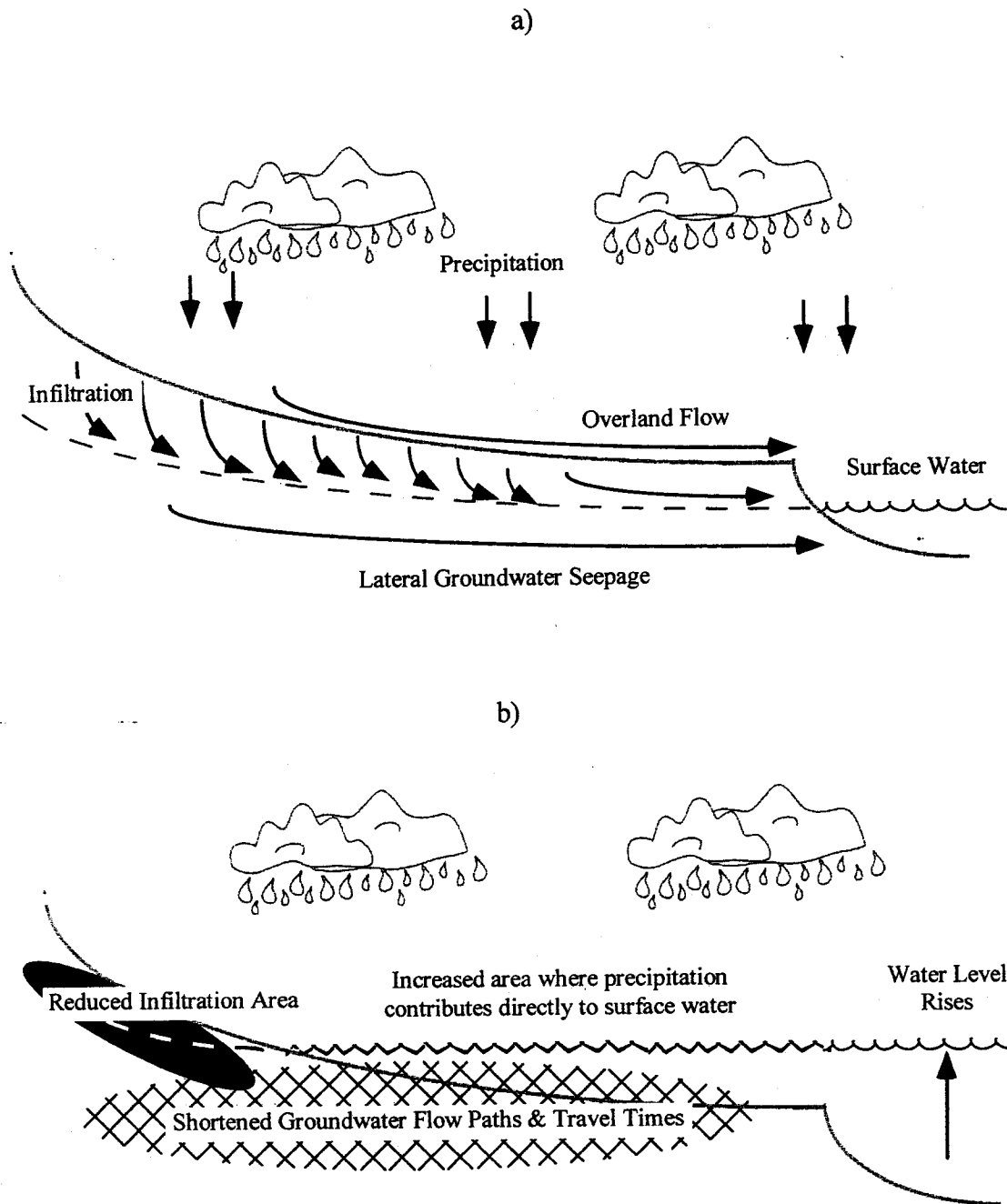


Figure 3. The Variable Source Area concept, where hydrology changes between; a) at the beginning of rainfall, and b) sometime after a), as rainfall continues. Groundwater tables (indicated by the dashed lines) increase, resulting in the changes named in b).

The Use of GIS in Hydrologic Modeling

Real watersheds possess complex patterns in the spatial arrangement of soils, land cover, and topography, and this heterogeneity determines hydrologic distributions.

Consequently, increased resolution of spatial attribute data is considered to improve hydrologic model performance, and to be a primary challenge to modeling efforts (Jakeman, 1999). In addition to improving model accuracy, finer spatial resolution can generate a greater quantity of simulated hydrologic information about specific locations in the study area. Two methods are generally used to create a more realistic representation of the hydrologic landscape, although neither is a truly spatially-distributed modeling method: a) the hydrologic system is partitioned into subsystems, to which lumped-parameter models are then applied; and b) a GIS is interfaced to an existing model (Olivera & Maidment, 1999). GIS technology has been applied to minimize the difficulty of storing and utilizing the large volume of data associated with increasing scale and accuracy, which reduces data pre-processing time, and for simulation output display and analysis in conjunction with other spatial data.

Interfaced Modeling Methods

The primary GIS function is to feed information from databases to surface hydrologic models by compiling and processing spatial data into input data formats. The spatial data layers, known as coverages, can also be referenced to an attribute table (sometimes called a "lookup table") to generate spatial attribute data that can be used as additional input. Examples of GIS-generated input parameters include basin areas, watershed boundaries, water features, flow-routing networks, spatial rainfall pattern,

slope, soil properties, runoff curve numbers, and event-mean concentrations (Warwick & Haness, 1994, He et al., 1994). Currently, the best-known example of a GIS interfaced to a surface hydrologic model is EPA's BASINS package, which links HSPF to ESRI's GIS software. BASINS significantly reduces both pre-processing time and information gaps that go into the data-hungry HSPF simulations (Jacomino & Fields, 1997). Groundwater models that utilize numerical finite-element or finite-difference methods are potential candidates for interfacing with GIS, since GIS layers can be stored as grid data, which is exactly analogous to a finite-difference grid. As a result, GIS has been interfaced to models such as MODFLOW, where spatial attribute data, such as hydraulic conductivity and aquifer thickness, satisfy the variables in the mathematical equations (Watkins et al., 1996). Other examples of interfaced GIS models include modeling of pollutants in the vadose zone (Corwin & Wagenet, 1996), interfacing with the WEPP model (Savabi et al., 1995), and a stormwater runoff and pollutant model adapted from SWMM for agricultural catchments (Xue & Zhang, 1997). Since GIS-traditional model interfaces are generally customized for an individual model or for specific model options, they are more inflexible and require additional licenses, dollars, and user familiarity with both the hydrologic model program and the GIS system.

Independent GIS Models

For hydrologic modeling independent of external models, GIS models employ overlay and spatial analysis functions. Overlay operations, as originally conceived by Ian McHarg, involve superimposing two or more spatial data layers that have been geo-referenced so that they coincide in space (McHarg, 1969). Qualitative and/or quantitative techniques can then be applied to the layers of the overlay, such as a when a descriptive

classification is conducted for particular combinations of multi-layer value sets. For example, one study modeled economic and structural damage to a Canadian town, for various flood sizes, by applying qualitative scalars to three variables: digital elevation data, historic flooding data, and land use (Boyle et al., 1998). Mathematical functions can be applied to the layers to generate a quantitative result, such as subtracting rainfall from evapotranspiration to generate an estimate of spatial irrigation water requirements in desert agriculture, as in Figure 4. GIS software is equipped with spatial analysis features including functions that generate statistics, calculate distances, derive slopes, allocate cells to distant locations, create flow-routing networks, and process flow-routing functions for both neighborhoods and zones defined in numerous ways. These features have allowed GIS to be used to model non-point source nutrient loading in watersheds. For the Lake Mendota watershed around Madison, Wisconsin, land use, topography, and precipitation data were used to model annual overland flow and phosphorus load, where phosphorus decay occurred as a function of flow distance (Soranno et al., 1996). A similar model used in the Florida panhandle added soils data to the input parameters and modeled phosphorus decay as an exponential function of flow length (Parker, 1998). Nitrogen loading in an agricultural catchment draining to a Danish fjord was modeled by overlaying terrain elevation, groundwater head, soils, and land-use derived nitrogen source data, and then applying a nitrate decay process that was first-order with respect to distance from sources (Skop & Sorensen, 1998).

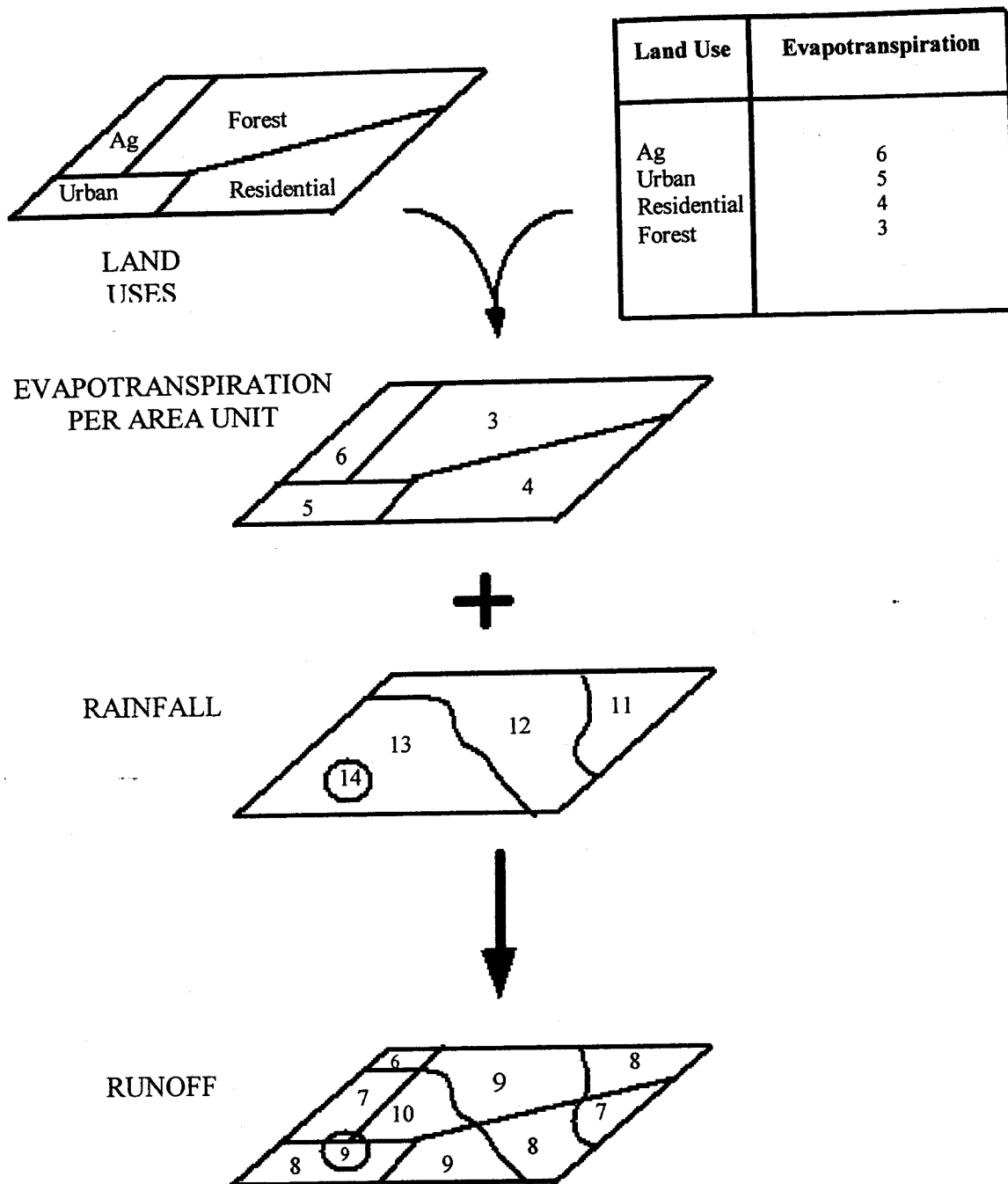


Figure 4. GIS overlay operations. The example shown is for determining runoff based on rainfall and. From top to bottom, a spatial coverage of land uses is referenced to a table of evapotranspiration data to generate a coverage of land use-dependent evapotranspiration on an area basis. This coverage is added to a layer of spatial rainfall values, generating the result of runoff per area.

Scale

A central issue for hydrologic models utilizing GIS, whether interfaced or not, concerns selection of spatial scale. Although increasing resolution can improve simulation accuracy, model resolution becomes limited by the availability of finer-scale data and computer processing power. The consequences of coarse spatial resolution are that the study area can be misrepresented by the raw data, and that data derived from this raw data will also be inaccurate. For example, a grid cell size of 700 meters or greater caused entire landscape features, namely tree island and brush mixture communities, to disappear from maps depicting the Florida Everglades, and the area-perimeter relationship changed rapidly as cell size was increased beyond 100 meters (Obeysekera & Rutchey, 1997). Stream characteristics such as length, sinuosity, and magnitude change depending on map scale (Tighe, 1988). Simulation results are significantly influenced by the accuracy and cell size of the Digital Elevation Model (DEM), which is the source of the flow-routing network. Since many of the DEM values are interpolated rather than actual, DEM accuracy is dependent upon the number and precision of interpolation points, and upon the interpolation method. Larger DEM cell size results in increased errors for variables that describe the flow-routing network, including watershed delineations, flow directions, slopes, and sinks, and single-point errors affect network variable values in all neighborhood cells (Garbrecht, 1994, Bennett & Armstrong, 1996, Wang & Yin, 1998). Flow directions, which are determined by comparing elevation values in adjacent cells, are particularly sensitive to DEM errors in low-relief landscapes, because even small errors can result in simulated flows traveling in opposite directions to those seen in reality (Verigin, 1997, Wang & Yin, 1998). DEM errors also result in false

sinks that short-circuit the flow-routing network and prevent flows from traveling downstream, although distinguishing between false and real sinks in karst, low-relief, or glacial areas is difficult and hydrologically important (Mark, 1988, Bennett & Armstrong, 1996). Figure 5 illustrates the effects of both a single-point DEM error on neighborhood cells and of false sinks. The accuracy of DEM-derived slope values has been found to decrease with increasing cell size, and in variable (i.e. hilly) terrain is actually more sensitive to cell size than to the algorithm used in it's calculation (Hodgson, 1998, Wang & Yin, 1998).

Dynamic Spatial Modeling

As mentioned above, since current models have fixed flow-direction paths, and cannot simulate variable source areas, they do not represent wetlands or sinks in the landscape accurately. Viewed from the time scale of land use change, urbanizing lands (which affect the magnitude and direction of non-point source flow) also behave similarly to variable source areas with respect to the changes in infiltration and runoff patterns, especially in low-relief landscapes. Examples of the relationship between VSA and dynamic flow direction are shown in Figure 6. The existing models, then, are inappropriate for urbanization studies in flat areas, since an important change from the existing conditions is unable to be predicted by the models and is therefore outside the valid problem domain (Van Straten, 1998). To capture landscape heterogeneity in time and space, models simulating hydrologic effects of land use change and wetlands in watersheds benefit from a dynamic spatial approach. Unfortunately, until very recently technology has not been favorable for the development of such models, leading one researcher to declare as recently as 1996, that:

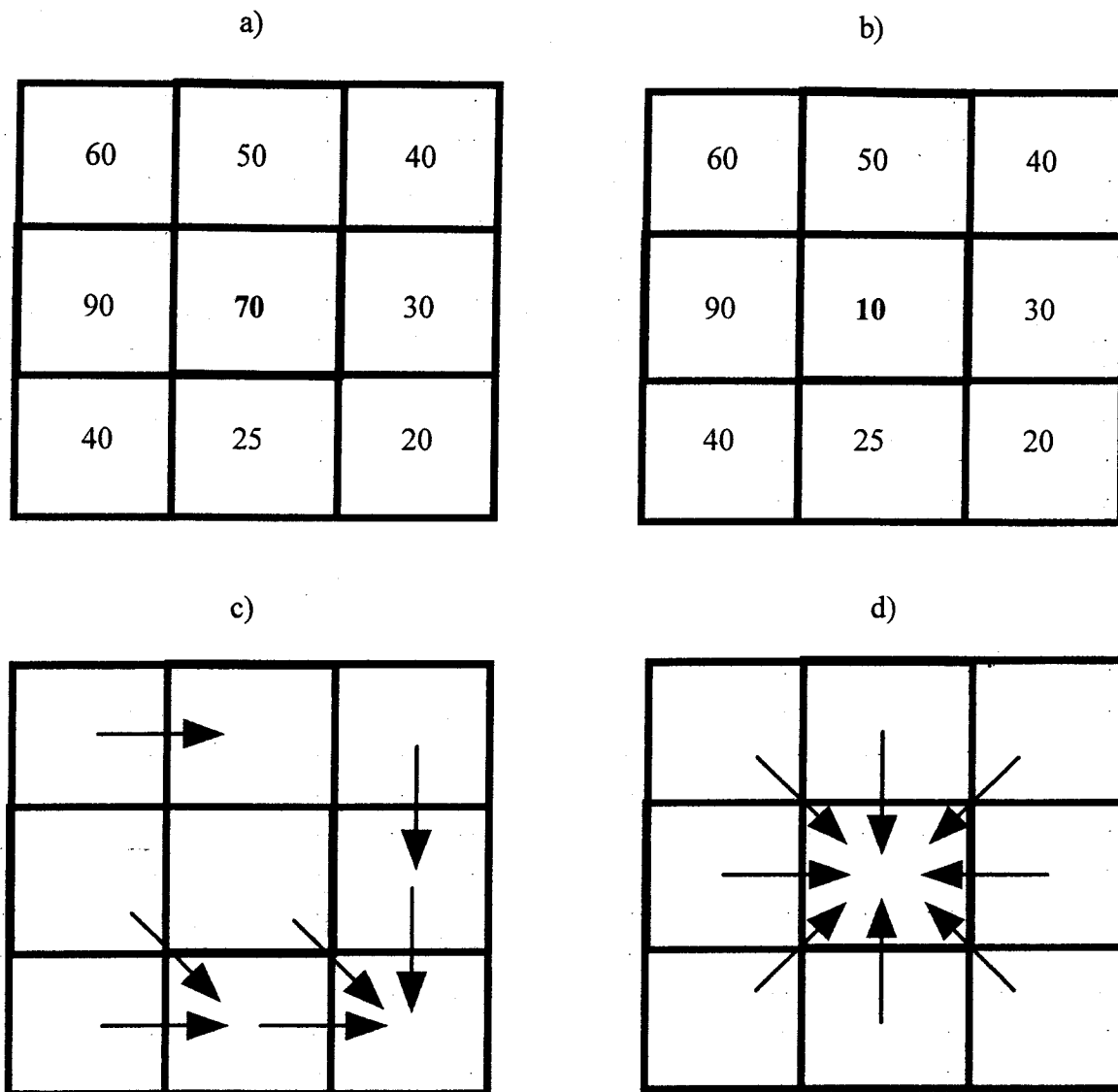


Figure 5. Effect of single point DEM error on flow directions, and an example of a false sink. When the elevation in the center cell of a) is changed from 70 to 10, as in b), the flow-routing network changes from c) to d). The simulated single point error results in changes to the flow direction path derived for all neighborhood cells. When a cell elevation is inaccurately lower than it's neighbors, a false sink is created, and the entire neighborhood is hydrologically misrepresented.

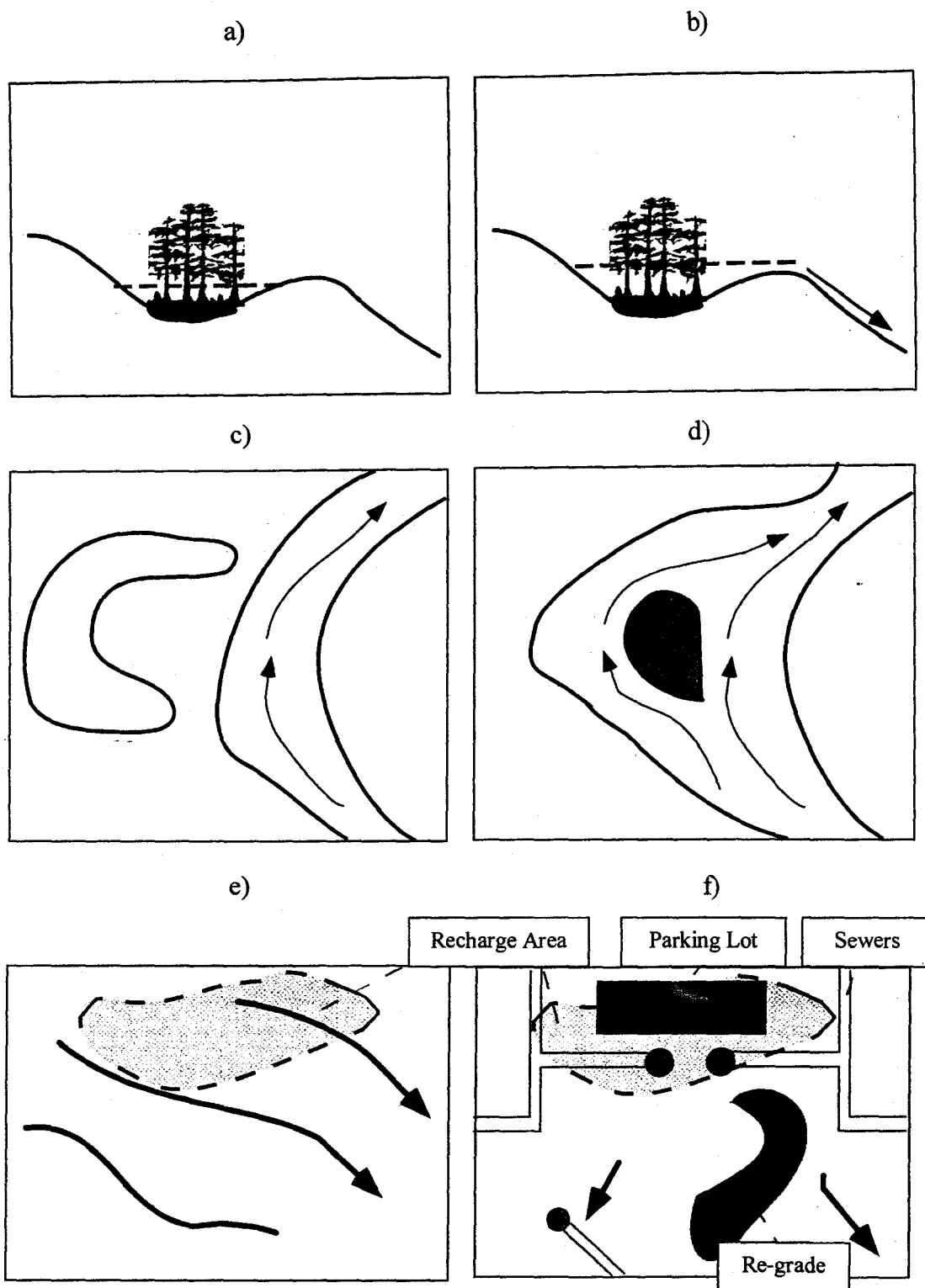


Figure 6. Dynamic relationships of Variable Source Areas to flow directions: 1. a) water flowing to a wetland & b) the full wetland contributing to downstream flow; 2. c) a river during low flow & d) a river at flood stage flowing through an oxbow; 3. e) an undeveloped landscape becomes f) urbanized, constricting flow through sewers. The paved recharge area also decreases local infiltration.

“no GIS has the data representation flexibility for space and time, together with the algorithmic capability, to be able to build process-based models internally; consequently, environmental models and GIS must be coupled (Corwin & Wagenet, 1996, p. 405).”

As of late, GIS software has become available that is equipped with object-oriented programming features, which allows spatial data to be manipulated dynamically. The object-oriented programming technique is to build processes from modular blocks of code that first define the system components from data, and then characterize component behavior and relationships between components based on algorithms. The modular code blocks are executed in sequence (where the sequence itself is a process defined by another code block), and since the code blocks are re-usable, the sequence can be repeated as many times as is desirable. In effect, then, the object-oriented code blocks are analogous to hydrologic processes such as rainfall, runoff, and evapotranspiration, each executed sequence represents all modeled processes in a time-step, and the iterative technique creates time. Although no modeling was done, general object-oriented analysis was used to develop design specifications for hydrologic software objects applicable in modeling the Kissimmee-Okeechobee-Everglades system (Tisdale, 1996). The inclusion of object-oriented programming in GIS may spawn an era in which GIS does not interface with other models to generate the temporal dimension. Object-oriented dynamic spatial models have been created to understand relationships of urban structure to sub-population activities (Janelle et al., 1998), and to locate shopping centers based on simulated temporal changes in the number and behavior of shoppers (Makin et al., 1997). Perhaps what is currently the most comprehensive grid-based dynamic spatial model is

the General Ecosystem Model (GEM), which is a process-based model that simulates ecosystem dynamics based on levels of water, nutrients, and environmental inputs (Fitz et al., 1996). To improve simulation results in a watershed with spatially-variable rainfall, the unit hydrograph method was modified for a heterogeneous grid cell environment to generate the time-history of runoff at the watershed outlet (Olivera & Maidment, 1999). While no examples exist of models that utilize dynamic flow direction grids, it has been suggested as a method for solving flow-routing problems caused by landscape sinks that fill when water levels rise (Bennett & Armstrong, 1996).

Description & History of Study Site

The region studied is composed of three sub-basins located in southeastern Miami-Dade county, Florida, which are drained by three canals, namely C-102, Military, and C-103 canals (see Figure 7). The sub-basins are part of the extensive hydrologic system including the Everglades. The C-102 sub-basin is 8,630 hectares, the C-103 sub-basin is 12,763 hectares, and the Homestead Air Force Base sub-basin is 1,021 hectares. The present-day climate of south Dade is sub-tropical, with mean July temperatures of 28°C and mean January temperatures of 19°C (Tilley & Brown, 1998). Annual rainfall averages 1473 mm, although wet and dry cycles lasting several years are common (Williams & Singleton, 1995). Rainfall is seasonal, as two-thirds of annual rainfall occurs from June through October (SFWMD, 1993). Wet season rainfall is generated by convective afternoon thunderstorms, and slow-moving cold fronts cause precipitation during winter months. Additional precipitation is provided when tropical cyclones pass through the area, which can occur between June and November. Annual Penman

potential evapotranspiration is 1389 mm, and actual evapotranspiration varies with water availability and land cover (Smajstrla et al., 1987). Monthly rainfall and evapotranspiration trends are shown as Figure 8.

The study area is a low-relief landscape, with elevations generally not exceeding 2 meters above Mean Sea Level (MSL). The highest elevations (about 6 meters above MSL) occur along the sandy Atlantic coastal ridge, which runs through the region from the central northern boundary of the C-102 sub-basin toward the south/southwest, as seen in Figure 9. Low strips of land averaging .8 km in width, known as transverse glades, occasionally cross the sand ridge and connect the lower elevation areas to the east and west of the ridge. Surface soils are composed of several inches to feet of Perrine marl in the transverse glades and along the coast, and have poor to moderate vertical permeability (Swain et al., 1996). Limestone of high hydraulic conductivity occurs at the surface adjacent to both sides of the sand ridge, and is interspersed within the marl near the coast. Sinkholes occur in areas with surficial limestone deposits. The area to the west-northwest of the sand ridge is known as the Rocky Glades, and was historically part of the Everglades system. Various types of limestone are layered beneath the surface soils, and these layers collectively comprise the surficial aquifer system (Fish & Stewart, 1991). The aquifer, known as the Biscayne Aquifer, has high hydraulic conductivity, due to the abundance of solution holes formed by groundwater percolation (Shaw, 1985).

Historically, surface and ground water flow patterns varied between the wet and dry seasons. Although modifications have occurred because of water management, the oldest documented hydrologic data resemble the general historic seasonal and spatial patterns (described below) enough to be useful for illustration. The average September

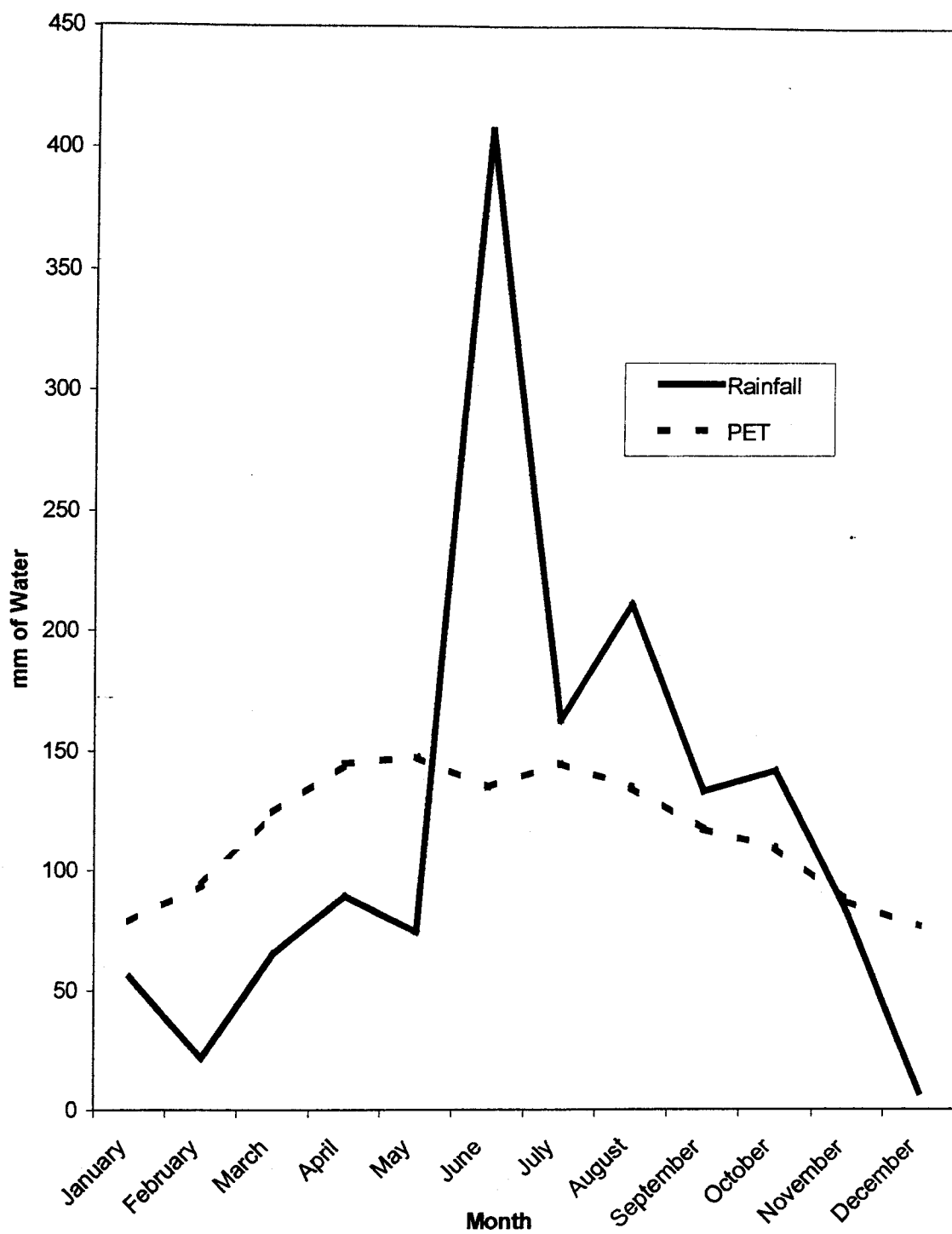


Figure 8. Monthly Penman potential evapotranspiration and average study area rainfall. Rainfall was averaged from 1989, 1992, and 1995 rain data taken at S-21A and S-331A stations.

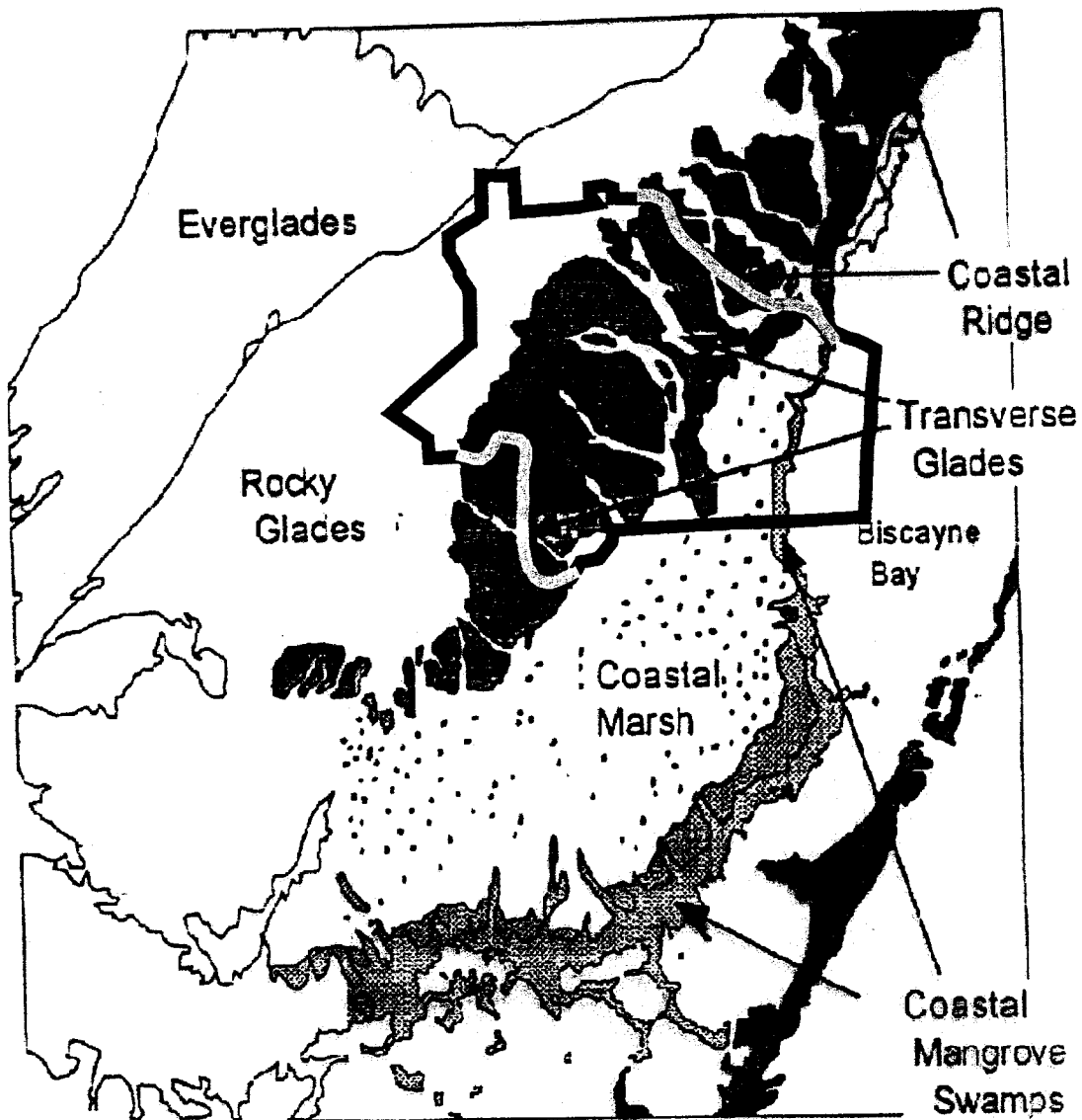


Figure 9. Location of major landscape features and pre-development land cover in southeastern Florida. The study area is outlined in black and grey. Adapted from Tilley & Brown, 1998, Figure 3, p. 136.

(wet season) water table configuration from 1974-1982 is included as Figure 10. During the wet season, water in the Everglades ponded behind the coastal ridge, which produced flow through the transverse glades (Fish & Stewart, 1991). Infiltration from summer rainfall increased the water table in the vicinity of the sand ridge. The seasonal groundwater mound generated subsurface flows away from the ridge in all directions, including the movement of water westward and away from the coast (Fish & Stewart, 1991). The increased surface flow through the transverse glades and the elevated water table of the ridge area resulted in sheetflow of water slowly moving toward Biscayne Bay, the estuary to the east of the study area. During the dry season, the groundwater mound under the ridge disappeared (see Figure 11) and a seaward water table gradient developed, causing groundwater to flow from the Everglades, through the ridge, and toward the coast (Fish & Stewart, 1991). The porous nature of the abundant limestone allowed free movement of water between the surface and surficial aquifer, and springs occurred throughout the coastal zone and in western Biscayne Bay (Fish & Stewart, 1991).

The hydrology produced distinct ecosystems in each of the areas shown in Figure 9, including: marl prairies and freshwater marshes in the transverse glades, flood-tolerant slash pine and sub-tropical hardwood hammock communities on the ridge, extensive marshes in the coastal area, and mangrove forests along the shore of Biscayne Bay (SFWMD, 1993). Seagrass communities colonized much of Biscayne Bay, a sand and mud-bottom estuary with depths of 3 to 9 feet (Williams & Singleton, 1995).

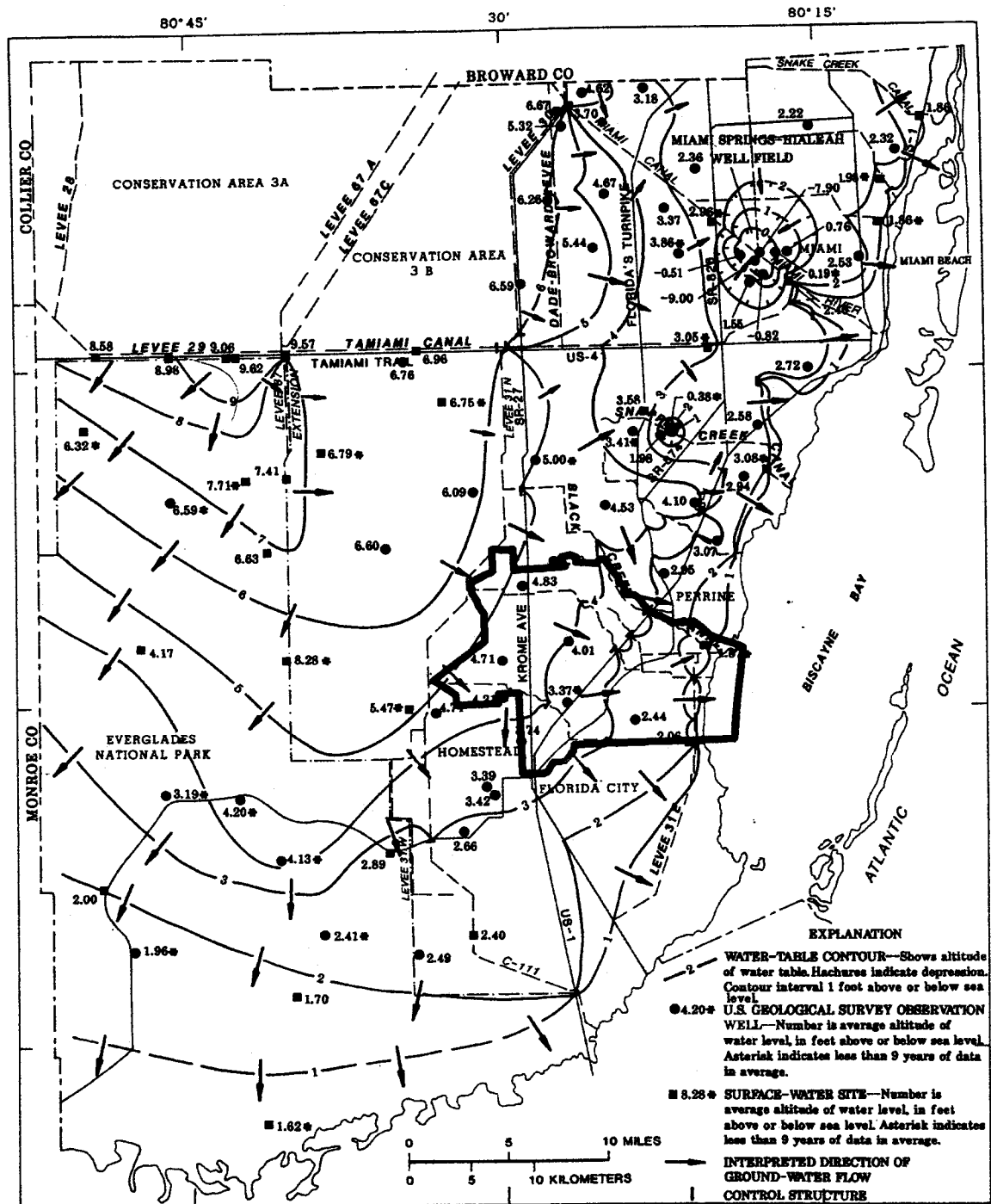


Figure 10. Average water table in Dade county, Florida during September from the period 1974-1982. Modified from Fish & Stewart, 1991, Figure 18, p. 41. The study area is outlined in dark black.

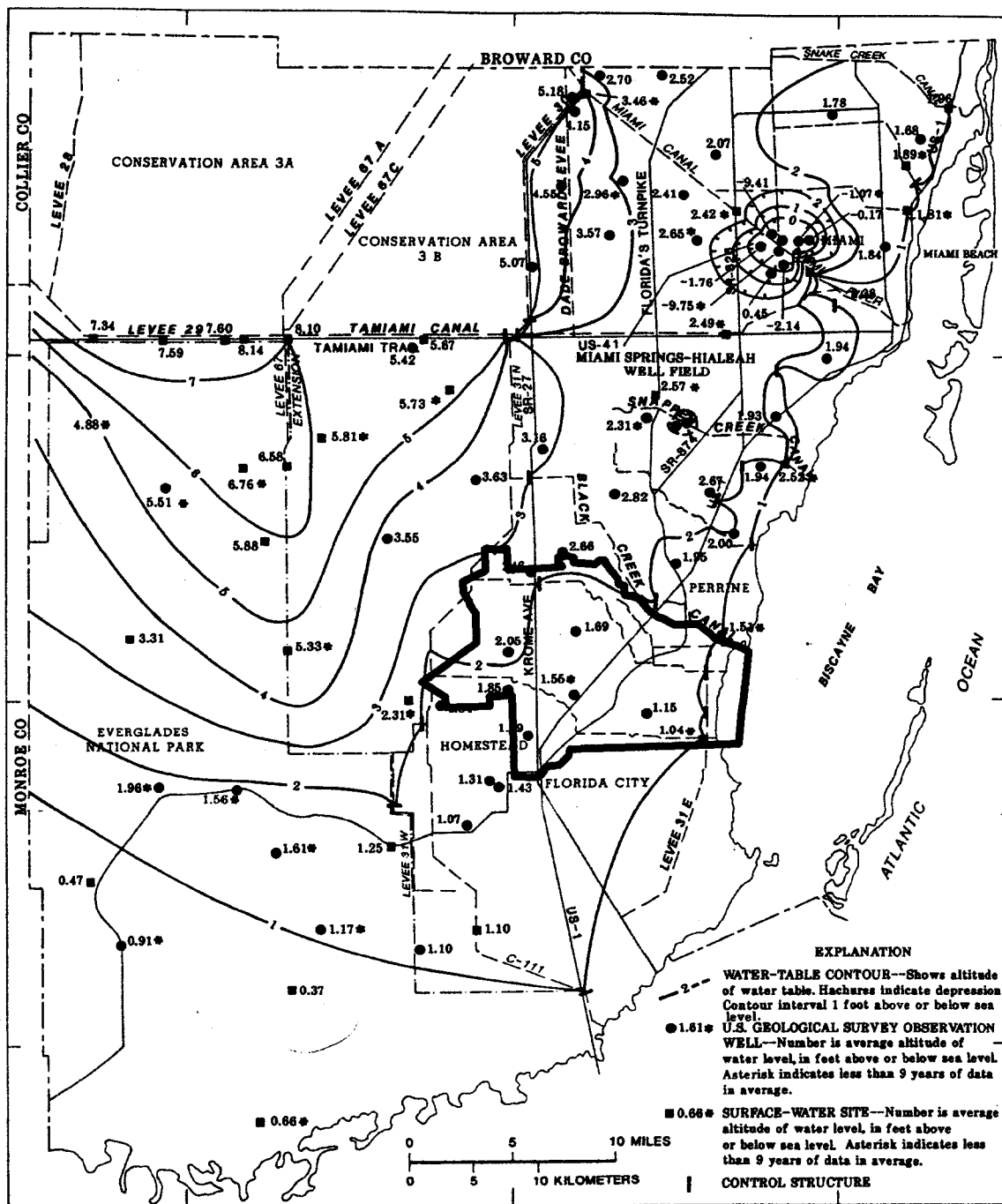


Figure 11. Average water table in Dade county, Florida during April from the period 1974-1982. Modified from Fish & Stewart, 1991, Figure 19, p. 42. The study area is outlined in dark black.

Significant changes in the study area occurred as a result of the Central & South Florida Flood Control Project. During the 1950's and 1960's, the US Army Corps of Engineers built canals, pumps, and structures (see Figure 8) for flood control in the urban and agricultural lands of the region. The C-102 and C-103 canals were dug into the transverse glades and extended to the coast, the L-31W canal brought surface waters from upstream, the C-111 canal drained water away to the south, and a hurricane protection levee (L-31E) was built just west of the shore of Biscayne Bay. The canals achieve flood control because the area has high hydraulic interconnection, and therefore they can regulate groundwater elevation at considerable distances. The water table 600 feet away from the canal has been found to be within .06 feet of canal levels (Shaw, 1985). The improvement of drainage has had several impacts, including: lowered seasonal groundwater ridges, reduced seasonal movement of groundwater, drainage of wetlands, shortened flow paths, potential for pollutants to enter the aquifer, and saltwater intrusion (Klein & Hall, 1978, Sonensheim & Hofstetter, 1990, Fish & Stewart, 1991). The area is also affected by South Florida Water Management District operations of canal gates and pumps throughout the rest of southeast Florida. Salinity pollution of the Biscayne aquifer has been a drinking water supply problem, which resulted in new wells tapped further away from the coast and the construction of salinity control structures in the canals (SFWMD, 1983, Alleman, 1995). Since flood control and drinking water protection are the primary objectives of the engineered system, the drainage network and water management decisions tend to disrupt historic hydrology. As a result, the magnitude, timing, and location of hydrologic loads discharged to Biscayne Bay have changed

(Williams & Singleton, 1995, Alleman, 1995). The current distribution of wetlands in the study area is shown in Figure 12.

Land use in the study area, based on a 1995 classification, is displayed as Figure 13. The predominant land use in the study area is agricultural, because the warm climate of southern Florida is conducive to the cultivation of crops and trees that cannot be efficiently grown elsewhere in the mainland United States. In general, most of the commercial and residential areas straddle the two major roads in the study area, US Highway 1 and the Florida Turnpike. Nurseries and agricultural lands are intermixed throughout the rest of the area. Most of the remaining natural systems are located near Biscayne Bay, although one larger wetland is located west of US 1 on the border of the C-102 and C-103 sub-basins (see Figure 12). Roughly 100,000 people lived in the study area in 1990, and the population is forecast to increase 20% by 2015. Most of the new residents will be settling on newly-converted agricultural lands.

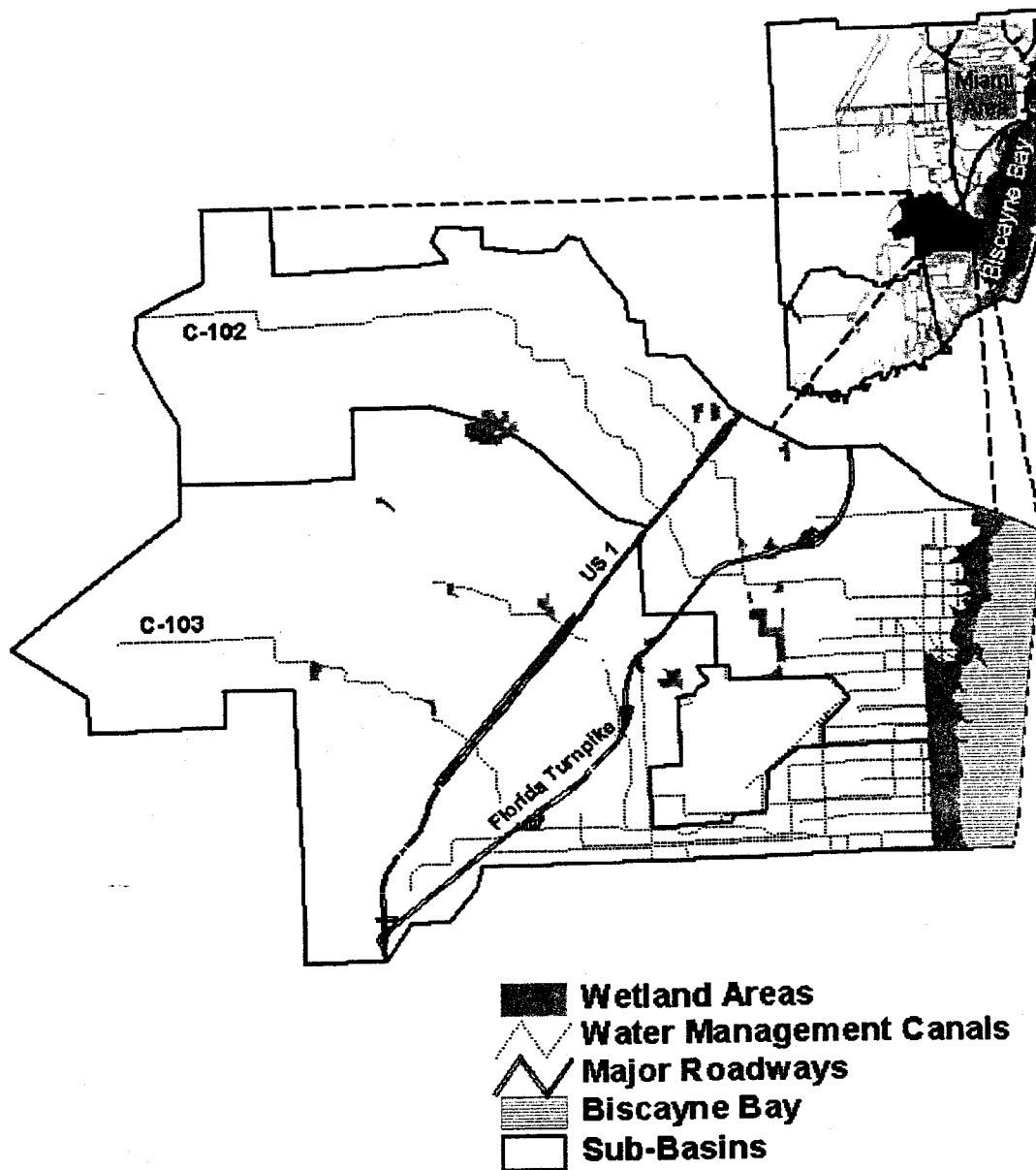
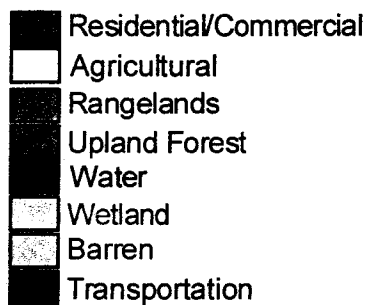
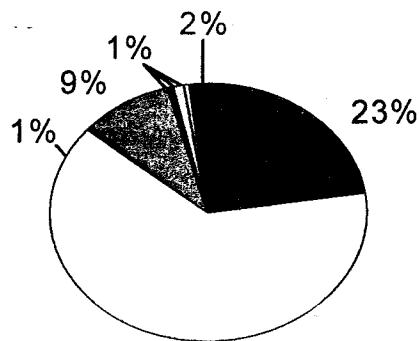
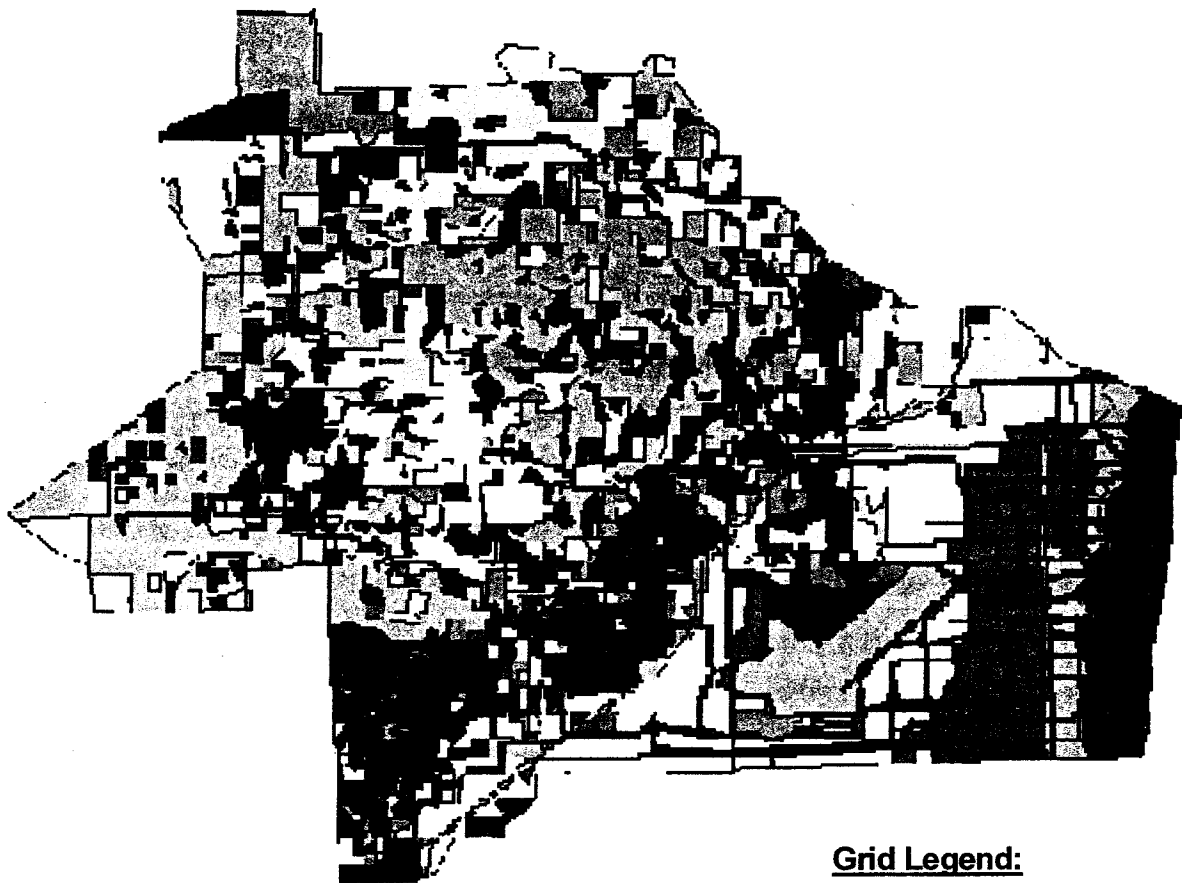


Figure 12. 1995 distribution of wetlands in the study area. Wetlands areas determined according to FLUCC (Florida Land Use Classification Code).



Grid Legend:

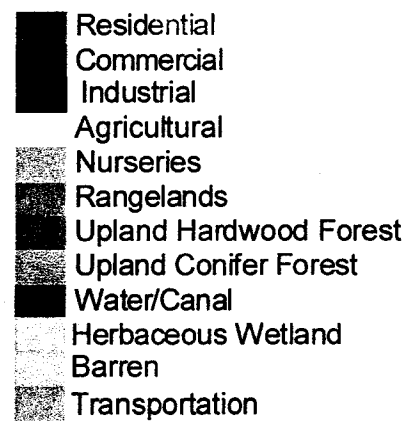


Figure 13. 1995 Level II Land Use/Land Cover in C-102, C-103, and Homestead Air Force Base sub-basins. Pie chart shows percentages of land area by Level 1 classification.

METHODS

The following section describes the methods used to study the hypothesized effects of spatial wetland configurations. A model of water quantity and quality was developed, and data to satisfy the model variables were collected. The model was then calibrated and validated. Sensitivity analyses were conducted to assess output sensitivity to particular model processes. A series of land use scenarios, including various stormwater wetland network designs, were constructed and tested using the model. Results from the scenarios were compared using maps and graphs to assess relationships of spatial land use/land cover patterns to hydrology and water quality.

General Methodological Approach

A temporal spatial model of hydrology and constituents was developed in a GIS environment, which took into account the unique characteristics of the study area. The model was designed to study the effect of wetland location in basins on water quality and quantity. Model development was particularly cognizant of the hydrologic processes at work in a flat landscape, most notably VSA (such as wetlands and urbanizing areas) hydrology. In addition, the model was developed to easily incorporate digital data about landscape characteristics to increase precision of spatial simulation results. The GIS software utilized was ESRI's ArcView and the ArcView extensions Spatial Analyst, Dialog Designer, and GeoProcessing Wizard.

The general approach in developing the model began with a conceptual characterization of how energy and matter were distributed, transformed, and stored

within the watershed, and of how they were exchanged across the system boundary. The driving forces, storages and flows of the conceptual model were then visually represented as a system diagram. The flow pathways on the diagram illustrated physical relationships that were then translated into mathematical equations. The equations were compiled into a numerical model that simulated the quantity of flows and storages of water, phosphorus, and sediments on a daily basis, for a period of one year.

To simulate the hydrology and constituent flows spatially, the study area was divided into a grid of square cells, where each individual cell was programmed with the mathematical equations representing the physical processes and flows. The grid cells used the local landscape attribute data as input for the equations. To generate the input data, GIS was used to reference study area coverages with attribute tables, resulting in spatial layers of attribute data values. These results were then converted into the analysis grid format, and the values in the grid cells satisfied the local equation variables. The cell size for all grids was 76x76 meters, so chosen to allow the South Florida Water Management District to compare the results of this project with those of other studies.

To simulate the time dimension, the spatial model used an iterative technique, illustrated in Figure 14, composed of time steps of one day. The daily processes were sequenced to perform a water budget and maintain constituent mass-balance. Based on a set of initial conditions and start-of-day storage values, the system processes were simulated, which yielded hydrologic and constituent flows. The simulated flows were overlayed in the appropriate manner (as an addition or subtraction) from the initial storage values to maintain mass-balance, which resulted in end-of-day storages. The end-of-day storages were then used as the next day's initial conditions. The sequence was

reiterated for the desired simulation length (generally one year). Avenue, the object-oriented programming language of ArcView, was used to create scripts that automated the calculation and iteration processes.

The spatial model routed water and constituent flows crossing each grid cell boundary into appropriate downstream cells. Several spatial functions provided by GIS software were utilized to allocate flows traveling laterally from cell to cell, most significantly the ArcView command "FlowAccumulation". When "FlowAccumulation" was executed (hereafter referred to as a "drain" event) with a flow direction and a flow volume grid as input, the volume values were distributed along flow paths determined from the direction grid, and so accumulate in the lower elevations of the watershed (see Figure 15).

Model Assumptions

Several assumptions were made about landscape properties and hydrology, as follows:

1. The study area is an ideal groundwatershed, with no groundwater seepage flows across the study area boundary.
2. The only surface water deliveries to the study area occur from L-31 W into C-102. The amount of water delivered to C-102 is a constant percentage of flow over S-331, located just upstream from the junction of C-102 and L-31 W.
3. Water management gates are operated precisely according to their regulatory schedule.

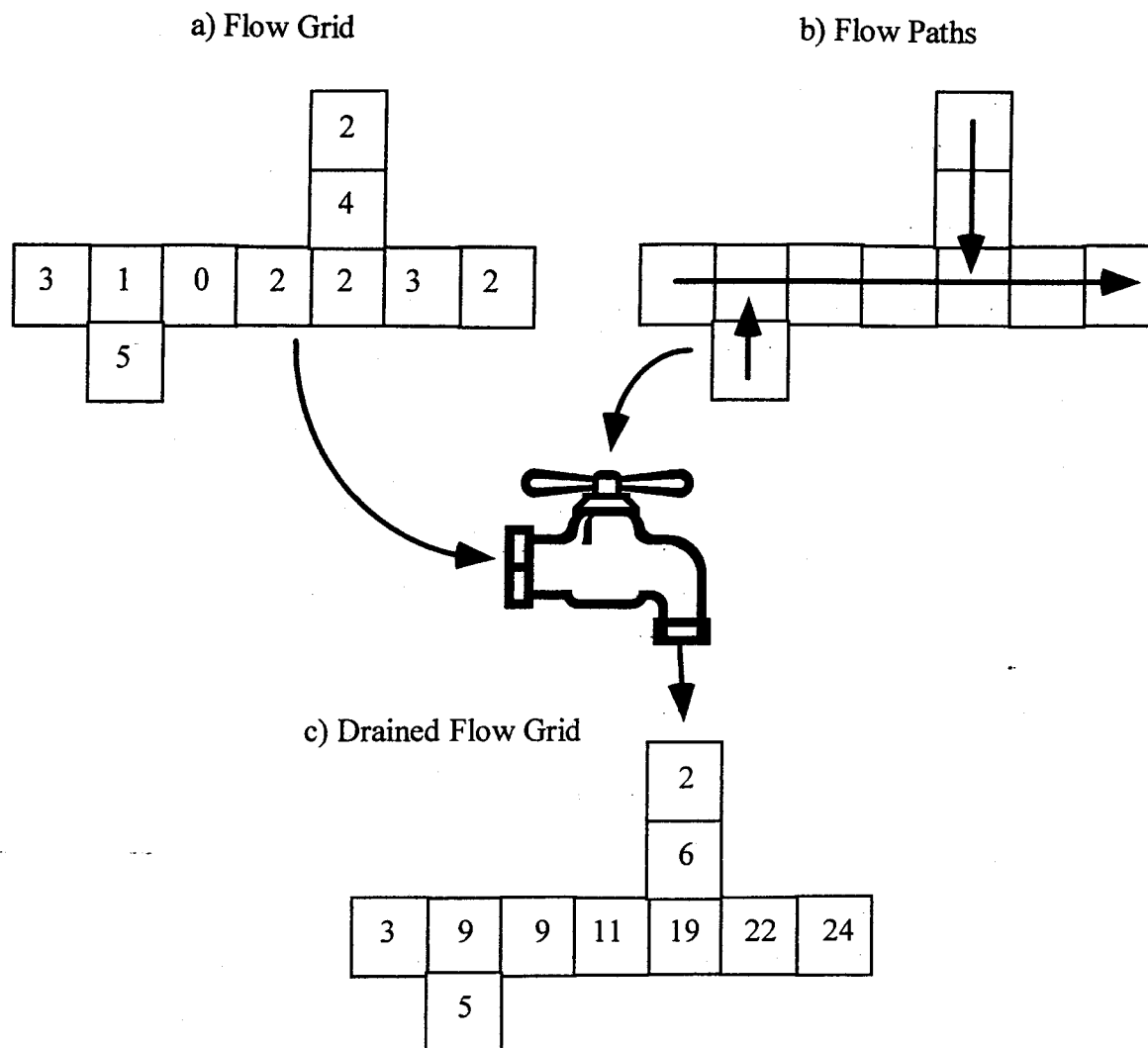


Figure 15. Demonstration of a drain operation. A flow grid a), is drained along the flow paths b) determined from a flow direction grid, resulting in the drained grid result in c). The total sum of all grid values in a) is 24, while the total from c) is 110. Without correction, drain then increases the actual amount of water involved in flow processes considerably, in this case over 450%.

4. No significant point source discharges occur in the study area.
5. Tidal influences on coastal area hydrology are negligible.
6. The soil column is a homogenous vertical extension of surface soil type.
Consequently, properties of the soil are uniform and identical to those of the surface soil throughout the soil column.
7. A function of soil permeability and water holding capacity can be used as a reasonable surrogate for hydraulic conductivity to calculate groundwater flow.
8. Pumping of groundwater for human purposes does not affect surface and shallow groundwater hydrology in the study area.
9. Rainfall in 2015 will be the same as rainfall in 1992.
10. Daily potential evapotranspiration equals $1/30^{\text{th}}$ of monthly potential evapotranspiration.
11. The soil water correction to potential evapotranspiration is a first order, linear equation.
12. Water flows only from one cell to one other during any individual time step.
Water cannot flow from one cell into two cells, for example where water travels in two different directions on a ridge or watershed divide.
13. The principle factors governing horizontal velocities of overland flow are slope and impervious surface.
14. Canals are uniform in depth.
15. Nutrient dynamics, constituent uptake, and evapotranspiration relationships related to plant growth are constant throughout the growing season.

16. The sole mechanism for transporting water quality constituents is surface runoff, and therefore the factors affecting overland flow velocity regulate nutrient transport.
17. Water entering the study area from groundwater contains no suspended solids.
18. Historic and future land cover Event-Mean Concentrations are the same as current values.

Data Collection & Processing

Primary spatial data were obtained from SFWMD, US Geologic Service (USGS), Miami-Dade Transit Agency, and the GeoPlan Center at the University of Florida. 1995 Land Use/Land Cover classifications, hydrography, and sub-basin delineation were obtained from SFWMD. A detailed, up-to-date road coverage was obtained from Miami-Dade Transit Agency. DEMs with cell sizes of 30x30 meters were obtained from GeoPlan Center and USGS. The USGS-obtained DEMs were more precise, but since quadrangles for all parts of the study area were not available at the time, the GeoPlan Center-obtained DEM provided values in remaining areas as necessary. Several coverages were obtained from GeoPlan Center, including general and detailed soils, geology, coastlines, and public land surveys. Some of these coverages were not explicitly used by the model, but enhanced study area visualization and development of model theory. Primary spatial data were projected into a standard coordinate system (NAD83, Albers projection), converted into 76x76 meter grid format, and clipped to the study basin area.

Climate-Dependent Parameters

Climate data were used to create daily grids of rainfall and evapotranspiration for 1989, 1992, and 1995. Daily rain data for each year were obtained from SFWMD for the S-21A structure, located at the terminus of the C-102 basin, and for the S-331 structure located just northwest of the C-102 basin. The study area was divided approximately in half along a north-south divide. Rainfall at S-331 was used for the western half of the basin, while rainfall at S-21A was used for the eastern half. Rainfall at S-331 was 1101, 1314, and 1608 mm for 1989, 1992, and 1995, respectively. Rainfall at S-21A was 1088, 1821, and 1761 mm for 1989, 1992, and 1995, respectively.

South Florida evapotranspiration rates are generally high, usually ranging from 908 to 1555 mm/yr, and constituting roughly 85% of the outflows of water from natural systems (Koch & Rawlik, 1993). Highly evaporative systems are estimated with reasonable accuracy by using Penman potential evapotranspiration (PET) (Calder, 1990). Penman PET relies on calculations based on net radiation, temperature, wind speed, and relative humidity (Jones et al., 1984). Smajstrla calculated PET using climate parameters for Miami (Smajstrla et al., 1985). Since Penman PET is based solely on weather conditions, it assumes sufficient soil water is present and cover of healthy growing grass (Bidlake et al., 1993).

PET for land covers other than grassland can be estimated by using consumption coefficients, sometimes referred to as "crop coefficients". Consumption coefficients are highest for upland forest and upland herbaceous systems. Wetland systems have consumption coefficients less than urban and open water areas. The consumption coefficients for agricultural lands were averaged from values for several different crops grown in south Florida (Jones et al., 1984, Lynne et al., 1987). Data were averaged for

upland forest, cypress, and herbaceous wetland from multiple literature values (Burns, 1978, Calder, 1990, Bidlake et al., 1993, Koch & Rawlik, 1993, Abtew, 1996). Upland and wetland coniferous land cover consumption coefficients were calculated using ET flows derived from research by Liu (1996). Where actual consumption coefficients were not available in literature, the coefficients were calculated by dividing annual evapotranspiration flow data by calculated annual PET, which was 1372 mm/year (Smajstrla et al., 1985). PET of standing surface water from urban and residential areas incorporates the "heat island" effect that results from increases in asphalt surface and generation of waste heat (Bornstein, 1968, Preston-Whyte, 1970). The effect of urban heat islands was determined to be 5% above open water PET, which assumes urban lands do not reduce ET from soil water. Wetland hardwood system PET was assumed to be equal to cypress PET. Grassland PET was assumed to be equal to calculated PET. The compiled PET consumption coefficients are given in Table 1.

Water Management Data

Daily deliveries from the L-31W canal into C-102 and C-103 were calculated by averaging daily flows from 1989 to 1997 over the S-331 structure, which is located just upstream from the source of C-102. Since much of this water bypasses C-102 and C-103, but the exact amount is unknown, a user-set coefficient was used to estimate the actual inflow. All model results seen later were obtained with a coefficient allowing 11% of the averaged S-331 flow to enter C-102 and C-103 combined.

Flows of water from the Florida City canal sub-basin into the eastern stretches of the C-103 canal were estimated, since the actual inflow is unknown. First, the number of

<u>LAND USE</u>	<u>FLUCCS CODE</u>	<u>K</u>
Residential (low)	110	1.05
Residential (medium density)	120	1.05
Residential (high density)	130	1.05
Commercial	140	1.05
Industrial	150	1.05
Extractive	160	1
Institutional	170	1.05
Recreational	180	1
Open Land	190	1
Cropland	210	1.02
Tree Crops	220	1.06
Feeding Operations	230	1.05
Nurseries	240	1.06
Specialty	250	0.88
Other Open Land	260	1
Herbaceous	310	1
Shrub/Scrub	320	0.83
Mixed Rangeland	330	1.05
Upland Conifer Forest	410	0.87
Upland Hardwood Forest	420	1.29
Upland Herbaceous	430	1.29
Tree Plantation	440	0.87
Streams/Canals	510	0.98
Lakes	520	0.98
Reservoir	530	0.98
Bays & Estuaries	540	0.98
Major Springs	550	0.98
Slough Waters	560	0.98
Wetland Herbaceous	610	0.87
Wetland Conifer Forest	620	0.87
Wetland Forest Mixed	630	0.87
Vegetated Wetland Mixed	640	0.84
Non-vegetated Wetland	650	0.98
Beaches	710	1
Sand Other	720	1
Exposed Rock	730	1
Disturbed	740	1
Transportation	810	1
Communication	820	1
Utilities	830	1

Table 1. Coefficients reflecting effect of land use on potential evapotranspiration.

cells in the Florida City sub-basin was determined. Next, a coefficient was set based on approximate losses of water from Florida City to pathways other than into C-103.

Specifically, the losses considered include deep aquifer recharge, evapotranspiration, and flows of water from Florida City into basins to the south of C-103 through L-31E. The final simulated inflow of Florida City canal water for each individual day was calculated by multiplying this coefficient by the number of cells, and then by the daily rainfall.

A water management regulatory schedule was provided by the SFWMD, which contained information about operation of water management gates. The operation of each gate depends upon canal stage, and is operated in open, closed, or static conditions. Differences in whether a gate was a spillway or a culvert were ignored.

Spatial Data Pre-processing

The spatial soil data were modified slightly from the coverage provided. A soil that was classified "urban" did not accurately represent soil behavior beneath the pavement. Since surface soil characteristics represented the entire soil column, groundwater dynamics would have been distorted using soils classified as urban. Urban soils were reclassified based on a spatial interpolation of adjacent soil types.

The source DEM is a map composite created from USGS and UF GeoPlan Center DEMs. The USGS DEM is a more detailed representation of the landscape, but is currently being modified in several south Dade quadrangles to correct errors. As a result, USGS DEMs did not exist for the entire study area, and the GeoPlan Center DEM values were used in areas not covered by USGS DEMs. The hydrography coverage was used to cut ten foot deep surface flow paths out of the composite DEM to simulate the water

management canals. Where roads occurred, one foot of elevation was added to the DEM to represent the slightly elevated road surfaces. The edge of the study area was increased by six inches to insure internal surface water drainage. Elevations were converted to the 1929 NGVD vertical reference datum. The resulting analysis DEM is shown in Figure 16. The DEM then was used to calculate a map of initial surface flow direction and land surface slope.

Attribute tables were used to create the grids shown in Figure 17. A map of impervious surface was created by correlating Land Use/Land Cover to literature values for impervious surface (Harper, 1994, Tilley, 1996). PET consumption coefficients for specific land covers were referenced to the Land Use/Land Cover coverage to create spatial grids for adjusting daily PET. Soils data were referenced to the SSURGO database to create grids of soil permeability, soil water capacity, and percent soil as calcium carbonate. Literature values and values provided by SFWMD for Event Mean Concentration (EMC) of Total Phosphorus (TP) and Total Suspended Solid (TSS) concentrations were referenced to Land Use/Land Cover to obtain constituent EMC grids. The EMCs used are given in Table 2.

To generate potential soil water storage, a control volume was constructed based on the lowest land surface elevation in the landscape. The lowest surface elevation was taken as ten feet below MSL, the assumed design depth of canals flowing into Biscayne Bay. A soil depth grid based on the land surface height above minus ten feet MSL was then determined by the soil depth to generate a soil volume grid. The soil volume grid was then multiplied by the soil water capacity grid to obtain a grid of potential soil water storage.

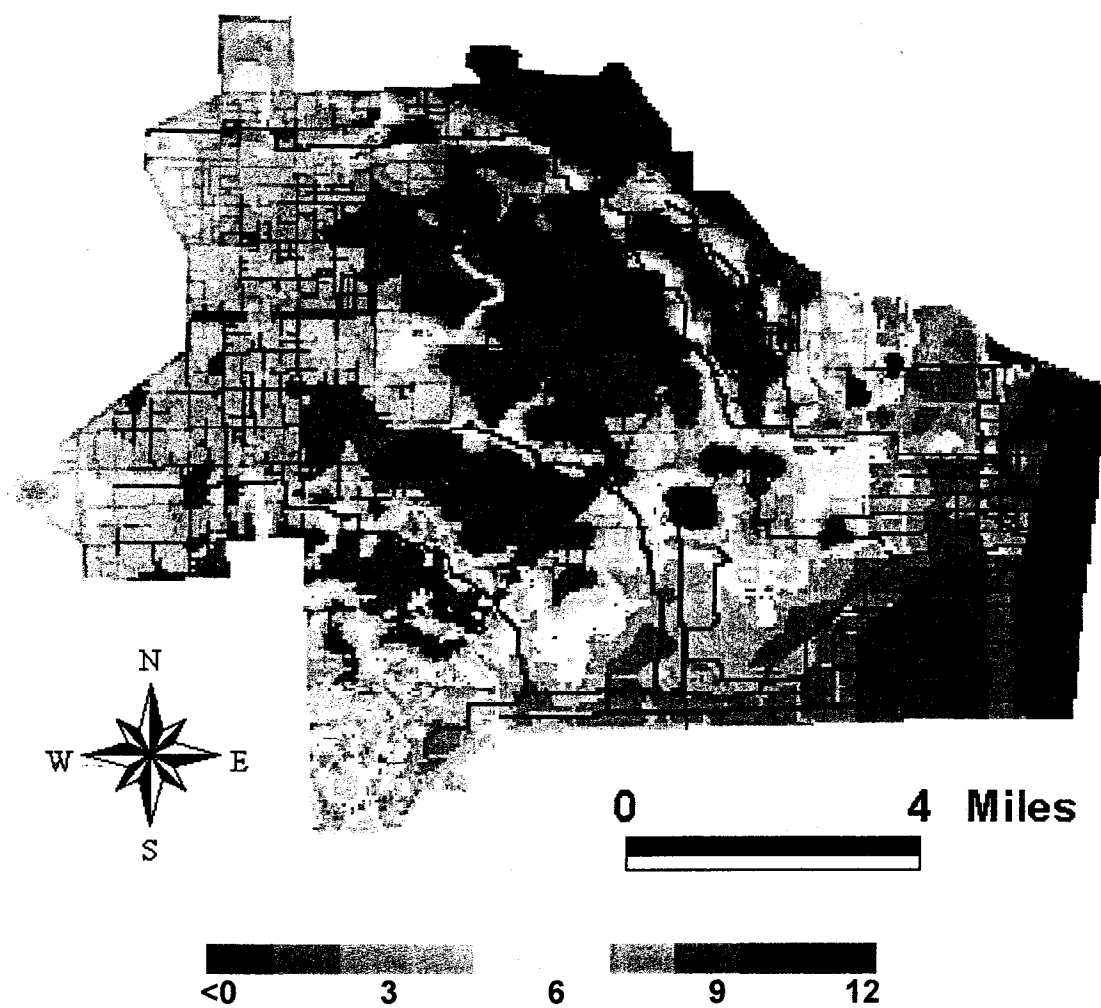


Figure 16. The analysis DEM used as model input for the land use scenarios. Units are in feet above MSL.

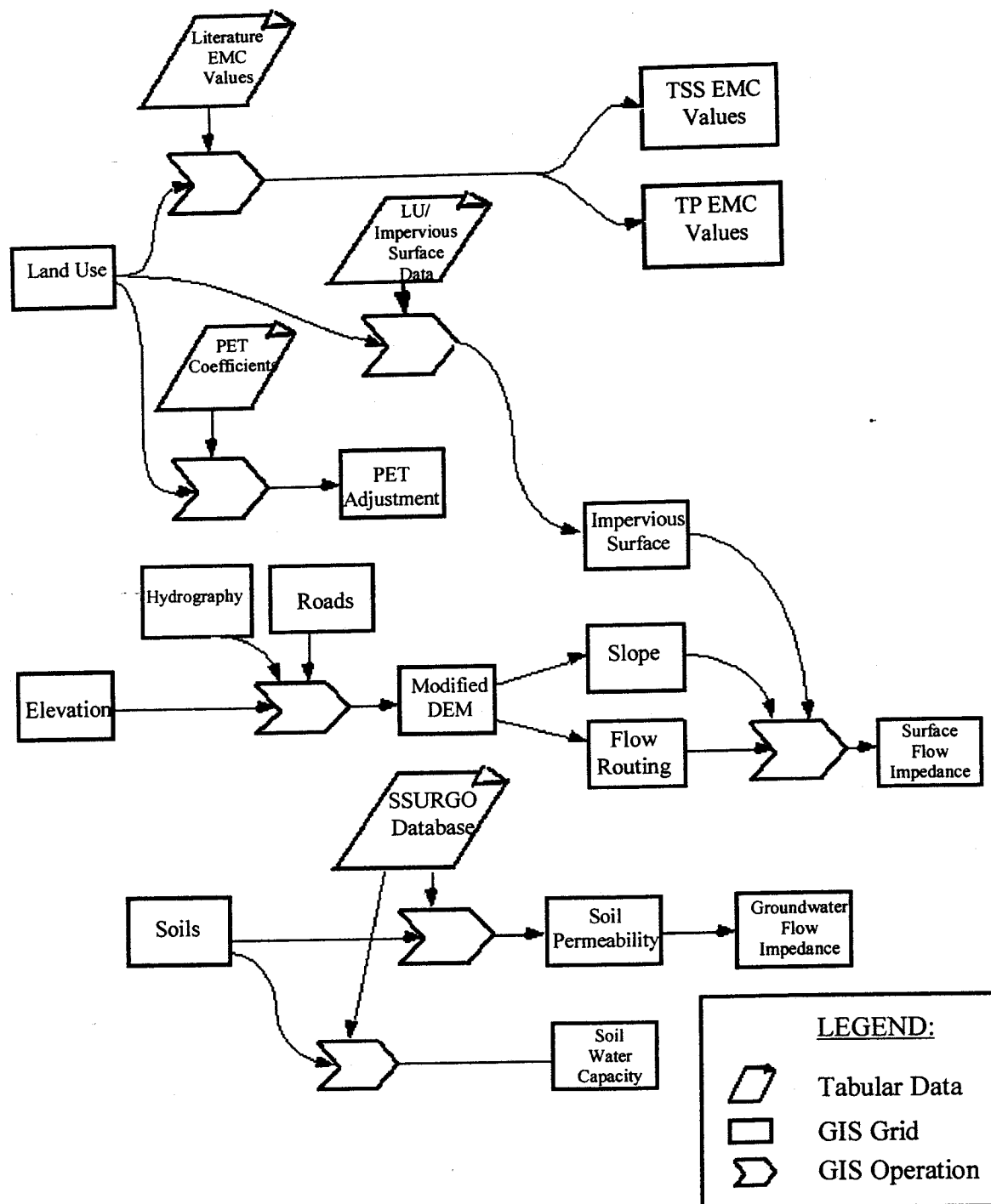


Figure 17. Creation of model input grids from primary data and attribute tables.

LAND USE	FLUCCS CODE	TP EMC	TSS EMC
Residential (low)	110	.337	40.1
Residential (medium density)	120	.337	40.1
Residential (high density)	130	.337	40.1
Commercial	140	.155	69.8
Industrial	150	.281	158.1
Extractive	160	.281	158.1
Institutional	170	.150	81.0
Recreational	180	.053	11.1
Open Land	190	.053	11.1
Cropland	210	.322	55.3
Tree Crops	220	.322	55.3
Feeding Operations	230	.322	55.3
Nurseries	240	.322	55.3
Specialty	250	.322	55.3
Other Open Land	260	.162	36.1
Herbaceous	310	.053	11.1
Shrub/Scrub	320	.053	11.1
Mixed Rangeland	330	.053	11.1
Upland Conifer Forest	410	.053	11.1
Upland Hardwood Forest	420	.053	11.1
Upland Herbaceous	430	.053	11.1
Tree Plantation	440	.053	11.1
Streams/Canals	510	.110	3.1
Lakes	520	.110	3.1
Reservoir	530	.110	3.1
Bays & Estuaries	540	.110	3.1
Major Springs	550	.110	3.1
Slough Waters	560	.110	3.1
Wetland Herbaceous	610	.190	10.2
Wetland Conifer Forest	620	.190	10.2
Wetland Forest Mixed	630	.190	10.2
Vegetated Wetland Mixed	640	.190	10.2
Non-vegetated Wetland	650	.190	10.2
Beaches	710	.162	36.1
Sand Other	720	.162	36.1
Exposed Rock	730	.162	36.1
Disturbed	740	.162	93.1
Transportation	810	.239	36.1
Communication	820	.162	36.1
Utilities	830	.162	36.1

Table 2. Event Mean Concentrations (EMCs) for various land uses. Land use/land cover classification scheme is the FLUCCS code level 2. Compiled from Harper, 1994, and Metropolitan Dade County DERM, 1996.

Initial Conditions

Grids representing initial conditions of surface water storage, soil water storage, groundwater table, TP, and TSS storage were constructed. Canal stage data were collected from S-194, S-195, S-165, S-21A, S-20G, S-196, S-167, S-179, S-166, and S-20F from January 1, 1992 (average rainfall year). The stage data were interpolated throughout the canal grids using an Inverse-Distance Weighted (IDW), nearest neighbor algorithm, creating initial surface water storages throughout the canal network.

Groundwater elevation data from USGS wells at G-757A, G-614G, G-1486, F-358, and three points on the shore of Biscayne Bay where groundwater was assumed to be zero feet above mean sea level were collected for January 1, 1992. The groundwater data were interpolated throughout the C-102 and C-103 basins using the IDW, nearest neighbor algorithm, resulting in basin-wide initial groundwater tables. The groundwater tables were then processed through an algorithm utilizing digital elevation data and potential soil water storage to generate an initial soil water storage grid. Average constituent concentrations at water quality monitoring points near S-194, S-195, S-165, S-21A, S-20G, S-196, S-167, S-179, S-166, and S-20F between 1985 and 1996 were compiled. The concentration data were interpolated throughout the canal grids using an Inverse-Distance Weighted (IDW), nearest neighbor algorithm, creating initial concentrations throughout the canal network. The concentrations were multiplied by water quantity to obtain initial constituent storages in the canals.

Travel Cost Grids

Travel cost grids were created to reflect the fact that certain values for physical properties (including slope, impervious surface, soil permeability, and soil water capacity) have a low resistance (travel cost) to horizontal flow. The travel cost grids were generated by inverting the property data values, normalizing the inverted values between 0 and 1, and then subtracting the normalized values from 1. The results were grids with low travel costs near 0 and high travel costs near 1. For example, high slope and/or impervious surface values led to lower (closer to 0) travel costs, whereas low slope and/or impervious surface values yielded travel costs nearer to 1. The travel cost grids were then drained to create a grid of basin impedance to surface and sub-surface lateral flow. The travel cost grids described both here and below were constructed according to the following equation:

$$Z_{\text{cost-property}} = 1 - (f(1 / x_{\text{property}})) \quad (1)$$

where,

$Z_{\text{cost-property}}$ = travel cost for landscape property

$f()$ = normalization of data values

x_{property} = landscape property value

Model Diagram & Mathematical Equations

A system diagram of the flows, storages, and processes affecting stormwater and stormwater constituents is shown in Figure 18. The diagram represents the landscape attributes, physical processes, flows, and storages of each cell in the spatial simulation grid. Inputs of water and constituents enter the cell from the top and left, and outflows to be routed to other downstream cells exit the system at right. The model storages include surface water, soil water, total phosphorus, and total suspended solids.

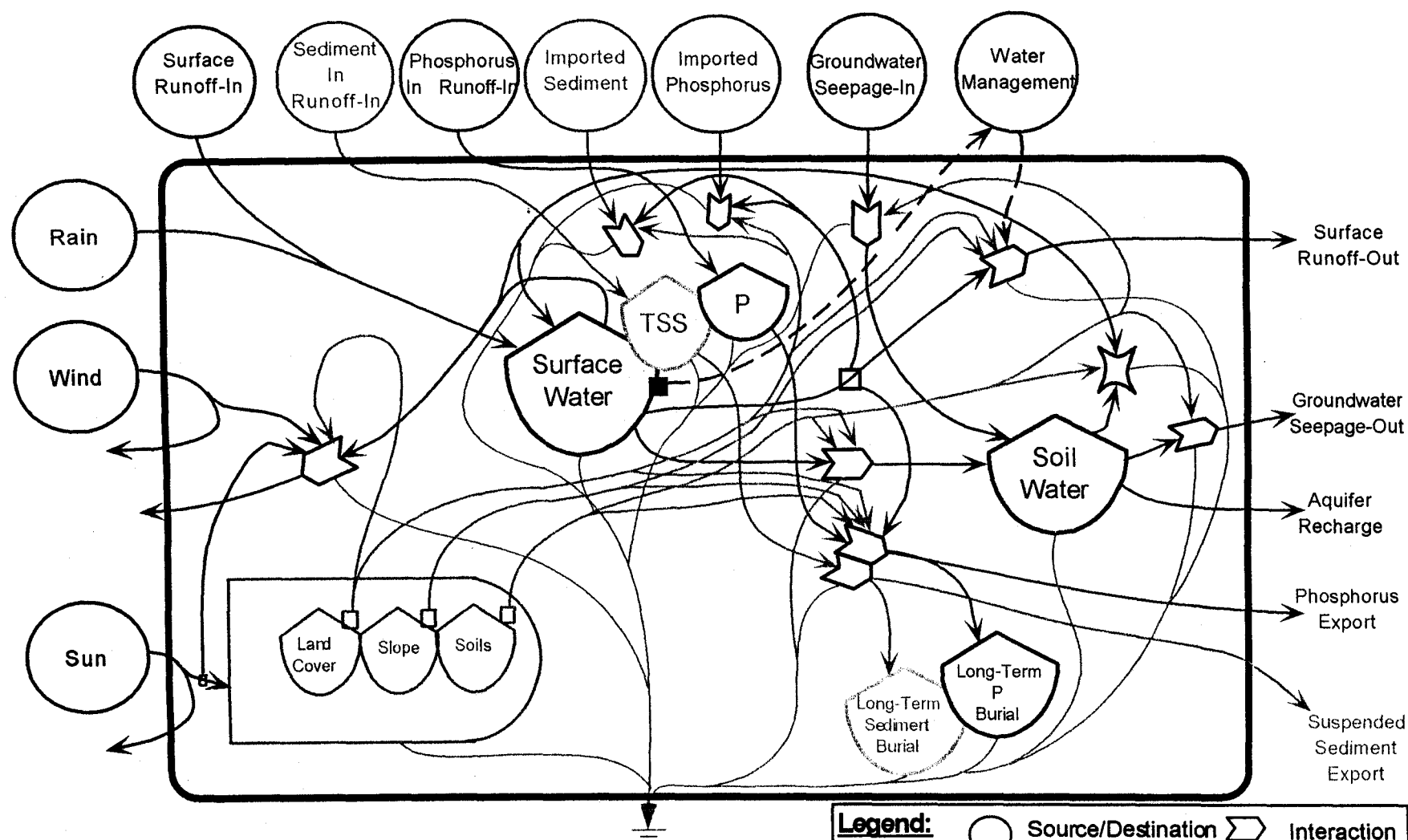


Figure 18. Systems diagram of the flows, storages, and processes affecting stormwater & stormwater constituents. The system organization is characteristic of each cell in the spatial simulation grid. Blue colors indicate hydrologic cycle, tan indicates sediment cycle, red indicates phosphorus cycle, green indicates ecosystem effects, purple indicates information, and grey indicates 2nd law losses.

The GIS water budget simulation model, as interpreted from the system diagram, is illustrated in Figures 19 and 20. Inflows of water in the form of rain, surface runoff-in, deliveries of water from the Florida City and L-31 W canals, and groundwater seepage-in were added to the cell water storages. Losses from the cell occurred from evapotranspiration, surface runoff-out, groundwater seepage-out, recharge to the surficial aquifer, and losses to Biscayne Bay. Movement of flows between surface and ground water storages occurred as a result of infiltration and overflow. Overflow occurred when the soil water control volume of a cell was exceeded. Groundwater elevations were calculated at the end of each simulation day based on the end-of-day soil water storage, the DEM, and the soil water capacity grid. The general equations describing the mass balances for surface and ground water storages are as follows:

$$Q_{surfaceend} = Q_{surfaceinitial} + \Delta Q_{surface} / \Delta T \quad (2)$$

$$Q_{groundend} = Q_{groundinitial} + \Delta Q_{ground} / \Delta T \quad (3)$$

where,

$Q_{surfaceinitial}$ = surface water storage at beginning of time step

$Q_{groundinitial}$ = ground water storage at beginning of time step

$Q_{surfaceend}$ = surface water storage at end of time step

$Q_{groundend}$ = ground water storage at end of time step

$\Delta Q_{surface} / \Delta T$ = change in surface water storage per time step

$\Delta Q_{ground} / \Delta T$ = change in ground water storage per time step

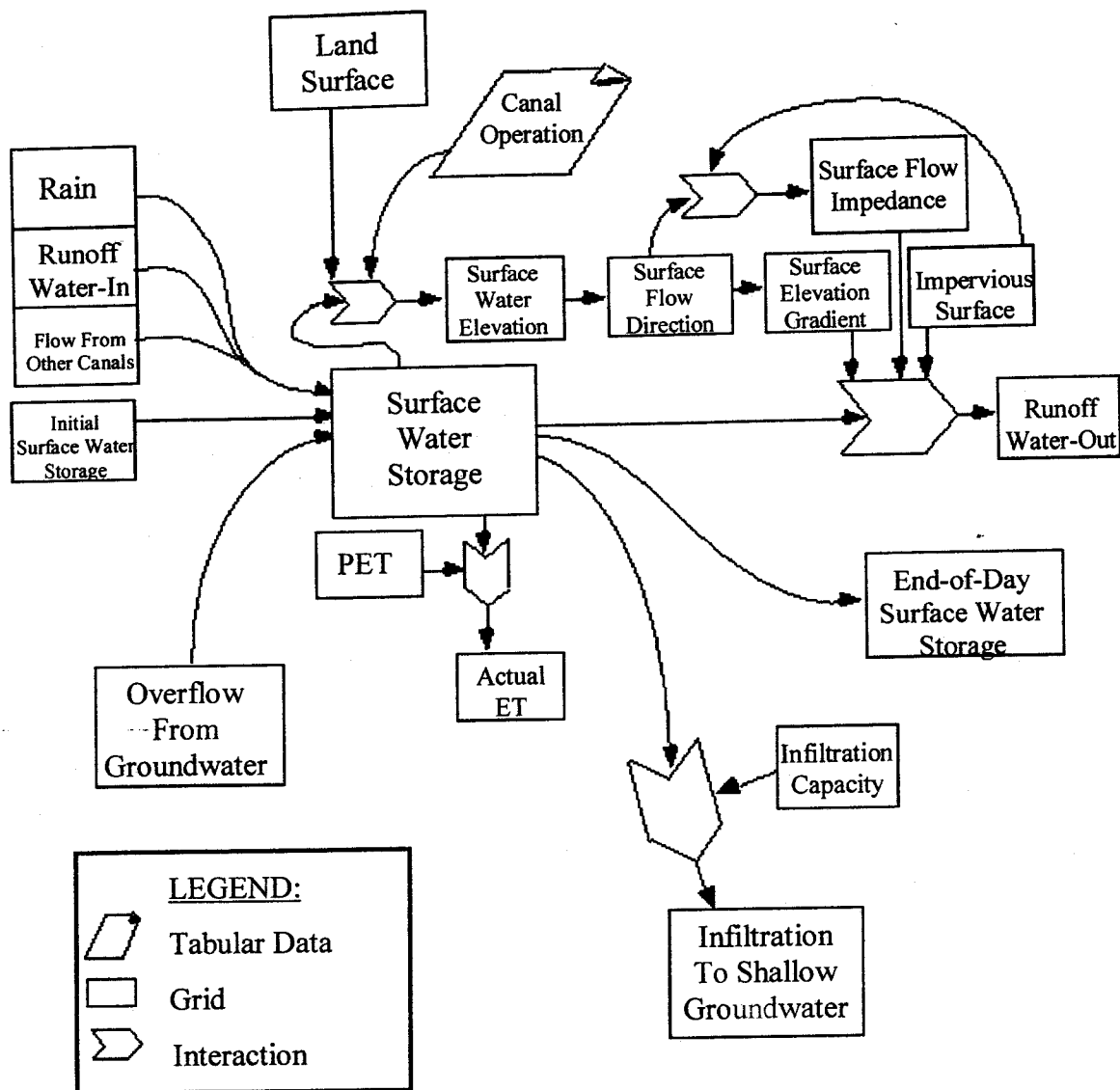


Figure 19. Grid manipulations performed to simulate surface water processes.

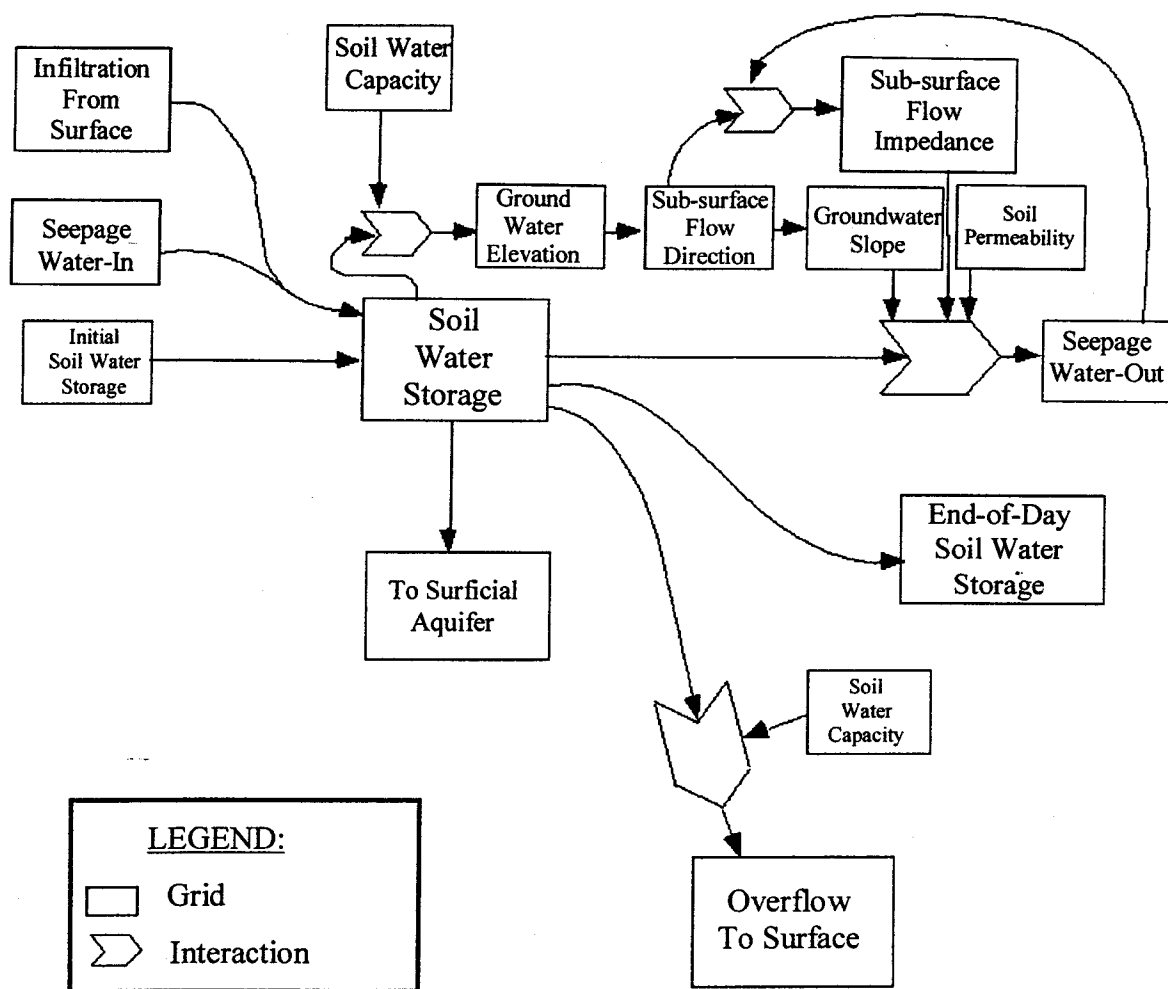


Figure 20. Grid manipulations performed to simulate groundwater processes.

Furthermore,

$$\Delta Q_{surface} / \Delta T = P + X_{in} + D + N - ET_{surface} - I - X_{out} - S_{lateral} - Z \quad (4)$$

$$\Delta Q_{ground} / \Delta T = S_{in} + I + S_{lateral} - ET_{soil} - S_{out} - N - R \quad (5)$$

where,

P = precipitation

X_{in} = surface runoff-in

D = water management deliveries from L-31 W & Florida City canals

N = spring flow from soil water in excess of soil water capacity

$ET_{surface}$ = evapotranspiration from surface water

I = infiltration from surface to soil

X_{out} = surface runoff-out

$S_{lateral}$ = lateral recharge of surface water to soil water

Z = discharge to Biscayne Bay

S_{in} = lateral groundwater seepage-in

ET_{soil} = evapotranspiration from soil water

S_{out} = lateral groundwater seepage-out

R = recharge to Biscayne aquifer

Evapotranspiration was calculated by first attempting to satisfy the required daily land use-adjusted PET with water from surface water storage. To correct the limitations

of the second assumption of Penman PET (adequate soil moisture availability) in places where surface water storage was insufficient to meet daily PET demand, a quantity of soil water storage was subtracted. The amount subtracted was based on the saturation deficit. Saturation deficit was expressed as the ratio of actual to potential soil water stored in the upper .8 meters of soil (Butts, 1988). Below this threshold depth, ET could no longer occur from the soil water storage. Mathematically, the ET functions are described by:

$$ET_{surface} = (PET_{Penman} * k_{PET}) \geq Q_{surface} = Q_{surface}, \quad (6)$$

$$(PET_{Penman} * k_{PET}) < Q_{surface} = PET_{Penman} * k_{PET} \quad (7)$$

$$ET_{soil} = (PET_{Penman} - ET_{surface}) > 0 = \omega, \quad (8)$$

$$(PET_{Penman} - ET_{surface}) \leq 0 = 0 \quad (9)$$

$$\omega = (PET_{Penman} - ET_{surface}) < Q_{soilavailableET} * \Omega = (PET_{Penman} - ET_{surface}), \quad (10)$$

$$(PET_{Penman} - ET_{surface}) \geq Q_{soilavailableET} * \Omega = Q_{soilavailableET} * \Omega \quad (11)$$

where,

$ET_{surface}$ = surface water evapotranspiration

ET_{soil} = soil water evapotranspiration

PET_{Penman} = daily Penman potential evapotranspiration

k_{PET} = PET consumption coefficient for cell land use

$Q_{surface}$ = surface water storage

Q_{soil} = soil water storage

$Q_{soilavailableET}$ = soil water storage available for evapotranspiration

Ω = ratio of available soil water to maximum available soil water

where,

$$Q_{soilavailableET} = \Psi > Q_{soil} = Q_{soil} - \Psi, \quad (12)$$

$$\Psi > Q_{soil} = 0 \quad (13)$$

$$\Omega = (Q_{soil} - \Psi) / Q_{potentialsoilmax} \quad (14)$$

and where,

$Q_{potentialsoilmax}$ = maximum potential soil water storage

Ψ = maximum potential soil water available for contributing to surface ET,

as determined from a user set fraction of soil water capacity

Surface runoff-out and groundwater seepage-out were derived by calculating the cell water storage fraction available for flow out, where the fraction was based upon the elevation gradient. Surface water elevation gradients were calculated separately for canal and non-canal areas. For canal areas, the canal grid was broken up into a series of zones. The zones were structured as segments upstream from the water management structures. The canal areas between the lowest water management structures and Biscayne Bay were made into one additional zone. The elevation gradient was defined as the cell elevation above the minimum canal stage from each zone. For non-canal areas, all cells were first allocated to the nearest canal cell using a GIS allocation function. Surface water elevation above the allocated canal cell stage was calculated. The elevation gradient

was then divided by the distance between the cell and the allocated canal cell, such as in a slope calculation. A GIS function was used to check for cells that were local sinks. Cells found to be sinks had elevation gradient values set to zero.

Water management operations were simulated by separately calculating surface runoff-out values for the canal structure cells. First, canal stage was compared to the gate regulatory schedule heights to see if the gate operation should be closed, static, or open. If canal stage was lower than the closed condition height, surface runoff-out of the cell was set to zero. If canal stage was in the static range (higher than the close height but lower than the open height), the cell elevation gradient was determined as the cell water elevation minus the local gate height. If canal stage was above the open condition height, the elevation gradient was calculated as the cell water elevation minus the gate height in the next downstream canal zone.

To determine groundwater elevation gradients, a GIS function used the daily water table grid to create sub-surface "watersheds". The watershed zones represented groundwater drainage basins, just as occurs on the surface. Next, the minimum water table values for each watershed zone were calculated using a GIS zonal statistic function. The minimum values were then subtracted from the water table grid to generate a grid of groundwater elevation gradients. The elevation gradients were then multiplied by a normalized soil permeability grid and by a normalized soil water holding capacity grid, resulting in a fraction of soil water storage available for groundwater seepage-out.

In areas where the soils were completely saturated, direct lateral recharge of groundwater could occur from surface water storage. The fraction of available soil water storage value was multiplied by a calibration coefficient to generate a fraction of surface

water storage available for lateral groundwater recharge. Since the phenomenon is principally associated with canals, and the magnitude of surface water storage is much larger in canals than in the rest of the study area, the surface fraction available was reduced by two orders of magnitude away from the canals. The equations describing horizontal surface runoff-out and groundwater seepage-out were:

$$X_{out} = \text{water management structures, } f(k_1 * Q_{surface}) \quad (15)$$

$$= \text{at most locations, } k_2 * \lambda_{surface} * Q_{surface} \quad (16)$$

$$S_{out} = k_3 * \lambda_{soil} * Q_{soil} \quad (17)$$

$$S_{lateral} = (Q_{soil} > Q_{potentialsoilmax}) = (\lambda_{surfacesoilrecharge} / k_4) * Q_{surface}, \quad (18)$$

$$(Q_{soil} \leq Q_{potentialsoilmax}) = 0 \quad (19)$$

where,

$\lambda_{surface}$ = fraction of surface water storage available for surface runoff-out

(normalized elevation gradient)

$f(k_2 * Q_{surface})$ = function dependent on water management regulatory schedule

λ_{soil} = fraction of soil water storage available for groundwater seepage-out

$\lambda_{surfacesoilrecharge}$ = fraction of surface water storage available for lateral groundwater recharge

where,

$$\lambda_{soil} = \theta_{soil} * \xi_{soilpermeability} * \xi_{soilwatercapacity} \quad (20)$$

$$\lambda_{surface\text{soilrecharge}} = \lambda_{soil} * Q_{surface} \quad (21)$$

Furthermore,

θ_{soil} = normalized groundwater slope

$\xi_{soil\text{permeability}}$ = normalized soil permeability

$\xi_{soil\text{watercapacity}}$ = normalized soil holding capacity

Surface runoff-in values were obtained by draining the surface runoff-out grid across a surface flow direction grid. To generate the daily flow direction grid, the daily surface water storage was added to the DEM. A GIS function then calculated the new flow directions. Since draining records all water that travels *through* a cell, rather than the water that actually *ends* up in a cell, an error, illustrated by Figure 15, would have been induced in the mass-balance if the drain result was not corrected. To simulate the physical landscape effects upon overland flow, and to correct the drain-induced mass-balance error shown in Figure 15, a two step process was used to adjust the raw drain result. First, an impedance map was created, where higher impedance values represented increased resistance to horizontal flow. To create the surface impedance grid, a travel cost grid was determined from the daily surface water elevation gradient (surface slope) grid. The surface slope travel cost grid was then multiplied by the impervious surface travel cost grid to generate a net travel cost grid. The travel cost grid was then drained across the daily flow direction grid to create the impedance grid. The drained surface runoff-out grid was then divided by the impedance grid to generate preliminary surface runoff-in.

Since impedance was only a relative index of resistance, a second drain adjustment was necessary to maintain mass-balance. The second drain adjustment used the quantity of actual runoff (surface runoff-out) to correct the relative runoff volume (impedance-adjusted surface runoff-in) by determining the ratio between the two (an index of the mass-balance error). Unique ratios were calculated within each distinct surface watershed throughout the study area. Surface watershed zones were delineated using ArcView functions and the surface water elevation grid as input. Within each watershed zone, the surface runoff-out and the impedance-adjusted surface runoff-in grids were spatially summed. The spatial sums were then divided, resulting in the ratio of the mass-balance error within each watershed zone. The ratio was then multiplied by the impedance-adjusted surface runoff-in grid to generate the final, corrected surface runoff-in grid. The process is clarified in Figure 21.

The groundwater seepage-out grid and the surface water lateral groundwater recharge grids were added together. The result of this addition was then drained across a groundwater flow direction grid (derived from the daily groundwater elevation grid) to generate values for groundwater seepage-in. The drain correction process described previously was then employed to modify the groundwater seepage-in grid, except using groundwater parameters. Travel cost grids were created based on daily groundwater slope, soil water capacity, and soil permeability. The sub-surface impedance grid was drained across the groundwater elevation grid. Sub-surface watershed zones were used for the mass-balance error ratio calculations.

The equations describing horizontal surface runoff-in and groundwater seepage-in were:

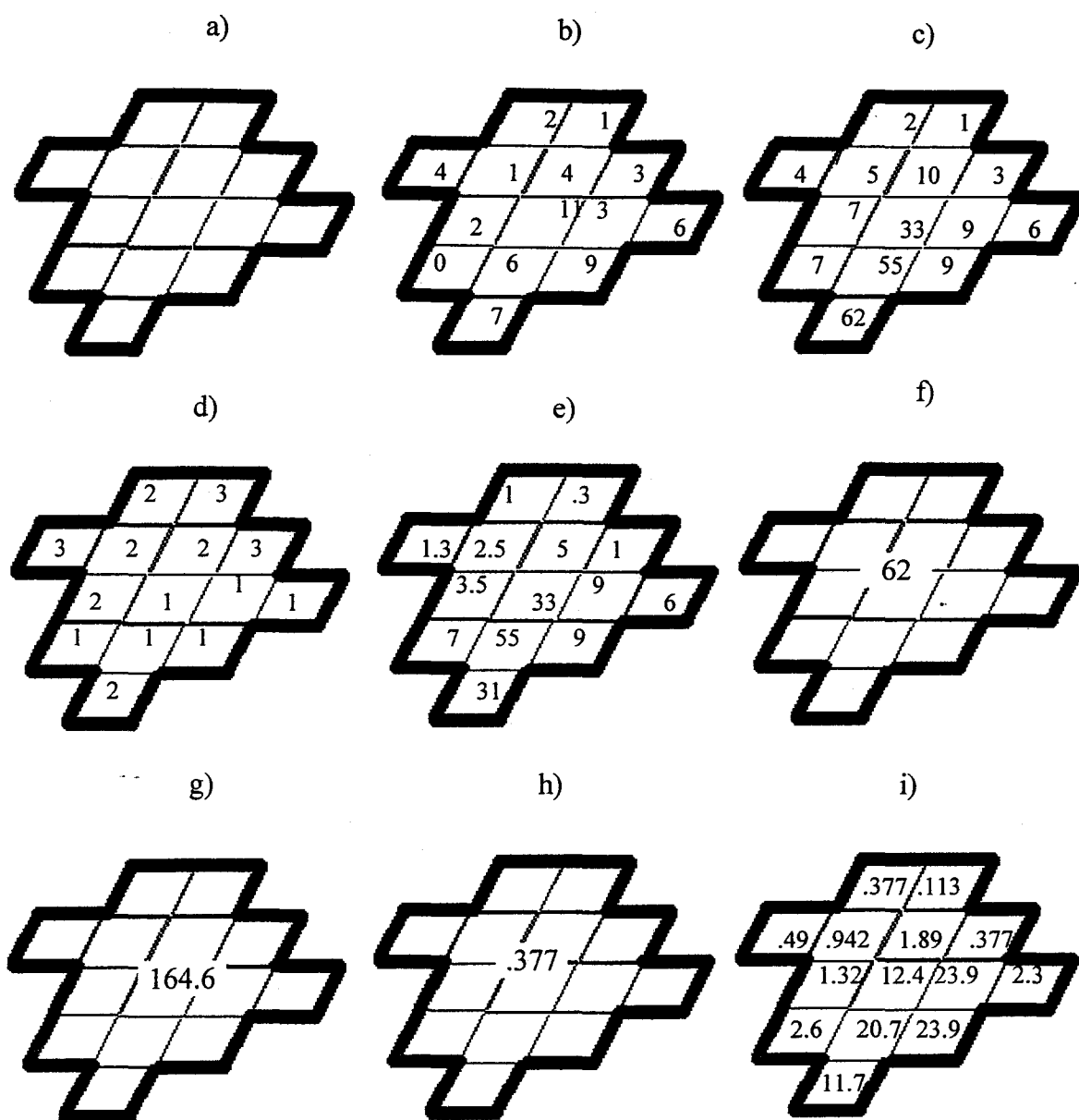


Figure 21. The drain adjustment process, illustrated for a single watershed zone in the entire analysis area. A watershed grid, a), is derived from the surface elevation grid. The daily runoff grid, b), is drained c), and divided by an impedance grid, d), to generate an impedance-adjusted drained runoff in, e). The second part of the process involves spatially summing daily runoff, f), and impedance-adjusted runoff in, g) within each watershed zone. A ratio is generated for each watershed zone, h), which is then multiplied by the impedance-adjusted runoff in grid for the final daily runoff in result, i).

$$X_{in} \cong f(X_{out}) / f(\beta_{surfacewater}) \quad (22)$$

$$S_{in} \cong f(S_{out} + S_{lateral}) / f(\beta_{soilwater}) \quad (23)$$

where,

$f(X_{out})$ = drained surface runoff-out as handled by GIS software

$f(\beta_{surfacewater})$ = drained surface flow impedance, based on surface elevation gradient and impervious surface travel costs

$f(S_{out} + S_{lateral})$ = drained groundwater seepage-out and surface lateral groundwater recharge as handled by GIS software

$f(\beta_{soilwater})$ = drained sub-surface flow impedance, based on groundwater elevation gradient, soil water holding capacity, and soil permeability travel costs

“ $\cong$ ” = operand signifying second drain-adjustment process

Furthermore,

$$\beta_{surfacewater} = k_5 / ((k_6 * \theta_{surface}) * (k_7 * \Gamma)) \quad (24)$$

$$\beta_{soilwater} = k_8 / ((k_9 * \theta_{soil}) * (k_{10} * \xi_{soilpermeability}) * (k_{11} * \xi_{soilwatercapacity})) \quad (25)$$

where,

$\theta_{surface}$ = normalized surface/surface water slope

Γ = normalized fraction impervious surface

Vertical movement was simulated by infiltration and groundwater overflow processes. The equations representing these processes were as follows:

$$I = (k_{12} * Q_{surface} * \xi_{soilpermeability}) / \Gamma \quad (26)$$

$$N = (Q_{soil} > Q_{potentialsoilmax}) = Q_{soil} - Q_{potentialsoilmax} \quad (27)$$

$$(Q_{soil} \leq Q_{potentialsoilmax}) = 0 \quad (28)$$

where,

I = infiltration to soil water

N = groundwater overflow into surface water

Aquifer recharge was calculated as a linear first-order function of soil storage.

The following equation represents aquifer recharge:

$$R = k_{13} * Q_{soil} \quad (29)$$

Ocean losses were derived by setting surface water storage to a constant value.

The value (120 inches) was set to result in a surface water elevation of 0 feet above MSL wherever Biscayne Bay occurs.

Figure 22 demonstrates the grid manipulations used to simulate the phosphorus flows from Figure 18. Phosphorus load-in and load-out grids were calculated. Pathways for TP fates via sequestration by calcium carbonate, physical sedimentation of particulate phosphorus, and long-term plant uptake were then simulated.

Phosphorus load was determined separately for cells that began the day with or without TP storage. For cells with TP storage at the beginning of the time step, TP

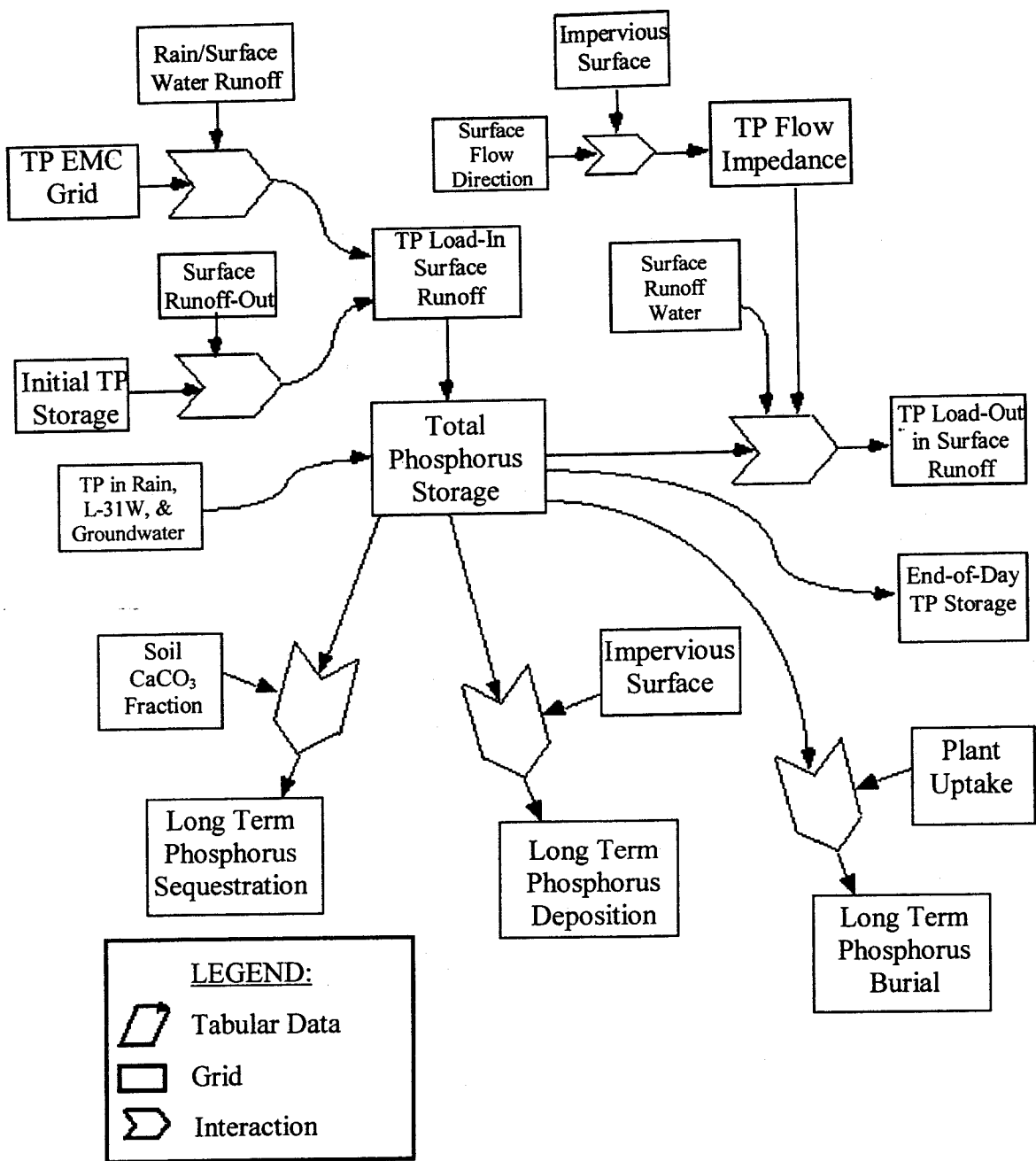


Figure 22. GIS spatial model of Total Phosphorus dynamics. Flows represent the TP processes on the system diagram in Figure 18.

storage was divided by water storage to obtain the cell TP concentration. The concentration was then multiplied by the surface runoff-out for TP load-out. For cells with no initial TP storage, daily surface water runoff-out grids were multiplied by the TP EMC grid. The two phosphorus load grids were merged to yield an area-wide potential TP load grid. Phosphorus load-out was drained across the daily flow direction grid, and the drain correction for phosphorus was handled by the same method used for surface water runoff, but with the TP appropriate parameters, resulting in a phosphorus load-in grid.

The general equations used to simulate TP mass-balance and lateral transport were:

$$Q_{TPend} = Q_{TPinitial} + \Delta Q_{TP} / \Delta T \quad (30)$$

where,

$Q_{TPinitial}$ = TP water storage at beginning of time step

Q_{TPend} = TP water storage at end of time step

$\Delta Q_{TP} / \Delta T$ = change in TP water storage per time step

Furthermore,

$$\Delta Q_{TP} / \Delta T = X_{TPin} + DL_{TP} + R_{TP} + GW_{TP} - X_{TPout} - Y_{TP} - G - U \quad (31)$$

where,

X_{TPin} = TP load-in

DL_{TP} = TP in water delivered from the L-31 W and Florida City canals

R_{TP} = TP in rain

GW_{TP} = TP in groundwater inflow

X_{TPout} = TP load-out

Y_{TP} = sedimentation of TP

G = chemical reaction of calcium carbonate with TP

U = long-term plant uptake of TP

Furthermore,

$$X_{TPout} = (Q_{TP} > 0) = k_{14} * (Q_{TP} / Q_{surface}) * X_{out}, \quad (32)$$

$$(Q_{TP} = 0) = \vartheta_{TP} * X_{out}$$

$$X_{TPin} \cong f(X_{TPout}) / f(\beta_{TP})$$

where,

ϑ_{TP} = TP EMC

$f(X_{TPout})$ = drained TP load-out as handled by GIS software

$f(\beta_{TP})$ = drained TP impedance, based on impervious surface travel cost grid

Furthermore,

$$\beta_{TP} = k_{15} / ((k_6 * \theta_{surface}) * (k_7 * \Gamma_{TP})) \quad (33)$$

where,

Γ_{TP} = fraction impervious surface normalized for TP

Loads of TP from the L-31W and Florida City canals, delivered water, rain, and groundwater inflows to the canals were determined by multiplying these water flows by concentrations derived from reference values. TP concentration in rain was taken from Ahn, 1997 as .0015 mg/L. The TP concentration of groundwater was taken to be .038 mg/L based on south Florida concentrations found in Herr & Shaw, 1989. TP concentration in water L-31W was taken as .03 mg/L based on values observed in the L-31W canal found in the STORET database.

Sequestration of TP by calcium carbonate was handled by multiplying the storage grid by a carbonate reaction factor. The carbonate reaction factor was obtained by combining the fraction calcium carbonate of surface soil by a rate determined for south Florida carbonate-rich soils (Gilliland, 1973). The reaction was mathematically represented by:

$$G = \chi * Q_{TP} \quad (34)$$

where,

χ = rate of carbonate/TP chemistry, dependent on soil carbonate %

Physical sedimentation was simulated by multiplying the TP storage by the inverse of the fraction of impervious surface in a cell. The inverse is used because physical resistance to horizontal movement across a landscape is lowest when impervious surface is highest, and vice versa. Horizontal velocity decreases the tendency of particles

to drop out of suspension. Therefore, sedimentation will be less likely to occur in areas of high impervious surface. Phosphorus sedimentation occurred as:

$$Y_{TP} = k_{16} * (1 - \lambda_{TP}) * Q_{TP} \quad (35)$$

Plant uptake of phosphorus was assumed to occur in wetland and upland forest systems. For purposes of this model, plant uptake here was defined as the amount of TP taken up by plants that eventually becomes deposited in long-term soil storage.

Consequently, the rate was lower than may be observed elsewhere, because short-term phosphorus cycling was ignored. A plant uptake rate of .24 mg TP/m²/day was used based on work done in southeast Florida wetlands (Kendall, 1996).

End-of-day TP storage was derived by adding and subtracting the phosphorus flows as appropriate from the initial TP storage. TP storage was set to zero where no end-of-day surface water storage existed. It was also reset to zero where Biscayne Bay occurs, to simulate TP dilution into the rest of the Atlantic Ocean.

The method for simulating Total Suspended Solids (TSS) was similar to the method used for simulating phosphorus. Figure 23 demonstrates the grid manipulations used to simulate daily suspended solids flows and storages. Load grids were calculated in the same manner, except using TSS instead of TP for EMC input parameters. In addition, TSS equations had calibration constants distinct from the TP equation calibration coefficients. Delivery of TSS from the L-31W and Florida City canals, and TSS sedimentation were simulated in the same manner as for TP, although TSS in rain, TSS from groundwater inflow, plant

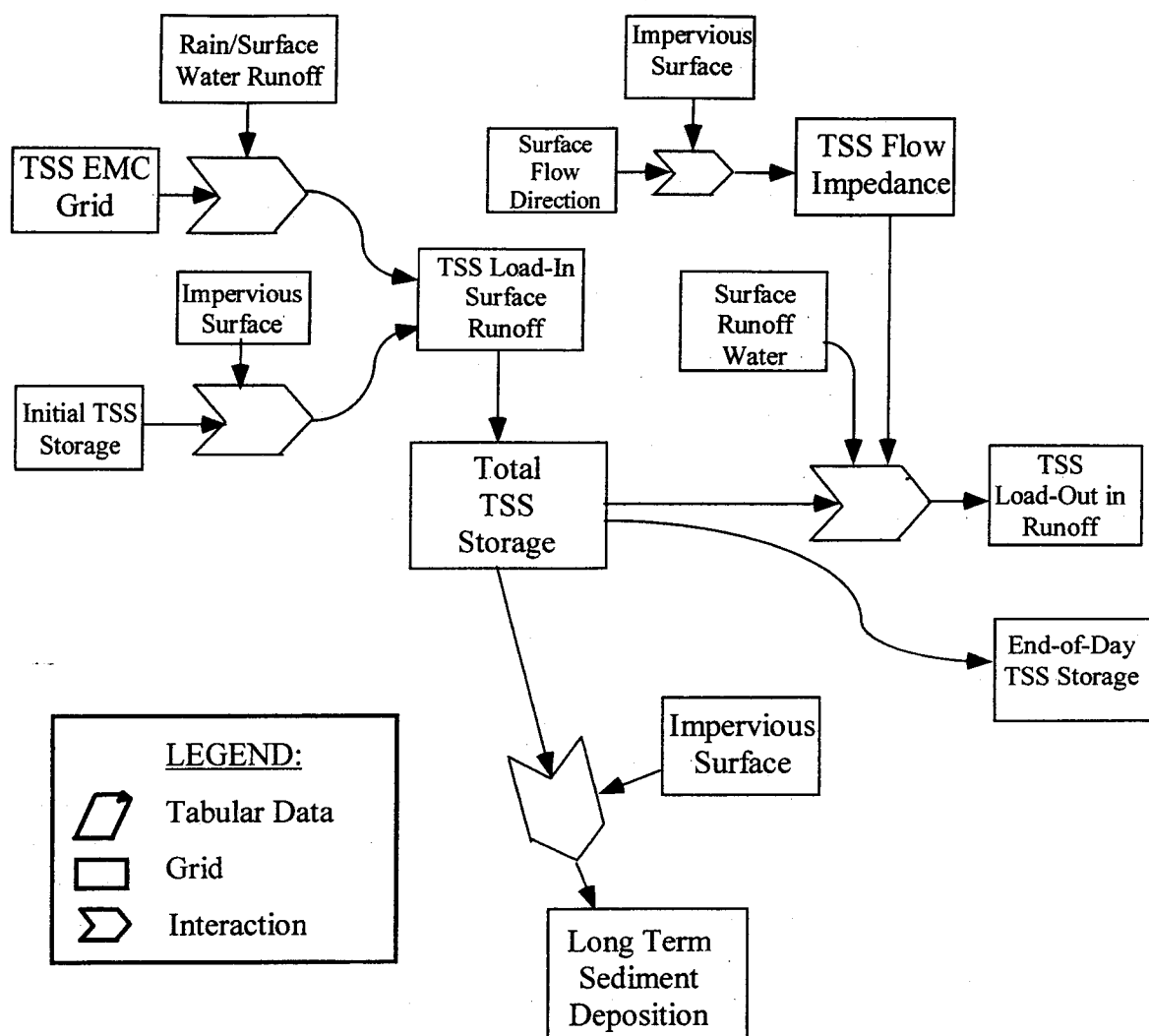


Figure 23. GIS spatial model of Total Suspended Solids dynamics. Flows represent the TSS processes on the system diagram in Figure 18.

uptake and carbonate chemistry algorithms were omitted. The general equations describing TSS processes were:

$$Q_{TSSend} = Q_{TSSinitial} + \Delta Q_{TSS} / \Delta T \quad (36)$$

where,

$Q_{TSSinitial}$ = TSS water storage at beginning of time step

Q_{TSSend} = TSS water storage at end of time step

$\Delta Q_{TSS} / \Delta T$ = change in TSS water storage per time step

Furthermore,

$$\Delta Q_{TSS} / \Delta T = X_{TSSin} - X_{TSSout} - Y_{TSS} \quad (37)$$

where,

X_{TSSin} = TSS load-in

DL_{TSS} = TSS in water delivered from the L-31W and Florida City canals

X_{TSSout} = TSS load-out

Y_{TSS} = sedimentation of TSS

Furthermore,

$$X_{TSSout} = (Q_{TSS} > 0) = k_{17} * (Q_{TSS} / Q_{surface}) * X_{out} \quad (38)$$

$$(Q_{TSS} = 0) = g_{TSS} * X_{out} \quad (39)$$

$$X_{TSSin} \cong f(X_{TSSout}) / f(\beta_{TSS}) \quad (40)$$

$$Y_{TSS} = k_{18} * (1 - \lambda_{TSS}) * Q_{TSS} \quad (41)$$

where,

Q_{TSS} = TSS EMC

$f(X_{TSSout})$ = drained TSS load-out as handled by GIS software

$f(\beta_{TSS})$ = drained TSS impedance, based on impervious surface travel cost grid

Furthermore,

$$\beta_{TSS} = k_{19} / ((k_6 * \theta_{surface}) * (k_7 * \Gamma_{TSS})) \quad (42)$$

where,

Γ_{TSS} = fraction impervious surface normalized for TSS

The preceding equations contained the coefficients that were adjusted during calibration. In addition to threshold data values, the equation coefficients that were calibrated included the following:

Threshold Values for Model Parameters

- Threshold surface elevation gradient
- Threshold groundwater slope
- Threshold impervious surface
- Threshold soil permeability
- Threshold soil water capacity
- Threshold size for surface watershed calculation

-Threshold size for sub-surface watershed caluclation

Water Quantity Calibration Coefficients

K1: Fraction of flow over water management gate

K2: Coefficient of surface water elevation gradient

K3: Coefficient of groundwater slope

K4: Resistance to surface water lateral groundwater recharge

K5: Weighting of surface flow impedance grid

K6: Influence of surface water elevation travel costs on surface impedance grid

K7: Influence of impervious surface travel costs on surface impedance grid

K8: Weighting of groundwater flow impedance grid

K9: Influence of surface water elevation travel costs on surface impedance grid

K9: Influence of groundwater slope travel costs on sub-surface impedance grid

K10: Influence of soil permeability travel costs on sub-surface impedance grid

K11: Influence of soil water capacity travel costs on sub-surface impedance grid

K12: Coefficient of surface infiltration rate

K13: Coefficient of surficial aquifer recharge

Water Quality Calibration Coefficients

K14: Impervious surface influence on TP load-out

K15: Coefficient of TP sedimentation during overland flow

K16: TP sedimentation rate adjustment

K17: Impervious surface influence on TSS load-out

K18: Coefficient of TSS sedimentation during overland flow

K19: TSS sedimentation rate adjustment

Calibration & Validation

Model calibration was the process of improving model accuracy by adjusting simulation operations. The process involved running the simulation model and then comparing output with expected results. The comparison suggested which operations needed adjustment. To adjust specific operations, calibration coefficients were included in the mathematical equations. These coefficients were tuned to correct the magnitude of inaccurate flows. The process was repeated until results were considered satisfactory.

The rainfall year used for calibration was 1992, which was considered an average rainfall year by the SFWMD. The calibration study area was the C-102 sub-basin.

Modeled flow and storage data were output as both study area grids for any simulation date(s), and as individual cell values written into database tables for the entire simulation period. The simulation period cell values were graphed to generate time-histories of flows and storages at specific points within the study area where actual data were available. Scripts were written to generate a user-interface for the model, which facilitated output data formatting and the setting of calibration coefficients.

The expected results used to calibrate the model included qualitative and quantitative measures. Qualitative assessments were used to coarsely calibrate and debug the model. The overall strategy was to generally replicate the study site hydrology described previously and to eliminate nonexistent phenomenon ranging from water levels that oscillated wildly to the manifestation of large, ephemeral geysers. To characterize spatial surface hydrology, a grid of hydroperiod index was created, where the index was

the percent of simulation time a cell was inundated. The hydroperiod index was calculated by tallying the number of days simulated surface water storage occurred in all cells, then divided by the simulation length in days. The hydroperiod index grid was then used to evaluate the spatial distribution of surface waters. Prolonged ponding of surface water in urban areas was minimized. The seasonal pattern of water tables seen in Figures 11 and 12 was used as a guide to calibrate groundwater.

As simulation accuracy improved, quantitative calibration methods were employed, which included comparisons between simulated results and observed data. Observed data for canal stages, flows, and groundwater elevations were extracted and compiled from the SFWMD DB/HYDRO database. Observed canal stage and flow data were compiled for stations in the C-102 sub-basin at S-194, S-165, S-21A, and groundwater elevation data at USGS well G-757A. Observed canal stage and flow data were compiled for stations in the C-103 sub-basin at S-196, S-179, and S-20F. Observed data for total phosphorus (TP) concentration and total suspended solids (TSS) concentration between 1985 and 1996 were compiled from the USEPA STORET water quality database. Total phosphorus concentration data were compiled for 17 surface water quality stations located in C-102 and C-103 sub-basins. TSS data were compiled for one station located near S-194.

The compiled observed data were graphed with simulation output taken at grid points that corresponded to the observation stations. Graphs were of canal stage, groundwater elevation, and constituent concentrations in the cell water storage. Simulated canal stages and groundwater elevations were actual simulation output data. Constituent concentrations were determined by dividing constituent load by water

quantity. Comparisons between observed data and simulation output were then conducted. The standard deviations between observed and simulated values for the dry and wet seasons were noted. The number of days simulated and observed values were within .25 meters was determined. To assess whether the simulations were generally leaving the basins too wet or too dry, the number of days simulated values were higher and lower than observed values was recorded for the wet and dry seasons. Based on these comparisons, qualitative assessments, and the spatial patterns of where errors were greatest, the coefficients were modified. Calibration continued until the comparisons indicated simulation output were not getting any closer to the target observed data.

In addition to the approach of matching simulation output to observed values at individual cells, a larger scale assessment of model results was conducted to aid calibration. Estimated actual (taken from Alleman, 1995) and simulated total annual constituent discharges to Biscayne Bay were calculated and tabulated. To calculate the simulated annual Bay discharges, a GIS function was developed to tally the flow or storage totals at the end of each day within specified grid cells for the entire simulation period (hereafter referred to as "annually summed"), an example of which is provided as Figure 24. The total discharge was then determined by adding up the values for each individual grid cell throughout Biscayne Bay. The annual summing technique was applied to the watershed grids for each constituent pathway as well. The annual simulated watershed pathway flows and total annual discharges into the ocean were then collectively tabulated. Comparisons of estimated actual and simulated total discharges to the Bay revealed how accurately the model simulated flows at the watershed scale, and was used to adjust coefficients appropriately.

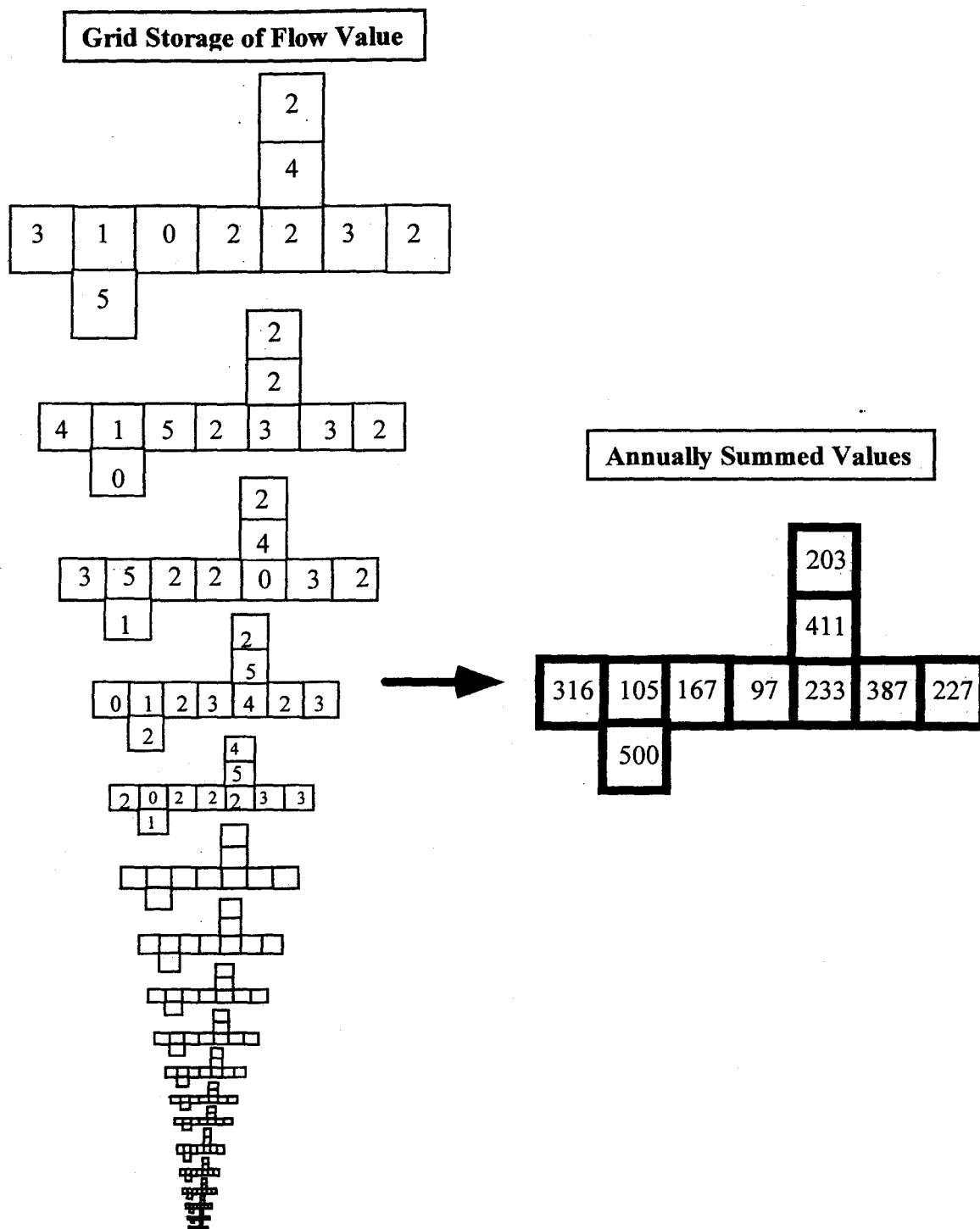


Figure 24. Illustration of annual summing. The simulation was run, generating values in the process flow or storage grids for each simulation date, as seen at left. The values were tallied up and included in a grid, seen at right.

Time-histories were compiled of daily total watershed constituent ocean discharges. These time-histories of loads of TP and TSS discharged into Biscayne Bay were then graphed.

Validation tested the model by using a new set of input conditions to see if it was an appropriate (valid) tool for simulating situations other than the calibration scenario. For this analysis, two types of validation were performed. First, the study area was changed to the C-103 sub-basin. The C-103 sub-basin is a more urbanized sub-basin than the C-102 sub-basin. A comparison of model accuracy between the sub-basins tested validity for different urbanization scenarios. Second, the rainfall year was changed to the dry (1989) and wet (1995) conditions. The two alternate rainfall year simulations assessed the validity of the model for conditions other than those of the 1992 rainfall year.

The comparisons conducted during calibration between actual data and simulated results were again used to determine model validity. Observed data and simulation results from specific points in the C-103 sub-basin were graphed and compared to each other. The discharge of constituents from the C-103 sub-basin into Biscayne Bay were calculated, tabulated, and compared to observed data.

The same comparative evaluations used during calibration were conducted for each rainfall year validation test. For the alternate rainfall year simulations, single-point simulation output was graphed against observed 1989 and 1995 C-102 and C-103 canal stages, groundwater elevations, and constituent concentrations. Time-histories of daily simulated constituent discharges into Biscayne Bay were graphed. The annual watershed

discharges of constituents into Biscayne Bay were calculated and compared to estimated values.

To further study the influence of rainfall on simulation output, additional assessments were conducted. Time-histories of cumulative rainfall at S-21A and S-337 were compiled and graphed. Simulated canal stage values at S-21 from all three rainfall conditions were graphed. The hydroperiod index grids for the dry and wet rainfall conditions were calculated and subtracted from the 1992 rainfall condition hydroperiod index grid.

The comparative effects of rainfall input on ground water volumes were evaluated next. To show temporal response to rainfall, simulated total watershed ground water storage for each day of the simulation year was compiled into a time-history. The time-histories from each rainfall condition were then graphed collectively. To show spatial effects of rainfall, groundwater table grids from the last date of the simulation year for the low, average, and high rainfall conditions were displayed. The low and high rainfall condition end-of-year groundwater elevation grids were then subtracted from the final date average rainfall groundwater elevation grid and displayed.

Time-histories of the simulated total cumulative watershed TP and TSS discharge into Biscayne Bay were compiled and graphed collectively.

Sensitivity Analysis

During the course of the model development, calibration, and validation tasks, certain model processes seemed to influence simulation results significantly. To quantify the importance of these processes to simulation results, a sensitivity analysis was conducted. Three processes were analyzed using the 1992 rainfall year: one for surface

water quantity, one for groundwater quantity, and one for each water quality constituent. The calibration coefficient for the equation simulating each process was first increased by 50%, and then decreased by 50%.

For the surface water process sensitivity analysis, simulated canal stage at S-194, S-21A, S-196, and S-20F time-histories for all three calibration coefficient settings were graphed. For each coefficient setting, a time-history of simulated total watershed surface water storage and hydrologic discharge into Biscayne Bay from each simulation date was compiled. The time-histories for surface water storage and oceanic discharge from all three settings were graphed to show temporal responses to coefficient settings. For each setting, simulated average daily watershed storages of surface water, ground water, and constituents were calculated and tabulated. Annual hydrologic and constituent discharged into Biscayne Bay were calculated from each coefficient setting simulation, and included in a table. The percentage changes in the average daily watershed storages and discharges of water and constituents from the calibration simulation were calculated and tabulated.

For the ground water process sensitivity analysis, time-histories of groundwater elevation at G-757 for each of the three coefficient settings were graphed. For each coefficient setting, a time-history of simulated total watershed ground water storage from each simulation date was compiled. The time-histories from all three settings were then graphed. For each setting, simulated average daily watershed storages of surface water, ground water, and constituents were calculated and tabulated. Annual hydrologic and constituent discharged into Biscayne Bay were calculated from each coefficient setting simulation, and included in a table. The percentage changes in the average daily

watershed storages and discharges of water and constituents from the calibration simulation were calculated and tabulated.

For the water quality sensitivity analysis, the time-histories of constituent concentrations at S-194, S-21A, S-196, and S-20F for each of the three coefficient settings were graphed. For each setting, simulated average daily watershed storages of surface water, ground water, and constituents were calculated and tabulated. Annual hydrologic and constituent discharged into Biscayne Bay were calculated from each coefficient setting simulation, and included in a table. The percentage changes in the average daily watershed storages and discharges of water and constituents from the calibration simulation were calculated and tabulated.

Simulation & Evaluation of Land Use Scenarios

The model was used to simulate different land use scenarios. The study area was simulated as it was just before human development (the "historic" condition), as it was in 1995 (the "current" condition), and how it would be after a 20% increase from the 1995 population. Since the forecast is for 20% growth from 1995 to 2015, this will be referred to as the 2015 scenario. The historic scenario was created by backfilling the canals and eliminating elevated road surfaces in the DEM, converting the land use into the pre-development land cover as shown in Figure 9, recreating grids with land use-specific attributes, and turning off the water management operations in the model. To mimic the pre-development land cover in Figure 9, all elevations greater than 9.5 feet above MSL were classified as upland coniferous forest (the coastal sand ridge), areas along the coast currently classified as wetlands were classified as forested wetlands (mangroves), and remaining lands were classified as herbaceous wetlands (rocky glades, transverse glades,

and coastal marsh). Since the canal adjacent to the western study area boundary did not exist historically, hydrologic connection to the main Everglades system maintained higher water tables in the western part of the study area. To simulate this effect for the historic land use scenario, the water tables along the western boundary were reset to 7 feet above MSL during the dry season, and 9 feet above MSL during the wet season.

The 2015 land use distribution was created by using GIS to buffer urban/residential polygons of the land use coverage, excluding those lands classified as urban recreational areas. The GIS buffer function is illustrated as Figure 25. The buffer was sized to increase the percentage of urban/residential lands by 20%. The newly urbanizing lands buffer was excluded from areas classified as either transportation or canals, since roads are not intentionally destroyed during urbanization. Grids with land use-specific attributes were then recreated using the new land use data.

Spatial Arrangements of Wetland Stormwater Treatment Areas (WSTAs)

To test theoretical effects of the spatial arrangement of WSTA networks, the 2015 scenario was modified to include different configurations of stormwater wetlands. The spatial distributions included WSTAs located: (1) randomly, (2) coincident with high constituent source/sink potential, (3) in environments of favorable hydrology, (4) according to distance from watershed discharge, and (5) with arranged size class distributions.

The 2015 scenario land use grid was altered for each WSTA network configuration, keeping total wetland area to 10% of the watershed area. All WSTA networks were built around any remaining wetlands shown in Figure 12. The percentage and location of urban/residential and transportation lands was kept constant, to

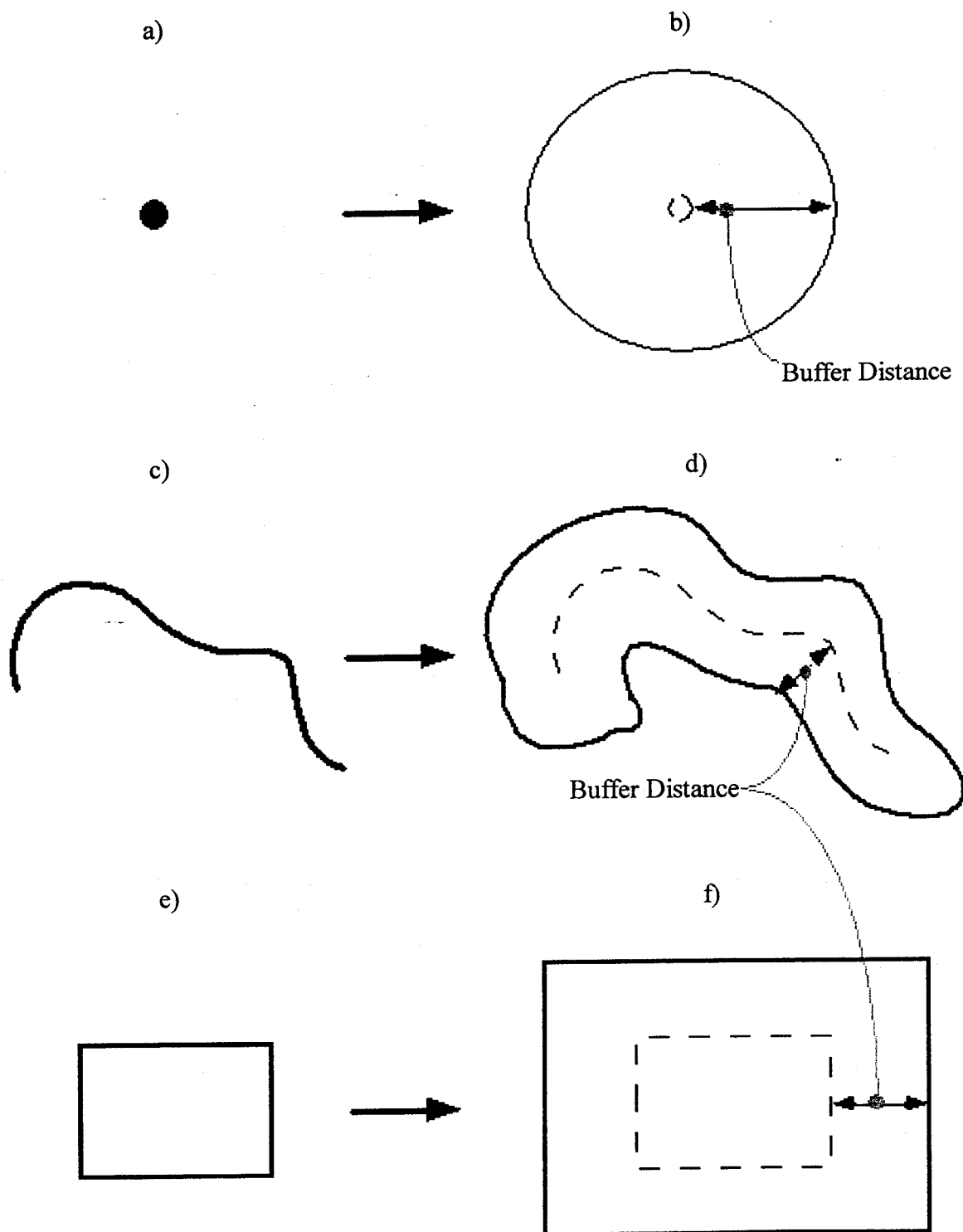


Figure 25. Illustration of GIS buffer operations. A buffer of a specified distance can be applied to a point coverage such as a) to generate buffer polygon b), a line coverage such as c) to generate buffer polygon d), or a polygon coverage such as e) to generate buffer polygon f).

eliminate changes in urban land area affecting network performance. First, a grid of the test WSTA network was created. The WSTA grid was then overlaid on the land use grid, and landuses were reclassified as a wetland WSTA (herbaceous wetland). When a wetland occupied a location that was originally a canal, the DEM values were increased by nine and a half feet. Grids with land use-specific attributes were then recreated using the new land use data. Techniques for creating each of the spatial arrangements of WSTAs are described next.

Random WSTA Networks

Two random network scenarios were constructed. X and y coordinates of wetlands were derived from a random number generator, and used to create points in a point coverage. The point coverage was then buffered 600 meters, and then converted into the analysis grid format. Points were added until 10% of the study area was covered in wetlands.

High Source/Sink Potential WSTAs

The creation of the WSTA network located in areas of high source/sink potential was a two-step process. First, a configuration was generated where WSTAs were sited in close proximity to large sources of constituent loads. WSTAs were located in cells with the highest values from the 2015 scenario annually summed constituent load inflow grids, excluding cells classified as urban/residential or transportation. Second, a configuration was created where WSTAs were located in potential constituent sink areas. Since limestone soils are conducive for chemical uptake of phosphorus, cells with the highest soil calcium carbonate percentages were incorporated into the network grid. Canals were excluded from the WSTA network.

WSTAs in Areas of Favorable Hydrology

The WSTA network within areas of favorable hydrology was based on the hydroperiod index grid output from the 2015 scenario. Cells with the highest hydroperiod index were included until the wetland fraction of the landscape was 10%. The water management canals were excluded from the network.

WSTA Distance From Watershed Discharge

Two networks were created where the design variable was distance to watershed discharge. The average distance to Biscayne Bay was 1000 meters for the first network, and 10000 meters for the second network. For each configuration, a line coverage was created where a line was positioned at the average distance and oriented parallel to the shore of Biscayne Bay. The line coverage was buffered such that wetlands occupied 10% of the landscape, and the buffer result was converted into the analysis grid format.

Size Class Distribution of WSTAs

Two size class distributions of WSTAs were created by locating many small wetlands randomly throughout the study area, medium size wetlands randomly along the canals, and large wetlands at the discharges of C-102 and C-103. The two distributions created were 70-20-10 and 5-40-55 of small, medium, and large wetlands, respectively. The size class distributions were calculated using all watershed wetlands, including both WSTA wetlands and existing wetlands, and WSTAs were added to generate the final distributions. A point coverage of wetland center points was created. Small wetland x,y coordinates were derived from a random number generator. Small wetland points were buffered 76 meters (the smallest cell size), medium wetland points were buffered 300 meters, and the large wetland points were buffered 900 meters. The first size class

distribution (with 10% as large wetlands) did not require any additional large WSTAs, since the existing coastal wetlands satisfied the necessary large wetland area. The second size class distribution small wetlands were created by eliminating small wetlands from the first distribution network, and shrinking the small wetland sizes to a standard width of one cell.

Land Use Scenario Analyses

Several analyses were applied to the results from each of the land use scenario simulations. The simulated total watershed flows of constituent pathways were calculated, and then tabulated along with the calculated total annual watershed flows of water and constituents discharged into Biscayne Bay. In addition, the hydroperiod index grid from the historic land use scenario simulation was calculated and displayed.

To support further evaluation of scenario output, results from multiple scenarios were compared as part of two collections. The historic, 1995, and 2015 scenario results were grouped into one collection, and the WSTA results were grouped into a second collection. For the first collection, annual watershed hydrologic and constituent flows were determined and tabulated together. For added comparison, the percentage changes in annual flows to the simulated 1995 condition results were calculated and included in the table.

For the WSTA network scenario collection, the percentage change of annual watershed constituent pathway flows and discharges to Biscayne Bay to the results from the 2015 land use scenario were calculated and included in the aforementioned flow tables. For each modeled pathway flow, the WSTA networks were then ranked from

greatest to least in magnitude, and the ranked networks were displayed in tables for each pathway.

RESULTS

The following section includes the results of the methods described previously. First, results of model development steps are shown, followed by results from the land use scenario simulations. Results are shown as graphs and tables. For all cases, wet season is considered to be between simulation dates 151 and 330 (180 days), and the dry season is considered to be the remainder of the year (185 days).

Calibration & Validation Results

Calibration

Figures 26-28 show the simulated and observed time-histories of canal stage at three locations in the C-102 sub-basin, specifically canal stages at S-194, S-165, and S-21A. Figure 26 shows the time-histories at S-194. Distinct seasonal patterns emerged from comparisons between dry and wet season simulation results. Simulated model canal stage values were mostly (147 days) below observed values during the dry season, but usually (119 days) higher during the wet season. During the dry season, the difference between simulated and observed S-194 stage values was within .25 meters on 181 days, and the standard deviation was .11 meters. During the wet season, the differences in stage values increased. Simulated and observed values remained within .25 meters of one another for 125 days, and the standard deviation increased to .21 meters.

Figure 27 includes the time-histories of simulated and observed canal stages at S-165. A similar pattern to that seen at S-194 emerged as the difference in stage values also increased from the dry season to the wet season. The standard deviation between simulated and observed stage values at S-165 increased from .07 to .30 meters from the dry to the wet season. Simulated and observed stage values were within .25 meters for

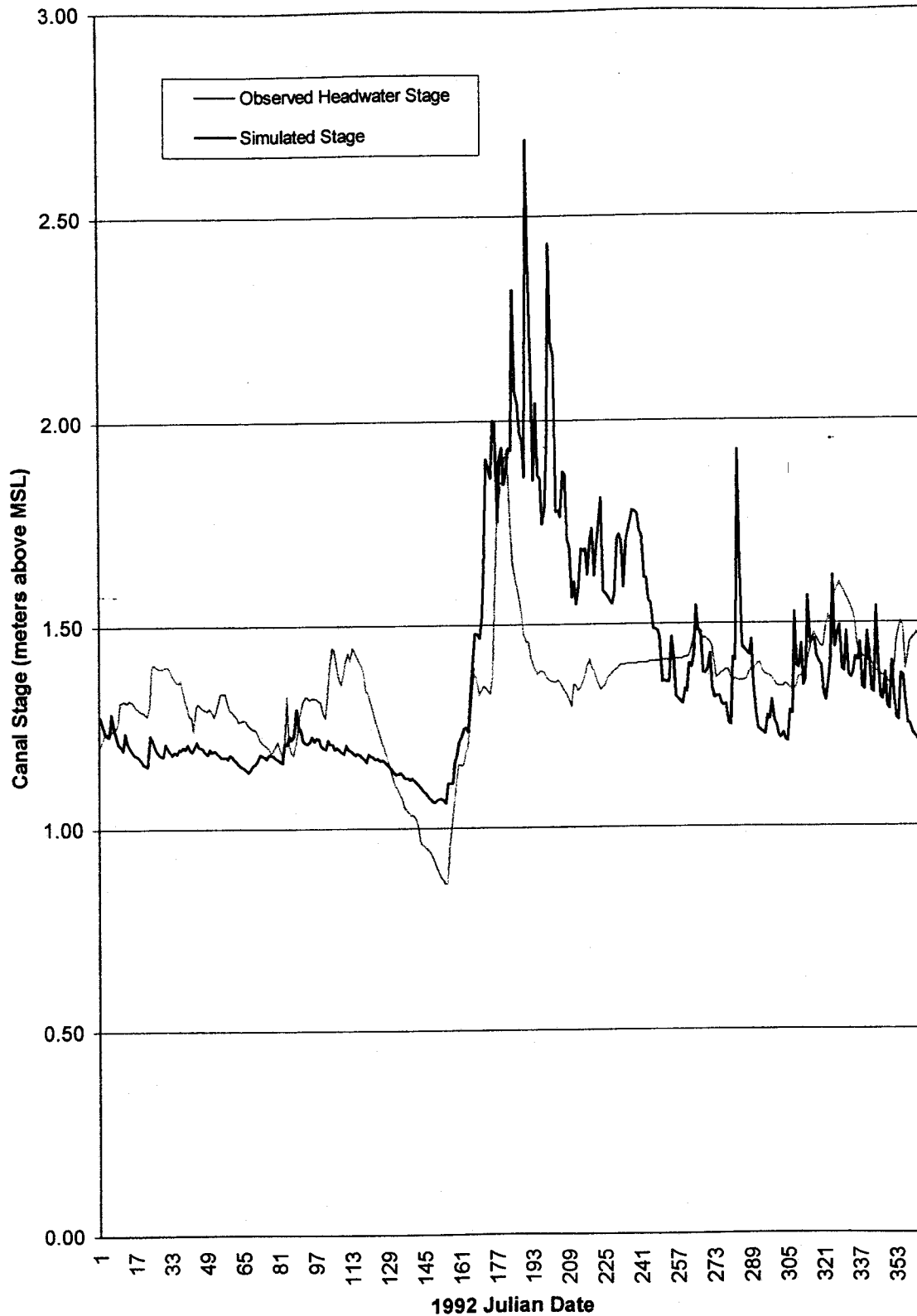


Figure 26. Simulated (black) versus observed (grey) canal stage at S-194 for 1992 rainfall year.

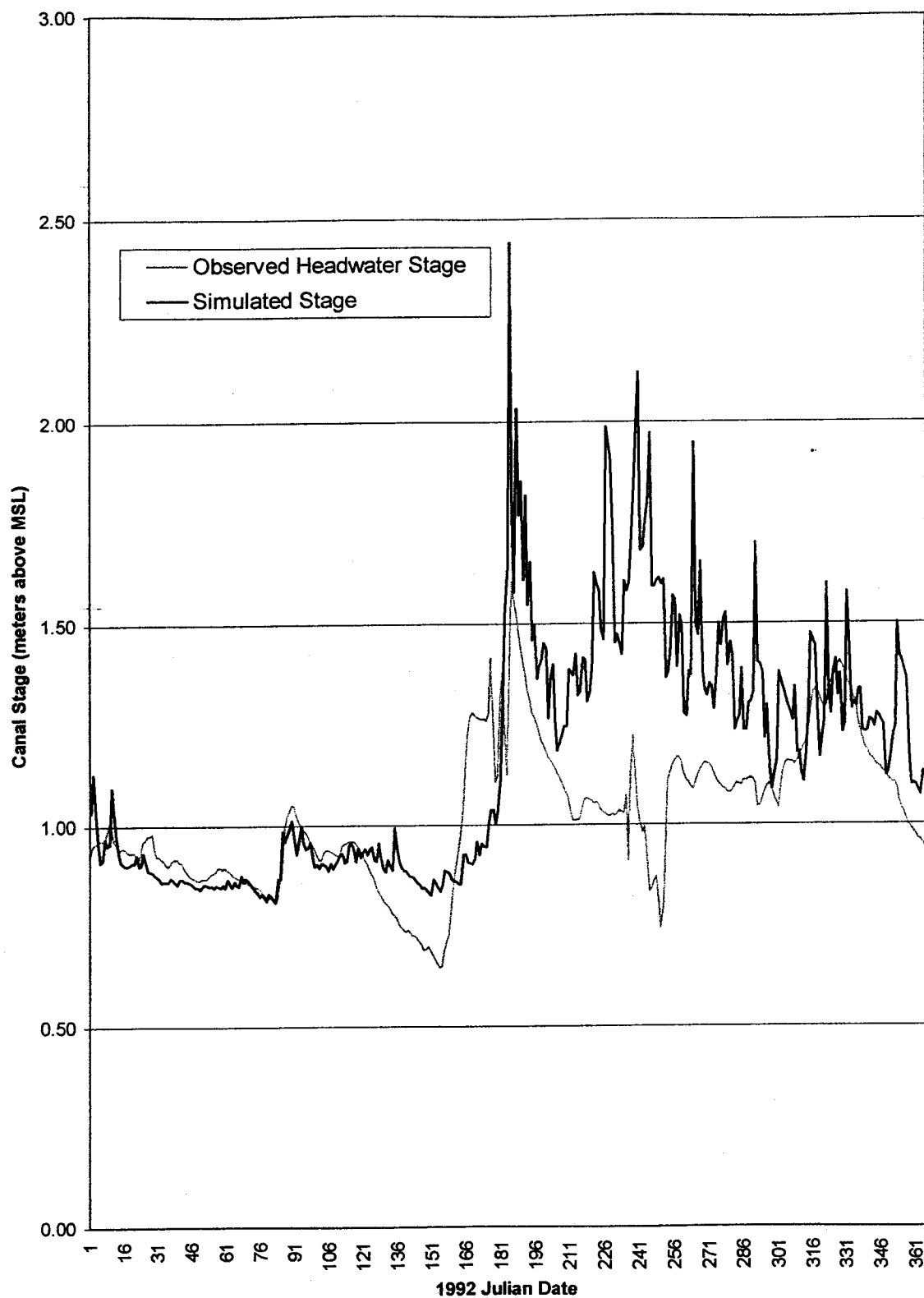


Figure 27. Simulated (black) versus observed (grey) canal stage at S-165 for 1992 rainfall year.

180 days during the dry season, which declined to 89 days during the wet season. The simulated dry season canal stages were not below observed values as frequently as at S-194, as simulated stages were below observed values on 103 days. Simulated wet season values were higher than observed values on 146 days.

Figure 28, which contains the time-histories at the furthest downstream canal gate in C-102, S-21A, reveals some contrast in the model behavior when compared to points upstream. Seasonal differences were not as pronounced at S-21A. The standard deviations between simulated and observed stage values were .11 and .12 meters for the dry and wet seasons, respectively. Simulated and observed stage values were no more than .25 meters apart for 167 dry season and for 165 wet season days. Simulated stages were primarily higher than observed stages during both seasons, as simulated values exceeded observed values for 119 days during the dry season, and for 94 wet season days.

Simulated groundwater elevations at G-757A are shown in Figure 29 along with the observed values at that location. The pattern of low groundwater elevations during the early months was followed by a rapid spike from infiltration of summer rains, and then by a gradual decline as water percolated downward and seeped laterally. The model slightly underestimated dry season groundwater elevations. Dry season water table values, which had a standard deviation of .18 meters from observed values, were lower than observed values during 164 days. In contrast, simulated values were greater than observed elevations during most of the wet season. Wet season simulated values, lower than the observed values for 143 days, had a standard deviation of .44 meters from observed values.

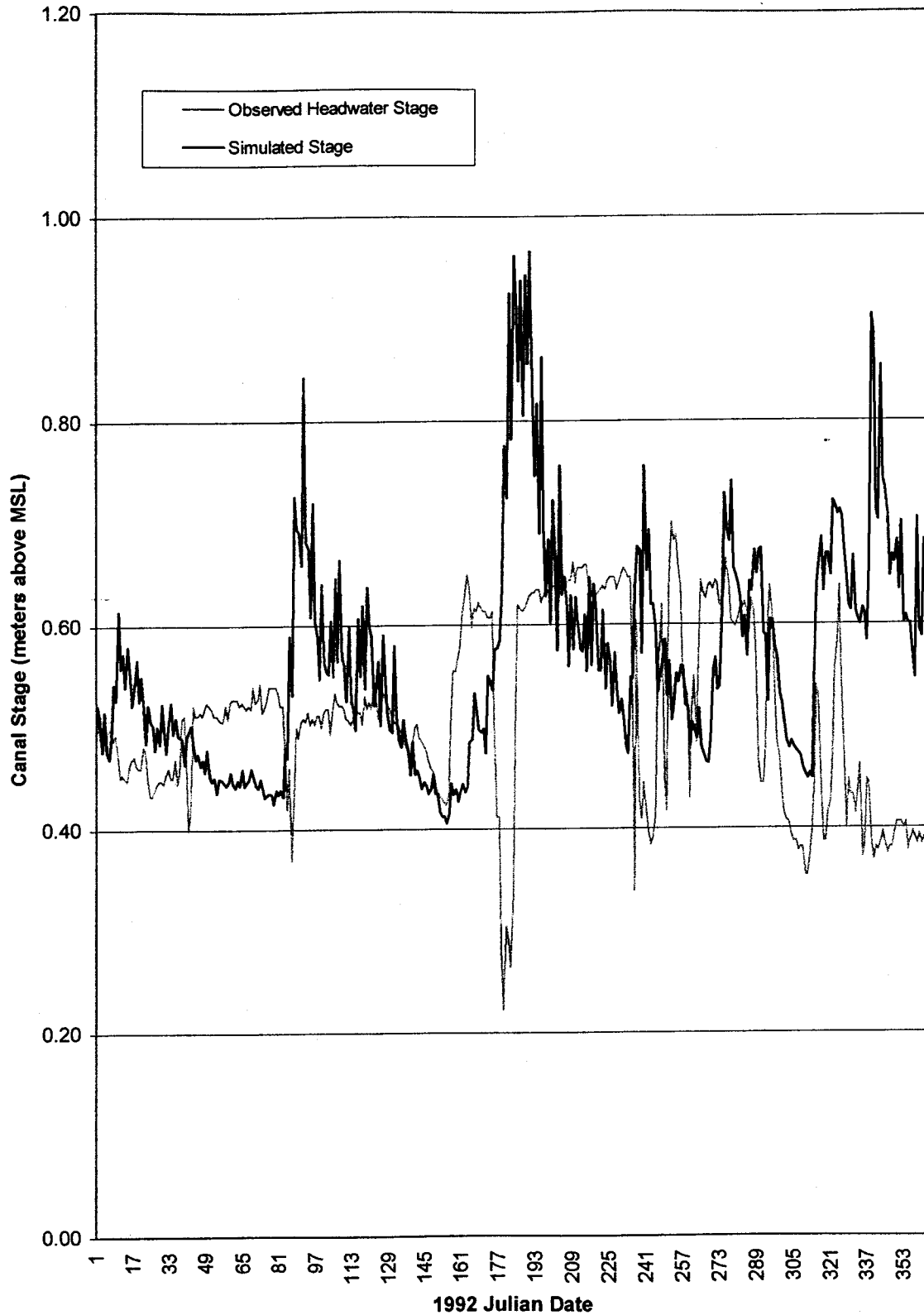


Figure 28. Simulated (black) versus observed (grey) canal stage at S-21A for 1992 rainfall year.

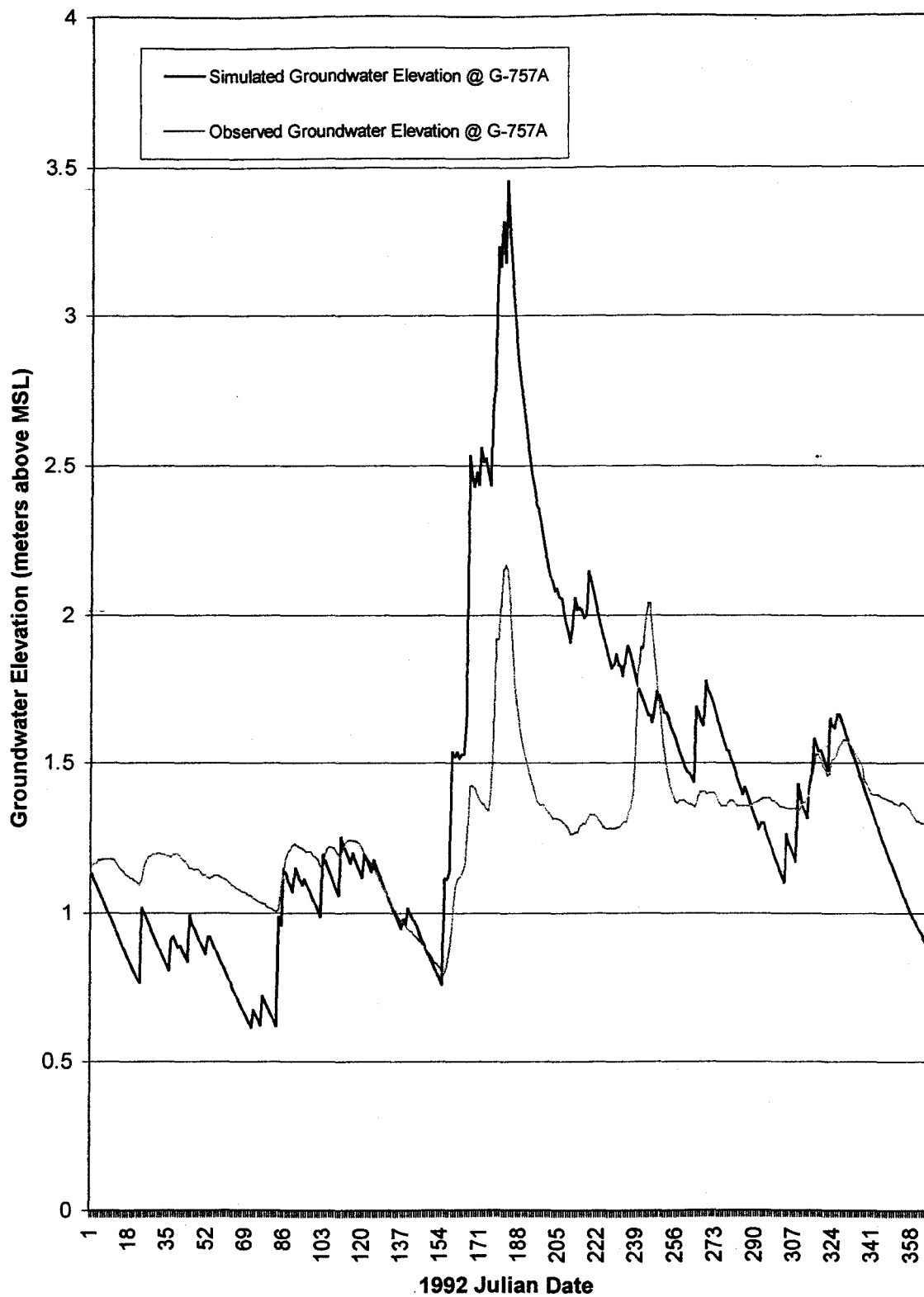


Figure 29. Simulated (black) versus observed (grey) groundwater elevations at G-757A for 1992 rainfall year.

A characteristic shared by the time-histories at all hydrologic calibration points was that day-to-day fluctuations in water levels were greater for simulated than for observed values. These fluctuations usually became more pronounced during the wet season.

Simulated time-histories of concentrations of constituents were graphed versus observed values at the C-102 calibration points, and are shown in Figures 30-31 and 32-33 for TP and TSS, respectively. The temporal resolution of simulated values was considerably higher than the observed values to which they were compared. Figure 30 shows the simulated TP concentration time-history at S-194 versus observed concentrations. Average simulated concentration values were .008 mg/L during the dry season, and increased to .033 mg/L during the wet season. Observed values ranged between .012 and .030 mg/L throughout the year, with the exception of a TP concentration of .060 mg/L on simulation date 189. The higher observed value corresponded to a period of higher simulated concentrations. The highest simulated TP concentration, .15 mg/L, was reached during one of two prolonged wet season TP concentration increases.

Figure 31 shows the time-history of simulated TP concentrations versus observed values at the S-21A structure near the terminus of the C-102 canal. As indicated by comparing values at S-194 and S-21A, both simulated and observed TP concentrations were lower in the downstream part of the canal, except for average simulated dry season values (nearly identical for S-194 and S-21A). In addition, seasonal variability between the wet and dry seasons was reduced. Simulated TP concentrations averaged .009 mg/L during the dry season, and .010 mg/L during the wet season. Observed TP concentrations

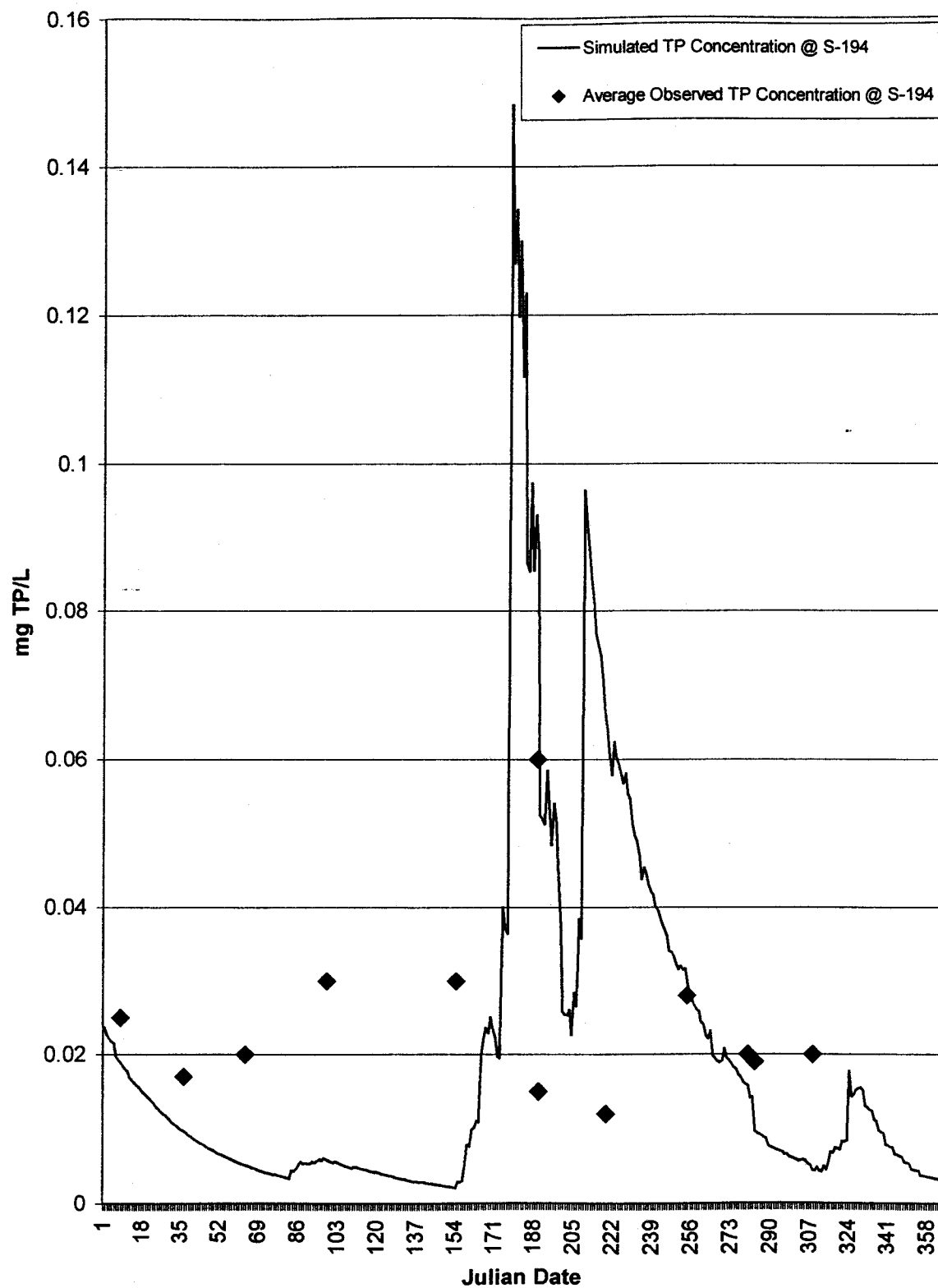


Figure 30. Simulated time-history of TP concentration versus individual observed concentration values at S-194 for 1992 rainfall year.

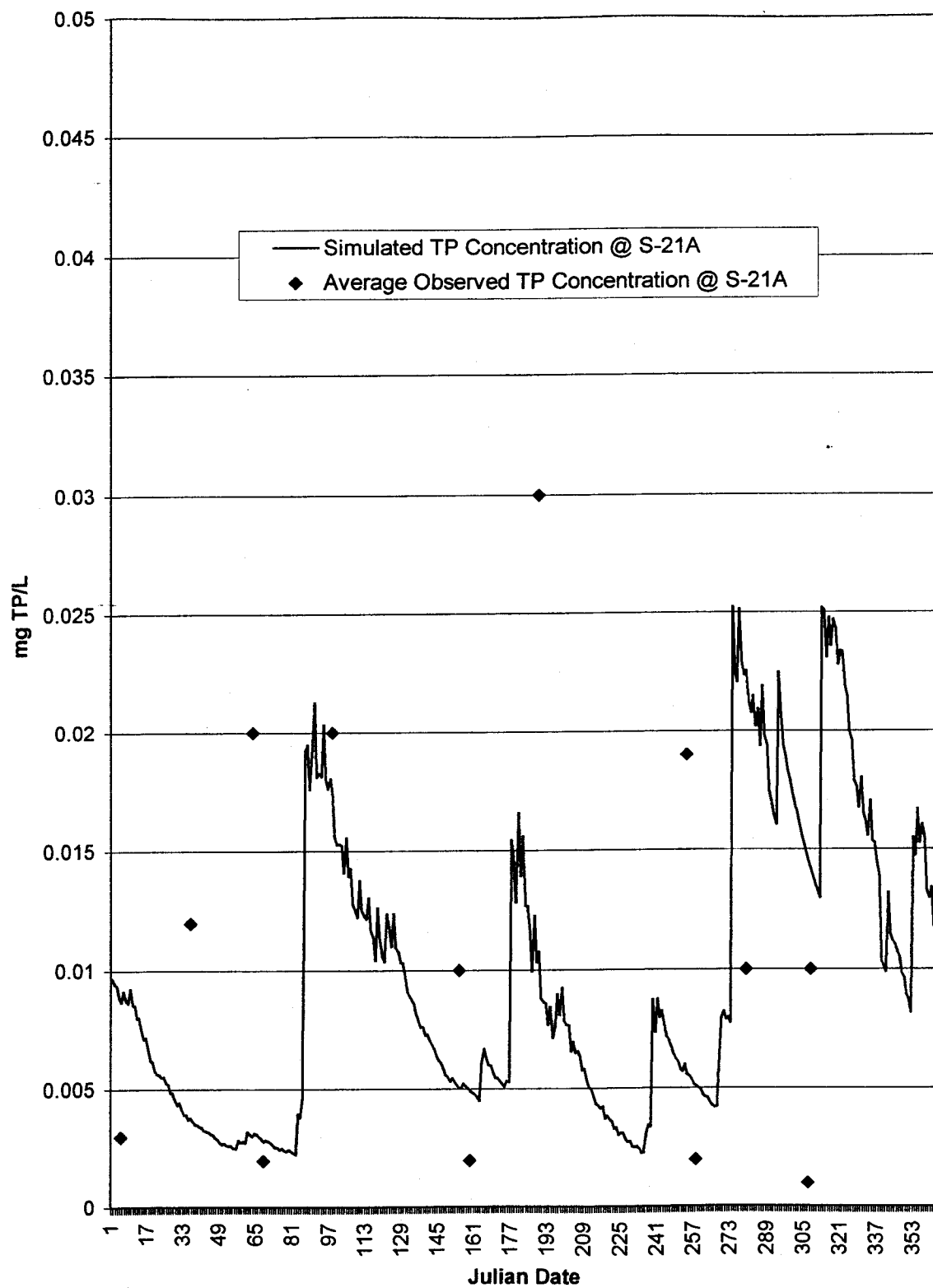


Figure 31. Simulated time-history of TP concentration versus individual observed concentration values at S-21A for 1992 rainfall year.

at S-21A averaged .011 mg/L during both seasons. The highest simulated TP concentration at S-21A was .025 mg/L, somewhat lower than the highest observed concentration of .030 mg/L.

Observed TSS concentration data were scarce, so all comparisons were made to a single point where data were available within the study area. Furthermore, no data were available from the heart of the wet season. The observed values ranged from 3 mg/L to 11 mg/L.

Simulated TSS concentrations had similar seasonal and watershed location patterns as those for TP. Figure 32, which shows the time-history of simulated TSS concentrations at S-194, shows a considerable increase in simulated concentration during the wet season after the onset of summer precipitation and associated EMC-based sediment loads. Simulated TSS concentrations averaged 3.6 during the dry season, and 22.4 mg/L during the wet season. The peak concentration reached during the wet season was 64.6 mg/L.

Figure 33 contains the time-history of the simulated TSS concentrations at the downstream structure S-21A. Simulated concentrations were considerably lower during the wet season, with an average value of 4.6 mg/L. As occurred with TP, the seasonal variation was much less at S-21A than upstream, as the dry season average concentration was 4.8 mg/L. The peak simulated concentration of 12.9 mg/L occurred during the brief part of the dry season at the end of the simulation year.

The annual simulated loads of constituents discharged from C-102 into Biscayne Bay were then tabulated, and are included as Table 3a. Incorporated into Table 3a are the estimates of actual sub-basin discharges into Biscayne Bay from Alleman, 1995. The

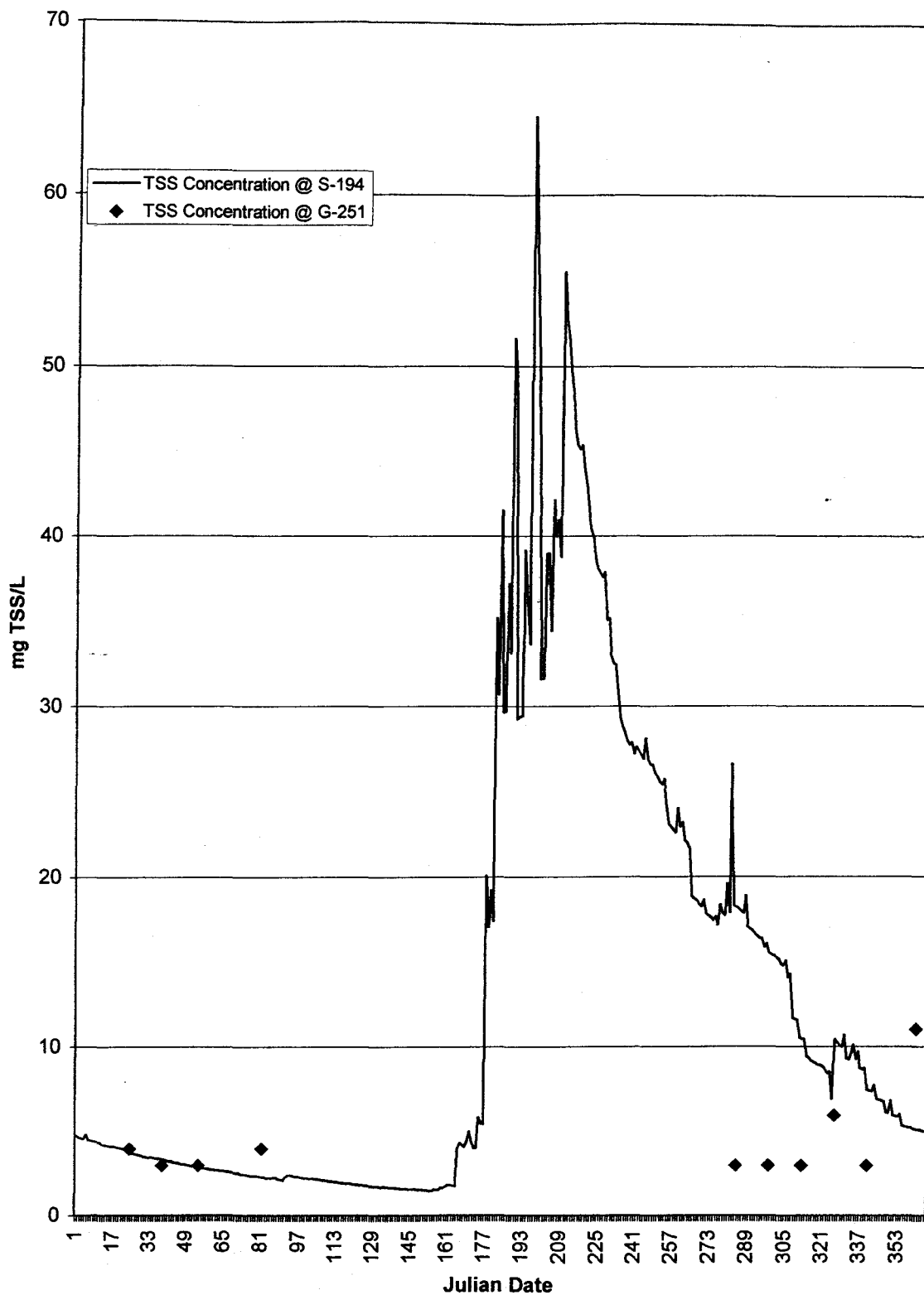


Figure 32. Simulated time-history of TSS concentration at S-194 versus individual observed TSS G-251 concentration values for 1992 rainfall year.

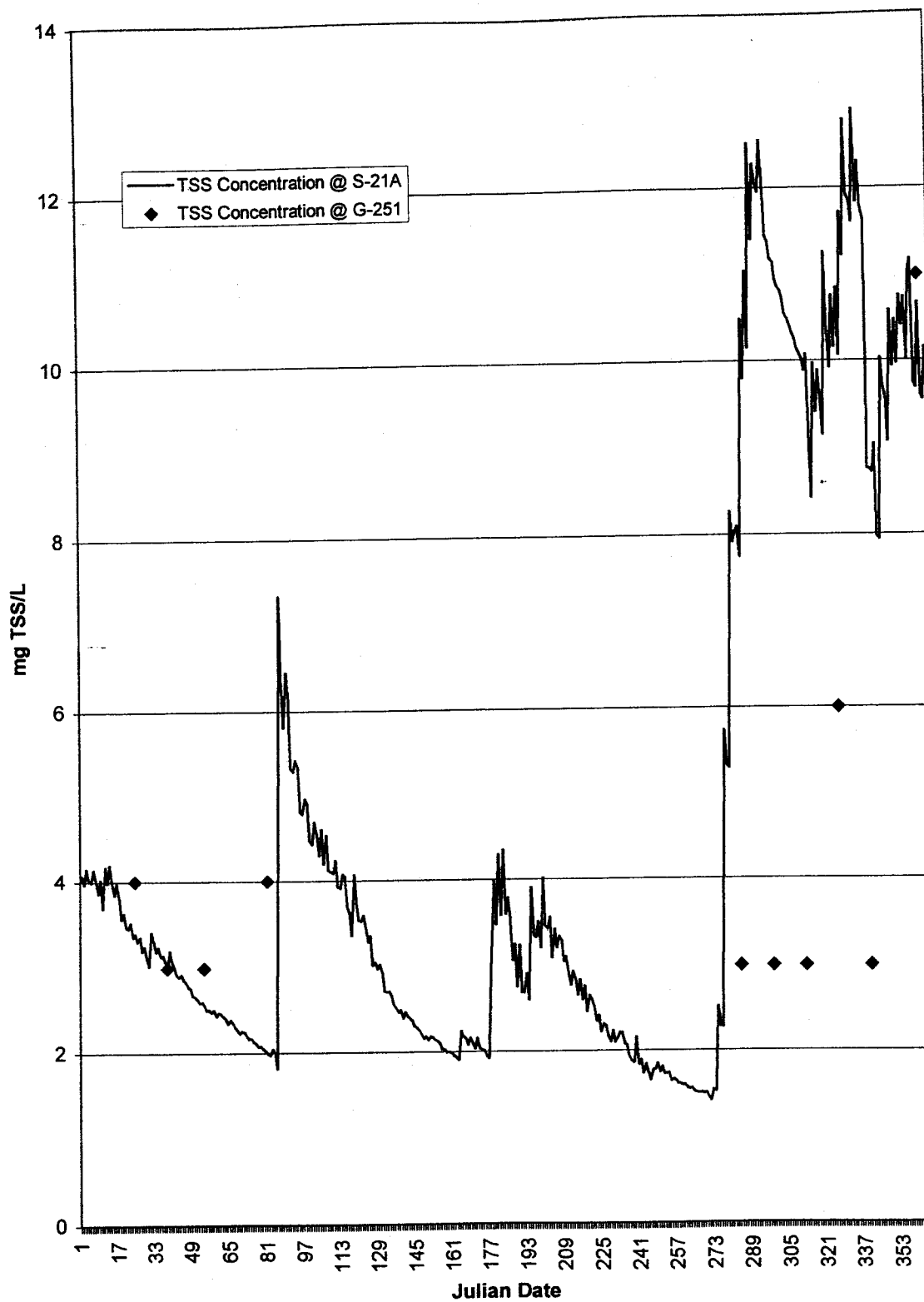


Figure 33. Simulated time-history of TSS concentration at S-21A versus individual observed TSS G-251 concentration values for 1992 rainfall year.

a)

<u>Constituent</u>	<u>Estimated Load Discharged to Bay</u>	<u>Simulated Load Discharged to Bay</u>	<u>Simulated/Estimated</u>
TP	1.53E+06	7.01E+05	0.46
TSS	5.37E+08	1.14E+09	2.11

b)

<u>TP Pathways</u>	<u>Grams/Year</u>	<u>% of Total</u>
<i>Chemical Uptake</i>	1.91E+07	64.0
<i>Sedimentation</i>	9.44E+06	31.7
<i>Discharge to Ocean</i>	7.01E+05	2.4
<i>Plant Uptake</i>	5.74E+05	1.9
<u>TSS Pathways</u>	<u>Grams/Year</u>	<u>% of Total</u>
<i>Sedimentation</i>	1.50E+09	57.0
<i>Discharge to Ocean</i>	1.14E+09	43.0

Table 3. a) Simulated versus estimated actual discharge of constituents from C-102 into Biscayne Bay. Units are in grams of constituent per year. Estimated actual values taken from Alleman, 1995. B) Simulated annual flows, in grams, for each constituent pathway. Percentages for each flow of total constituent flows are also given.

simulated load of TP was 46% of the estimated actual annual load. In contrast, simulated TSS load was 211% of the total estimated actual discharge of TSS into the ocean. These results suggest the model underestimates TP and overestimates TSS discharge into Biscayne Bay.

Time-histories of daily simulated TP and TSS loads discharged from the C-102 sub-basin into the Bay are shown in Figures 34 and 35. The time-history figures reveal that the model discharged phosphorus into the estuary more as a pulsed occurrence than for TSS, loads of which were delivered into the Bay more evenly over time. Peak discharge of TP occurred between simulation dates 155 and 208, although considerable loads were discharged during several other shorter intervals. Simulated TSS discharge was greatest in the early part of the year, although a second peak occurred at the beginning of the wet season.

Table 3b contains annual totals for specific constituent pathways, including sedimentation, plant uptake, and chemical uptake. Most of the TP leaving the watershed flowed through the chemical uptake pathway, with sedimentation as the other primary outflow pathway. Simulated discharge of TP into the ocean was proportionally small, at 2.4%. Sedimentation was the largest pathway for TSS, accounting for 57% of the total flow leaving the watershed. Simulated discharge into the ocean, however, was a much more significant pathway for TSS than for TP.

Validation Using Alternate Sub-basin

The results of the validation test of model appropriateness for alternative land use conditions, performed by simulating a different sub-basin, are described next. Figures 36-38 show the simulated and observed time-histories of canal stage at three locations in

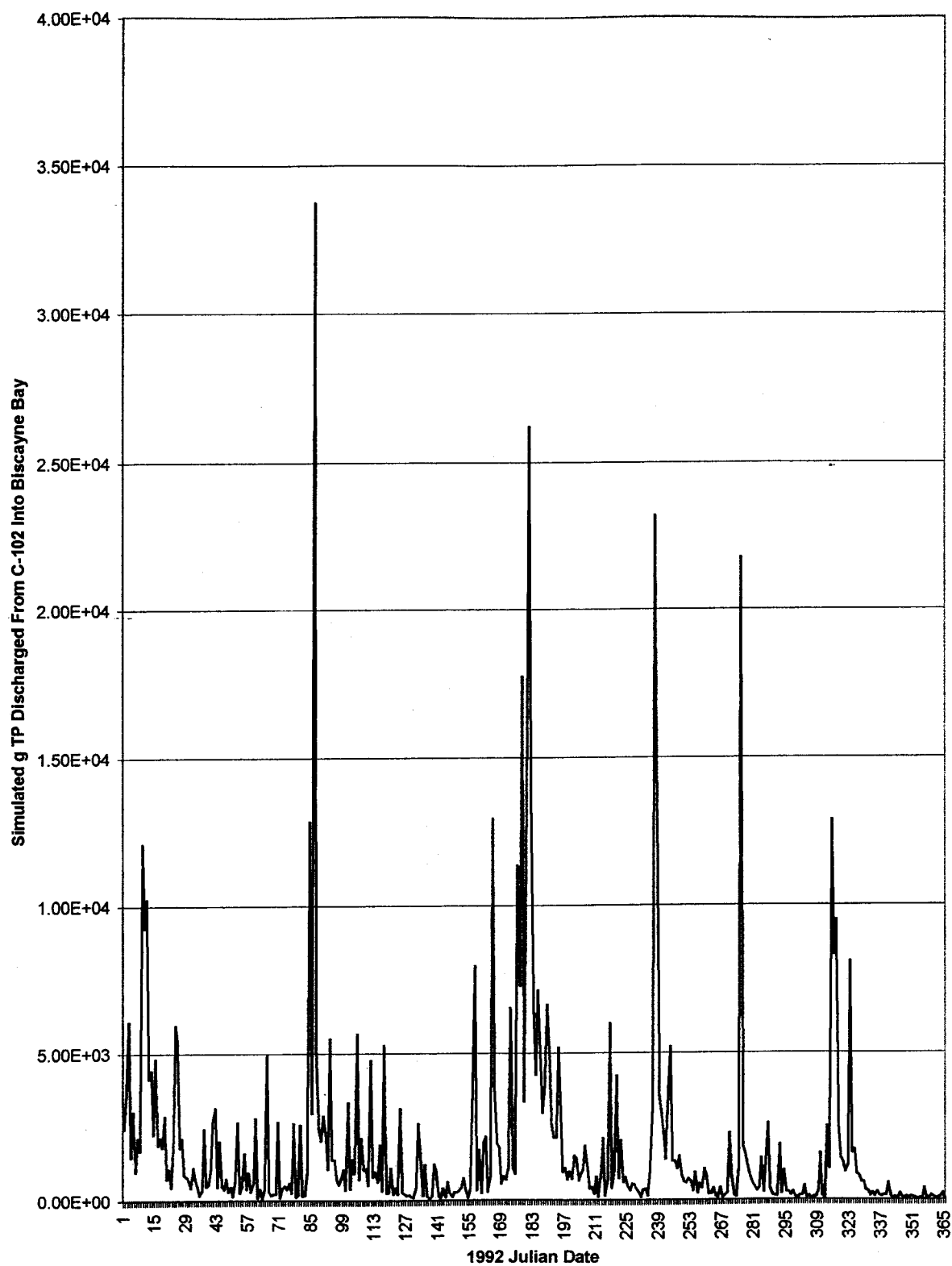


Figure 34. Time-history of simulated C-102 watershed TP discharge into Biscayne Bay for 1992 rainfall year.

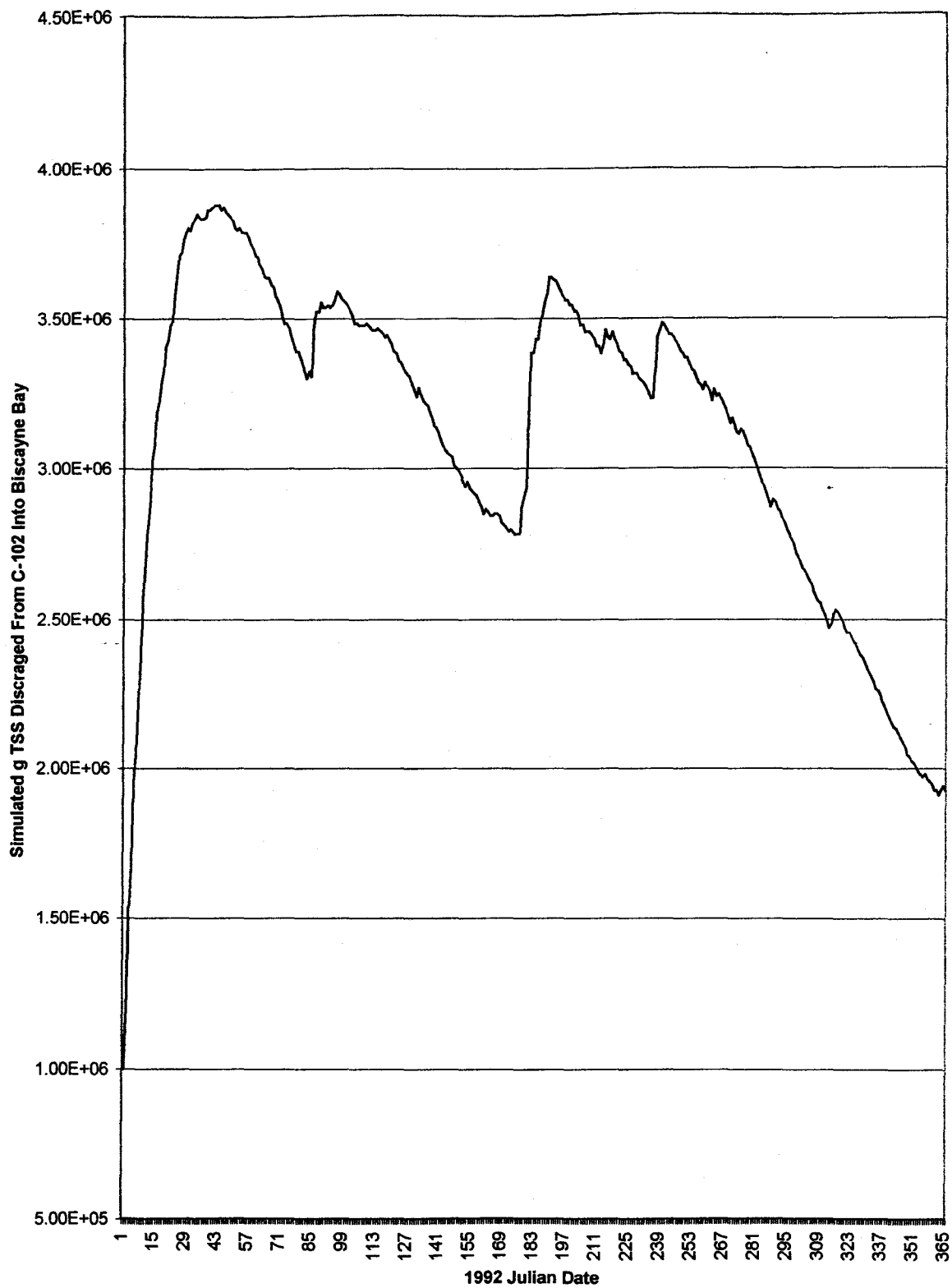


Figure 35. Time-history of simulated C-102 watershed TSS discharge into Biscayne Bay for 1992 rainfall year.

the C-103 sub-basin, specifically at S-196, S-179, and S-20F. As in the C-102 sub-basin, the model results exhibited characteristic seasonal and watershed position patterns.

Figure 36 shows the time-history of simulated canal stage at S-196, the westernmost structure in C-103. During the dry season, simulated stages were below observed stages on 121 days. Observed and simulated dry season stages were less than .25 meters apart on 175 days, with a standard deviation of .10 meters. Increased variability occurred during the wet season, as the standard deviation of observed and simulated values increased to .21 meters, and stages were less than .25 meters different on 124 days. The simulated wet season stages at S-196 were generally higher than those observed, as higher simulated values occurred on 160 days. These results were very similar to those for the westernmost structure in the C-102 sub-basin, S-194.

Figure 37 contains the time-histories of simulated and observed stages at S-179, in the middle of the C-103 canal sub-basin. The simulated results from the dry season at S-179 were the closest to the observed values among all the analysis points. Standard deviation during the dry season was .04 meters, and was within .25 meters for every day (185 days) of the dry season. Wet season errors between simulated and observed values were larger than for the dry season, with a standard deviation of .12 meters and a period where stages were less than .25 meters of 152 days. In contrast to the pattern at other points, simulated stages were usually higher during the dry season and lower during the wet season. Simulated dry season stages were higher than observed stages for 141 days of the dry season and lower than observed 147 days of the wet season.

Simulated canal stage results from the downstream structure, S-21A, are shown in Figure 38. The dry season results were similar at S-20F to those for S-21A, the terminus

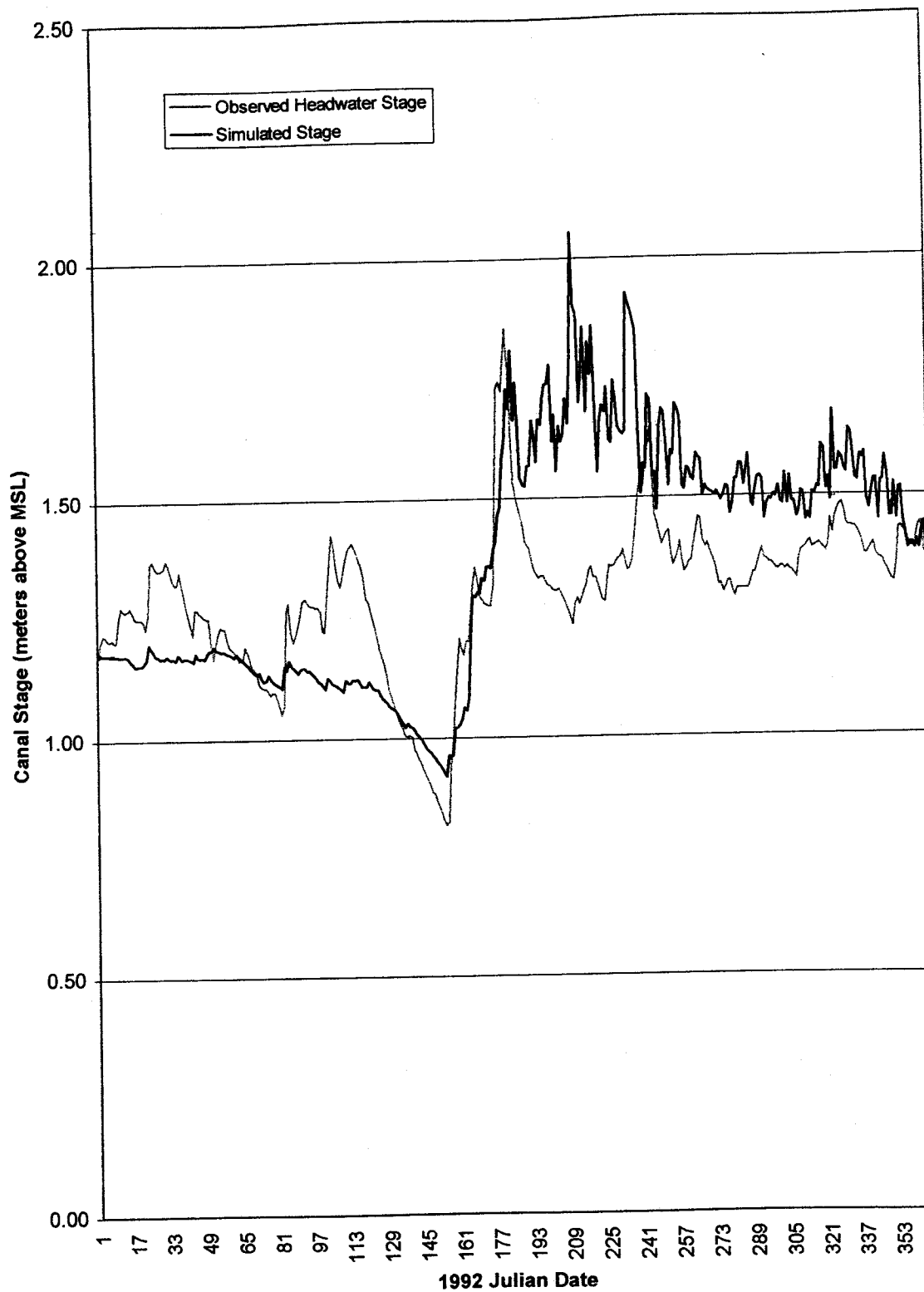


Figure 36. Simulated (black) versus observed (grey) canal stage at S-196 for 1992 rainfall year.

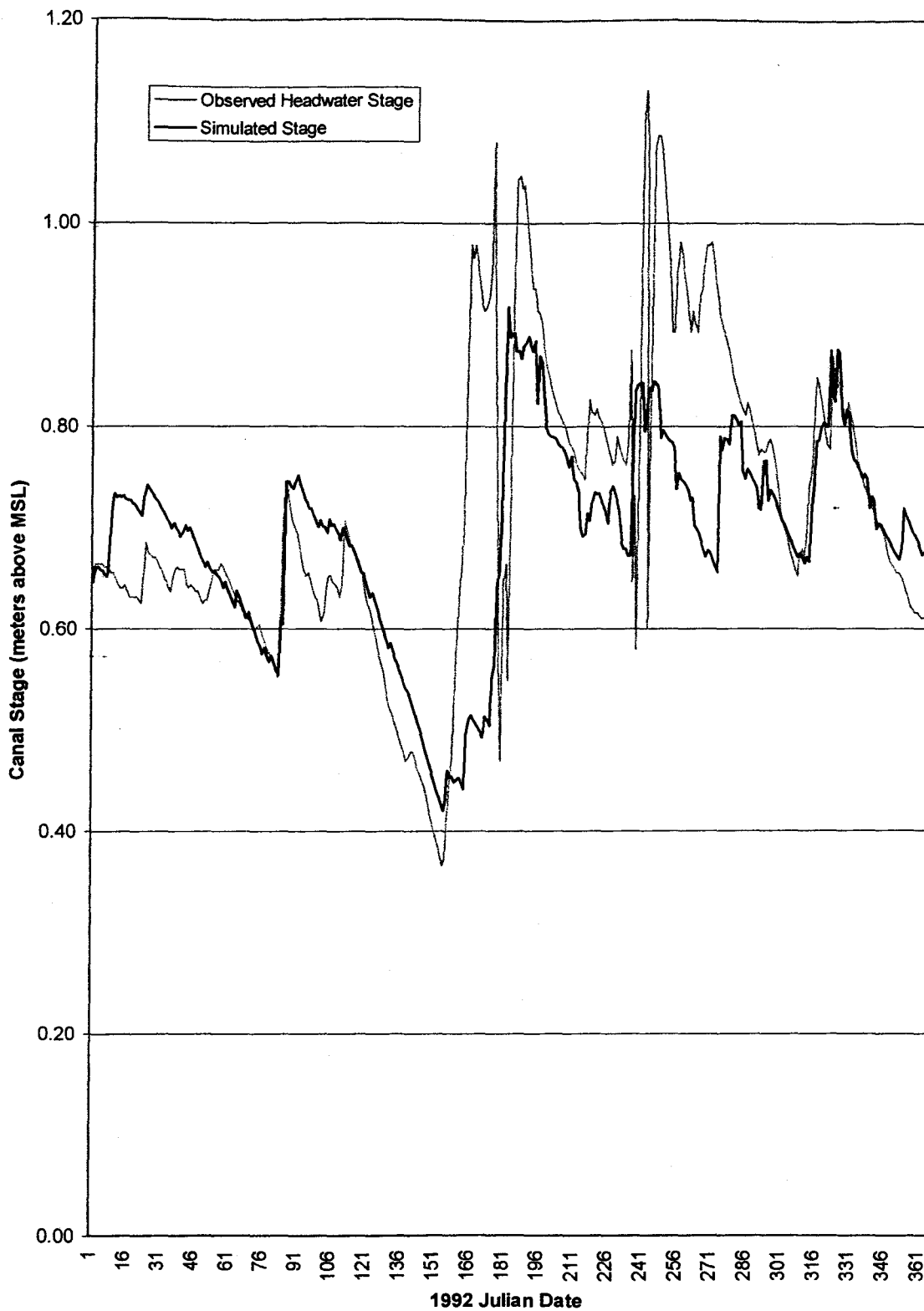


Figure 37. Simulated (black) versus observed (grey) canal stage at S-179 for 1992 rainfall year.

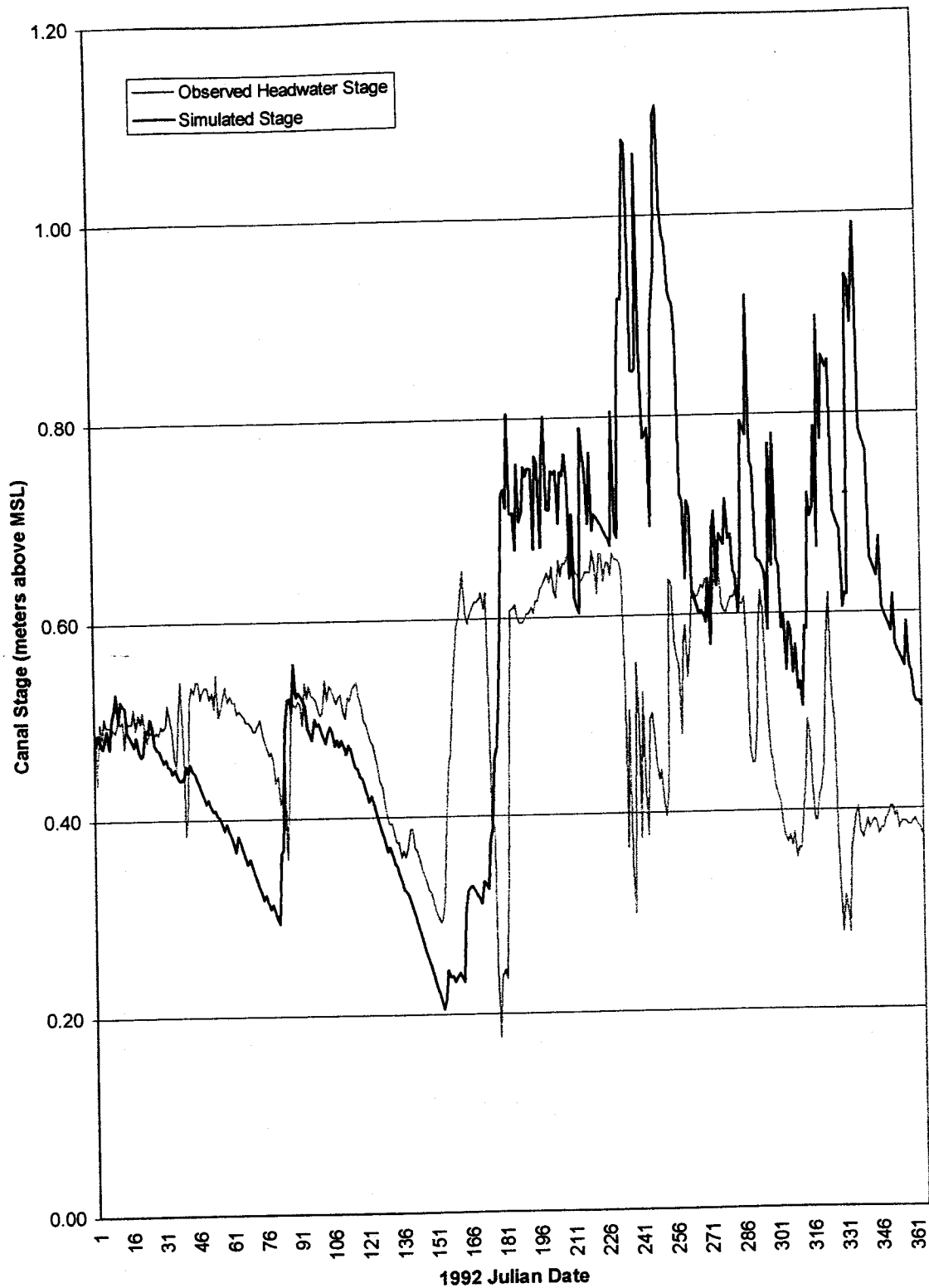


Figure 38. Simulated (thick black) versus observed (dashed grey) canal stage at S-20F for 1992 rainfall year.

structure in C-102. Dry season simulated stages were within .25 meters of observed stages for 166 days, where the standard deviation was .11 meters. As with all points, errors increased during the wet season, with a standard deviation of .20 meters and 118 days where simulated and observed stages were within .25 meters. Simulated stages were lower than observed for 127 days of the dry season and higher than observed on 140 days during the wet season.

The overall hydrologic response simulated by the model in C-103 was similar to that seen in C-102. With the exception of S-179 (which simulated hydrology nearly identically to observed hydrology), simulated dry season stages were lower than observed stages, and the wet season simulated stages were higher than observed stages. The hydrologic response in C-103 shared another characteristic with the response in C-102. Day-to-day fluctuations in simulated water levels became more pronounced during the wet season than during the dry season. In addition, the accuracy of the hydrologic model (where accuracy is measured by the standard deviation and number of days where simulated and observed values were less than .25 meters) was nearly equal at S-196 and S-20F to the spatially analogous points in C-102. The mid-canal point (S-179) was more accurate than the mid-canal point in C-102.

The similar hydrologic response and improved accuracy of simulated values in C-103 as compared to C-102 suggests the model was valid for studying hydrology in watersheds of increased urbanization from the 1995 C-102 sub-basin condition, since the C-103 sub-basin is more urbanized than the C-102 sub-basin. The results at S-179 offer additional evidence supporting the validation of the model under increased urbanization. S-179 is the structure located near the most urbanized areas (Homestead and Florida

City) in the analysis extent, and had the smallest discrepancies between simulated and observed stage values of any station within either sub-basin.

Figures 39-41 (TP) and 42-44 (TSS) show the simulated time-histories versus observed values of constituent concentrations at the C-103 canal observation points. In Figure 39, the time-history of simulated TP concentrations is graphed with observed values at S-196. Average simulated TP concentrations at S-196 were .009 mg/L during the dry season, and .020 mg/L during the wet season. The highest simulated TP concentration, .082 mg/L, was the peak value reached during the one period (at the onset of summer rains) where concentrations were considerably higher than at any other time of year. Other than this concentration spike, simulated concentrations remained between .005 mg/L and .011 mg/L. Observed values at S-196 ranged from .001 mg/L to .030 mg/L, and averaged .007 mg/L. Observed and simulated concentrations were generally lower than concentrations at the spatially analogous point in the C-102 sub-basin, S-194.

Figure 40 contains the time-history of simulated TP concentrations at S-179. Simulated S-179 concentrations underwent a consistent pattern of slow decline followed by occasional sudden increases, which was followed by another period of slow decline. Average simulated TP concentrations at S-179 were .007 mg/L during the dry season, and .015 mg/L during the wet season. The highest simulated TP concentration, .033 mg/L, was lower than the highest observed value of .048 mg/L. Observed values at S-179 ranged averaged .007 mg/L, as at S-196. The results suggested that the model simulated TP concentrations quite accurately at S-179 during the dry season, but that wet season peak concentrations were dampened, and average wet season concentrations were somewhat higher than actual values.

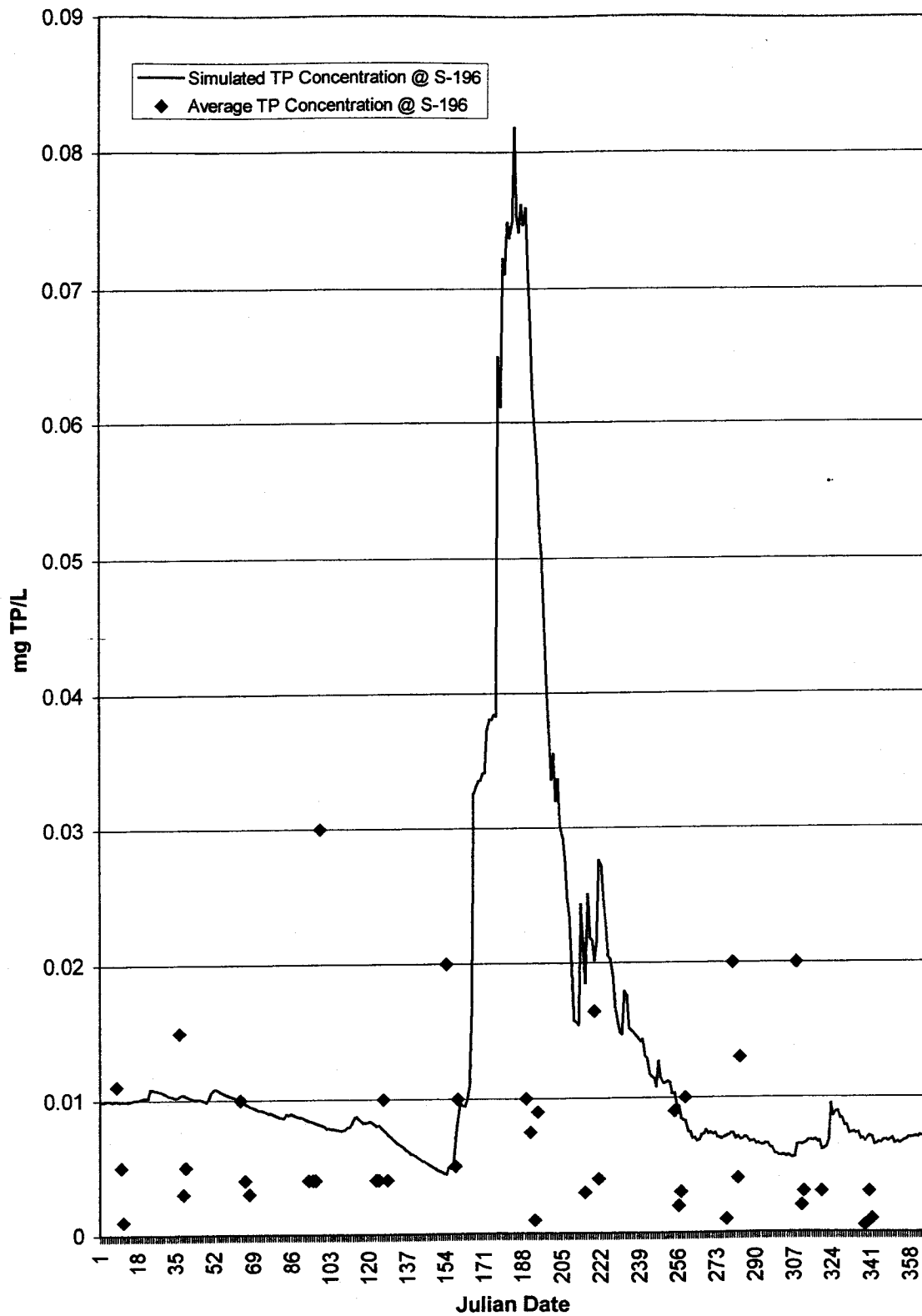


Figure 39. Simulated time-history of TP concentration versus individual observed TP concentration values at S-196 for 1992 rainfall year.

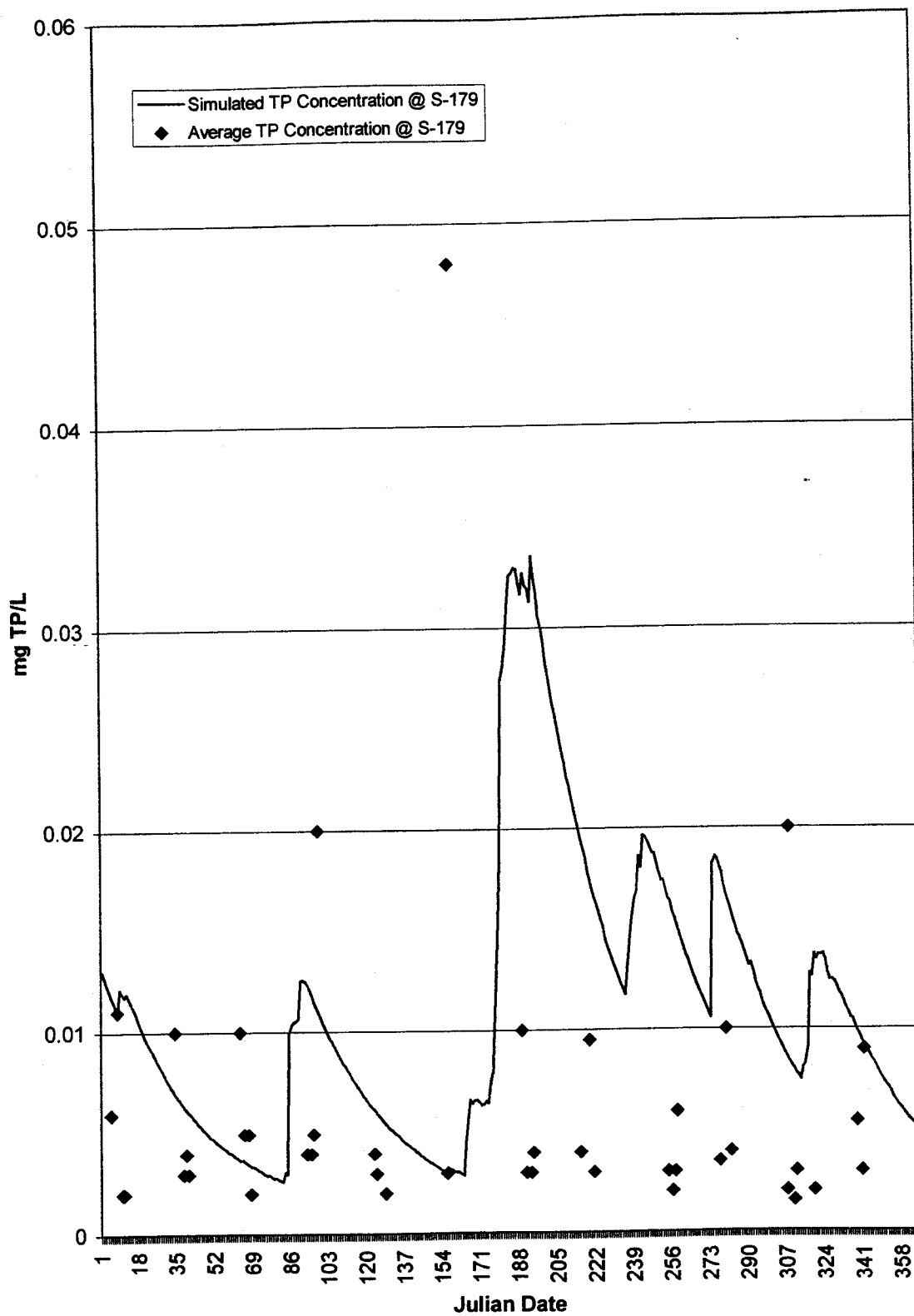


Figure 40. Simulated time-history of TP concentration versus individual observed TP concentration values at S-179 for 1992 rainfall year.

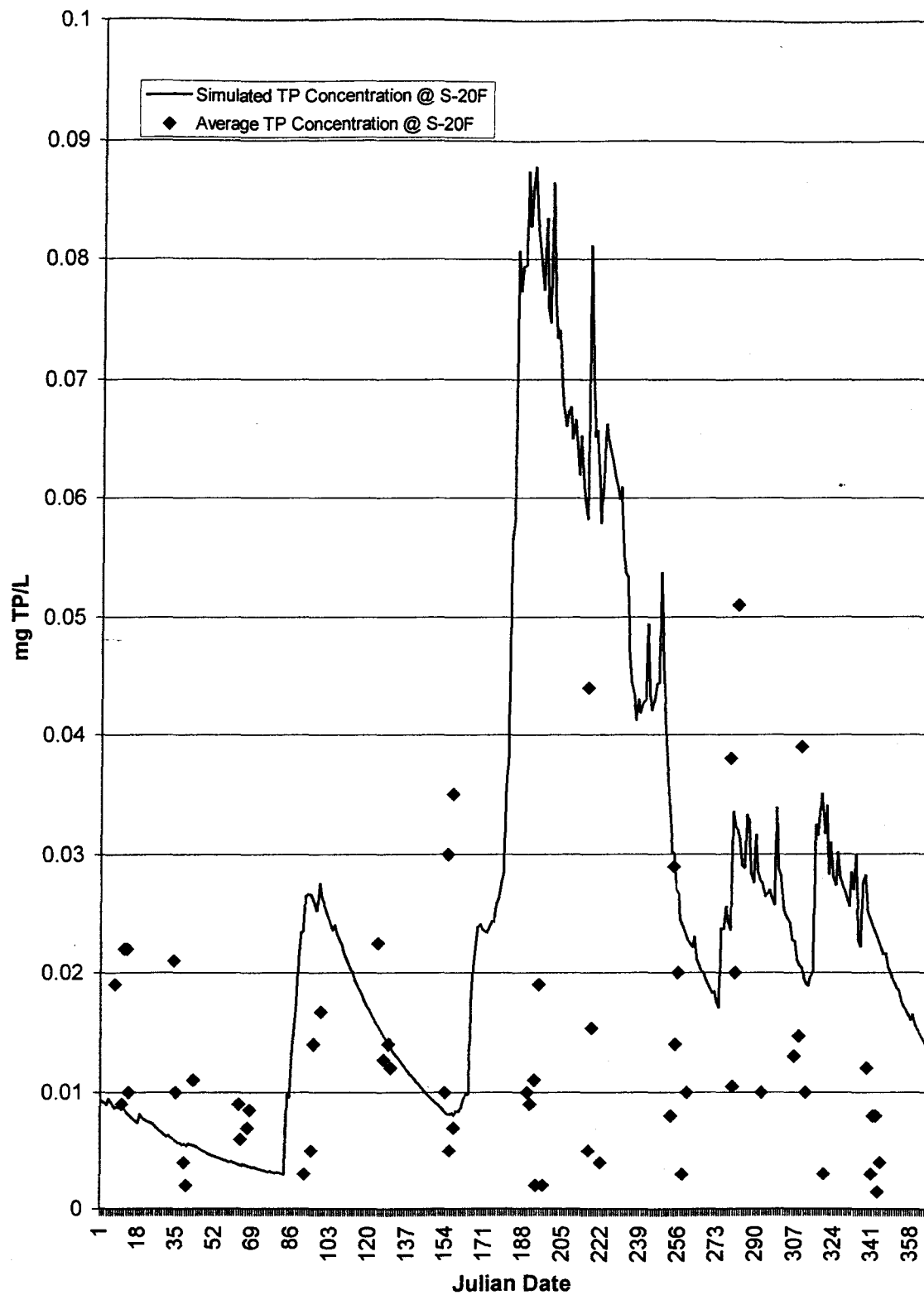


Figure 41. Simulated time-history of TP concentration versus individual observed TP concentration values at S-20F for 1992 rainfall year.

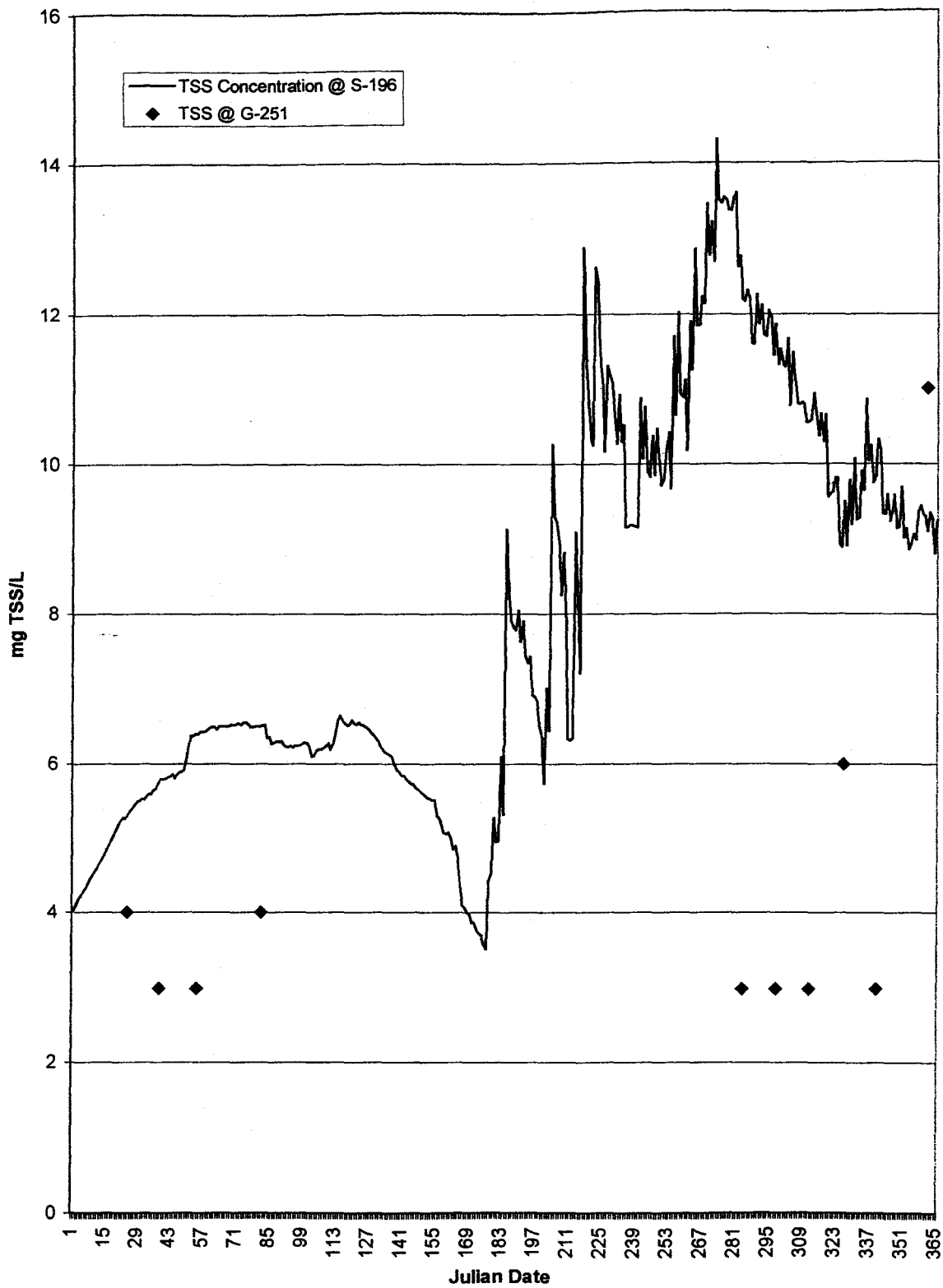


Figure 42. Simulated time-history of TSS concentration at S-196 versus individual observed TSS G-251 concentration values for 1992 rainfall year.

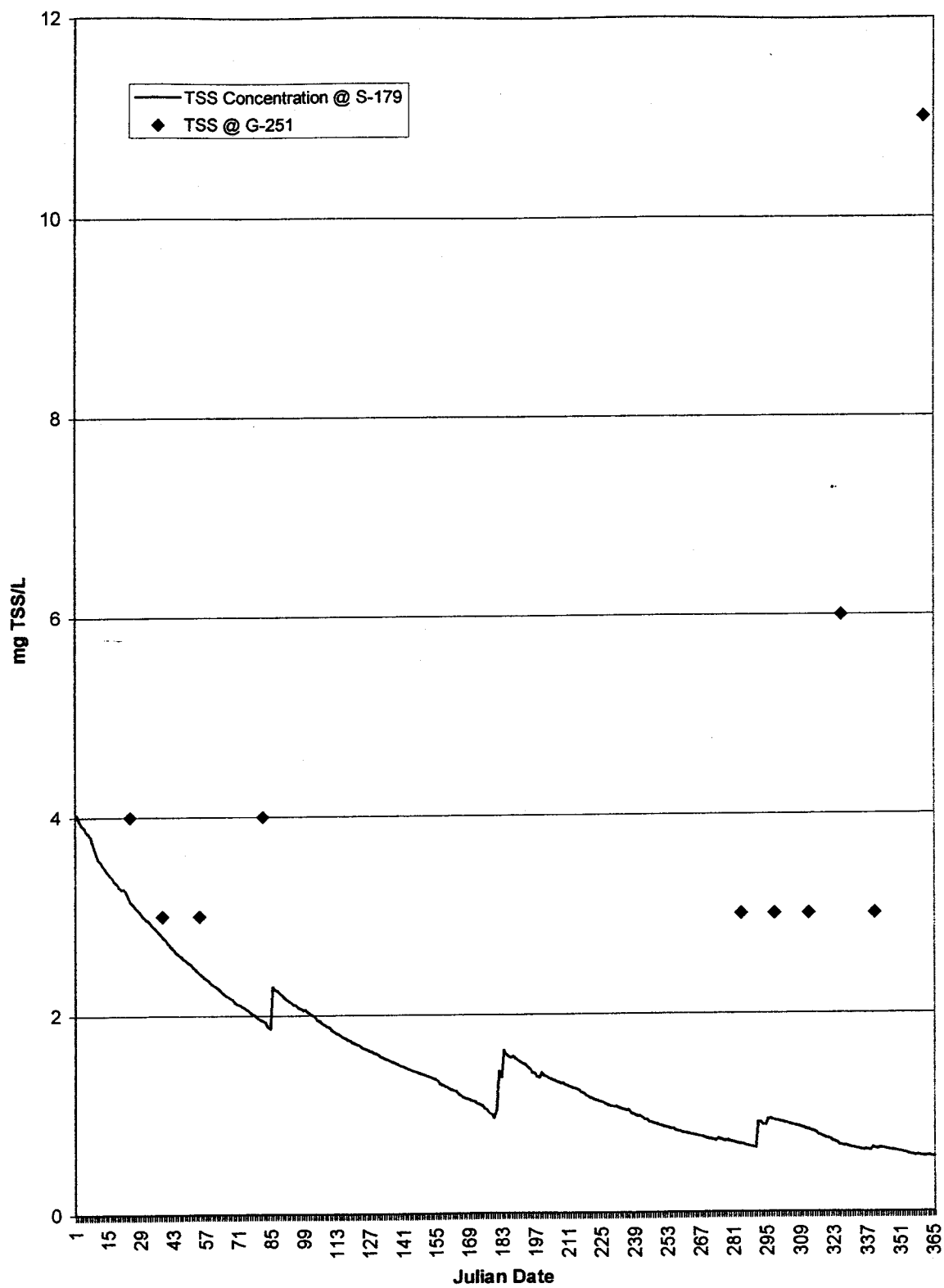


Figure 43. Simulated time-history of TSS concentration at S-179 versus individual observed TSS G-251 concentration values for 1992 rainfall year.

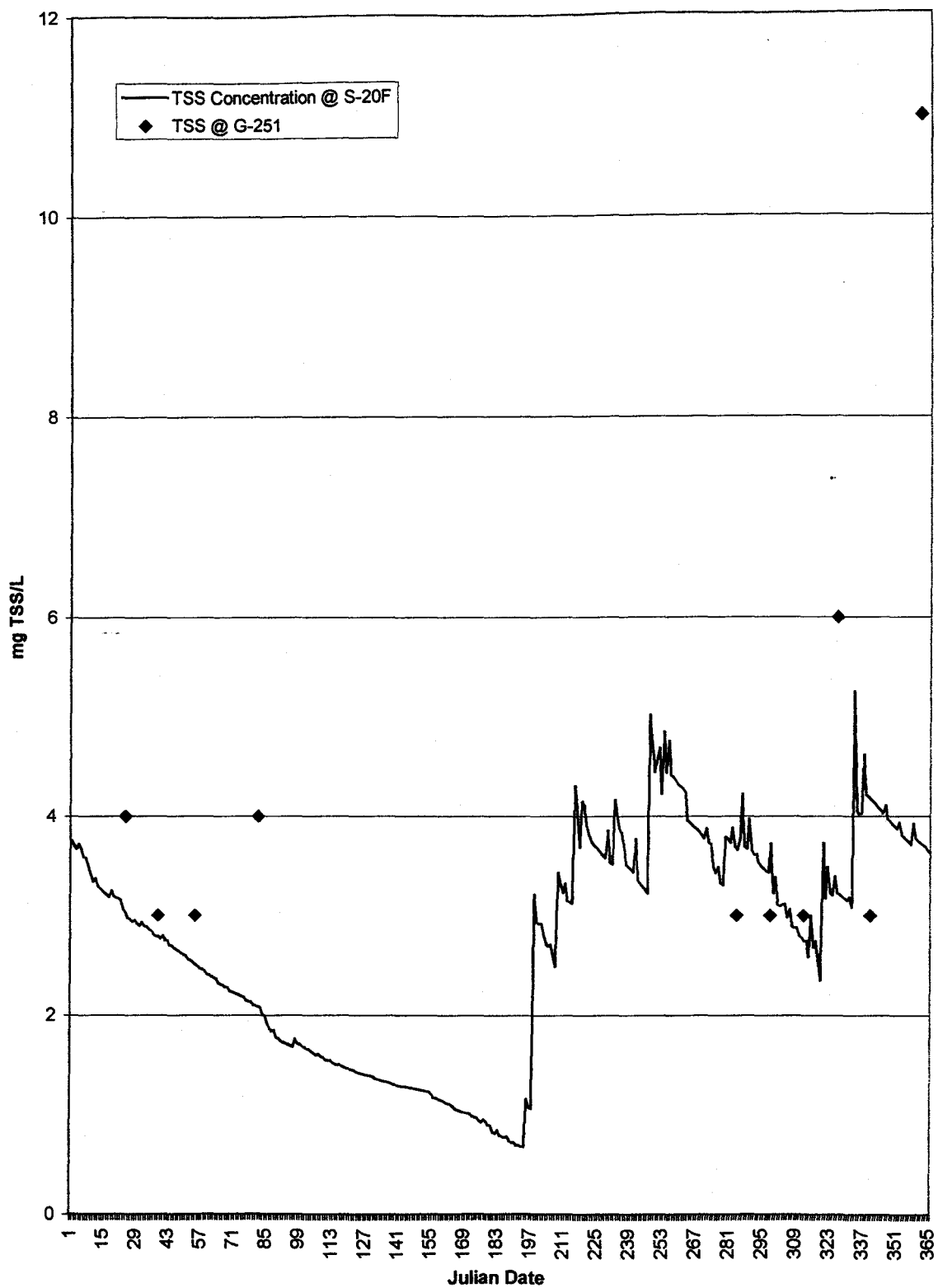


Figure 44. Simulated time-history of TSS concentration at S-20F versus individual observed TSS G-251 concentration values for 1992 rainfall year.

Figure 41 shows the time-history of simulated TP concentrations at S-20F.

Average simulated TP concentrations at S-20F were .013 mg/L during the dry season, and .039 mg/L during the wet season, which were the highest average TP concentrations throughout the analysis extent. Observed values at S-20F averaged .014 mg/L. The highest simulated TP concentration, .088 mg/L, was reached during the concentration increase corresponding to the onset of the wet season, and compares to a peak observed concentration of .051 mg/L. Observed and simulated concentrations were generally higher than concentrations upstream, and compared to the spatially analogous point in the C-102 sub-basin, S-21A.

The time-history of TSS concentration at S-196 is shown in Figure 42. Simulated concentrations increased from the dry season, with an average value of 6.8 mg/L, to the wet season, where values averaged 9.3 mg/L. The peak simulated concentration was 14.3 mg/L.

Figure 43 shows the time-history of simulated TSS concentration at S-179. The time-history curve here reveals a unique pattern, where TSS concentration underwent a fairly consistent decline from the initial concentration of 4 mg/L. The simulated concentration declined to a low of .54 mg/L on simulation date 365.

Figure 44 shows the simulated TSS concentration time-history graph for S-20F. After an overall decline in simulated concentrations similar to what occurred at S-179, concentrations rebounded during the wet season. Simulated concentrations increased from the dry season, with an average value of 2.6 mg/L, to the wet season, where values averaged 2.9 mg/L. The peak simulated concentration was 5.2 mg/L.

Simulated total annual load of TP from C-103 into Biscayne Bay was $1.56 \text{ E}+06$ grams, 61% of the total estimated actual TP load of $2.57 \text{ E}+06$ grams. Simulated total annual load of TSS from C-103 into Biscayne Bay was $2.53 \text{ E}+09$ grams, 211% of the total estimated actual TSS load of $1.15 \text{ E}+09$ grams. As in C-102, the simulated total annual load of TP was lower, and load of TSS was higher, than the estimated values.

In general, the water quality model simulated accurately in C-103. Simulated values were usually close to average values seen at each observation point. Simulated TP concentrations had similar averages at S-196 and S-179, just as occurred in observed values. The increase in observed values at S-20F also appeared in simulated concentrations at that structure. Since simulated annual loads distributed into the Bay from both sub-basins were nearly identical in percentage to the estimated values, the water quality model was equally valid for the more urbanized condition.

Validation Using Alternate Rainfall Years

The results from the two alternate rainfall years are described next. The dry (1989) rainfall condition results are described first, followed by wet (1995) rainfall results, and then followed by cross-rainfall year comparison results.

Low Rainfall Conditions

Figures 45-55 show time-histories of simulated canal stages, groundwater elevations, constituent concentrations, and total watershed discharges into Biscayne Bay. Figure 45 displays the time-histories of simulated canal stages at S-194 for the dry rainfall condition and the observed 1989 stages. The 1989 observed stages were mostly lower than the 1992 stages, with a 1989 peak value of 1.37 meters in contrast to the maximum 1992 stage of 1.91 meters. Simulated S-194 stages exhibited less variation

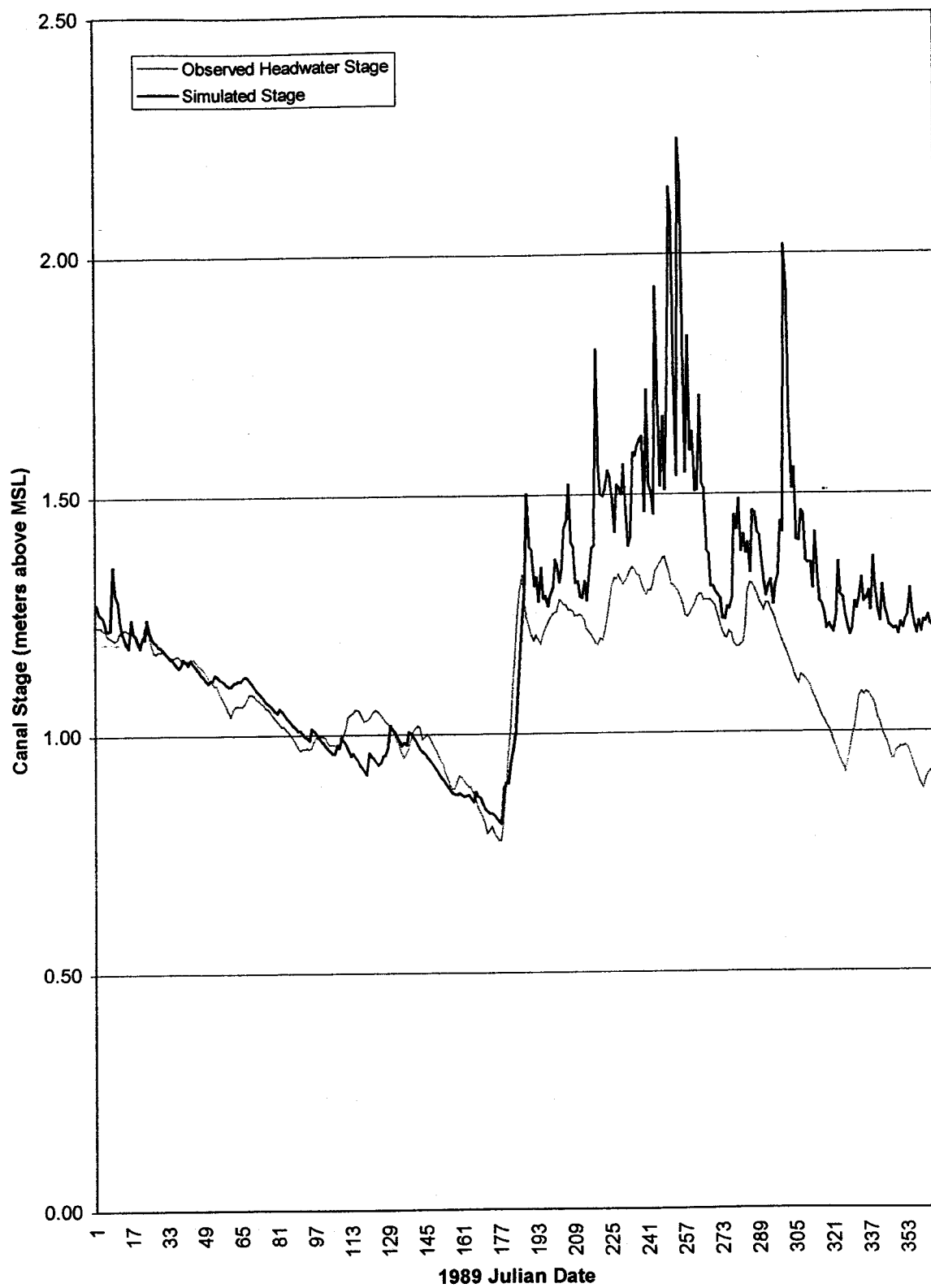


Figure 45. Simulated (black) versus observed (grey) canal stage at S-194 for 1989 rainfall year.

from observed stages than did the calibrated condition results. The dry season standard deviation of .08 meters increased to .19 meters during the wet season. Values were within .25 meters for most of the year, specifically for 162 dry season and 131 wet season days. Simulated were higher than observed stages during the majority of both the dry season (124 days) and the wet season (131 days).

Figure 46 shows the time-histories of observed and simulated 1989 rainfall conditions canal stages at S-21A. Significant contrast appeared between the simulated and observed time-histories. The observed canal stage time-history was quite flat, staying near .55 meters during the dry season and .65 meters during the wet season. The simulated stages oscillated considerably. The difference in behavior was reflected in the standard deviations, which were .30 and .22 meters for the dry and wet seasons, respectively. A clear effect of the low rainfall conditions was that simulated stages were lower than observed values on all but 2 days of the year. Simulated stages even dropped below MSL for some of the dry season.

The time-histories of simulated and observed canal stages for 1989 rainfall at S-196 are included in Figure 47. Simulated and observed stages were in general agreement until late in the wet season (around simulation date 270). Simulated stages then remained fairly constant throughout the remainder of the year, while observed values staged an end-of-year decline of about .55 meters to .83 meters above MSL. Values were within .25 meters for most of the early part of the year, and overall for 149 dry season and 128 wet season days. Standard deviation of simulated and observed values in the dry season was .15 meters, and increased to .18 meters during the wet season. Simulated stages were higher than observed stages for a similar amount of time as at S-194, except for the

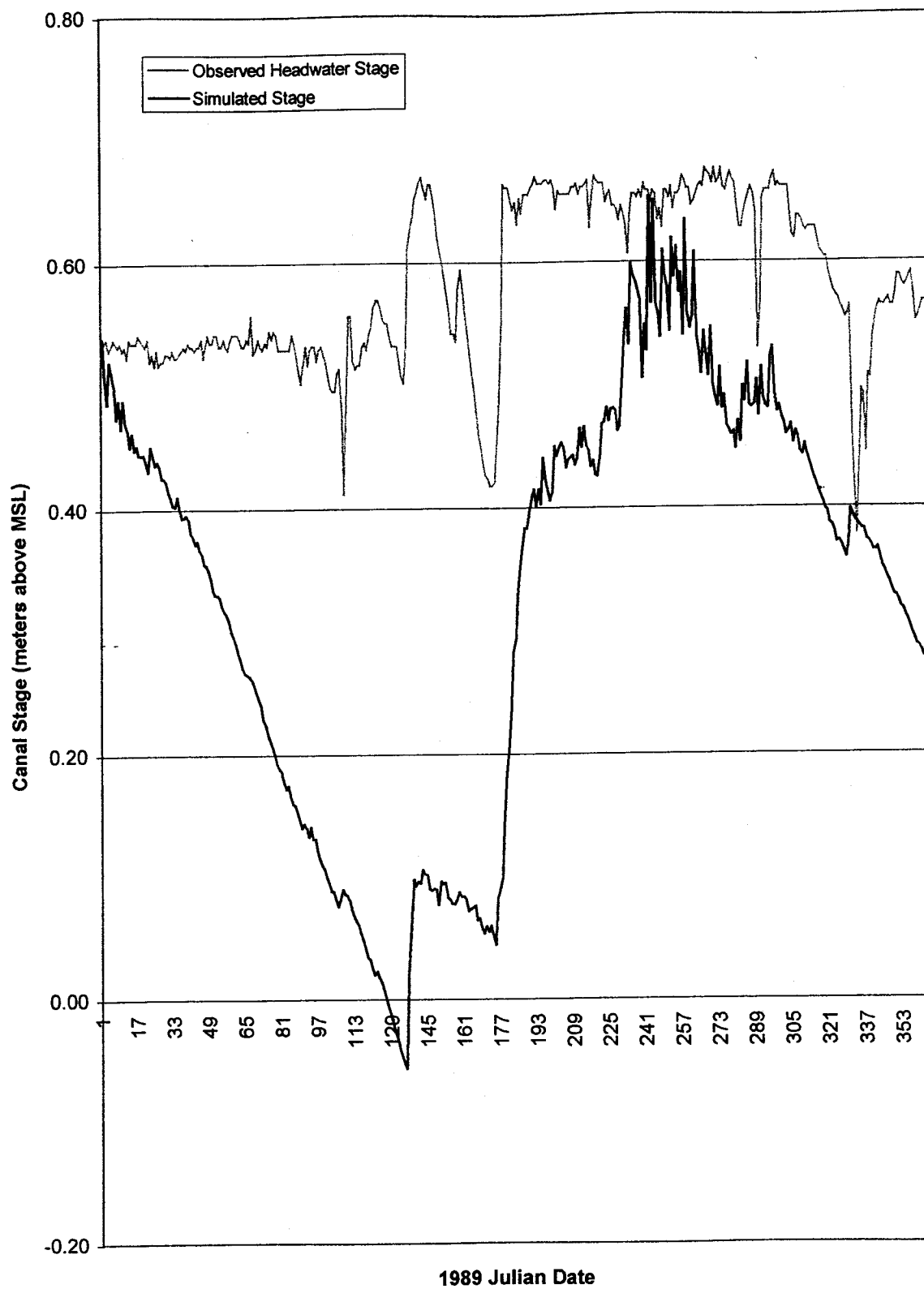


Figure 46. Simulated (black) versus observed (grey) canal stage at S-21A for 1989 rainfall year.

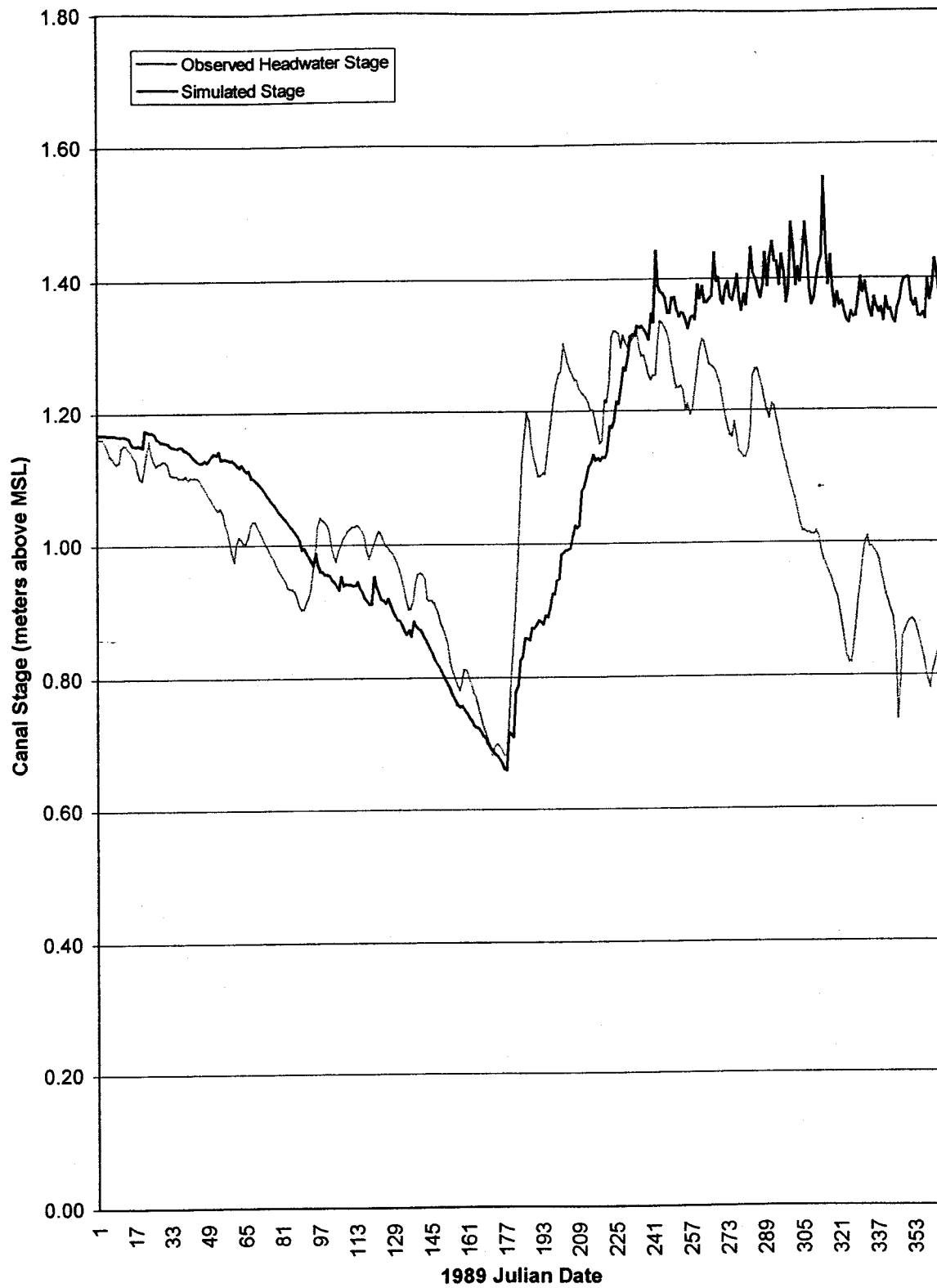


Figure 47. Simulated (black) versus observed (grey) canal stage at S-196 for 1989 rainfall year.

drier period at S-194 for the first 30 days of the wet season. Simulated stages were higher for 131 days of the dry season and 99 days of the wet season.

Figure 48 includes graphed time-histories of simulated and observed stages at S-20F for the 1989 rainfall year conditions. The effect of reduced rainfall was considerable at S-20F. Simulated canal stages dropped below sea level, as at S-21A, but for a longer time at S-20F. Simulated stages were lower than observed stages for every day of the simulation year. The discrepancies between simulated and observed values resulted in higher standard deviations than for S-20F during the calibration conditions. Dry season standard deviation was .45 meters, and the wet season standard deviation was .38 meters.

Simulated and observed 1989 condition groundwater elevations at G-757A, compiled into time-histories, are shown in Figure 49. The behavior was very similar to that seen in G-757A groundwater elevations during 1992 rainfall. Three significant differences in behavior occurred. First, simulated groundwater elevations during the first part of the dry season (days 1-150) ran about .45 meters lower than they had than during the 1992 conditions. The minimum value reached during the 1989 conditions was .10 meters above MSL, compared to a minimum value of .59 meters above MSL for the 1992 condition. The second difference was that the peak elevation reached during the wet season (2.43 meters above MSL) was more than a meter lower than that of the 1992 simulation (3.45 meters above MSL). Groundwater elevations from the 1989 and 1992 simulation runs ended the year with similar values. The simulated groundwater elevations had greater error than during the 1992 conditions. Standard deviations between simulated and observed elevations were .51 and .62 meters during the dry and wet seasons, respectively.

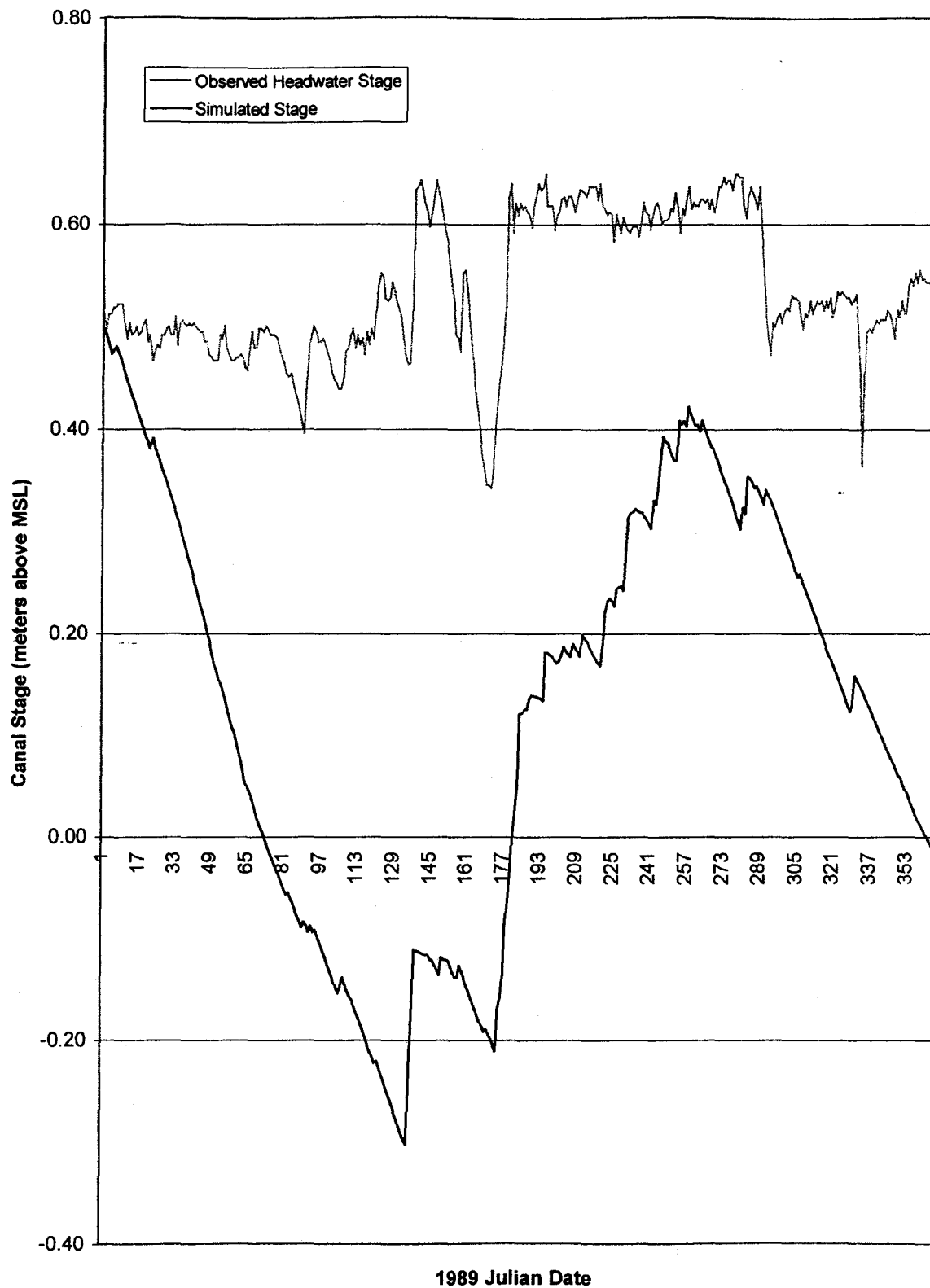


Figure 48. Simulated (black) versus observed (grey) canal stage at S-20F for 1989 rainfall year.

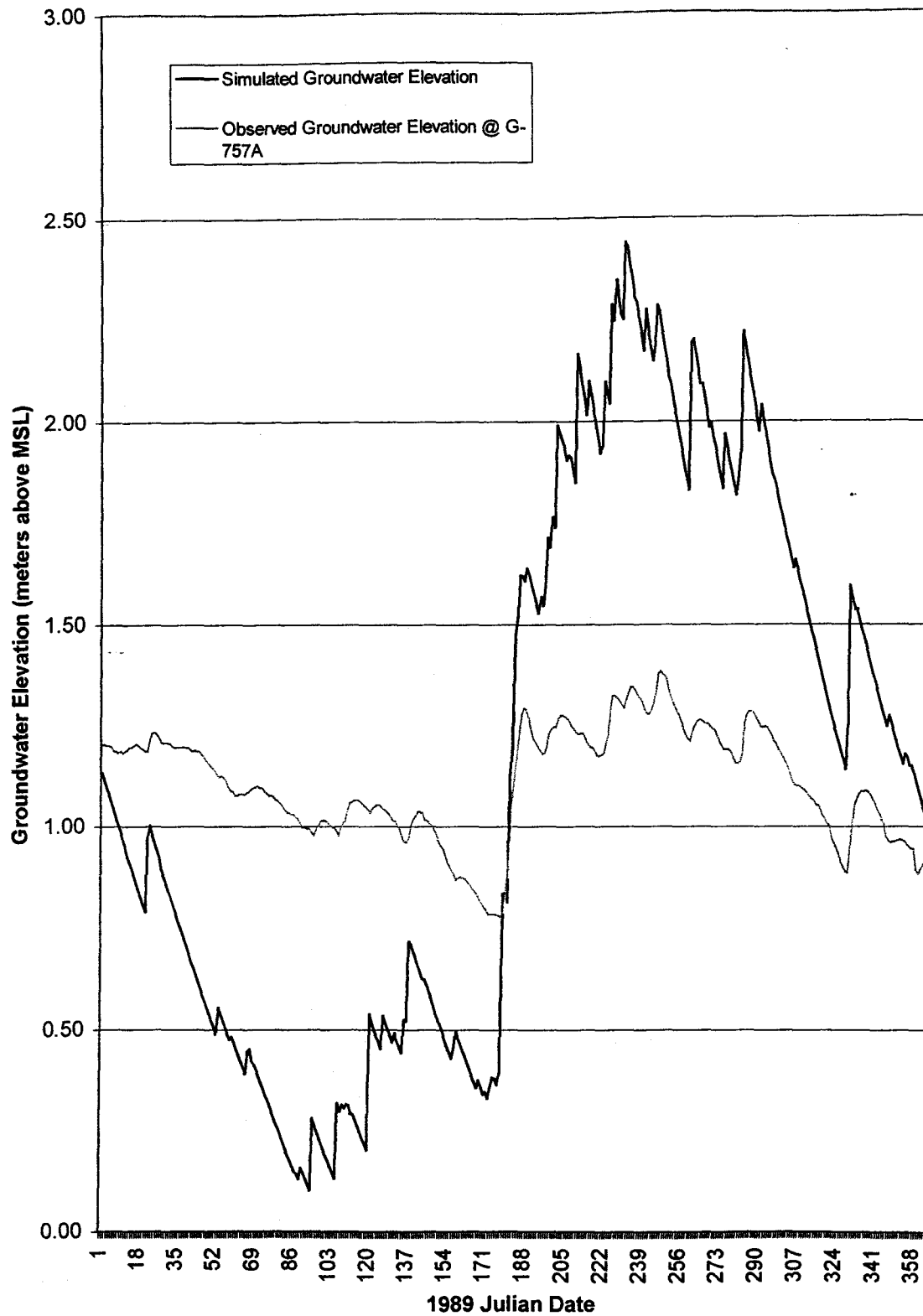


Figure 49. Simulated (black) versus observed (grey) groundwater elevations at G-757A for 1989 rainfall year.

Figure 50 shows the time-history of TP concentration at S-21A for 1989 rainfall conditions. Sudden increases in simulated TP concentrations occasionally occurred due to rainfall-spawned EMC load events. Simulated TP concentrations at S-21A averaged .005 mg/L during the dry season and .009 mg/L during the wet season, lower than for 1992 conditions. The maximum TP concentration reached was .024 mg/L, also lower than the peak reached under 1992 rainfall conditions.

Reducing the rainfall input had a similar influence at S-20F, as seen in Figure 51, which shows time-history of TP concentrations during 1989 rainfall. The average simulated dry season concentrations were lower and the simulated wet season concentrations were nearly equal to the simulated 1992 condition concentrations. Average simulated dry season TP concentration at S-20F was .006 mg/L, and wet season concentrations averaged .040 mg/L for the 1989 rainfall conditions. The peak concentration reached was .072 mg/L, which was lower than the highest 1992 simulation TP concentration at S-20F.

Figure 52 shows the time-history of simulated TSS concentrations at S-21A for 1989 rainfall conditions. Concentration values were depressed from the levels from the higher rainfall conditions of the calibration run. The overall temporal pattern was a general decline occasionally interrupted by slight concentration increases triggered by large rainfall events. The average simulated TSS concentration was 2.5 mg/L, and wet season concentrations averaged 1.3 mg/L. The highest TSS concentration reached was 4.2 mg/L, which was lower than the average concentration from the 1992 rainfall condition. The effects of reduced rainfall on simulated TSS concentration temporal patterns were similar at S-20F. Simulated TSS concentrations were also lower at S-20F,

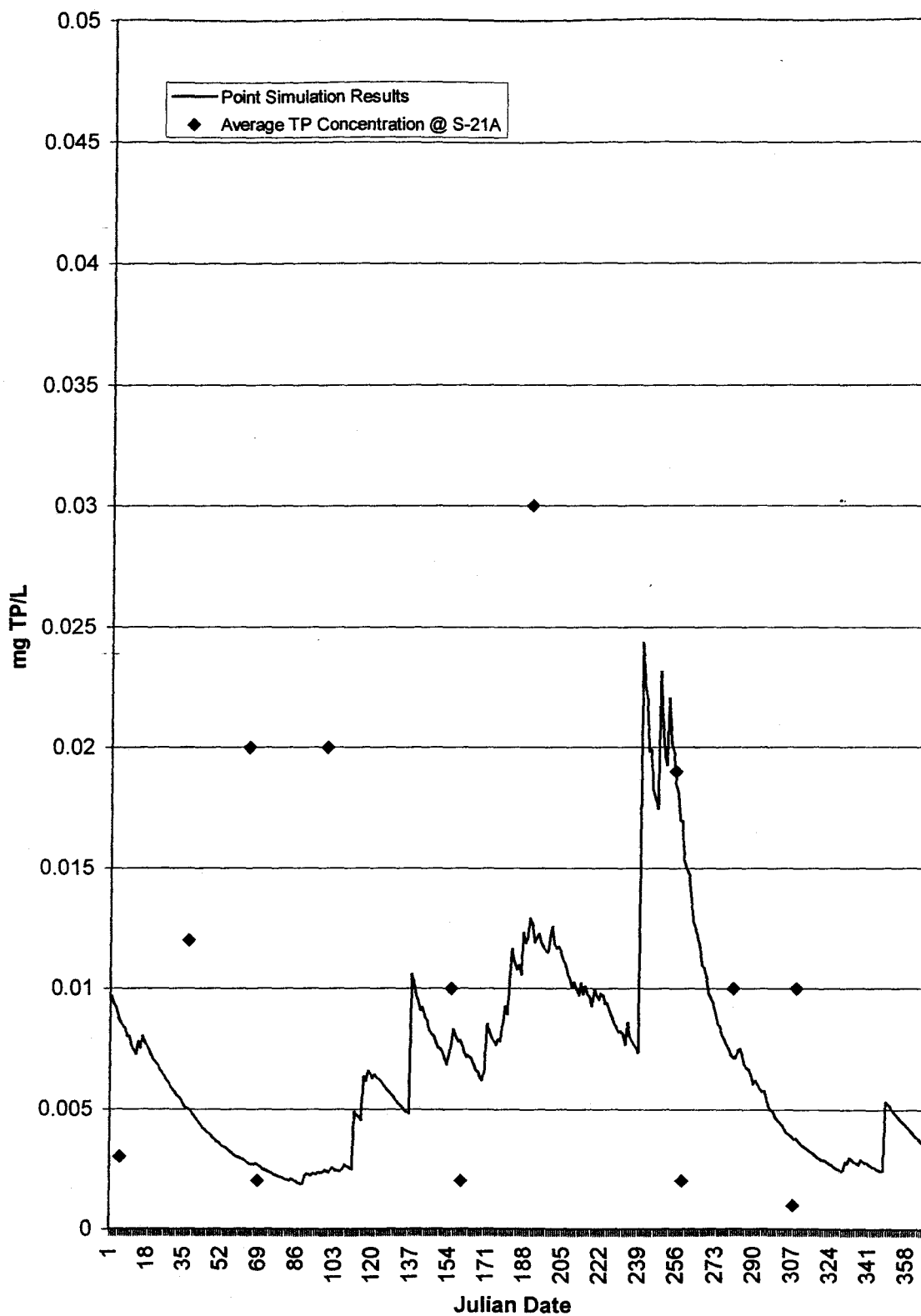


Figure 50. Simulated time-history of TP concentration versus individual observed concentration values at S-21A for 1989 rainfall year.

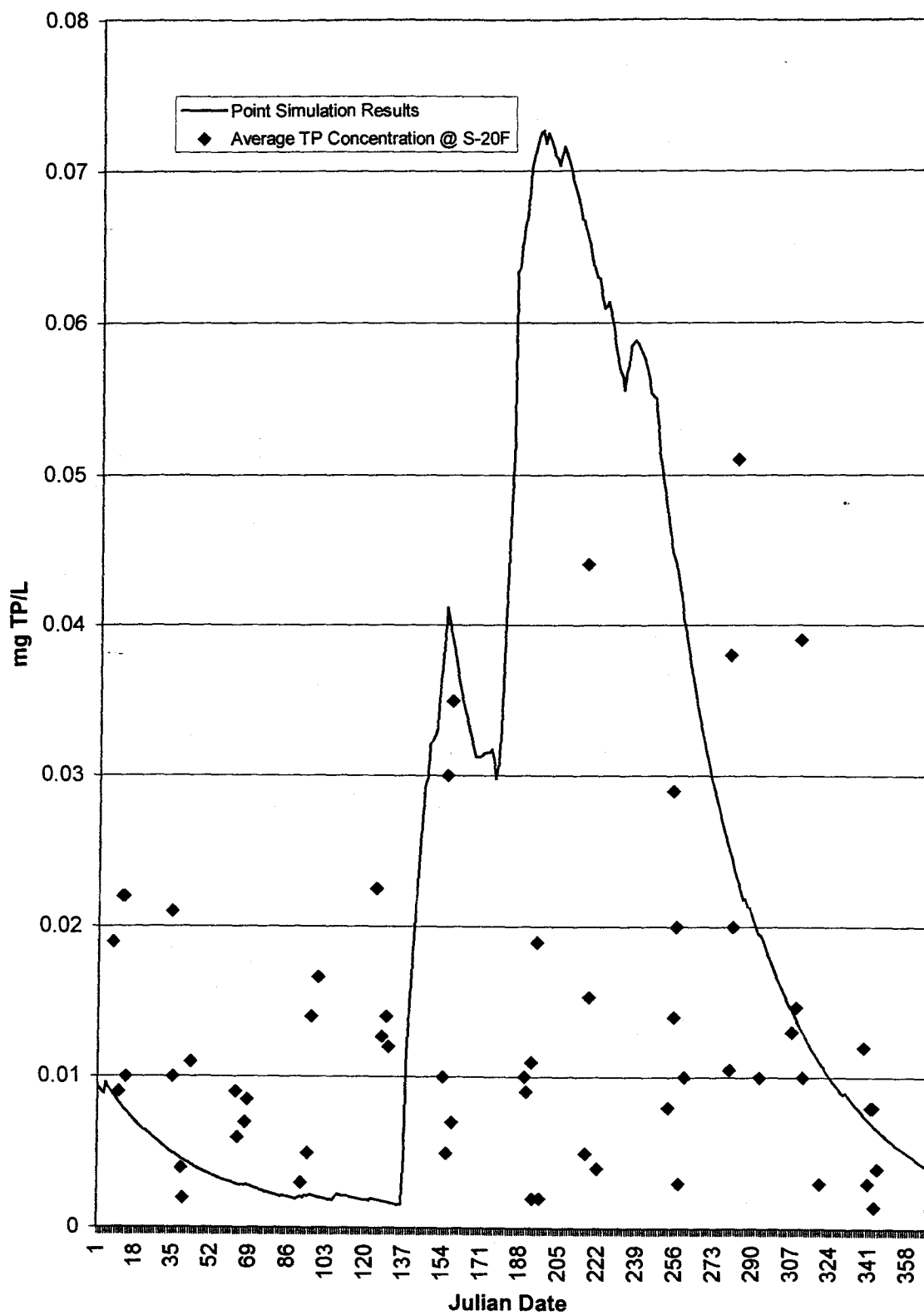


Figure 51. Simulated time-history of TP concentration versus individual observed concentration values at S-20F for 1989 rainfall year.

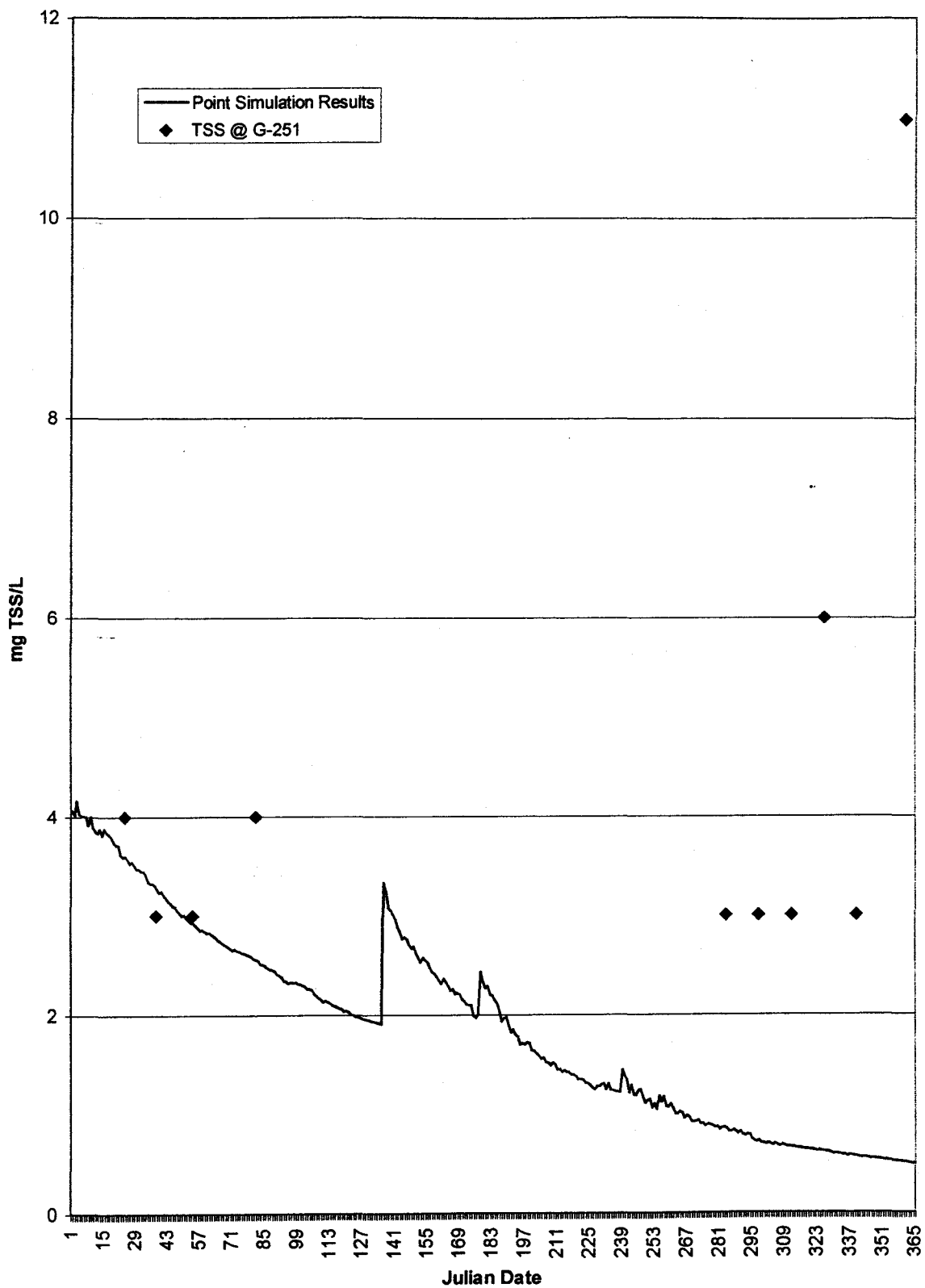


Figure 52. Simulated time-history of TSS concentration versus individual observed concentration values at S-21A for 1989 rainfall year.

as shown in Figure 53. The lower rainfall resulted in average S-20F simulated concentrations of 2.4 mg/L and 1.0 mg/L for the dry and wet seasons, respectively.

Time-histories of total daily flows of constituents discharged from the C-102 and C-103 sub-basins into Biscayne Bay are shown in Figures 54 (TP) and 55 (TSS). The temporal pattern of TP load discharged into the Bay was very similar for the two rainfall conditions. The total simulated load of TP from the study area into the Bay was $1.19 \text{ E}+06$ grams, which was 52.3% of the load from the average rainfall conditions. For 1989 rainfall conditions, simulated discharge of TSS into the ocean achieved its maximum early in the year, and then slowly declined. Total simulated load of TSS into the ocean was $1.94 \text{ E}+09$ grams, which was 52.9% of the average condition TSS load.

High Rainfall Conditions

Figures 56-66 show time-histories of simulated canal stages, groundwater elevations, constituent concentrations, and total watershed discharges into Biscayne Bay for the simulation of high rainfall conditions. Figure 56 displays the time-histories of simulated canal stages at S-194 for the high rainfall condition and the observed 1995 stages. The accuracy of the S-194 simulated stages was about equal to that of the simulated stages from the calibrated condition. Discrepancies between dry season simulated and observed stages slightly increased, with a standard deviation of .13 meters and values within .25 meters for 160 days. Wet season error metrics were nearly identical to those from average rainfall, with a standard deviation of .21 and S-194 stage values no more than .25 meters apart on 116 days.

Figure 57 shows the time-history of simulated and observed stages at S-21A for the high rainfall conditions. Simulated hydrologic responses were not as accurate as

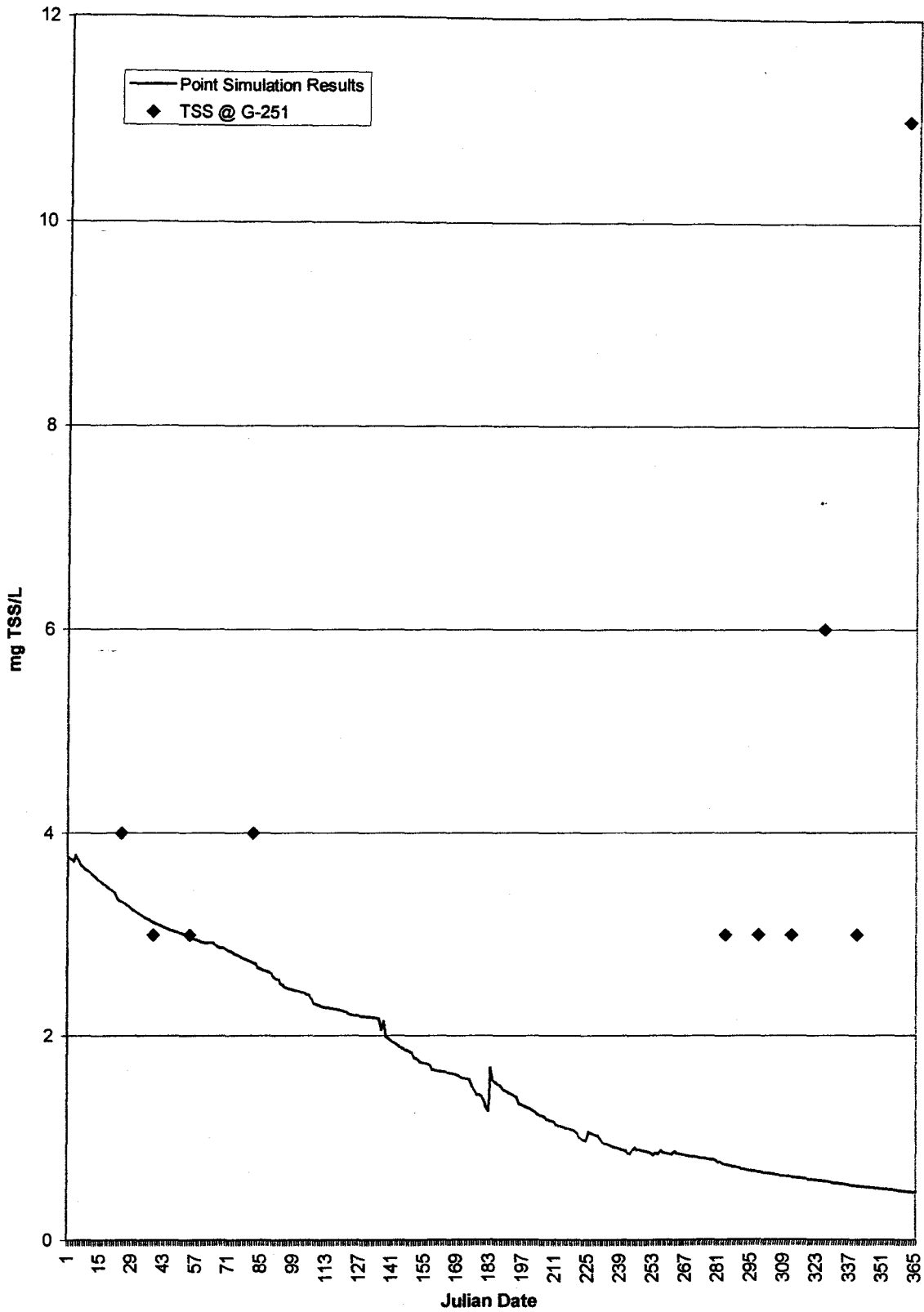


Figure 53. Simulated time-history of TSS concentration versus individual observed concentration values at S-20F for 1989 rainfall year.

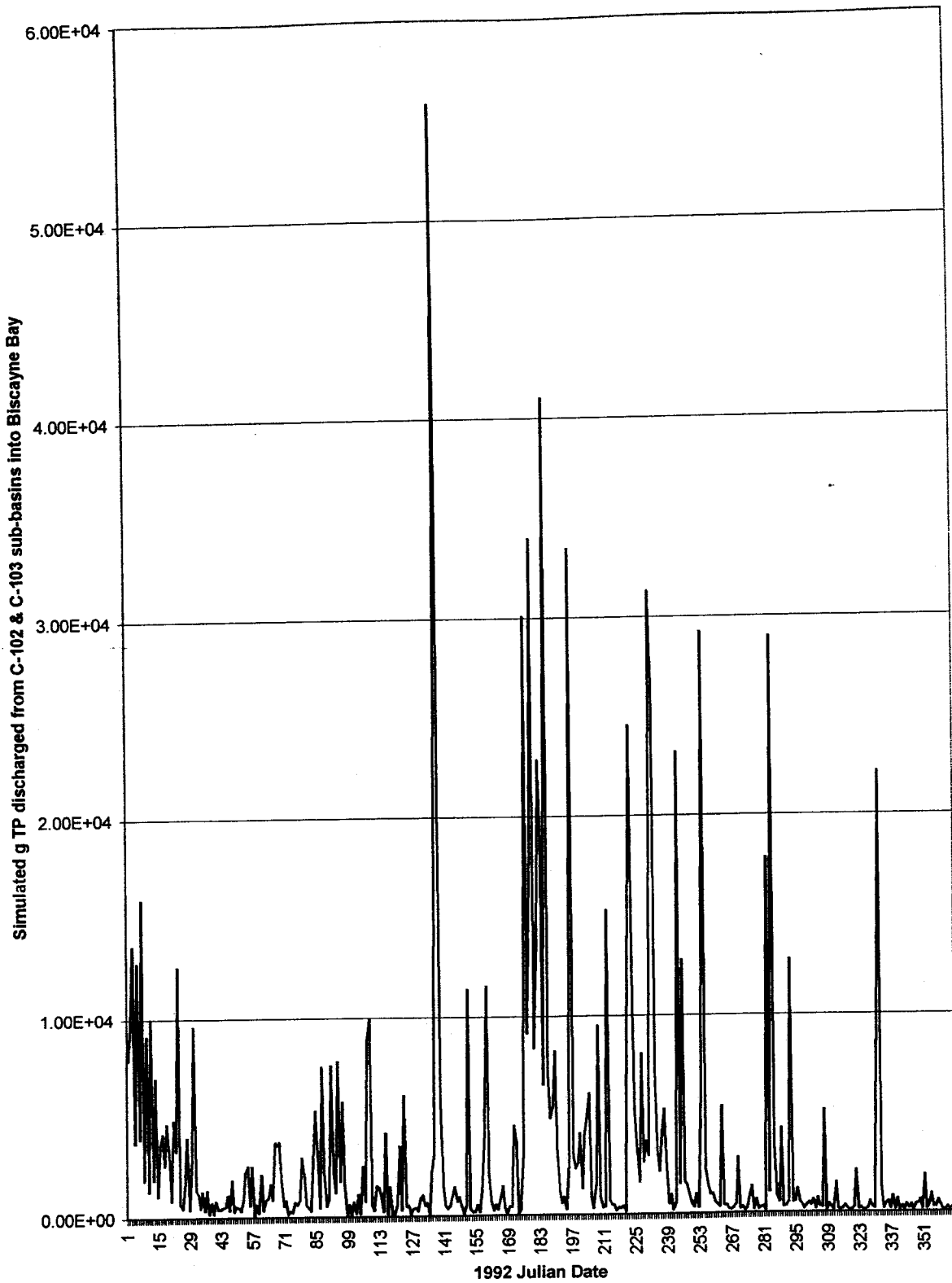


Figure 54. Time-history of simulated TP discharge from the C-102 and C-103 sub-basins into Biscayne Bay for 1989 rainfall year.

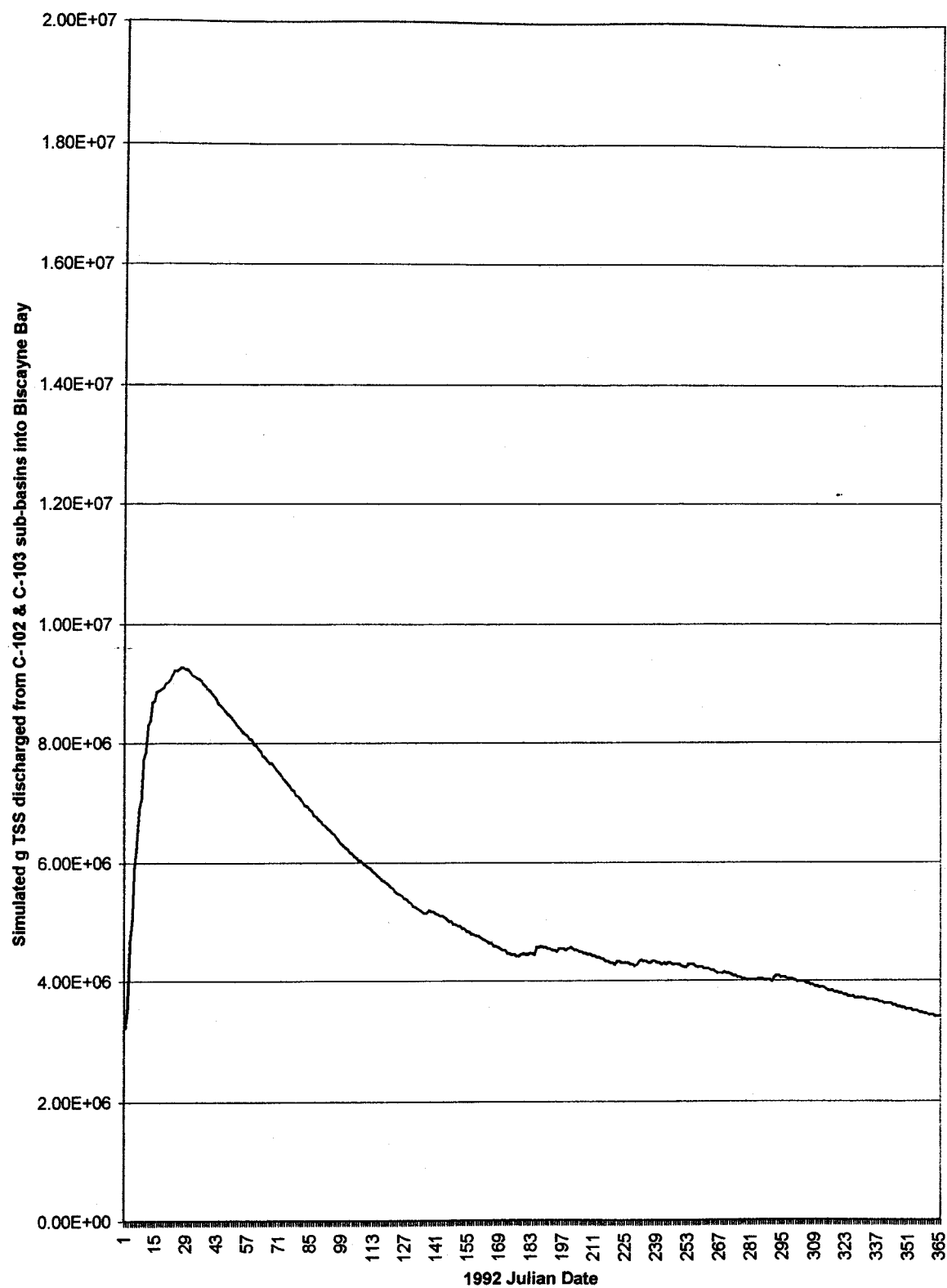


Figure 55. Time-history of simulated TSS discharge from the C-102 and C-103 sub-basins into Biscayne Bay for 1989 rainfall year.

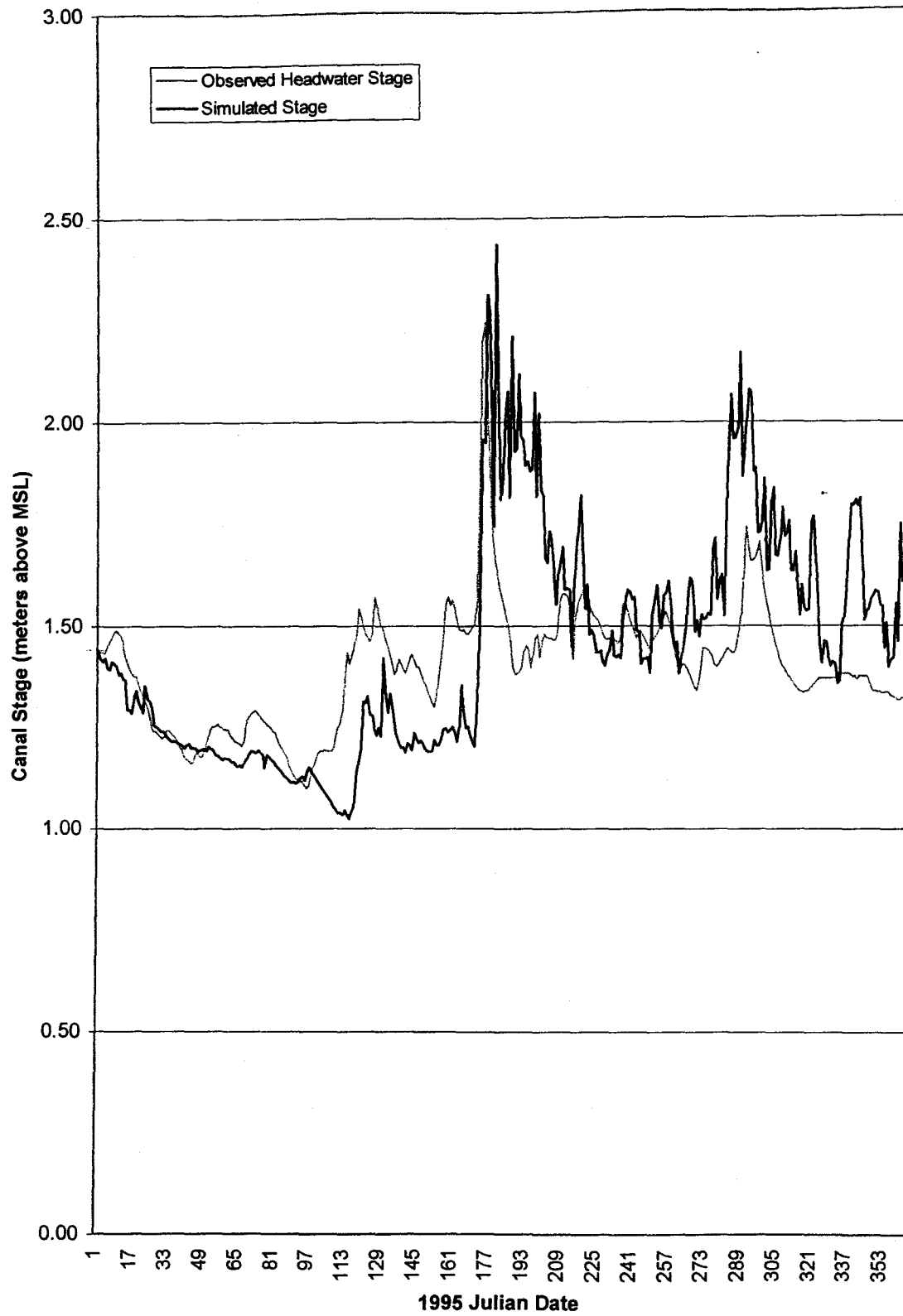


Figure 56. Simulated (black) versus observed (grey) canal stage at S-194 for 1995 rainfall year.

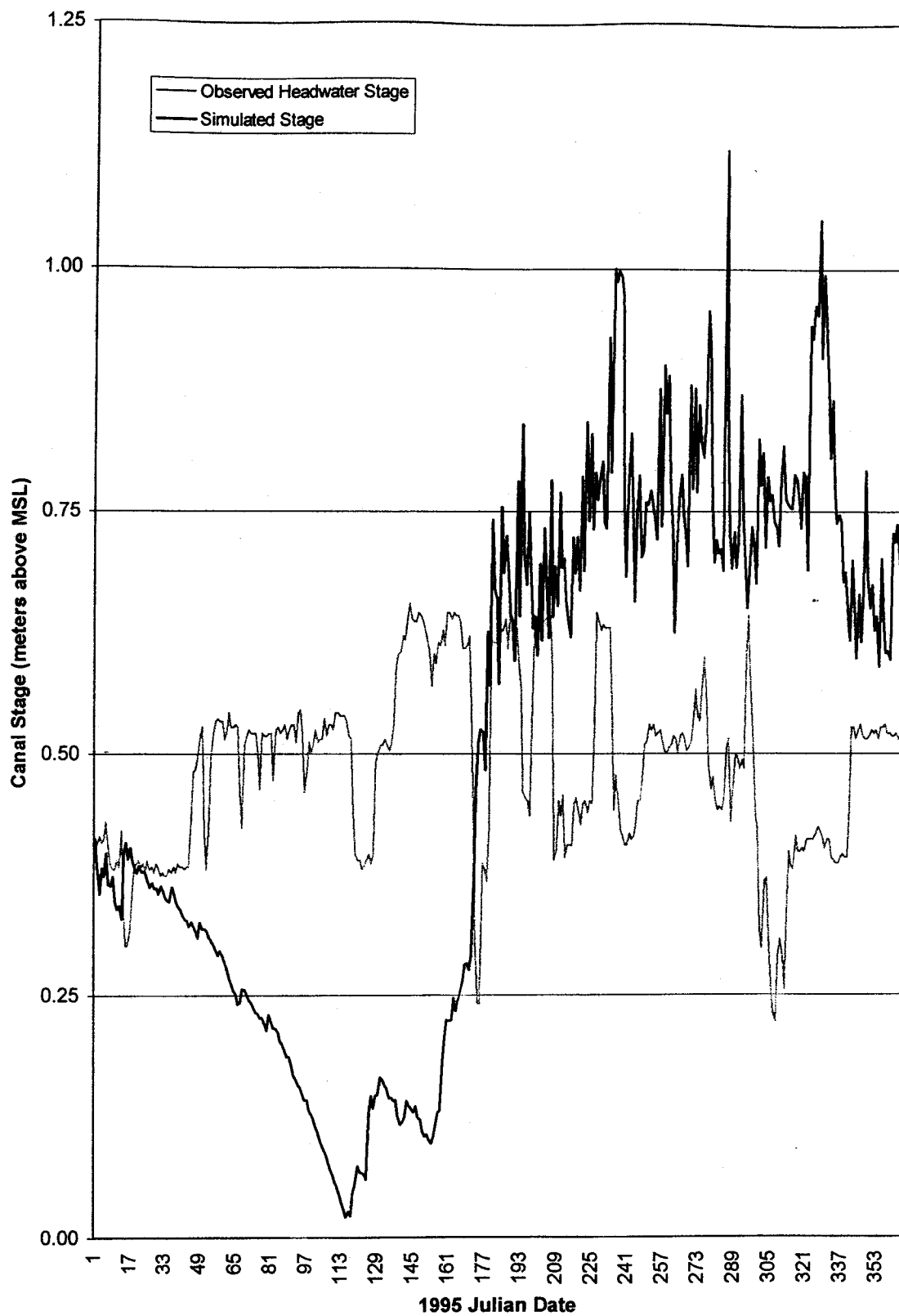


Figure 57. Simulated (black) versus observed (grey) canal stage at S-21A for 1995 rainfall year.

those of the calibrated condition. The simulated dry season stages were considerably lower than observed stages, and the day-to-day wet season water level fluctuations were greater. Standard deviations were .25 and .29 meters for the dry and wet seasons, respectively. The maximum simulated canal stage at S-20F was 1.12 meters above MSL, while the minimum simulated stage was .02 meters above MSL.

Figure 58 contains the time-histories of simulated and observed S-196 canal stages for 1995 rainfall conditions. Simulated stages were lower than observed values for 149 days of the dry season, and higher than observed for 156 days of the wet season. Hydrologic response to high rainfall was similar at S-196 to the calibrated conditions, but with greater error. Standard deviations between simulated and observed stages were .18 and .40 meters for the dry and wet seasons, respectively.

The time-history of simulated and observed canal stages at S-20F from the high rainfall condition runs is shown in Figure 59. Simulated stages during the early part of the dry season were lower than observed values, and for a brief period dropped below sea level to a minimum of .20 meters below MSL. Dry season simulated values were usually lower than observed values on 159 days, and wet season simulated values were generally (143 days) higher than observed stages. Unlike most points, differences between observed and simulated values decreased during the wet season. The standard deviations between simulated and observed stages at S-20F were .33 and .22 meters for the dry and wet seasons, respectively.

Figure 60 displays the time-histories of simulated and observed groundwater elevations at G-757A for the 1995 rainfall conditions. The temporal pattern of groundwater elevations was slightly different from the 1992 conditions, as the simulated

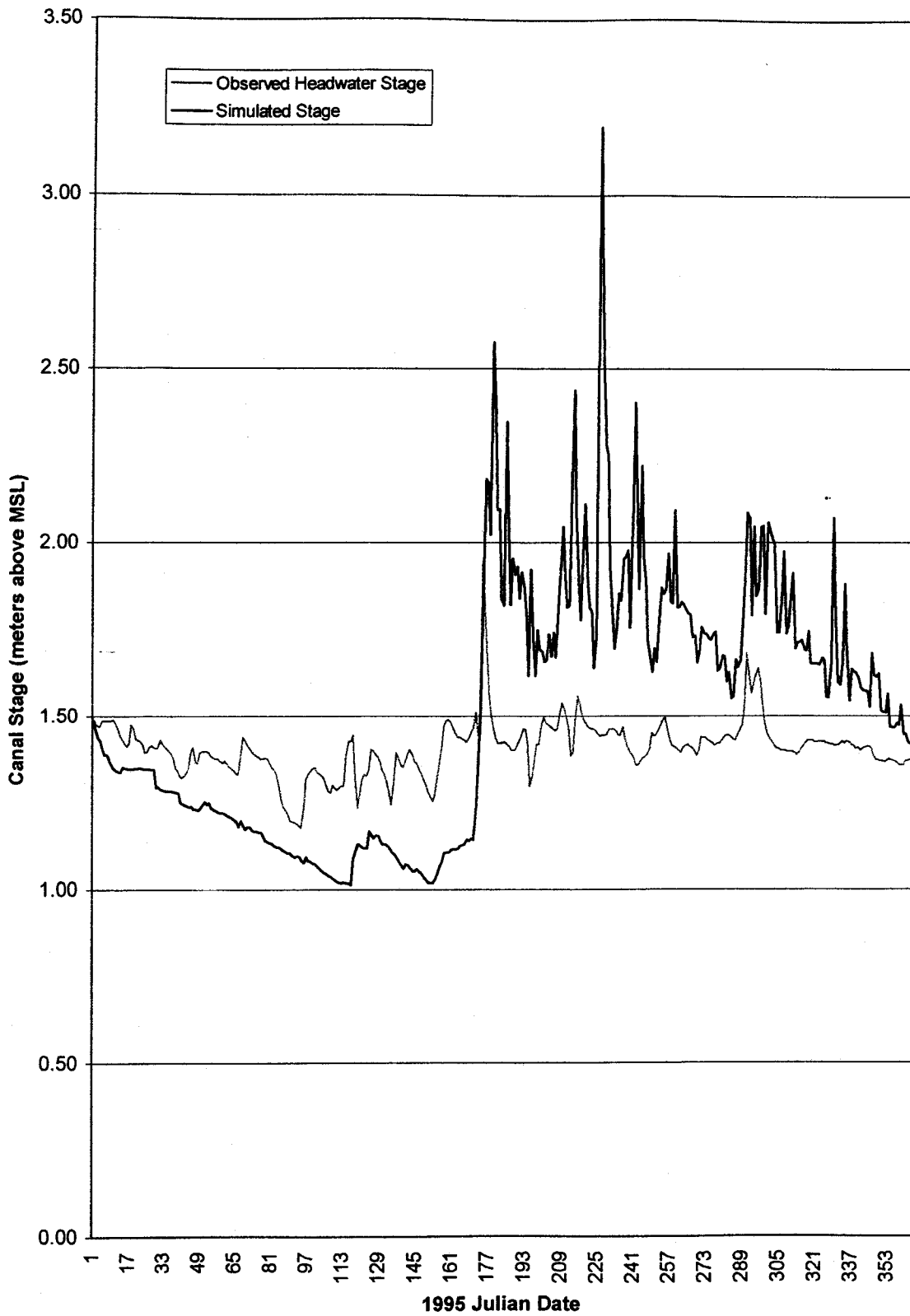


Figure 58. Simulated (black) versus observed (grey) canal stage at S-196 for 1995 rainfall year.

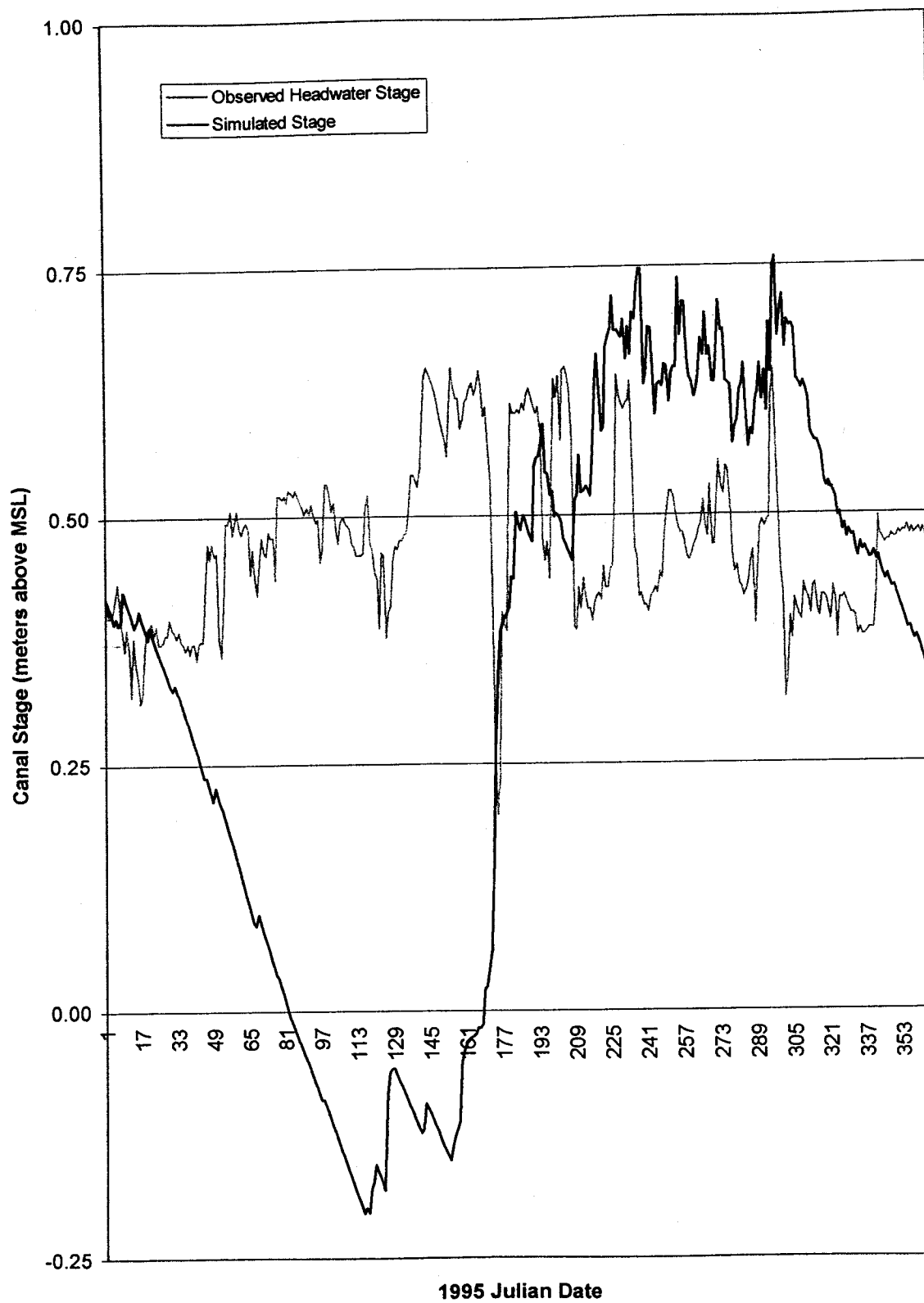


Figure 59. Simulated (black) versus observed (grey) canal stage at S-20F for 1995 rainfall year.

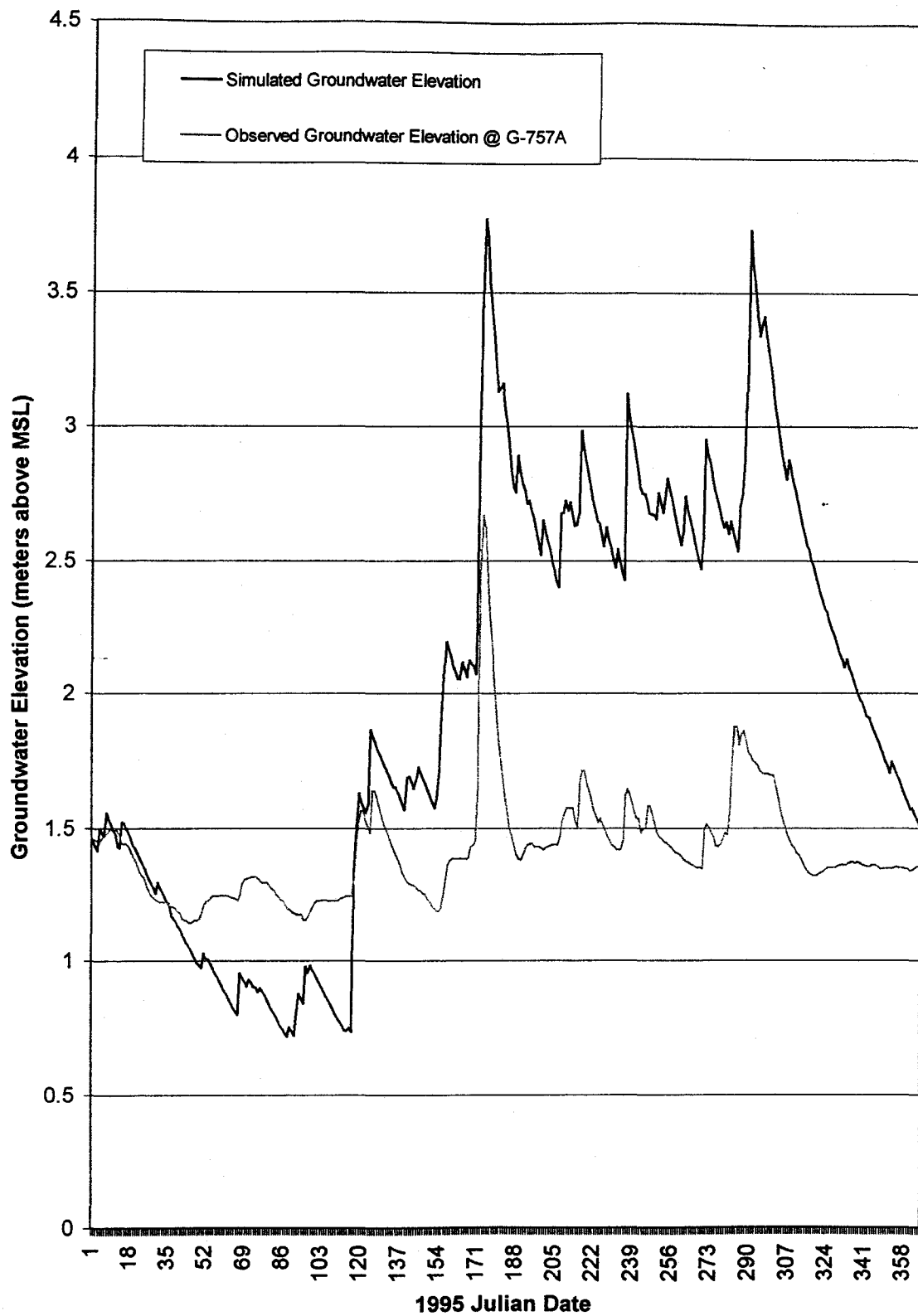


Figure 60. Simulated (black) versus observed (grey) groundwater elevations at G-757A for 1995 rainfall year.

and observed stages were slightly higher during 1995 conditions. Maximum 1995 rainfall groundwater elevations were 2.67 and 3.77 meters above MSL for observed and simulated values, respectively (compared to 2.15 and 3.45 meters above MSL for observed and simulated 1992 elevations). In addition, simulated elevations were higher than the observed values a greater percentage of the time under conditions of higher rainfall. The degree to which simulated elevations were overestimated was also larger. Simulated dry season groundwater elevations were higher than observed values for 97 days, with an average standard deviation of .29 meters. Simulated groundwater elevations were higher than observed values for the entire wet season, with a larger standard deviation of 1.15 meters.

Figure 61 shows the time-history of TP concentration at S-21A for 1995 rainfall conditions. Concentration fluctuations increased during the higher rainfall conditions, and there was an increase in peak TP concentration. Simulated TP concentrations at S-21A averaged .005 mg/L during the dry season (lower than the 1992 condition concentration of .009 mg/L) and .019 mg/L during the wet season (higher than the 1992 condition concentration of .010 mg/L). The maximum TP concentration registered was .072 mg/L, higher than the peak reached under 1992 rainfall conditions.

Figure 62 shows time-history of TP concentrations during 1995 rainfall at S-20F. The overall temporal pattern was very similar to that seen in the calibrated simulation results, with lower dry season concentrations followed by a sharp, sustained concentration increase. During the 1995 rainfall simulation, the duration of elevated concentrations was longer, with values remaining above .03 mg/L for 193 days. As at S-21A, the higher rainfall resulted in wider fluctuations in TP concentration and in higher

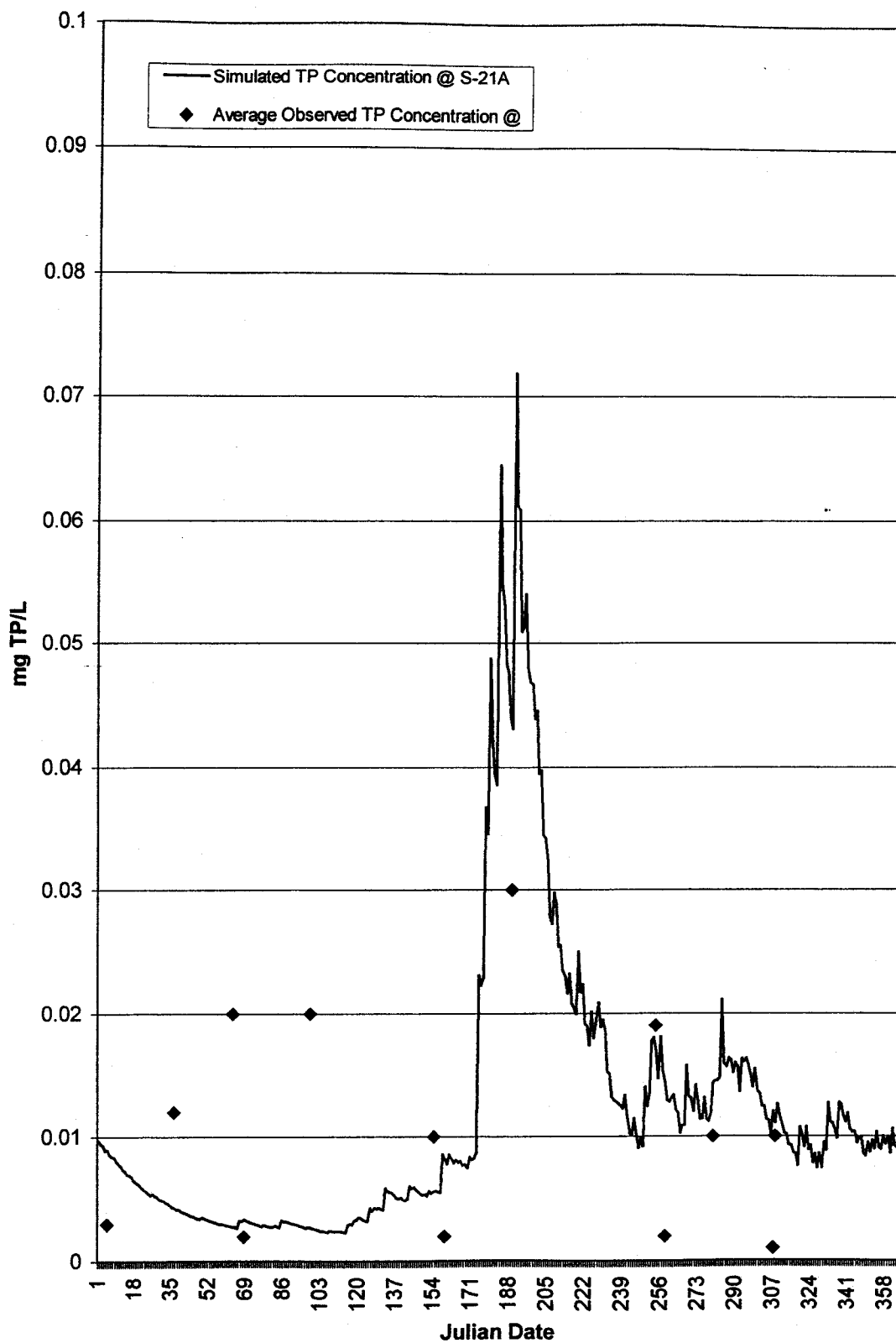


Figure 61. Simulated time-history of TP concentration versus individual observed concentration values at S-21A for 1995 rainfall year.

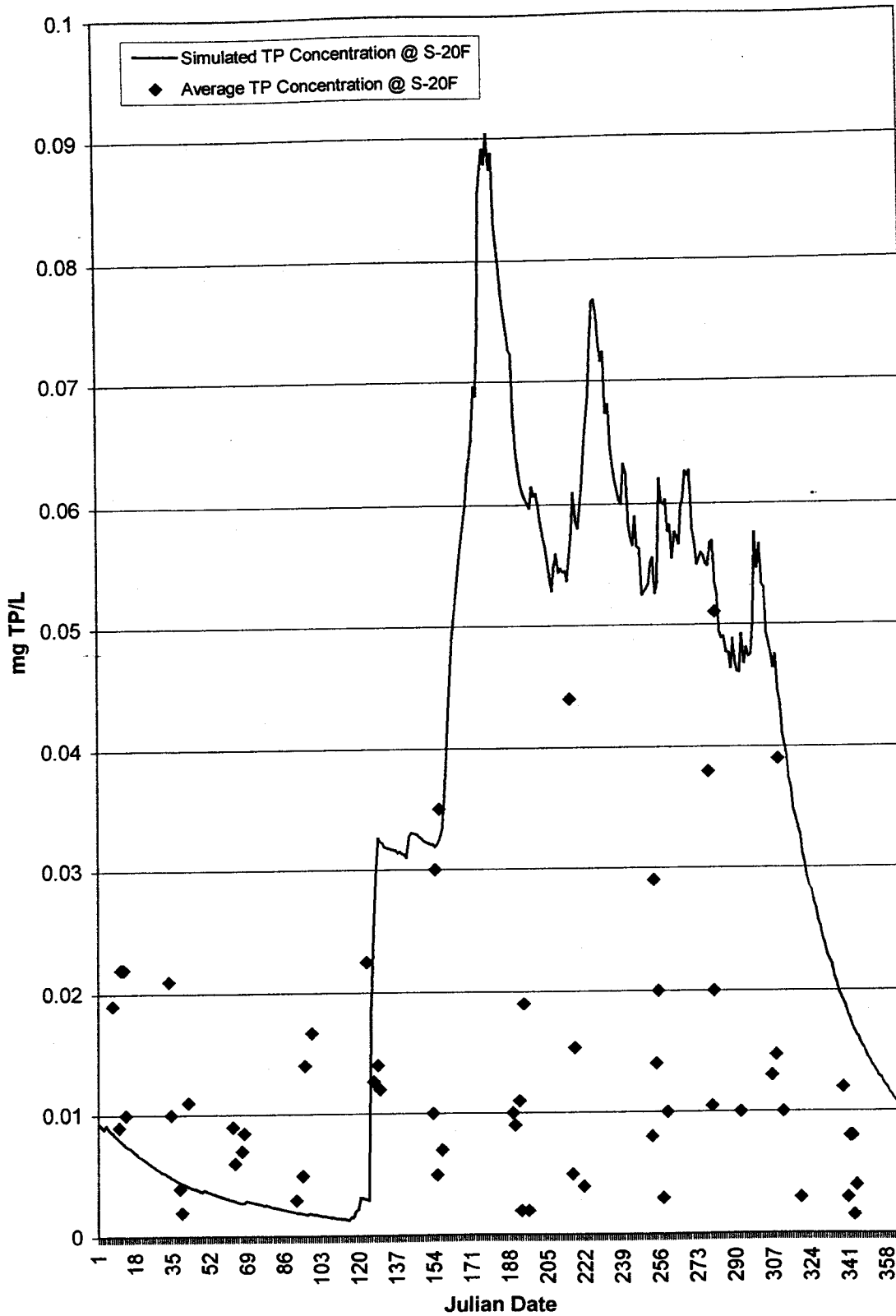


Figure 62. Simulated time-history of TP concentration versus individual observed concentration values at S-20F for 1995 rainfall year.

average and maximum values. The peak concentration reached during the wet season was .090 mg/L, which was higher than the maximum 1992 simulation S-20F TP concentration. The average simulated dry season TP concentration at S-20F was .010 mg/L, and wet season concentrations averaged .055 mg/L for the 1995 rainfall conditions.

Figure 63 shows the time-history of simulated TSS concentrations at S-21A for 1995 rainfall conditions. The temporal pattern of the concentration values was composed of a general early dry season decline followed by occasional periods of rapid increases during the latter half of the year. Average and maximum simulated 1995 rainfall S-21A TSS concentrations were reduced from the levels found in the lower rainfall conditions of the calibration run. Average dry season S-21A TSS concentration was 3.2 mg/L, and wet season concentrations averaged 3.0 mg/L. The highest TSS concentration reached was 8.9 mg/L, which was lower than the average concentration from the 1992 rainfall condition.

Simulated TSS concentrations were also lower than they were during than 1992 rainfall conditions at S-20F, as shown in Figure 64. A gradual drop in concentrations occurred throughout the simulation year. The higher rainfall resulted in average S-20F simulated concentrations of 2.2 mg/L and 2.6 mg/L for the dry and wet seasons, respectively. Maximum simulated S-20F TSS concentration for high rainfall conditions was 3.8 mg/L.

Time-histories of total high rainfall condition simulated daily flows of constituents discharged from the C-102 and C-103 sub-basins into Biscayne Bay are shown in Figures 65 (TP) and 66 (TSS). The temporal pattern of TP load discharged into

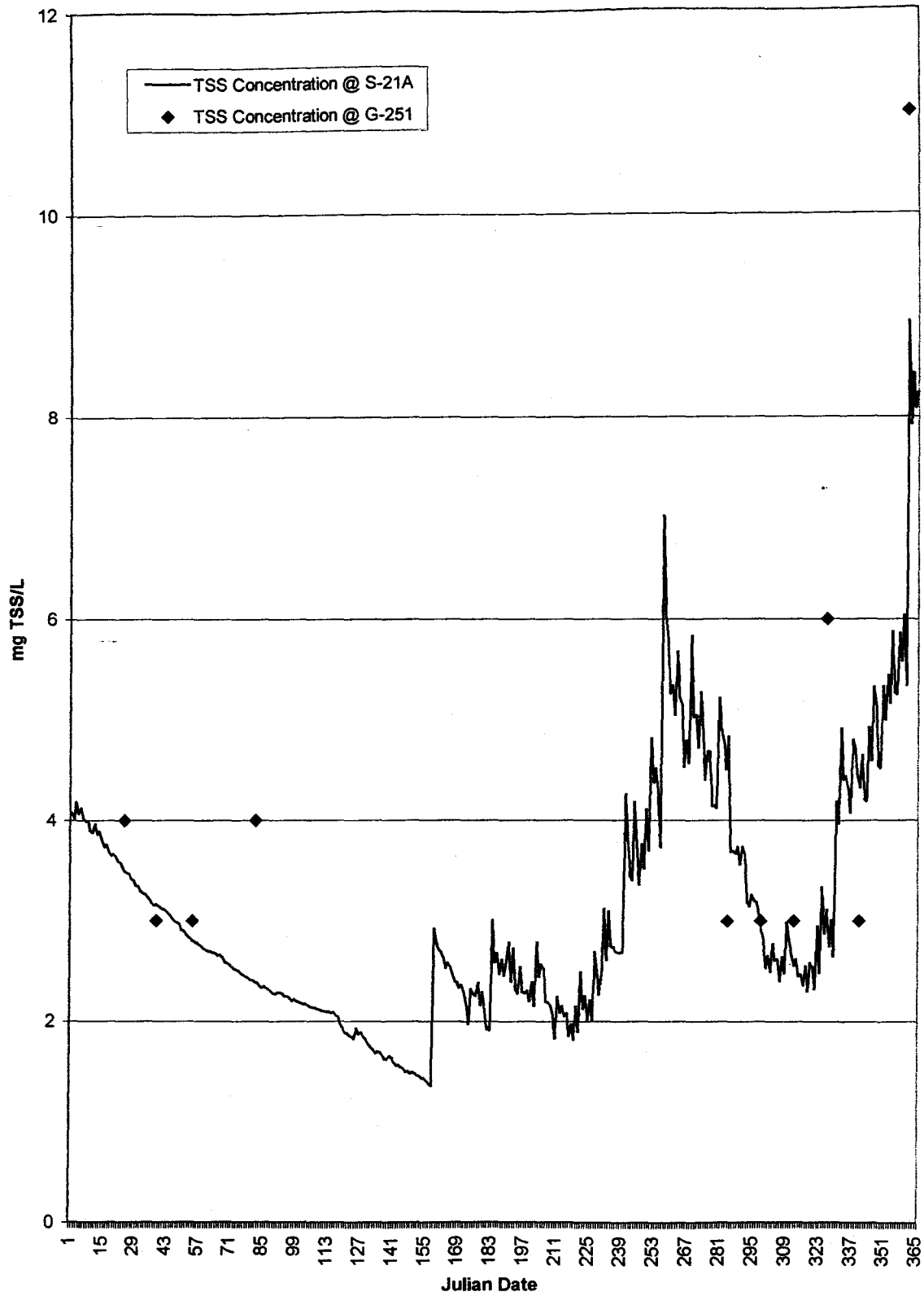


Figure 63. Simulated time-history of TSS concentration versus individual observed concentration values at S-21A for 1995 rainfall year.

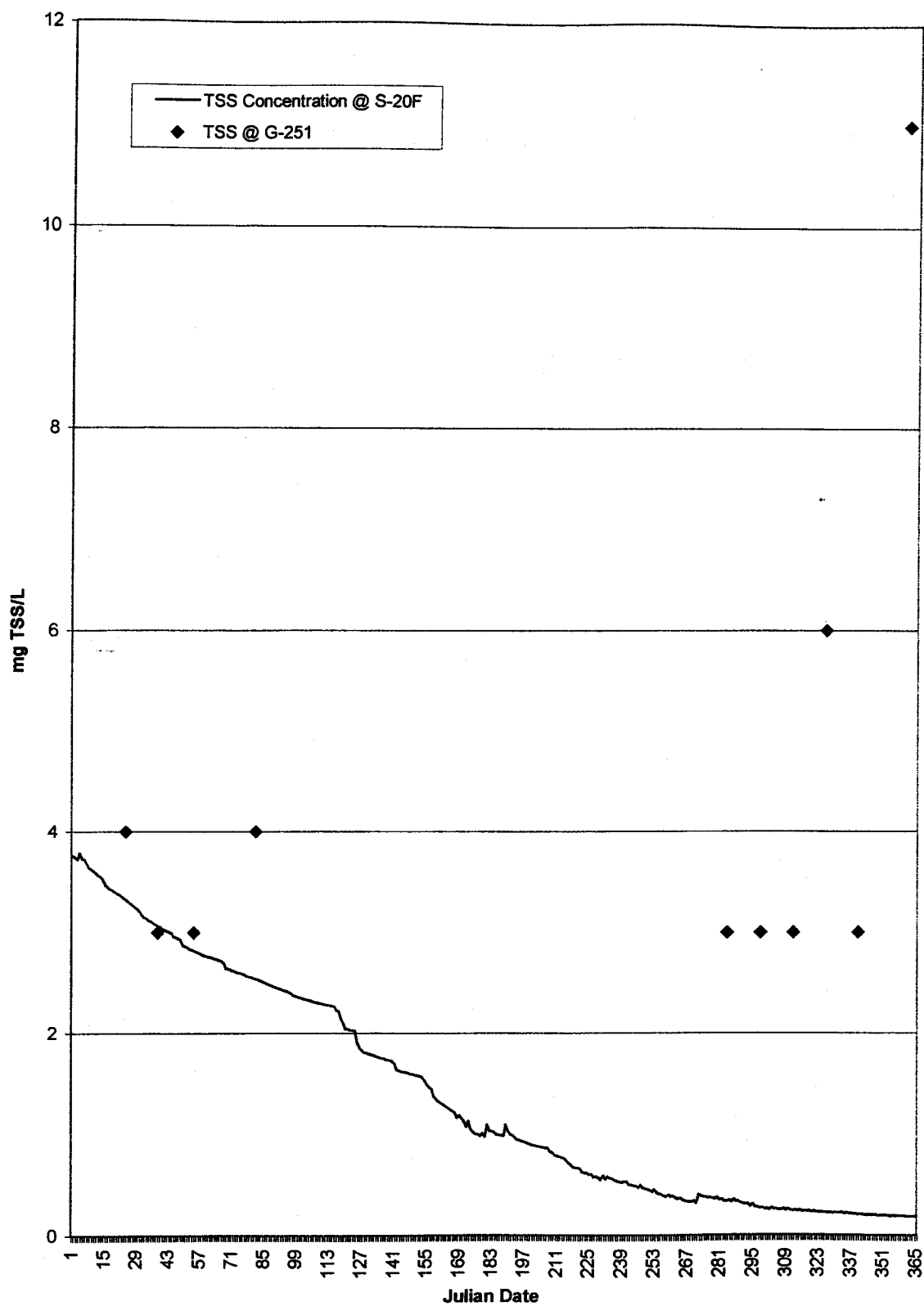


Figure 64. Simulated time-history of TSS concentration versus individual observed concentration values at S-20F for 1995 rainfall year.

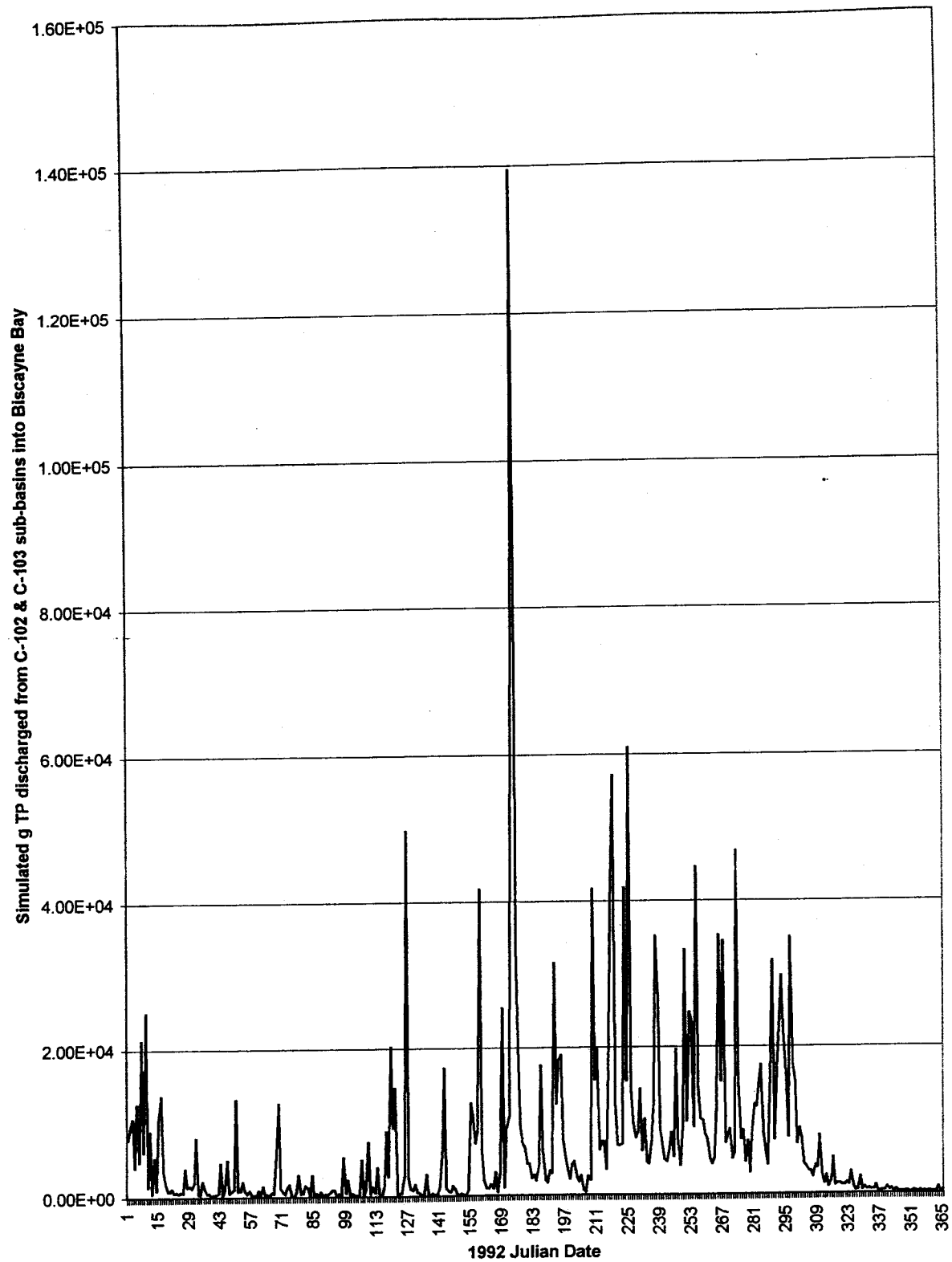


Figure 65. Time-history of simulated TP discharged from the C-102 and C-103 sub-basins into Biscayne Bay for 1995 rainfall year.

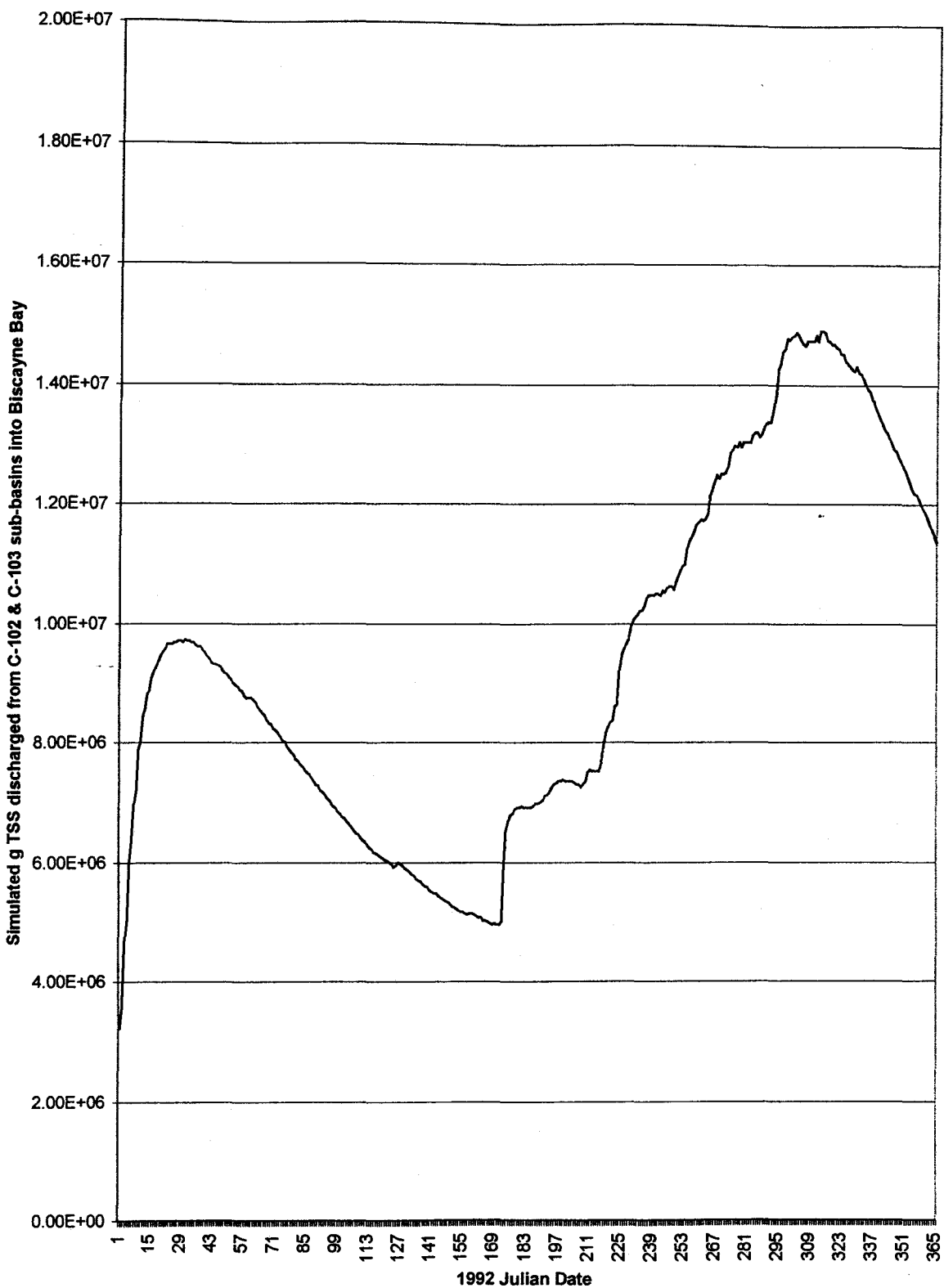


Figure 66. Time-history of simulated TSS discharged from the C-102 and C-103 sub-basins into Biscayne Bay for 1995 rainfall year.

the Bay was very similar to the patterns for the other two rainfall conditions. The total simulated load of TP from the study area into the Bay was $2.52 \text{ E}+06$ grams, which was 111.2% of the load from the average rainfall conditions. The temporal profile of total simulated high rainfall TSS discharge into the ocean resembled the low rainfall condition discharge during the dry season. However, unlike the low rainfall condition simulation, 1995 rainfall conditions increased TSS discharge throughout the wet season (300% between simulation dates 172 and 303). Total simulated load of TSS into the ocean was $3.42 \text{ E}+09$ grams, which was 92.9% of the average condition TSS load.

Cross-Rainfall Year Comparisons

The results of additional assessments conducted to better understand the influence of rainfall on temporal and spatial hydrology and water quality patterns are included below. Time-histories of canal stage, watershed storage, and concentration results from all three rainfall year conditions were graphed together for comparison. Furthermore, simulation output grids from each rainfall condition were compared to one another to assess effects of rainfall conditions on the spatial distributions of water and constituents.

The cumulative rainfall that had fallen by each day of the year, using both the S-21A and S-337 rain gauge data, is shown in Figure 67. 1989 rainfall cumulative values were lower on every day of the simulation year. However, although annual rainfall was highest during 1995, cumulative 1992 rainfall totals were highest for all but two of the first 224 days. The cumulative rainfall comparison revealed that dry season precipitation was highest in 1992. The annual total from 1995 was highest because of greater wet season rainfall.

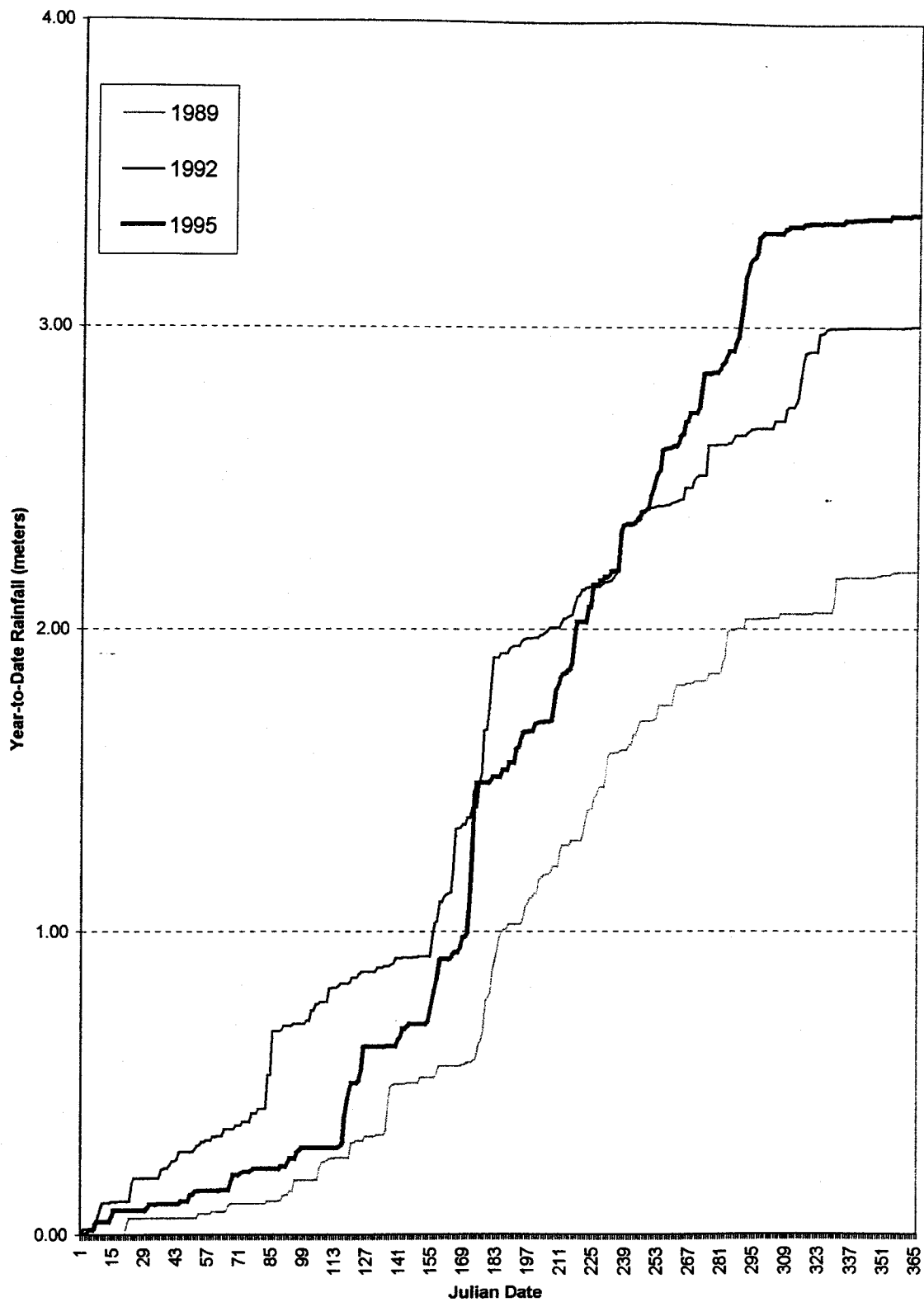


Figure 67. Time-histories of observed cumulative rainfall at S-21A and S-337 stations for 1989, 1992, and 1995.

The effect the rainfall volume and seasonality had on surface hydrology can be seen in the time-histories of simulated canal stage at S-21A, shown in Figure 68. During the dry season, simulated S-21A canal stages were highest during 1992. Low rainfall briefly dropped simulated stages during both 1989 and 1995 to near or below MSL. The increased wet season rainfall in 1995 resulted in the highest simulated dry season stage values of the three years.

The spatial effects of rainfall of surface hydrology were investigated next. The calculated hydroperiod index from the 1992 rainfall simulation is shown in Figure 69. The lower index values indicate areas where standing surface water was not as common. As would be expected, the highest index values corresponded to cells in the canal network and Biscayne Bay. Low spots adjacent to canals or urban areas had intermediate values.

The hydroperiod index calculated for the 1989 rainfall simulation is shown in Figure 70a. The general spatial pattern was comparable to the 1992 hydroperiod index grid. Some of the intermediate areas were somewhat smaller in spatial extent and had smaller index values. To better identify spatial and magnitude changes between the two conditions, the 1992 index grid was subtracted from the 1989 grid, and is shown in Figure 70b. Cells with negative index values had less frequent simulated standing surface water. In other words, the lowest index values indicated locations that were drier during the lower rainfall conditions. Most of the areas that were drier when annual rainfall was lower were in the southern and eastern parts of the study area, between the sandy ridge and the ocean.

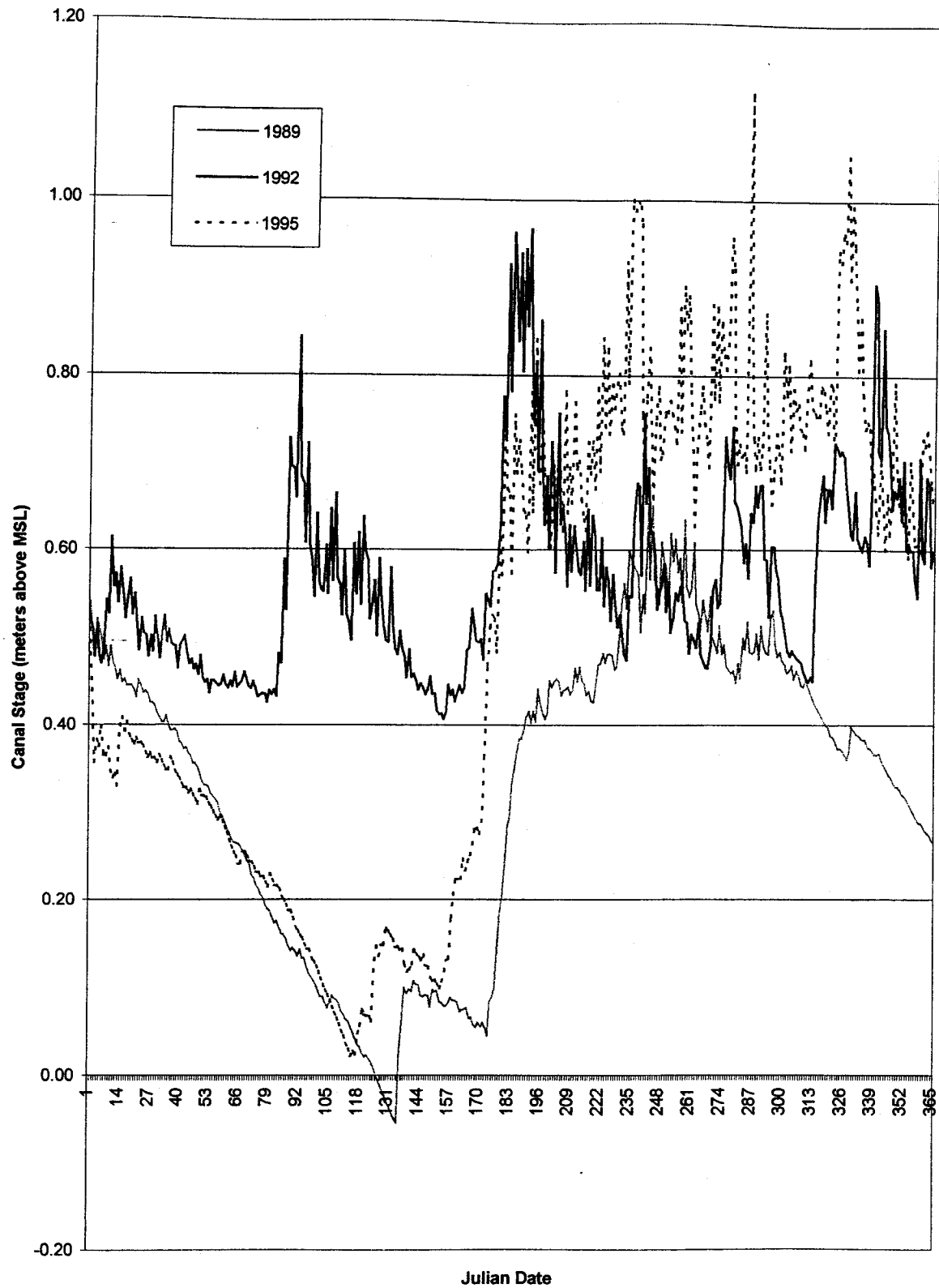


Figure 68. Time-histories of simulated canal stage at S-21A for low (1989), average (1992), and high (1995) rainfall conditions.

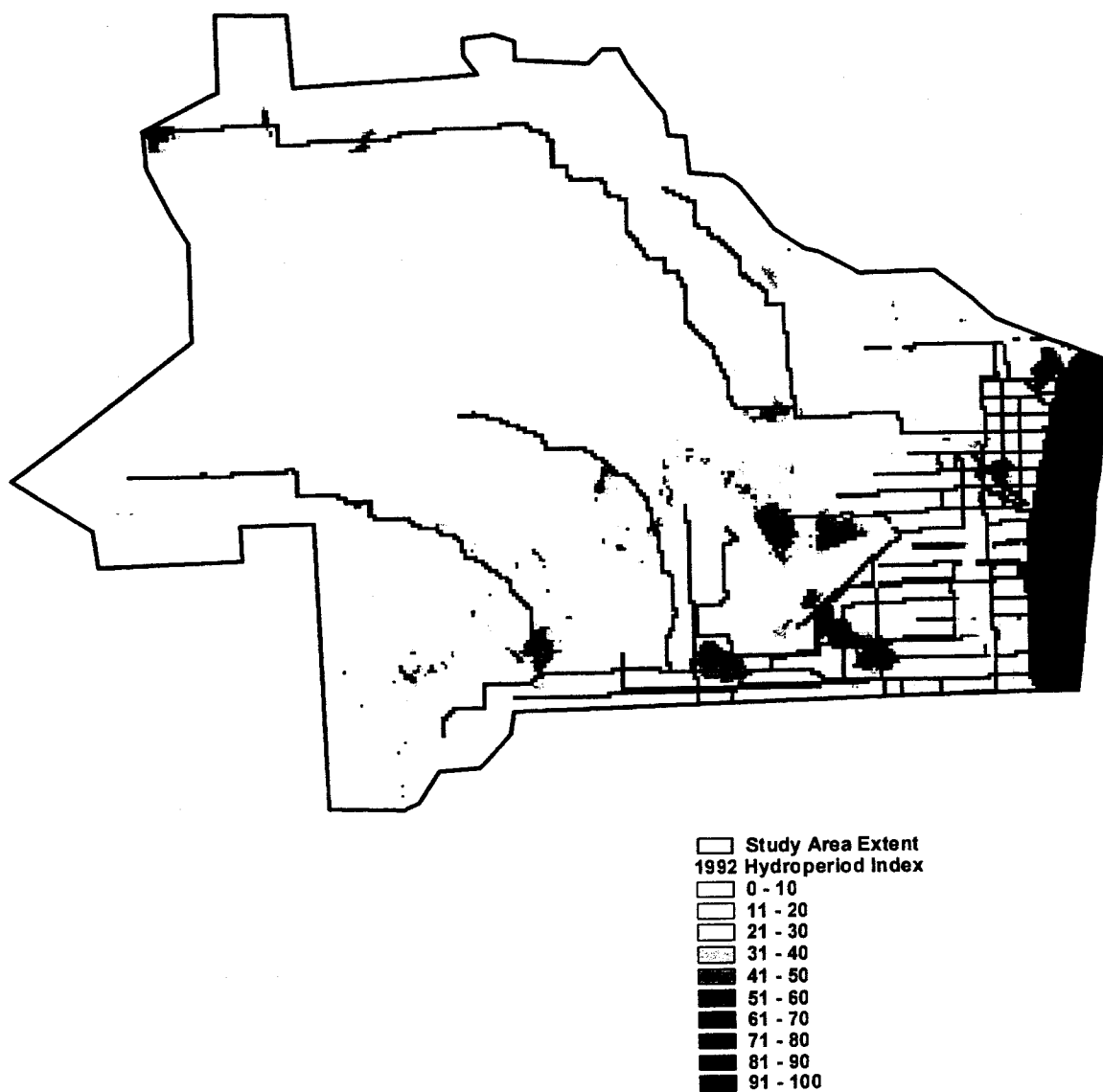


Figure 69. Hydroperiod index grid from simulation with 1992 rainfall conditions. Index values indicate percentage of year surface water storage occurred in a cell.

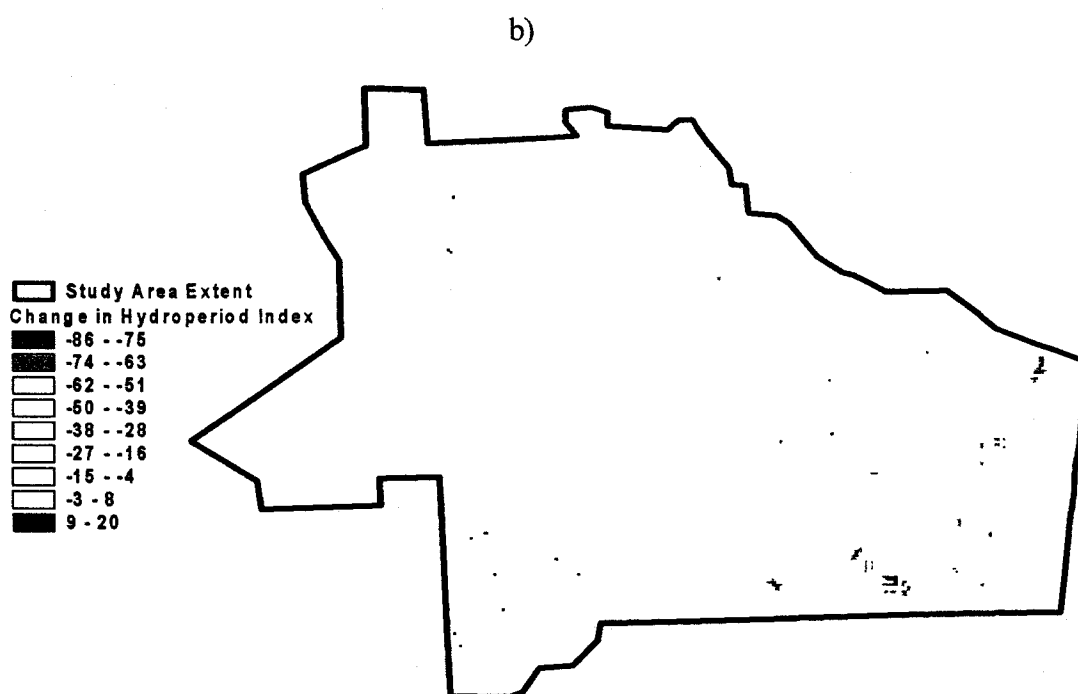
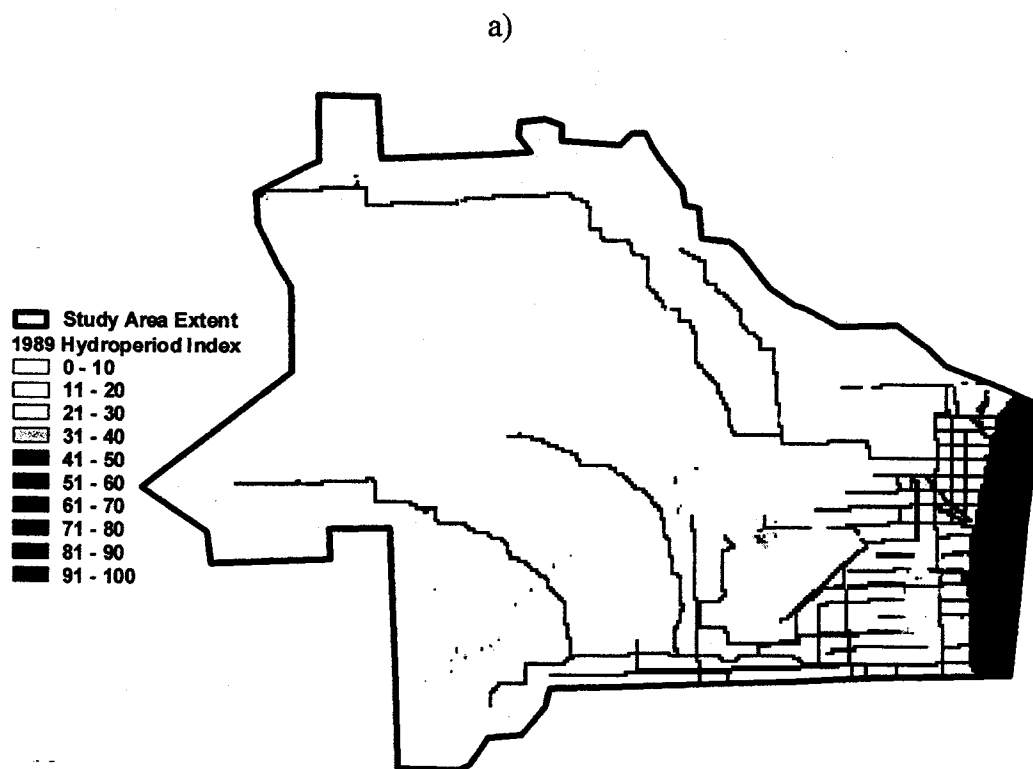


Figure 70. A) Hydroperiod index grid from simulation with 1989 rainfall conditions. B) Grid of change in hydroperiod index between 1989 and 1992 rainfall conditions. Positive values indicate higher index values in 1989, negative values indicate lower index values in 1989.

The hydroperiod index calculated for the 1995 rainfall simulation is included in Figure 71a. The general spatial patterns were comparable to the 1992 hydroperiod index grid. Areas with groups of cells having intermediate index values were greater in spatial extent. To better identify spatial and magnitude changes between the two conditions, the 1992 index grid was subtracted from the 1995 grid, and is shown in Figure 71b. Cells with higher positive index values indicated locations that were wetter during the higher rainfall conditions. The main areas that were wetter in the 1995 rainfall condition simulation were to the east and south of the sand ridge, along canals, and in the west in the historic East Everglades. Additional wetter areas occurred near the south central boundary in the vicinity of the city of Homestead. Some areas were drier, especially near the coast and in the immediate vicinity of the L-31 W/C-102 junction.

The effects rainfall had on groundwater hydrology were illustrated by the time-history of watershed soil water storage, shown in Figure 72. The effects were similar to impacts of rainfall on surface hydrology, as can be seen by the consistent patterns. The 1989 soil water volumes were the lowest of the three conditions. The 1992 time-history indicated that relatively higher dry season rainfall resulted in the highest soil water storage volumes during the early part of the year, until the simulated storages during 1995 conditions increased in response to the higher 1995 wet season rainfall.

The spatial effects of rainfall of groundwater hydrology were investigated next. The grid of groundwater elevations from simulation date 365 from the 1992 rainfall simulation is shown in Figure 73. Simulated groundwater elevations on the last day of the 1992 conditions were highest in the sandy ridge area in the western half of the study area, and lowest near the coast.

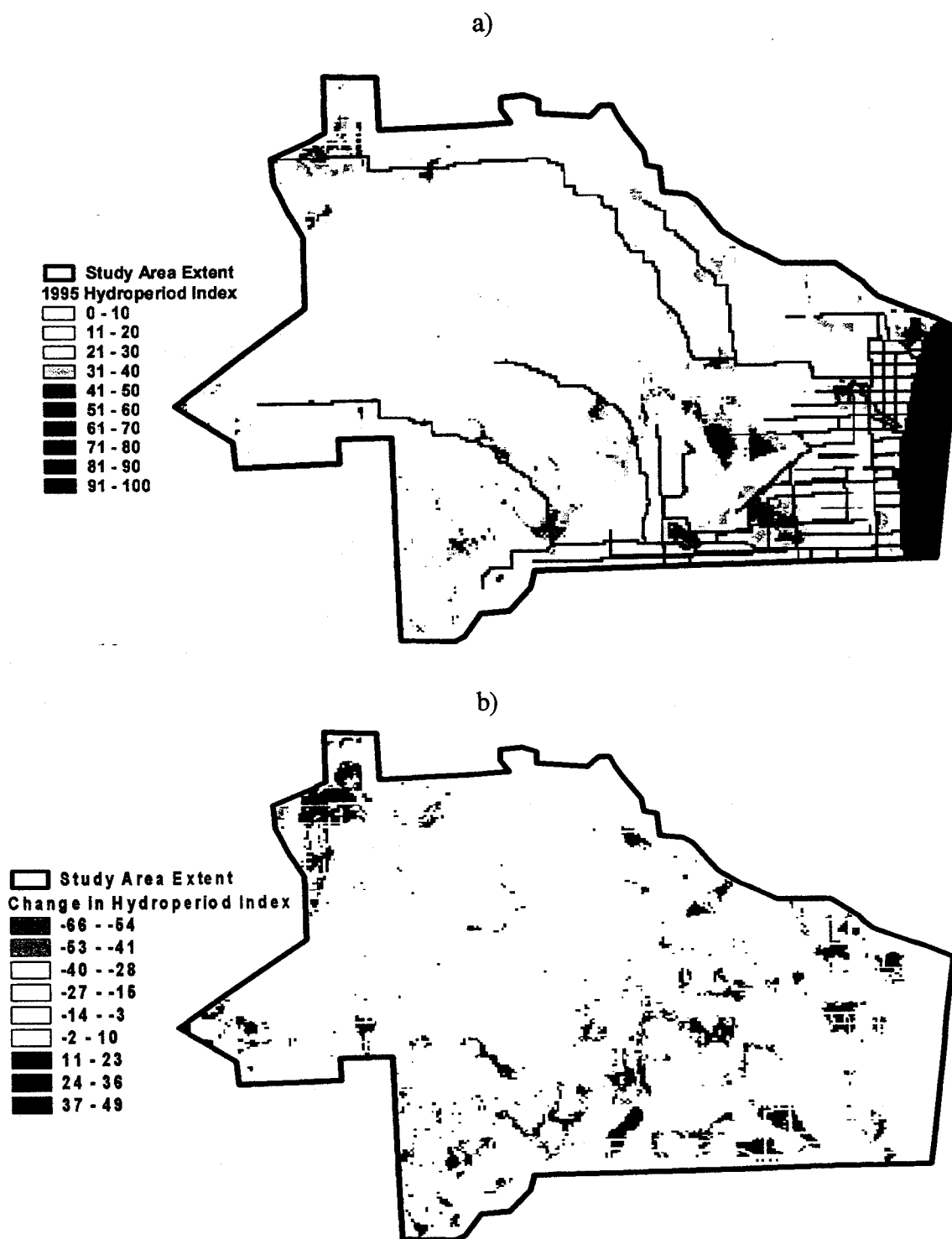


Figure 71. A) Hydroperiod index grid from simulation with 1995 rainfall conditions. B) Grid of change in hydroperiod index between 1995 and 1992 rainfall conditions. Positive values indicate higher index values in 1995, negative values indicate lower index values in 1995.

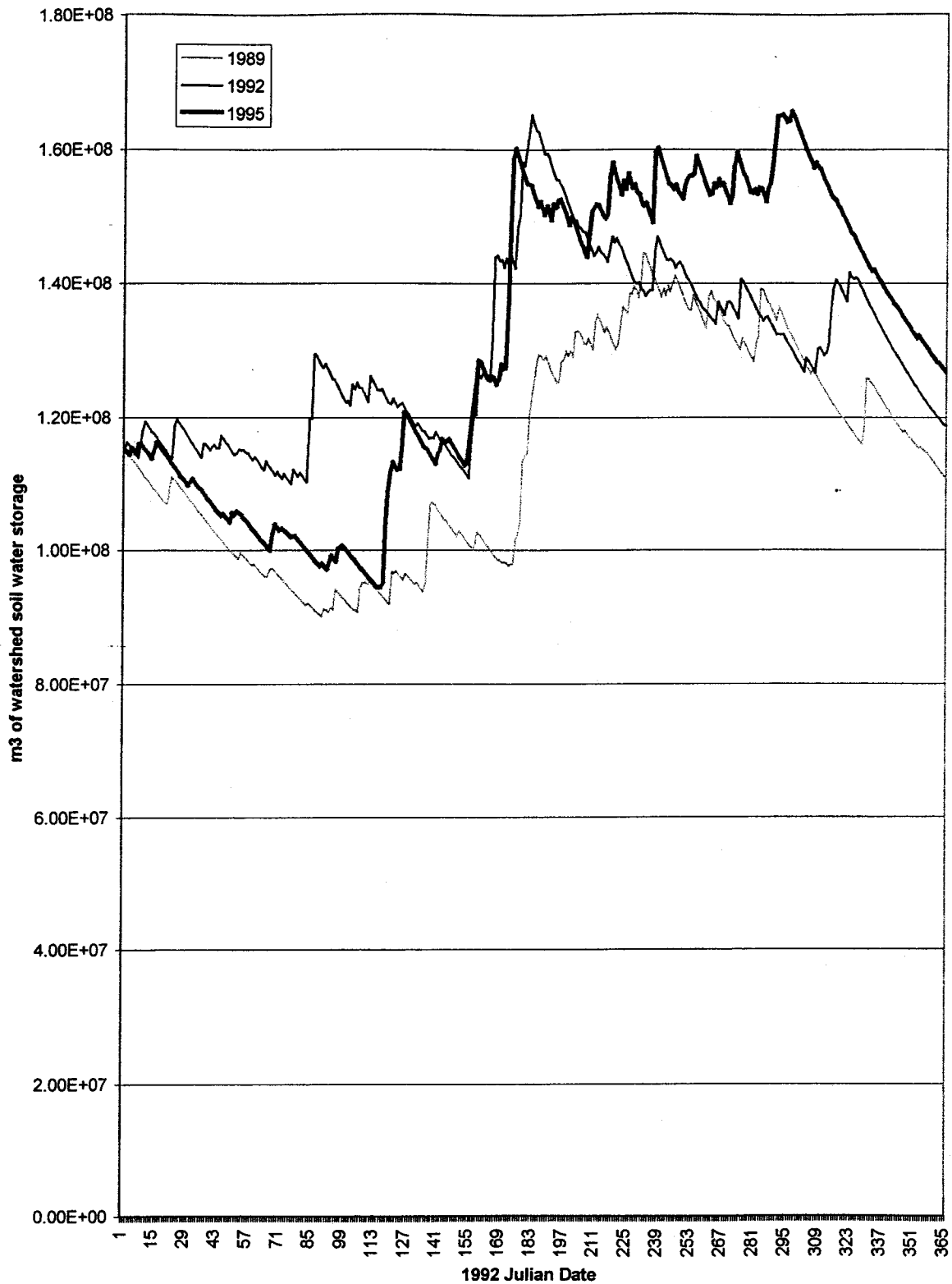


Figure 72. Time-history of simulated total watershed soil water storage for low (1989), average (1992), and high (1995) rainfall conditions.

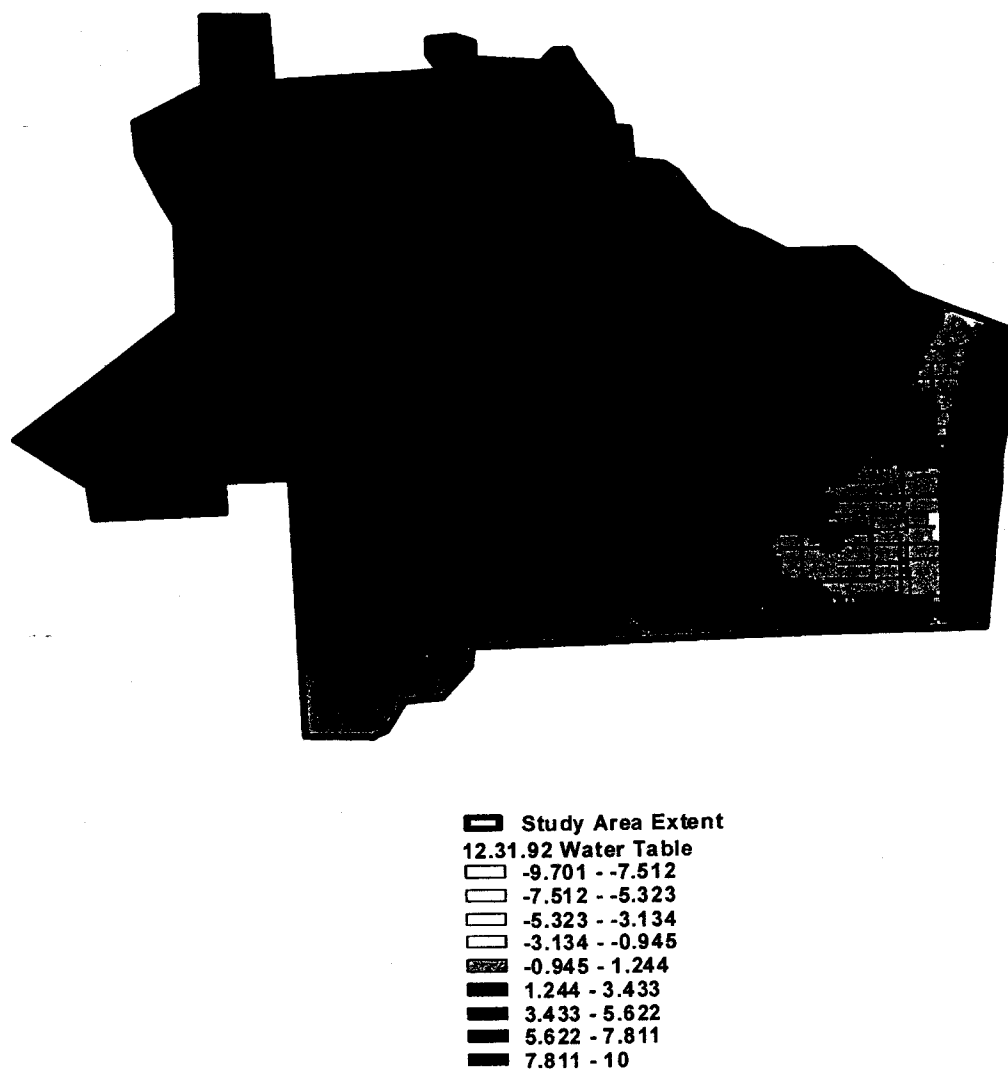


Figure 73. Grid of groundwater elevations on simulation date 365 for 1992 rainfall conditions. Units are in meters above MSL.

The grid of groundwater elevations from simulation date 365 for the 1989 rainfall conditions is shown in Figure 74a. The general spatial pattern was comparable that seen for the last day of the 1992 conditions. To better identify spatial and magnitude changes between the two simulation conditions, the 1992 groundwater elevation grid was subtracted from the 1989 groundwater elevation grid, and is shown in Figure 74b. Negative values indicate lower simulated 1989 groundwater elevations. Simulated water tables were generally lower in much of the eastern part of the study area, in some cases more than 3 meters. A mixture of lower and higher groundwater levels occurred in the western half of the region.

The grid of groundwater elevations from simulation date 365 for the 1995 rainfall condition is shown in Figure 75a. The general spatial pattern was again comparable that seen for the last day of the 1992 conditions. The 1992 groundwater elevation grid was then subtracted from the 1995 groundwater elevation grid, and is shown in Figure 74b. Positive values indicate higher simulated 1995 groundwater elevations. Simulated water tables were higher in most of the western part of the study area. Higher simulated groundwater levels also occurred in the eastern half of the region, although the magnitudes of change were decreased.

The effect of rainfall conditions on water quality discharge was evaluated next. To better illustrate differences, the cumulative simulated daily constituent discharges into Biscayne Bay were calculated and graphed as time-histories. Figure 76 shows the cumulative discharge time-history for TP. The temporal pattern was very similar to that seen for the cumulative rainfall time profile. Total simulated cumulative TP discharge

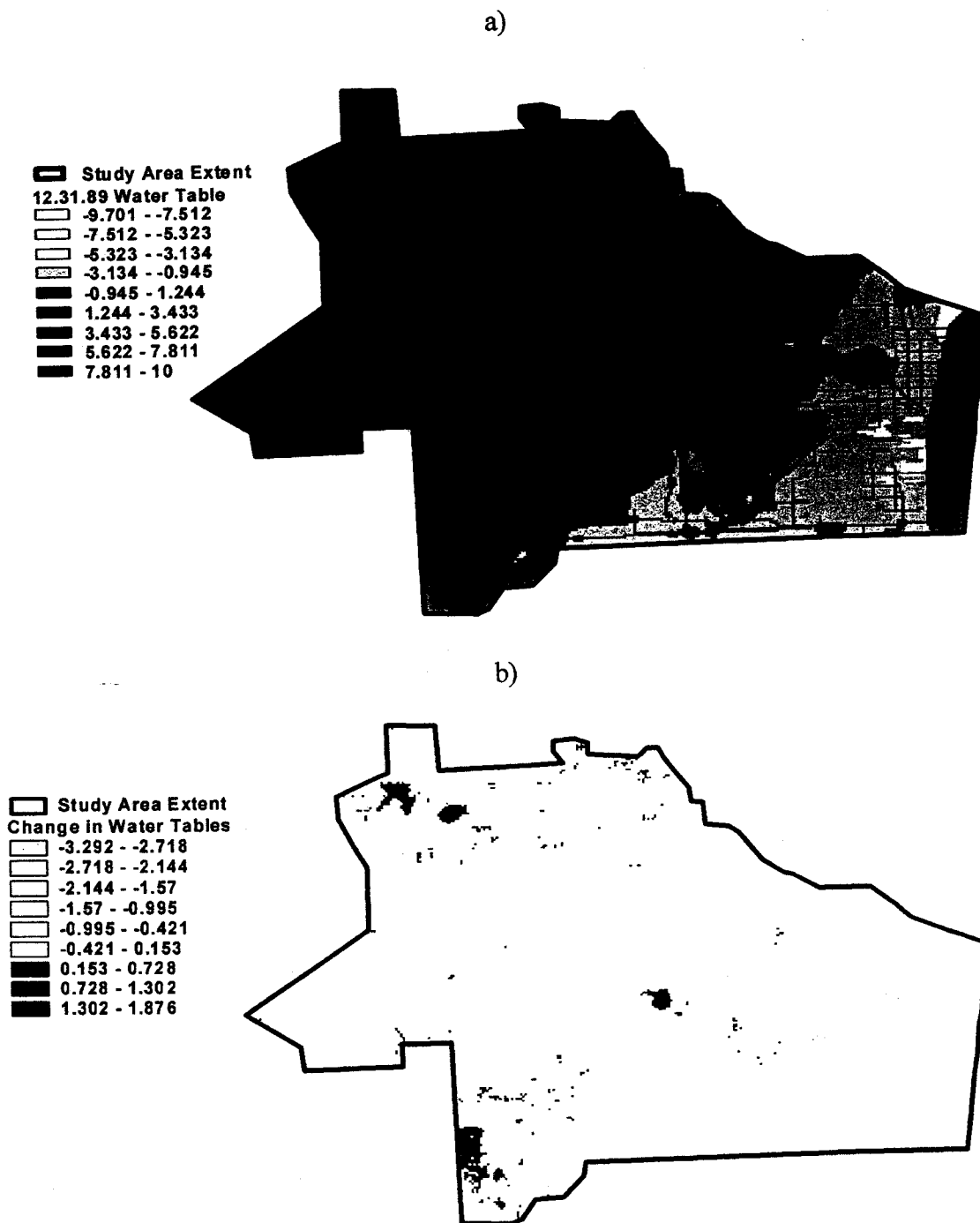


Figure 74. A) Grid of groundwater elevations on simulation date 365 for 1989 rainfall conditions. Units are in meters above MSL. B) Change in groundwater elevations between low rainfall (1989) and average rainfall (1992) conditions. Units are in meters, where positive values indicate higher values for 1989, and negative values indicate higher 1992 elevations.

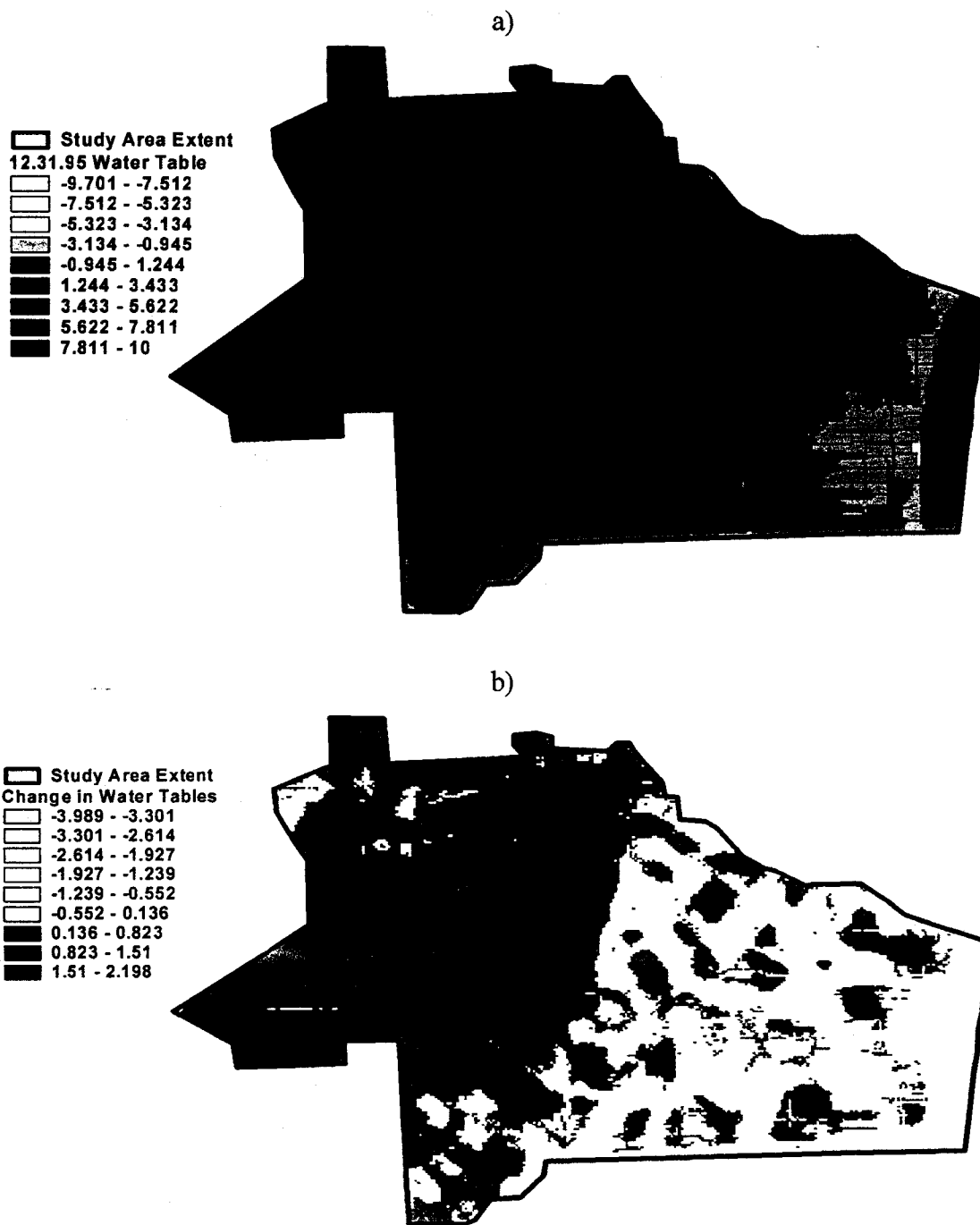


Figure 75. A) Grid of groundwater elevations on simulation date 365 for 1995 rainfall conditions. Units are in meters above MSL. B) Change in groundwater elevations between high rainfall (1995) and average rainfall (1992) conditions. Units are in meters, where positive values indicate higher values for 1995, and negative values indicate higher 1992 elevations.

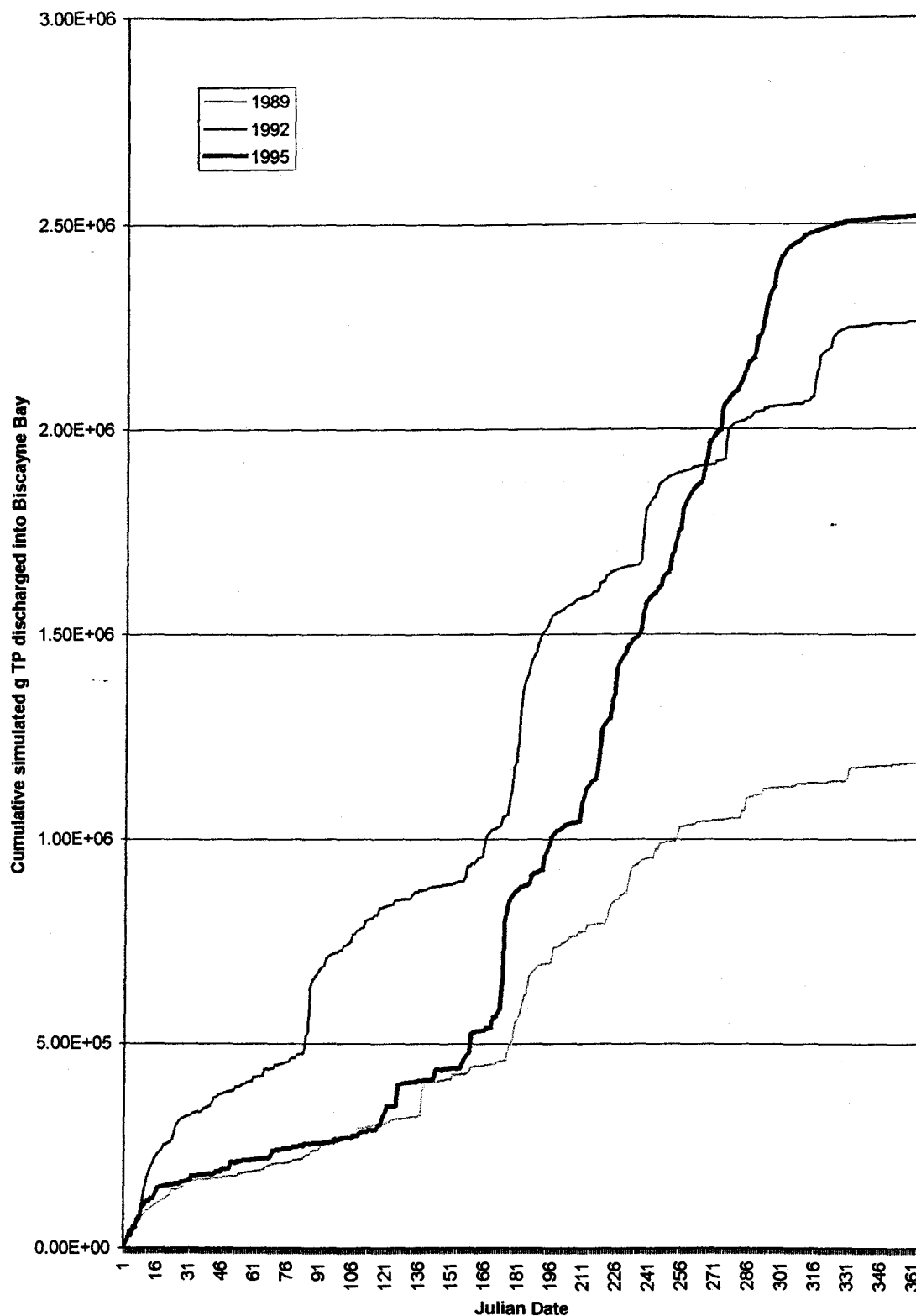


Figure 76. Time-history of cumulative TP discharge from C-102 and C-103 sub-basins into Biscayne Bay for low (1989), average (1992), and high (1995) rainfall conditions. Units are in grams.

was lowest for nearly every simulation date during the low rainfall conditions.

Cumulative simulated TP discharge was largest for the average rainfall conditions until simulation date 265, after which point the 1995 condition simulated TP discharge was greatest. For comparison, the 1995 cumulative rainfall exceeded 1992 cumulative rainfall on every day after date 249.

Figure 77 displays the time-history of cumulative simulated TSS discharge from the C-102 and C-103 sub-basins into Biscayne Bay. The cumulative discharge was lowest for 1989 rainfall, and highest for 1992 rainfall. Further clarification of TSS discharge patterns is shown in Figure 78. Although cumulative TSS discharge was never greatest for 1995 conditions, daily TSS discharge was higher under 1995 rainfall conditions after simulation date 251, as seen in Figure 78.

Sensitivity Analysis

The following section includes the results from simulations testing model sensitivity to key flow processes. The sensitivity analyses included tests of one process simulating surface water flow, one simulating a ground water flow, and one simulating constituent dynamics.

Surface Water Process Sensitivity Analysis

The results testing model sensitivity to a key surface water flow process are included as Figures 79-84, and in Table 4. Specifically, the tested process was one of the algorithms involved in simulation of overland flow. During model development, it was noticed that the setting of the threshold size (in number of grid cells) used to determine surface watersheds had considerable influence on the time-histories generated at the

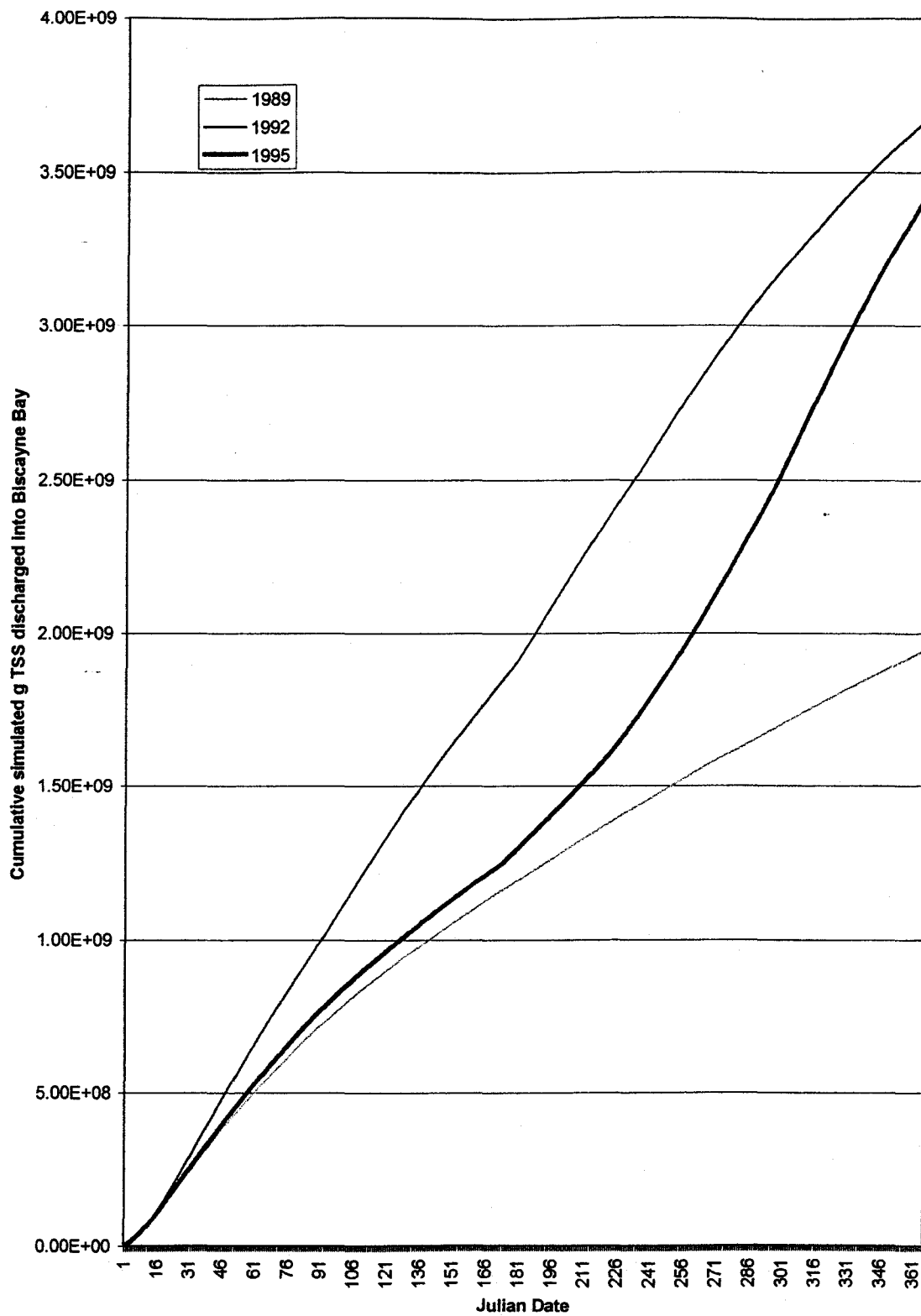


Figure 77. Time-history of cumulative TSS discharge from C-102 and C-103 sub-basins into Biscayne Bay for low (1989), average (1992), and high (1995) rainfall conditions. Units are in grams.

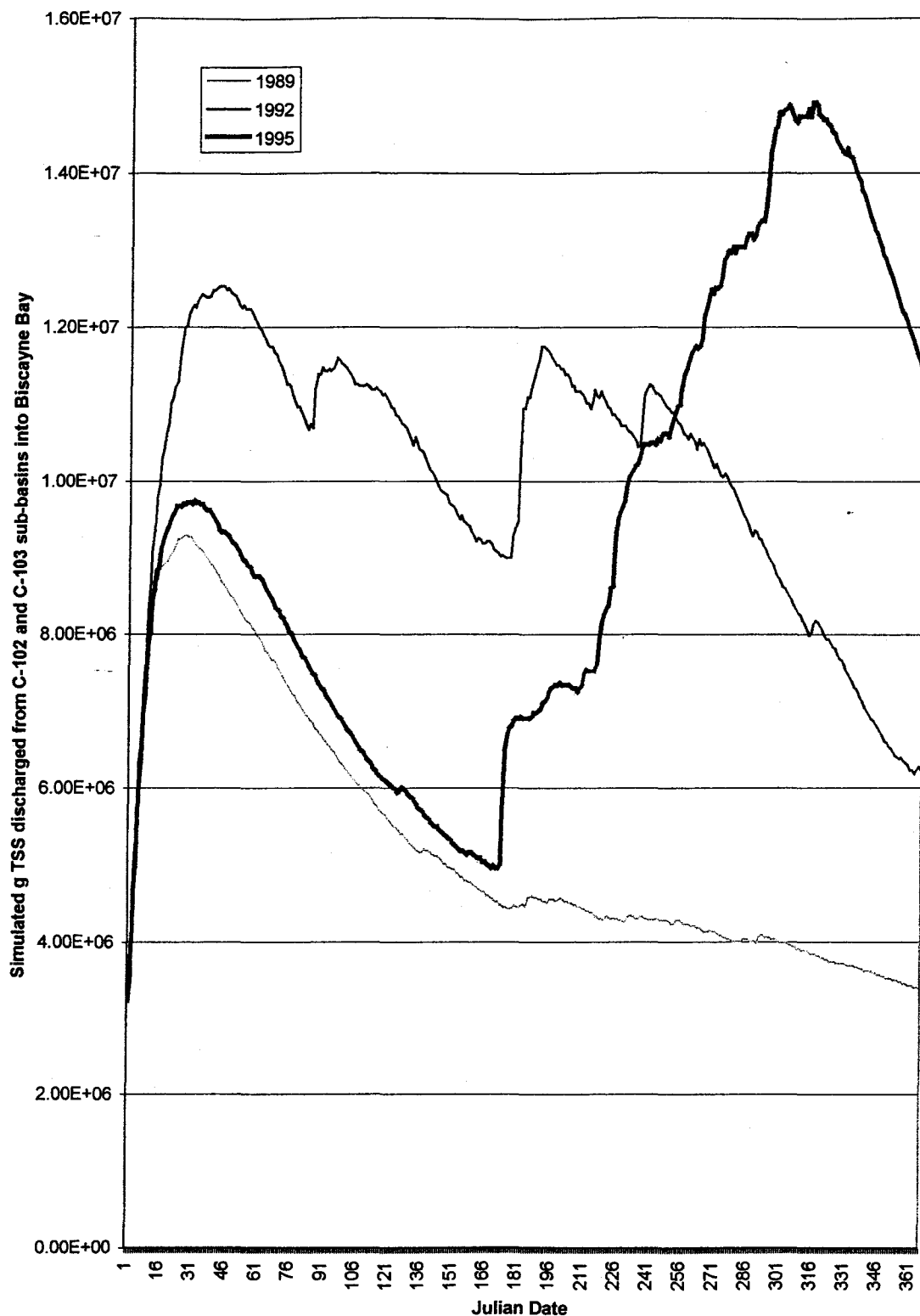


Figure 78. Time-history of simulated daily TSS discharge from C-102 and C-103 sub-basins into Biscayne Bay for low (1989), average (1992), and high (1995) rainfall conditions. Units are in grams.

individual calibration points. To quantify the effect, simulations were run where the calibrated setting (20 grid cells) was first decreased, and then increased, by 50%.

Figures 79-82 includes the time-histories generated from canal stages at S-194, S-21A, S-196, and S-20F from simulations run at each setting. At both upper-watershed points (S-194 and S-196), the time-histories (seen in Figures 79 and 80) indicated generally similar temporal response from simulations run at the decreased and the calibrated setting. However, simulated S-194 canal stages after the onset of the wet season were conspicuously higher when the threshold watershed size setting was raised. Maximum simulated S-194 stages were 2.50, 2.61, and 3.63 meters above MSL from the simulations with the decreased, calibrated, and increased threshold settings, respectively. Simulated S-196 stages showed little sensitivity to the setting value.

Time-histories of simulated canal stages at the downstream locations, S-21A (shown in Figure 81) and S-20F (shown in Figure 82), indicated increased sensitivity to the threshold setting, although the effects were opposite between the two points. At S-21A, higher settings decreased the simulated stages for most of the year, whereas higher settings increased simulated S-20F stages. At S-20F, simulated stage determined with the higher setting finished the year more than .6 meters higher than end-of-year S-20F stage simulated with the calibration setting.

To better understand the overall influence of the setting on surface water flows at the watershed-scale, time-histories were generated of daily watershed surface water storage for all three settings, shown in Figure 83. During the dry season, daily total surface water storages were nearly identical between the simulations run at all three settings. Some divergence in total simulated storage values occurred during the wet

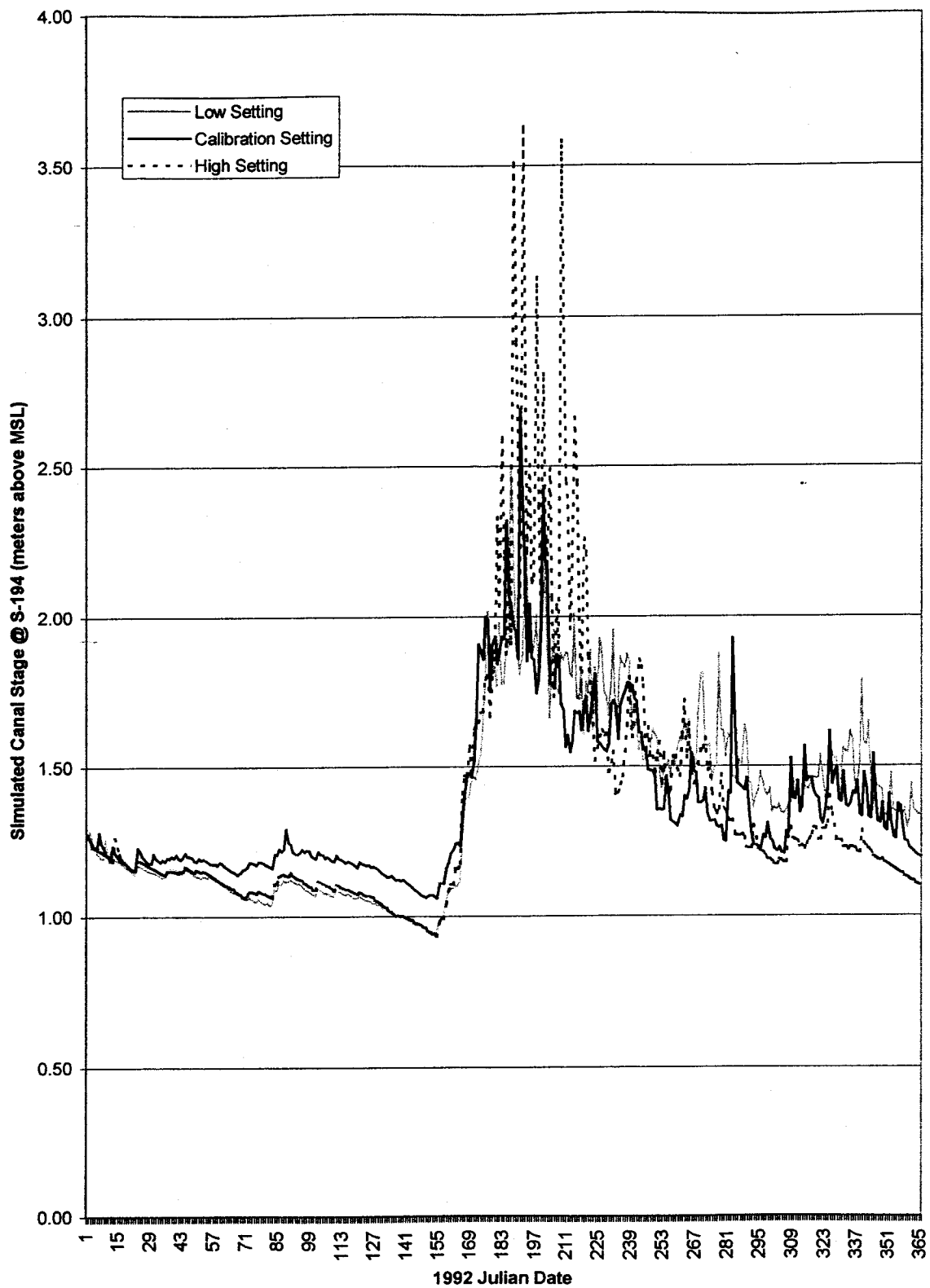


Figure 79. Time-histories of simulated canal stage at S-194 for three settings of surface water sensitivity analysis process. Units are in meters above MSL.

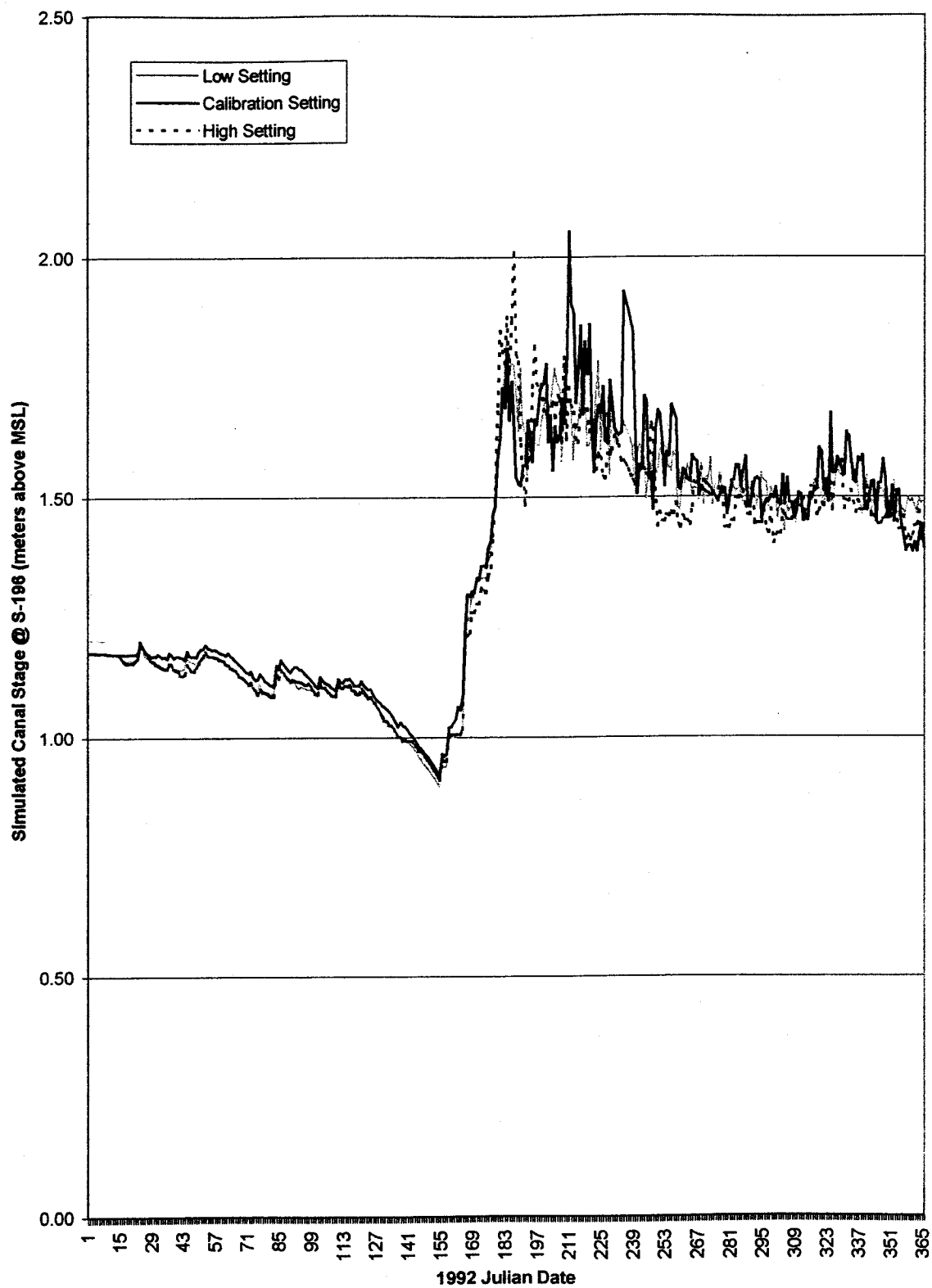


Figure 80. Time-histories of simulated canal stage at S-196 for three settings of surface water sensitivity analysis process. Units are in meters above MSL.

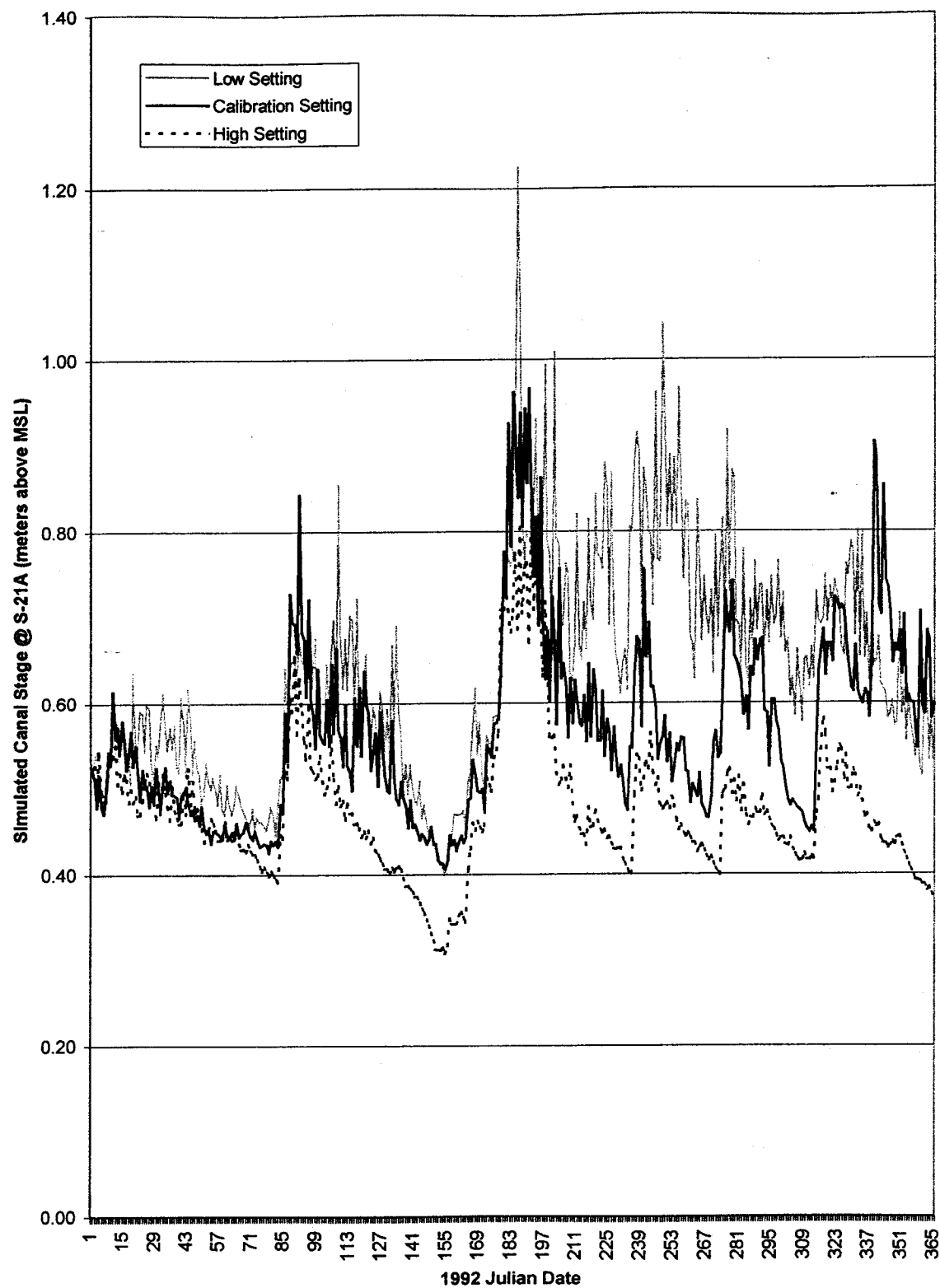


Figure 81. Time-histories of simulated canal stage at S-21A for three settings of surface water sensitivity analysis process. Units are in meters above MSL.

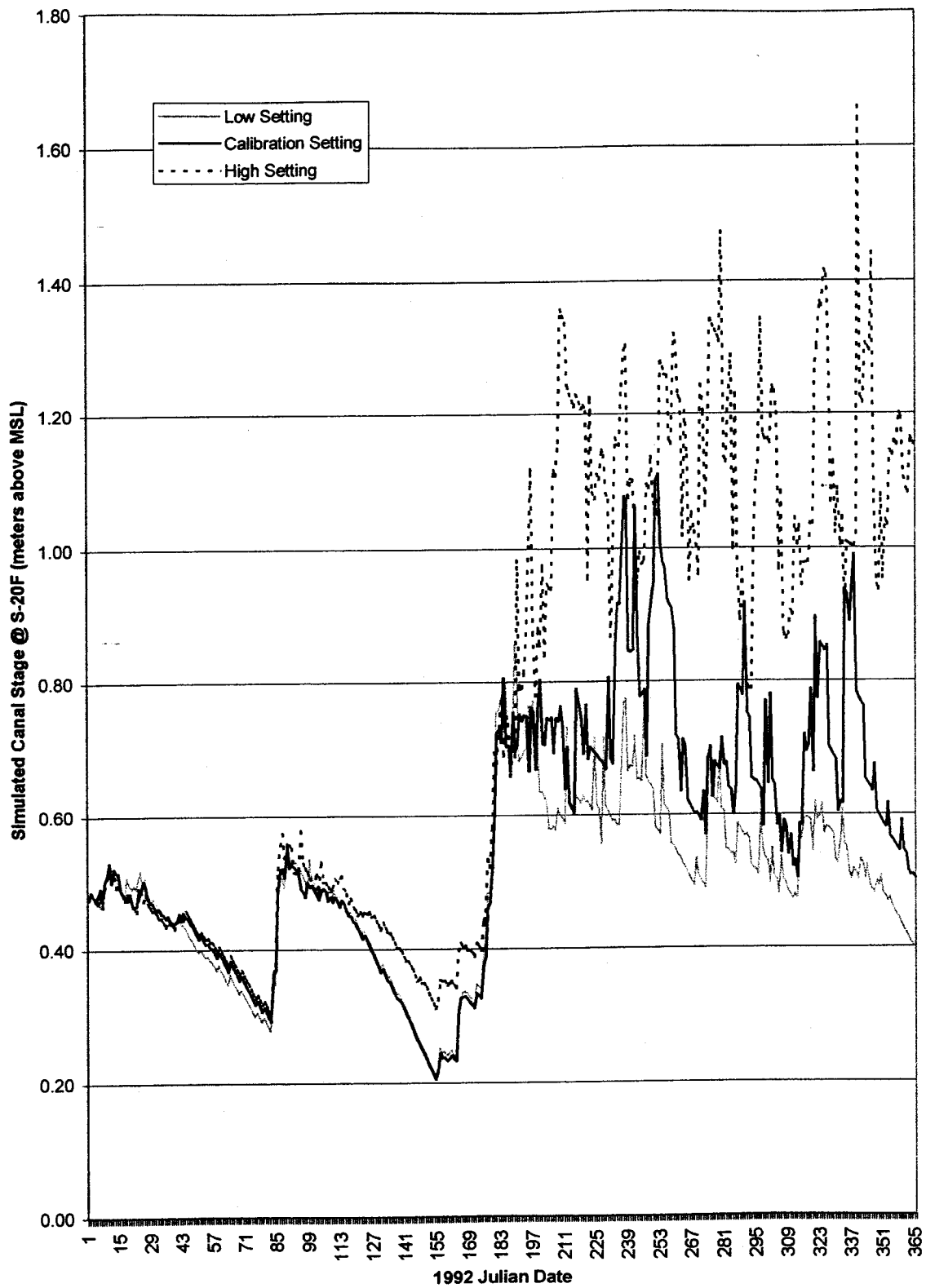


Figure 82. Time-histories of simulated canal stage at S-20F for three settings of surface water sensitivity analysis process. Units are in meters above MSL.

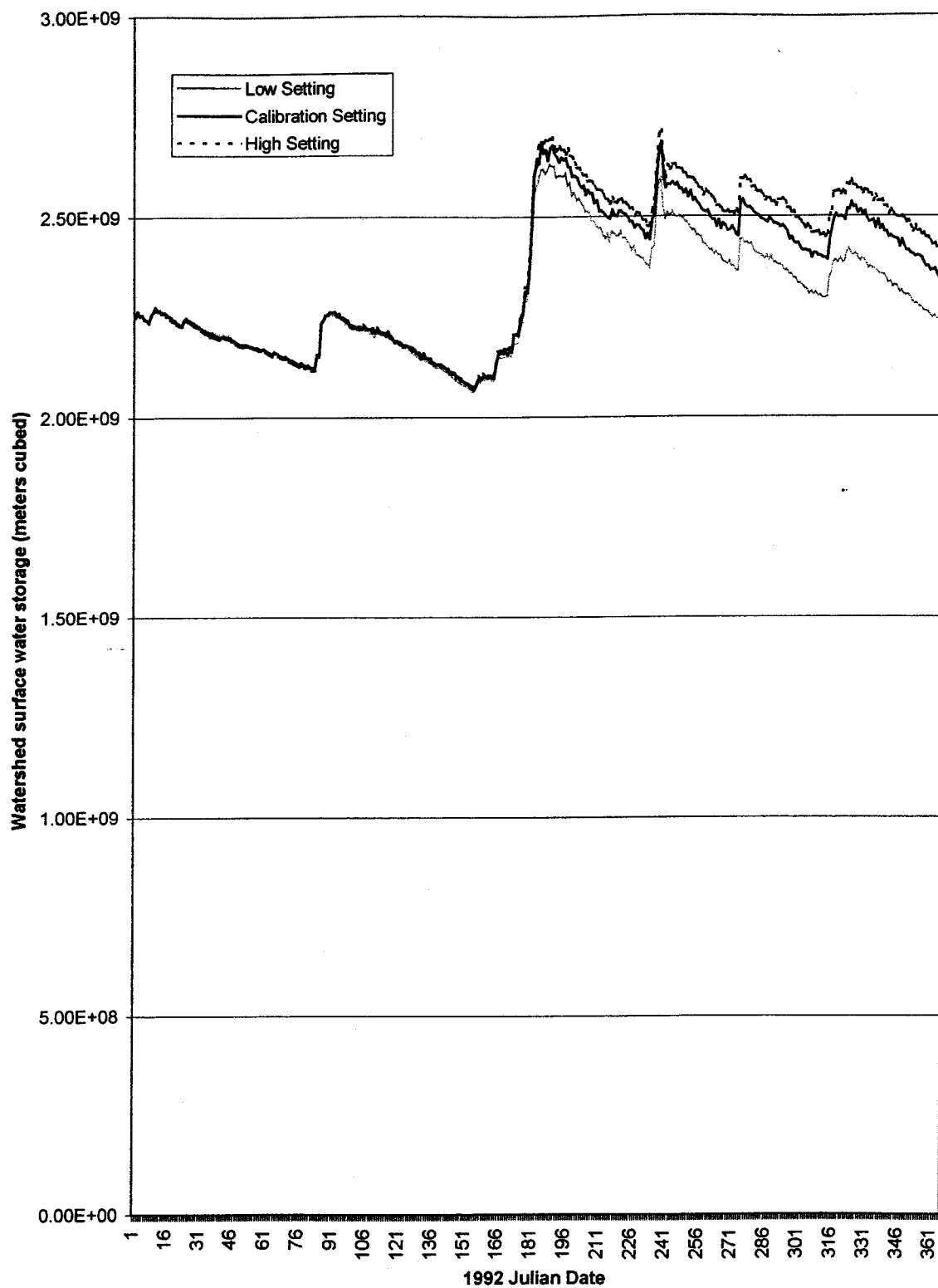


Figure 83. Time-histories of simulated total watershed surface water storage for three settings of surface water sensitivity analysis process. Units are in meters cubed.

season. By the end of the simulation year, the lower setting resulted in a 4.7% reduction in total watershed surface water storage value from the calibration run result. The higher setting increased the end-of-year watershed surface water storage value 3.0% above the calibration simulation result.

The threshold setting did not significantly influence the total simulated discharge of surface water into Biscayne Bay. Time-histories compiled of daily discharges to the ocean from simulations conducted at all three settings are included in Figure 84. The average daily flow for all three simulation runs was identical at $1.01 \text{ E}+06 \text{ m}^3$.

A summary table, Table 4, includes the average daily storages of surface water, ground water, TP, and TSS simulated with each threshold setting. In addition, the percentage change in storage between results of simulations run with the alternate and calibrated threshold setting values were calculated and included in the table. The average daily storage data indicated that higher setting values resulted in greater watershed surface water storage, while ground water storages were basically unchanged. Average daily watershed storage of constituents also increased when the setting was higher.

In addition, Table 4 includes the total annual flows of water and constituents discharged from the analysis extent into the ocean from the simulations run with each threshold setting. The percentage change in annual ocean discharges of water and constituents from the calibrated setting was also calculated and included into the table. Simulated ocean discharges were lower when the threshold setting was reduced, although the decrease was only significant in the case of TSS (a 19.6% reduction). Simulated discharge of water into the Bay was very slightly (.7%) reduced by raising the threshold setting. The raised threshold setting increased the discharges of both constituents into the

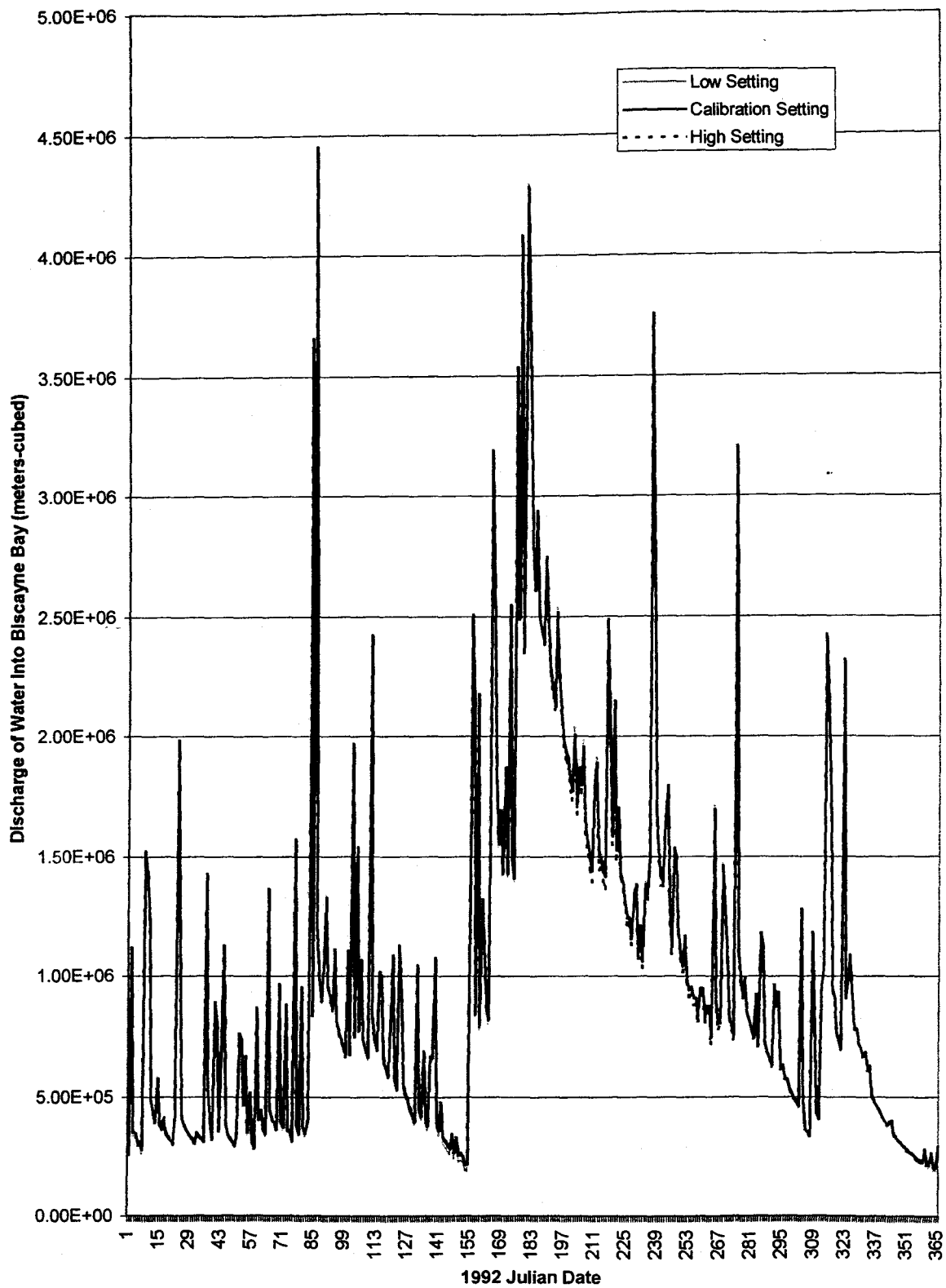


Figure 84. Time-histories of simulated total watershed discharge of surface water into Biscayne Bay for three settings of surface water sensitivity analysis process.

Average Daily Watershed Storage

	<u>Low Setting</u>	<u>Calibrated Setting</u>	<u>High Setting</u>
Surface Water (m3)	2.30E+09	2.35E+09	2.37E+09
Ground Water (m3)	5.09E+09	5.10E+09	5.10E+09
TP (grams)	8.80E+06	1.01E+07	1.09E+07
TSS (grams)	1.06E+09	1.44E+09	1.66E+09

% Change in Average Value From Calibrated Result

	<u>Low Setting</u>	<u>Calibrated Setting</u>	<u>High Setting</u>
Surface Water	-1.9	N/A	1.1
Ground Water	-0.2	N/A	-0.1
TP	-12.9	N/A	8.2
TSS	-26.9	N/A	15.0

Annual Watershed Discharge To Biscayne Bay

	<u>Low Setting</u>	<u>Calibrated Setting</u>	<u>High Setting</u>
Surface Water (m3)	3.70E+08	3.70E+08	3.68E+08
TP (grams)	2.32E+06	2.34E+06	2.62E+06
TSS (grams)	3.01E+09	3.74E+09	4.91E+09

% Change in Annual Discharge From Calibrated Result

	<u>Low Setting</u>	<u>Calibrated Setting</u>	<u>High Setting</u>
Surface Water	-0.2	N/A	-0.7
TP	-0.7	N/A	12.2
TSS	-19.6	N/A	31.1

Table 4. Summary table of watershed-scale surface water process sensitivity analysis simulation results.

Bay considerably, with simulated TP discharge increasing 12.2%, and simulated TSS discharge increasing 31.1%, from the calibrated conditions.

Ground Water Process Sensitivity Analysis

The results testing model sensitivity to a key ground water flow process are included as Figures 85-86, and in Table 5. Specifically, the tested process was part of the lateral groundwater flow algorithms. The algorithms were governed by a calibration coefficient which regulated the effect water table gradients had in calculations of the lateral groundwater flow volume. To quantify the effect, two simulations were run where the coefficient value was first reduced, and then increased, by 50% from the calibrated value. In effect, lower settings allowed more cell-to-cell lateral flow for a given gradient.

The effect of varying the calibration coefficient is illustrated in Figure 85, which shows the time-history of water tables at G-757A from the simulations run with each coefficient setting. The time-histories indicate a similar temporal response throughout the year for each setting. The groundwater elevation values were almost identical on every day of the year for the simulations run with the calibrated and the higher setting. However, the simulation run with the reduced setting resulted in a slight decrease in water tables. In addition, short-term fluctuations increased when the setting was lowered. Day-to-day groundwater elevations were 33, 8, and 9 days for the simulations run at the low, calibrated, and high settings, respectively.

The watershed-scale effect of changing the setting can be seen in Figure 86, which contains the time-history of total analysis extent ground water storage values from the simulations run with the low, calibrated, and high settings. The temporal responses

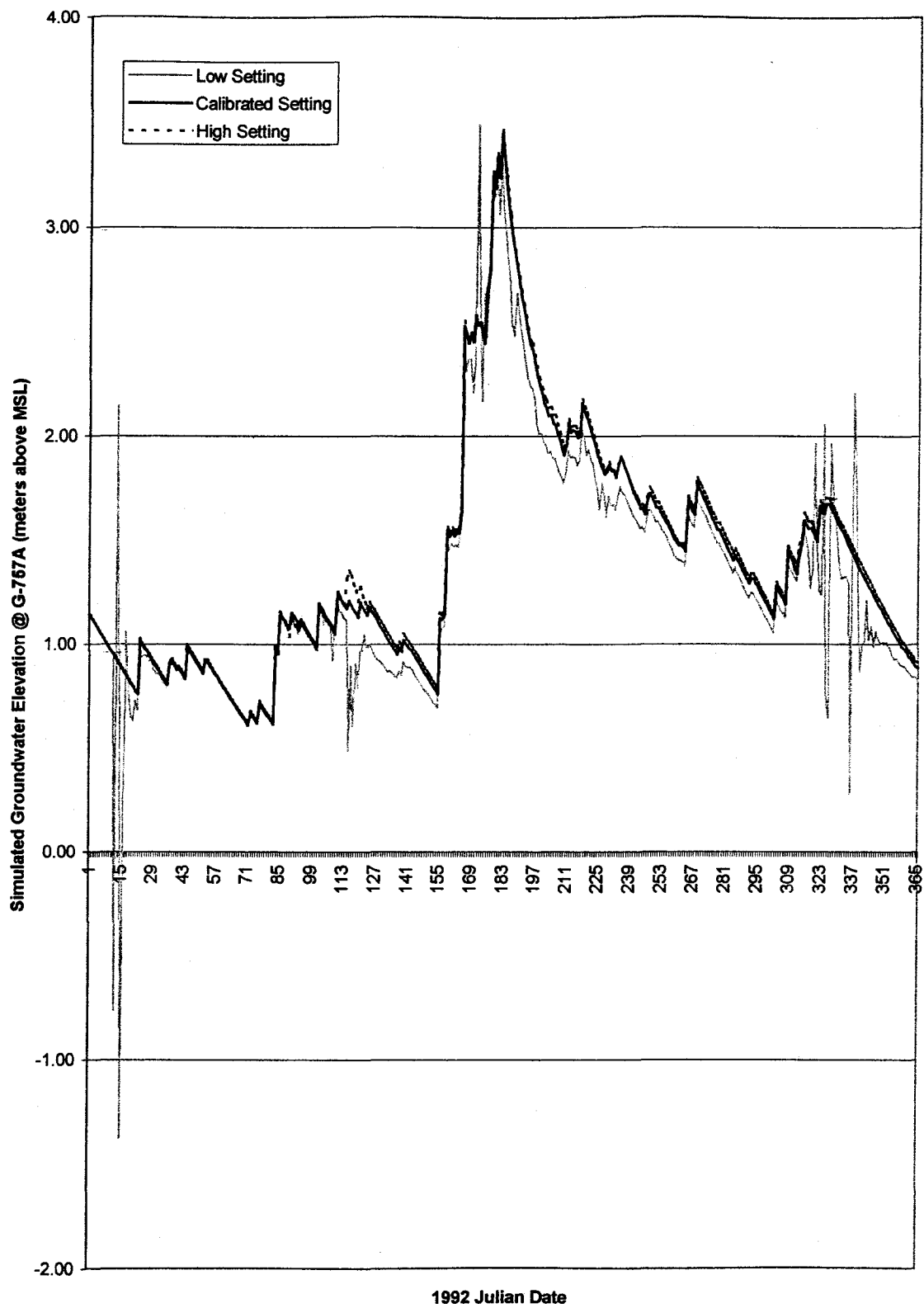


Figure 85. Time-histories of simulated groundwater elevations at G-757A for three settings of ground water sensitivity analysis process. Units are in meters above MSL.

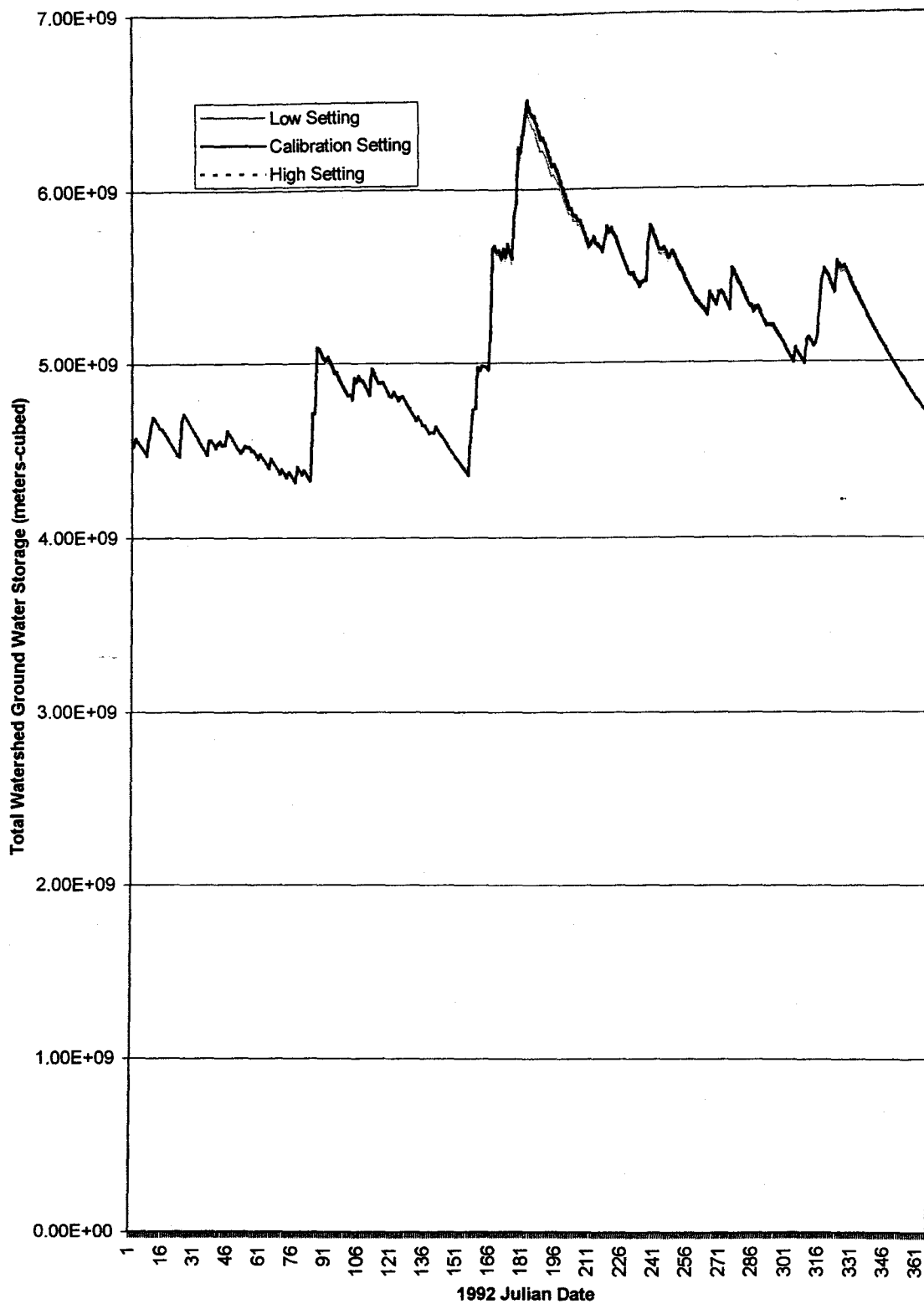


Figure 86. Time-histories of simulated total watershed ground water storage for three settings of ground water sensitivity analysis process. Units are in meters cubed.

and storage magnitudes were nearly identical for all three coefficient settings throughout the simulation year.

To further compare watershed-scale effects of coefficient setting, the average daily analysis extent storages of surface water, ground water, TP, and TSS simulated with each setting were calculated and tabulated as Table 5. In addition, the percentage change in storage between results of simulations run with the alternate and calibrated setting values were calculated and included in the table. Raising the calibration coefficient setting reduced the average daily total watershed surface water storage slightly. In contrast, higher settings increased average daily total watershed ground water storage values. Higher settings significantly reduced the average TP storage, but had little influence on average daily TSS storage. Average daily TP storages were reduced 68.4% between the simulation with the low setting as compared to the simulation run with the high setting.

In addition, Table 5 includes the total annual flows of water and constituents discharged from the watershed into Biscayne Bay from the simulations run with each setting. The percentage changes in annual watershed discharges of water and constituents into the ocean from the calibrated setting were also calculated and included in the table. Simulated annual discharge of water to the ocean was essentially unaffected by the coefficient setting. In contrast, the coefficient setting more significantly influenced simulated annual constituent discharge, especially for TP. Lower settings resulted in increased simulated ocean discharges. For the lower setting, simulated TP discharge increased 41.5% from the calibrated setting result. TSS discharge only increased 1.4% from the calibrated setting simulation result. For the higher setting,

Average Daily Watershed Storage

	<u>Low Setting</u>	<u>Calibrated Setting</u>	<u>High Setting</u>
Surface Water (m3)	2.37E+09	2.34E+09	2.33E+09
Ground Water (m3)	5.09E+09	5.10E+09	5.11E+09
TP (grams)	1.33E+07	1.01E+07	7.90E+06
TSS (grams)	1.52E+09	1.53E+09	1.51E+09

% Change in Average Value From Calibrated Result

	<u>Low Setting</u>	<u>Calibrated Setting</u>	<u>High Setting</u>
Surface Water	1.3	N/A	-0.4
Ground Water	-0.2	N/A	0.1
TP	31.5	N/A	-21.8
TSS	-0.7	N/A	-0.9

Annual Watershed Discharge To Biscayne Bay

	<u>Low Setting</u>	<u>Calibrated Setting</u>	<u>High Setting</u>
Surface Water (m3)	3.71E+08	3.71E+08	3.72E+08
TP (grams)	3.31E+06	2.34E+06	2.07E+06
TSS (grams)	3.72E+09	3.67E+09	3.23E+09

% Change in Annual Discharge From Calibrated Result

	<u>Low Setting</u>	<u>Calibrated Setting</u>	<u>High Setting</u>
Surface Water	-0.2	N/A	0.3
TP	41.5	N/A	-11.3
TSS	1.4	N/A	-11.8

Table 5. Summary table of watershed-scale ground water process sensitivity analysis simulation results.

simulated TP discharge declined 11.3%, and simulated TSS discharge dropped 11.8%, from results obtained with the calibrated conditions.

Water Quality Process Sensitivity Analysis

The results testing model sensitivity to a key water quality process are included as Figures 87-94, and in Table 6. Specifically, the tested process was the algorithm utilized for simulation of constituent sedimentation. The algorithm employed a calibration coefficient, which controlled sedimentation rates, and, ultimately, aquatic constituent concentrations. To quantify the sensitivity to coefficient setting, simulations were run where the calibrated setting was first reduced, and then increased, by 50%.

Time-histories were compiled of TP concentrations at S-194, S-196, S-21A, and S-20F from simulations run at each setting, and are included in Figures 87-90. The effects of changing the coefficient setting were the same at all points. As indicated in the figures, the temporal profiles (i.e. timing of increases and decreases) of the simulated concentration values were essentially identical at all locations. At each observation point, the effect coefficient setting had on magnitude was also the same. When the setting was lowered, (i.e. reducing the TP sedimentation flow) simulated TP concentrations were increased. When the setting was raised, more sedimentation occurred and the simulated TP concentrations were reduced.

Figures 91-94 displays the time-histories of simulated TSS concentrations at S-194, S-196, S-21A, and S-20F from the simulations run at each coefficient setting. As seen in the figures, changing the sedimentation coefficient setting had the same basic impacts on simulated TSS concentrations, and was again consistent at all locations. The temporal profiles of simulated TSS concentrations were essentially identical for all three

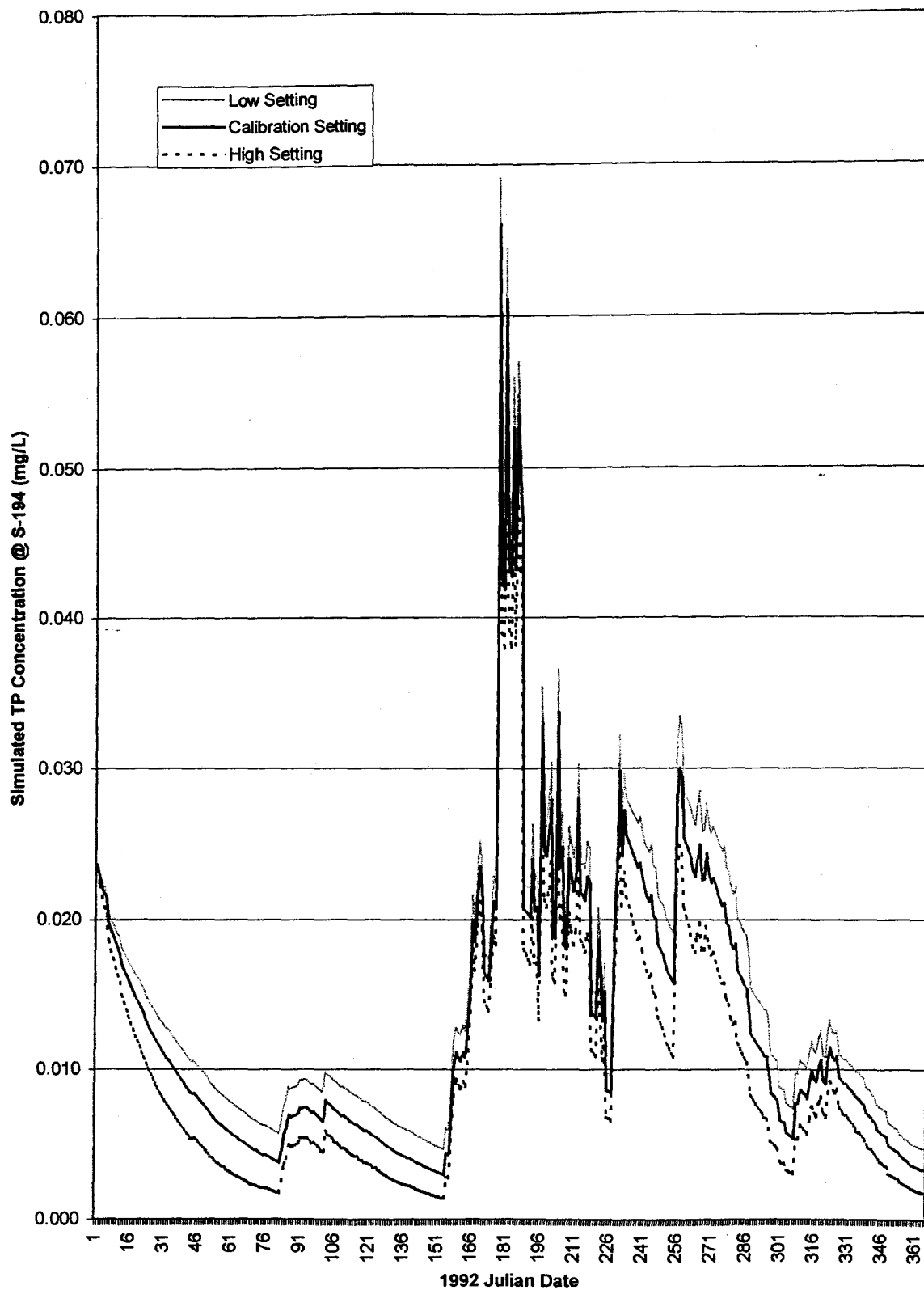


Figure 87. Time-histories of simulated TP concentration at S-194 for three settings of water quality sensitivity analysis process. Units are in mg TP/L.

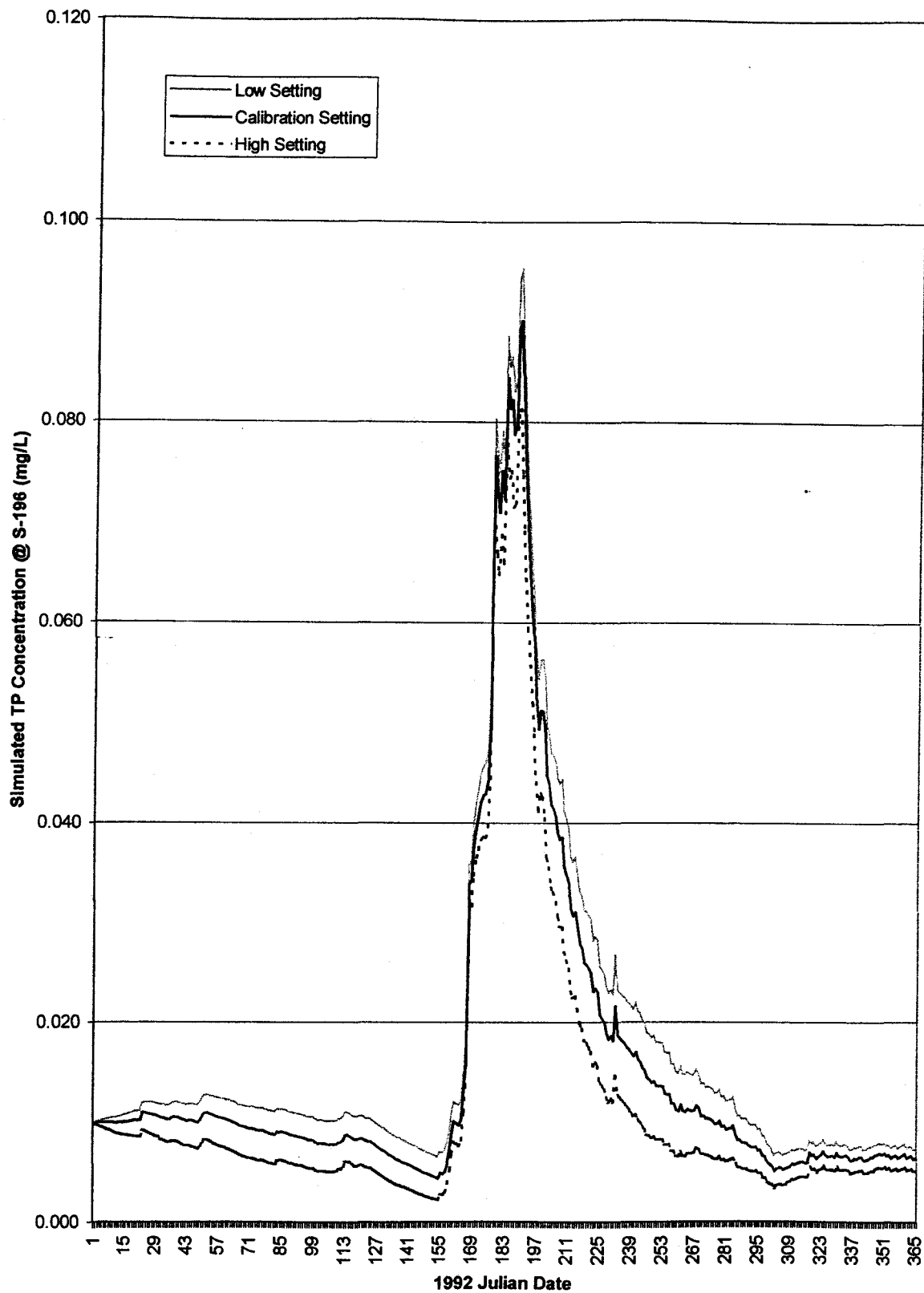


Figure 88. Time-histories of simulated TP concentration at S-196 for three settings of water quality sensitivity analysis process. Units are in mg TP/L.

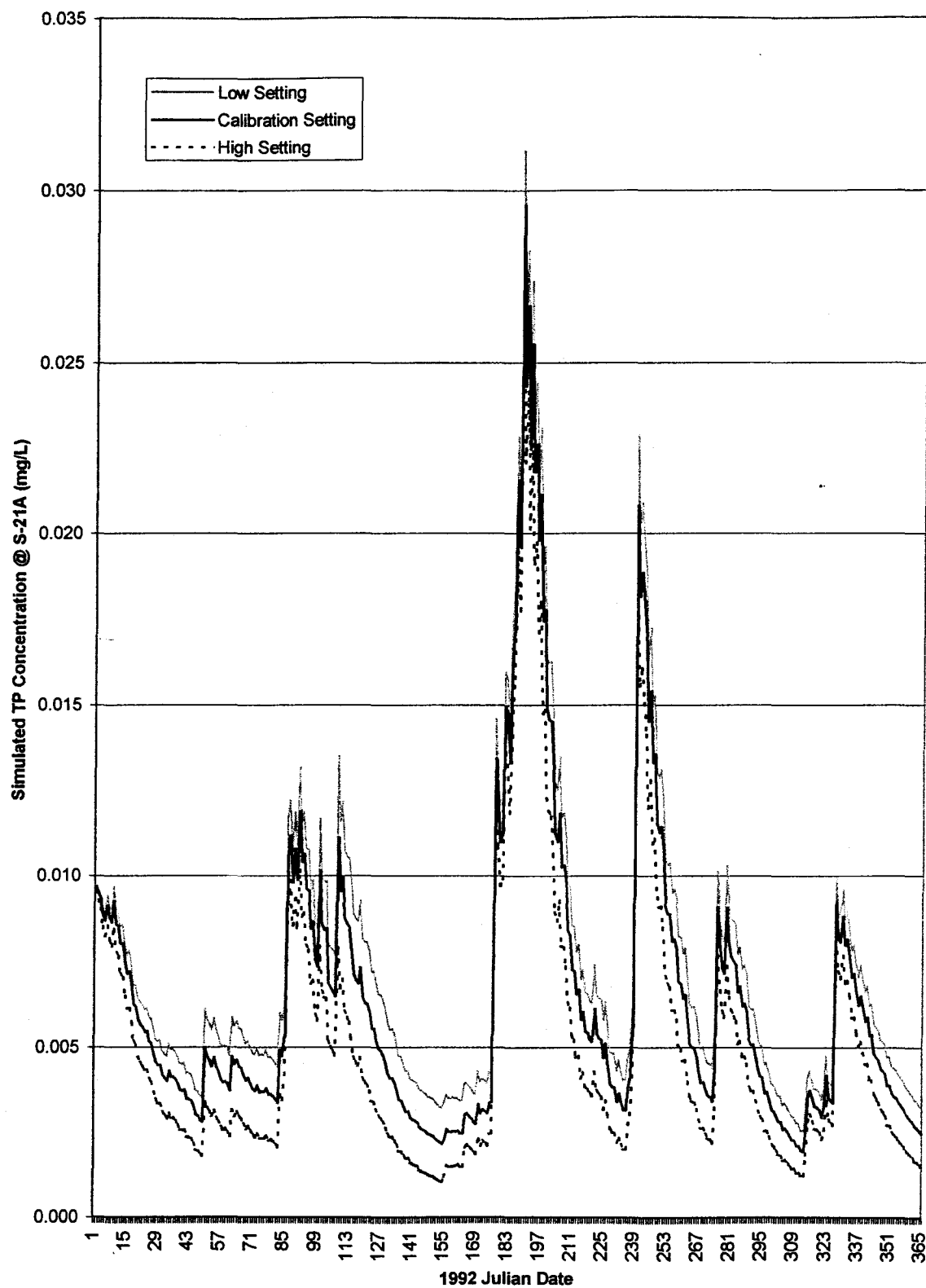


Figure 89. Time-histories of simulated TP concentration at S-21A for three settings of water quality sensitivity analysis process. Units are in mg TP/L.

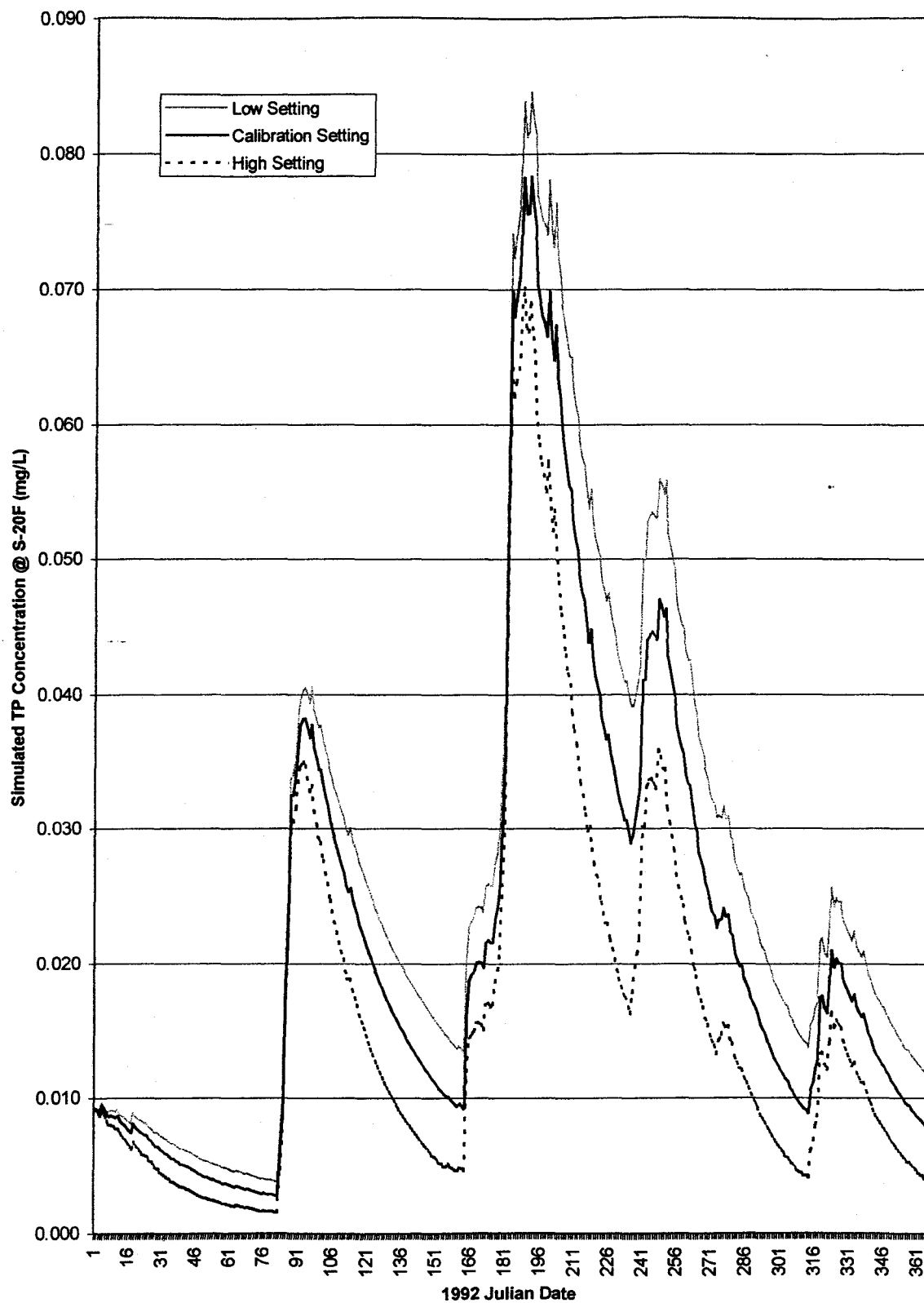


Figure 90. Time-histories of simulated TP concentration at S-20F for three settings of water quality sensitivity analysis process. Units are in mg TSS/L.

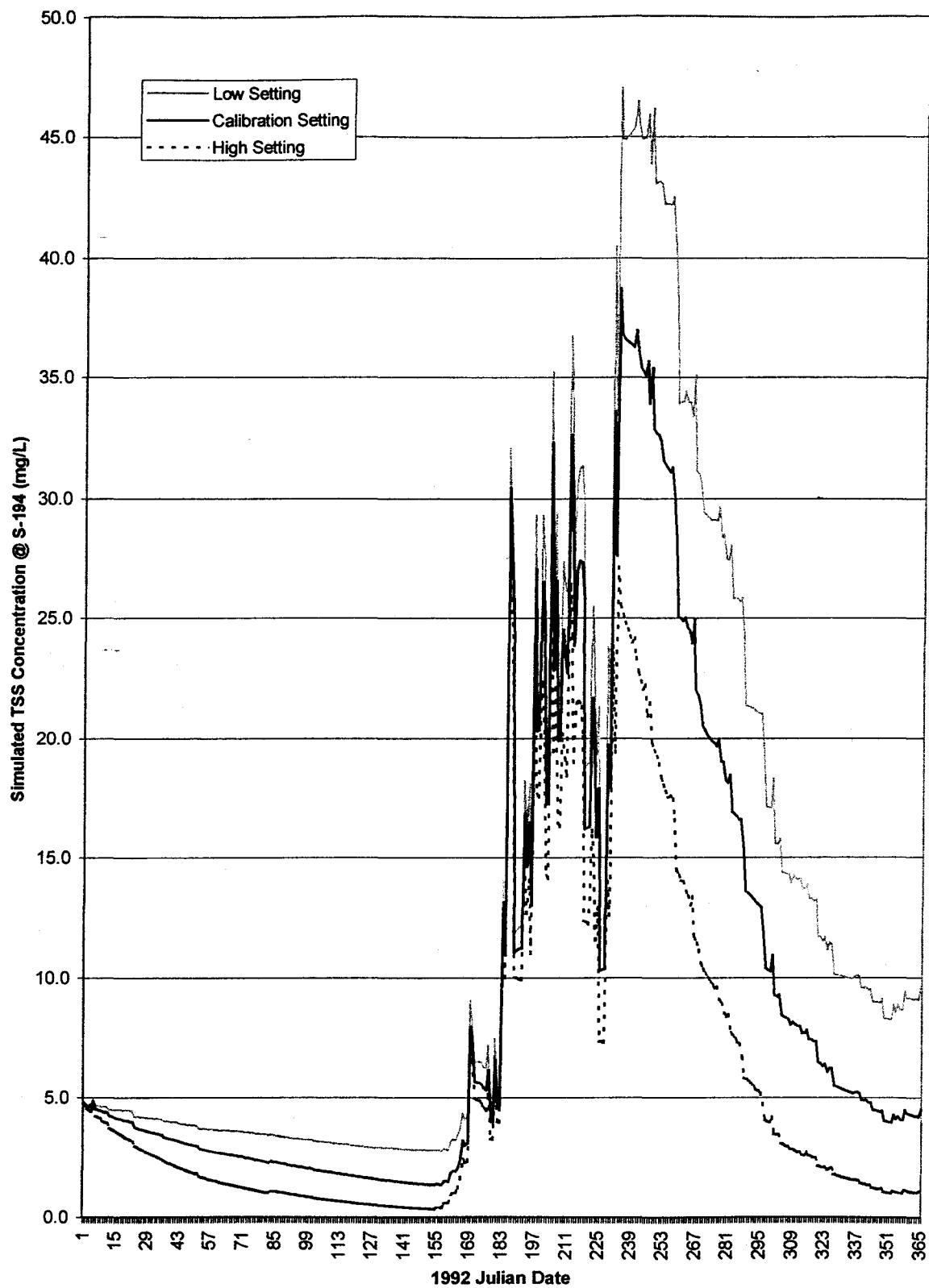


Figure 91. Time-histories of simulated TSS concentration at S-194 for three settings of water quality sensitivity analysis process. Units are in mg TP/L.

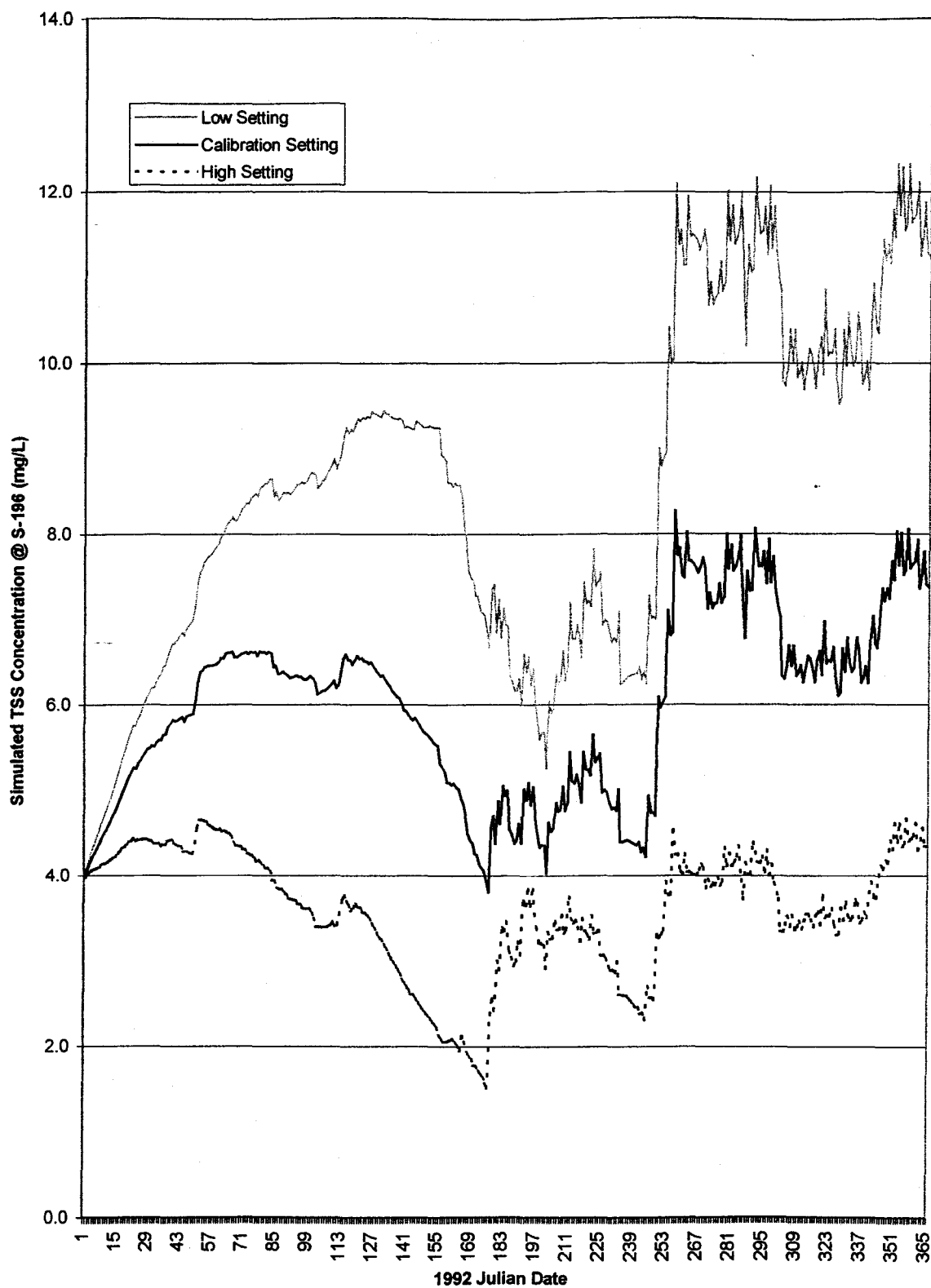


Figure 92. Time-histories of simulated TSS concentration at S-196 for three settings of water quality sensitivity analysis process. Units are in mg TSS/L.

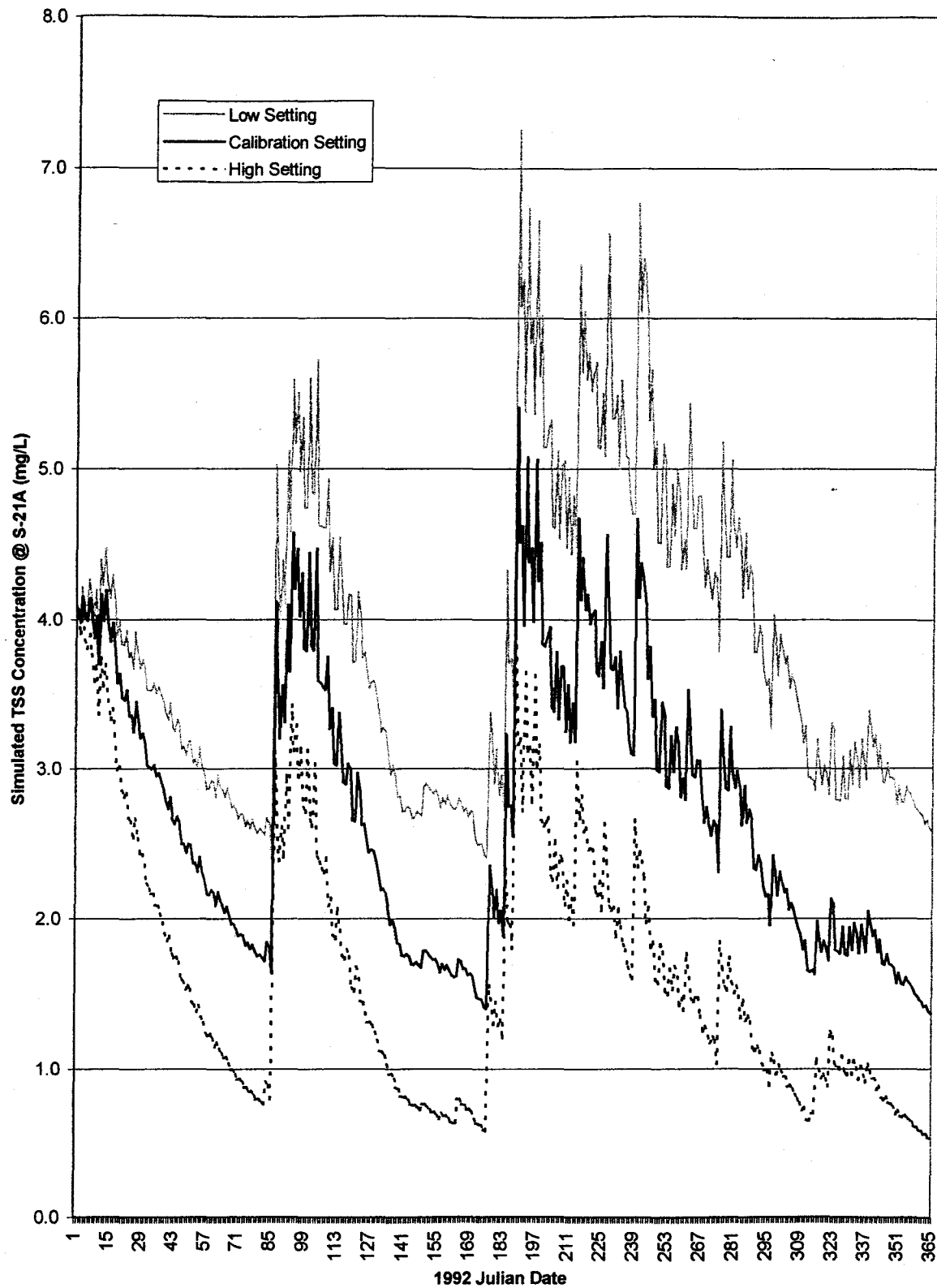


Figure 93. Time-histories of simulated TSS concentration at S-21A for three settings of water quality sensitivity analysis process. Units are in mg TSS/L.

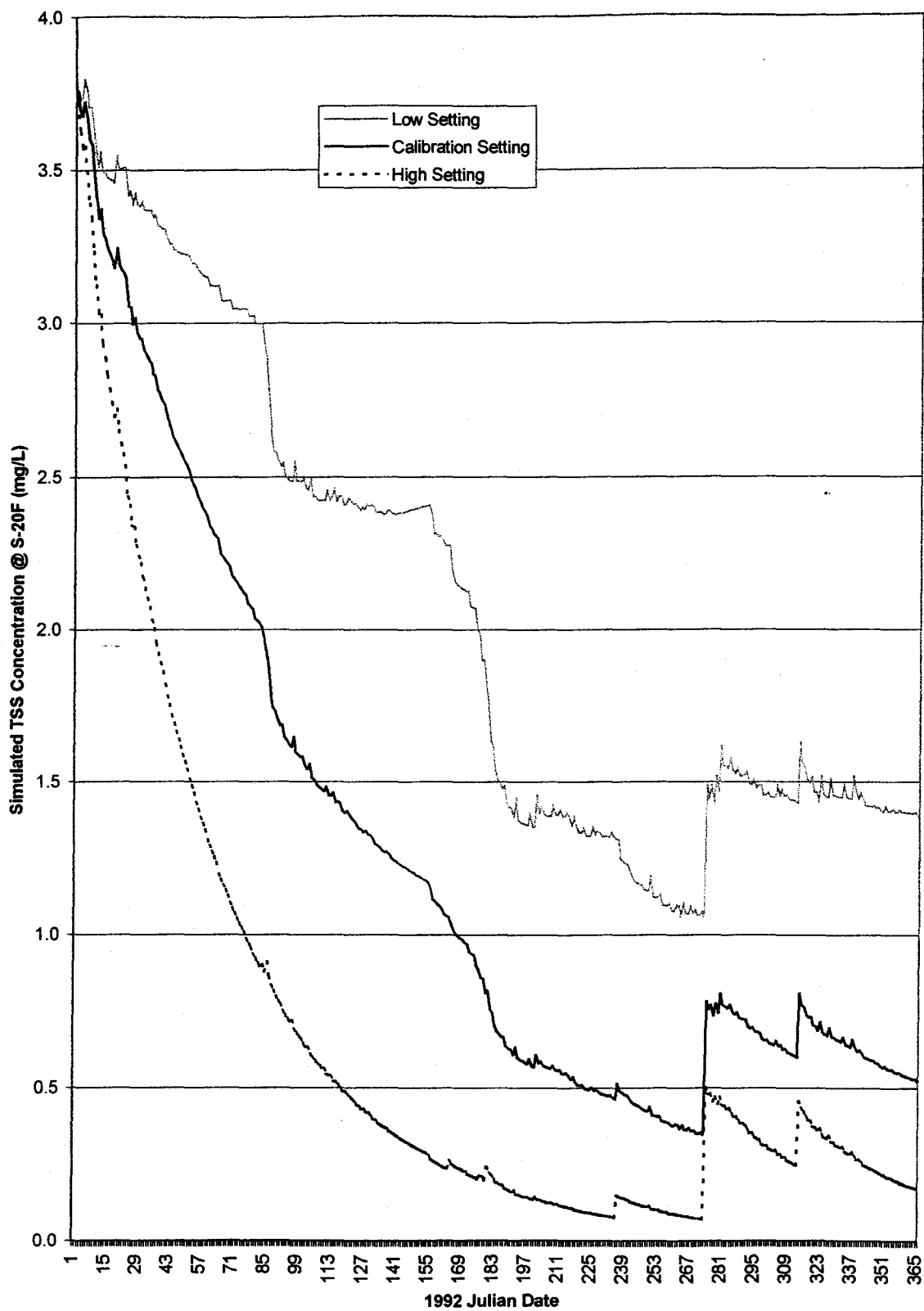


Figure 94. Time-histories of simulated TSS concentration at S-20F for three settings of water quality sensitivity analysis process. Units are in mg TSS/L.

settings. Simulated TSS concentrations were lower when the calibration coefficient setting was increased.

To further quantify the sensitivity to coefficient setting, average daily total watershed storages of each constituent were calculated for the simulations conducted at each setting value, and then compiled into Table 6. The percentage change in storage between results of simulations run with the alternate and calibrated setting values were then calculated and included in the table. As expected based on the results at the individual observation grid cells described above, the watershed-scale relationship between the sedimentation coefficient and watershed constituent storage was such that as the setting was reduced to reduce sedimentation, the constituent storage increased.

Table 6 also includes the total annual flows of water and constituents discharged from the watershed into Biscayne Bay from the simulations run with each sedimentation coefficient setting. The percentage changes in annual watershed discharges of water and constituents into the ocean from the calibrated setting were calculated and included into the table. The results suggested that TSS discharge was much more sensitive to the coefficient setting than was TP. The percentage changes between the TP discharge results from the calibrated setting and from the lower and higher settings were 2.0% and 3.2%, respectively. By contrast, simulated TSS ocean discharge increased 80.1% when the setting was lowered, and declined 60.0% when the setting was raised.

Simulation of Land Use Scenarios

Land Use Scenario Grids

The calibrated model was then used to simulate two sets of land use scenario grids. The land use scenario grids that were constructed included urbanization

Average Daily Watershed Storage

	<u>Low Setting</u>	<u>Calibrated Setting</u>	<u>High Setting</u>
Surface Water (m3)	2.34E+09	2.34E+09	2.34E+09
Ground Water (m3)	5.10E+09	5.10E+09	5.10E+09
TP (grams)	1.12E+07	1.01E+07	8.59E+06
TSS (grams)	1.90E+09	1.53E+09	1.13E+09

% Change in Average Value From Calibrated Result

	<u>Low Setting</u>	<u>Calibrated Setting</u>	<u>High Setting</u>
Surface Water	0.0	N/A	0.0
Ground Water	0.0	N/A	0.0
TP	10.6	N/A	-14.9
TSS	24.5	N/A	-26.0

Annual Watershed Discharge To Biscayne Bay

	<u>Low Setting</u>	<u>Calibrated Setting</u>	<u>High Setting</u>
Surface Water (m3)	6.19E+09	6.19E+09	6.19E+09
TP (grams)	2.38E+06	2.34E+06	2.26E+06
TSS (grams)	6.74E+09	3.74E+09	1.72E+09

% Change in Annual Discharge From Calibrated Result

	<u>Low Setting</u>	<u>Calibrated Setting</u>	<u>High Setting</u>
Surface Water	0.0	N/A	0.0
TP	2.0	N/A	-3.2
TSS	80.1	N/A	-54.0

Table 6. Summary table of watershed-scale water quality process sensitivity analysis simulation results.

scenarios, shown in Figures 95-96, and land use scenarios with WSTA networks arranged to test design alternatives, shown in Figures 97-105.

The land use grid with no agricultural or urban lands that was used to simulate the historic scenario is shown in Figure 95. The land covers were created as close approximations of those seen in Figure 9. The majority (56%) of the land area for this simulation was in wetlands, with the remainder of the area in forested uplands. The land use grid designed to evaluate the condition after a 20% increase from the 1995 population is shown in Figure 96.

Two WSTA network arrangements were developed with wetlands located randomly. The first random WSTA network (hereafter referred to as the "Random #1" network) was comprised of randomly-located wetlands sited as shown in Figure 97. The second random WSTA network arrangement (hereafter referred to as the "Random #2" network) is shown in Figure 98. These wetland networks represent possible spatial manifestations of ad hoc wetlands management approaches.

Wetlands located on lands near high sources of TP comprised another WSTA network (hereafter referred to as the "Source Proximity" network), as shown in Figure 99. Many of these wetlands occurred in more upland areas than networks that are shown below. Most of these wetlands were smaller wetlands located far from the coast, and were frequently in areas that were not historically wetlands. Another WSTA network consisting of wetlands placed in areas likely to offer chemical uptake of TP is depicted in Figure 100 (hereafter referred to as the "Sink Co-Location" network). The majority of the wetland area in this WSTA network was located within the spatial extent of the historic coastal marsh or along the primary canal stretches.

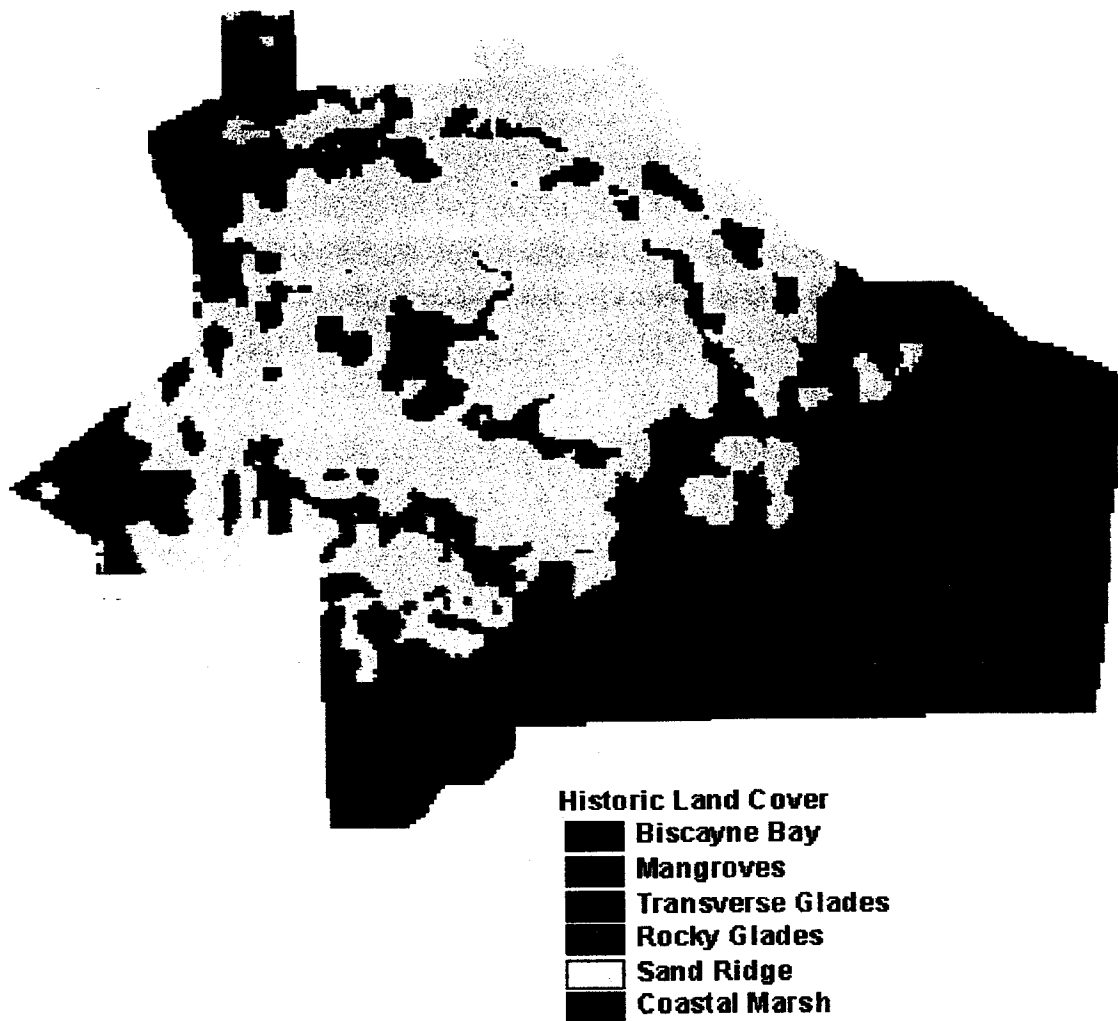


Figure 95. Grid of land cover in study area prior to extensive modern development used to simulate historic conditions.

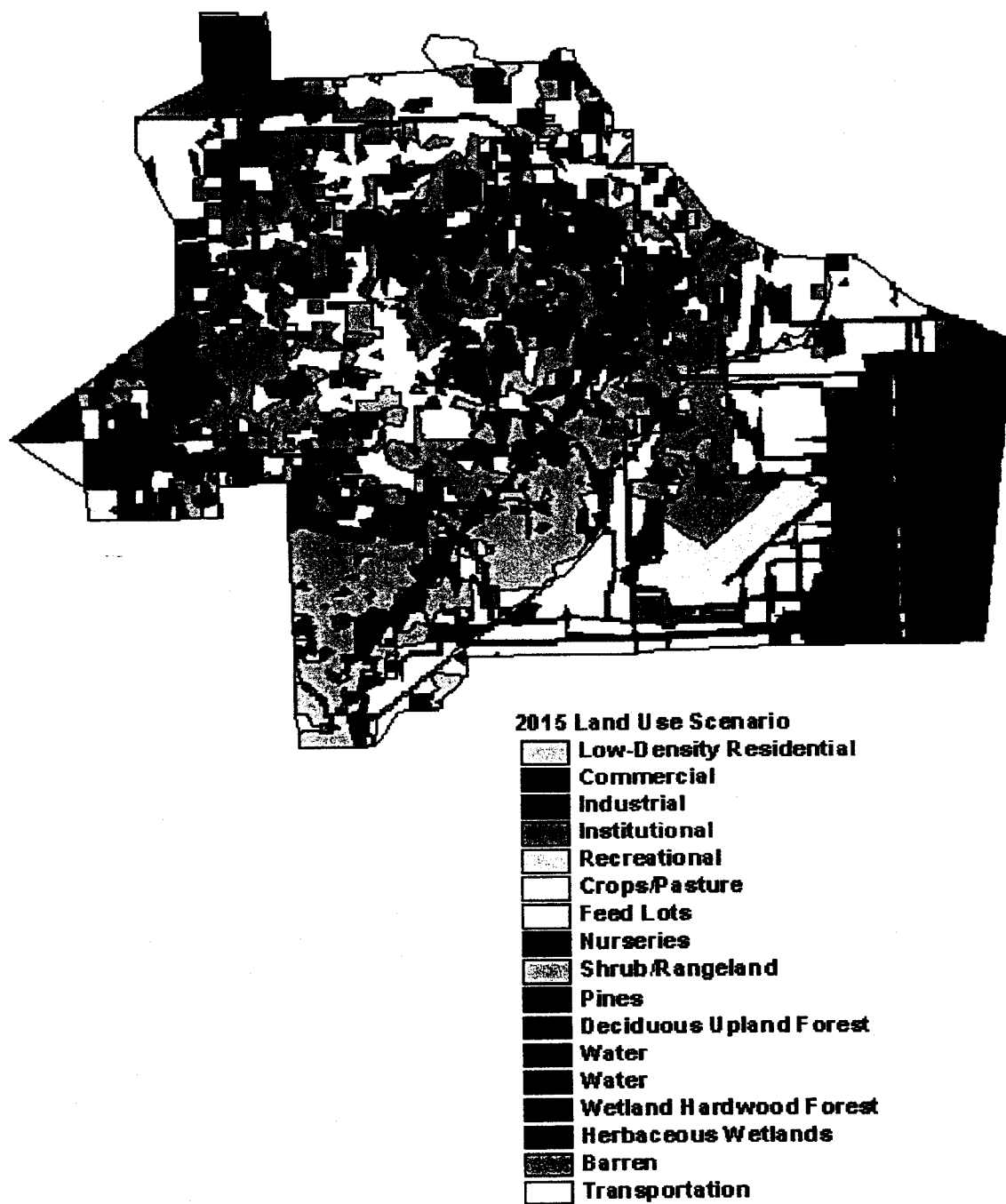


Figure 96. Land Use/Land Cover grid after 10% increase in urban lands from 1995 conditions used to simulate the 2015 land use scenario.

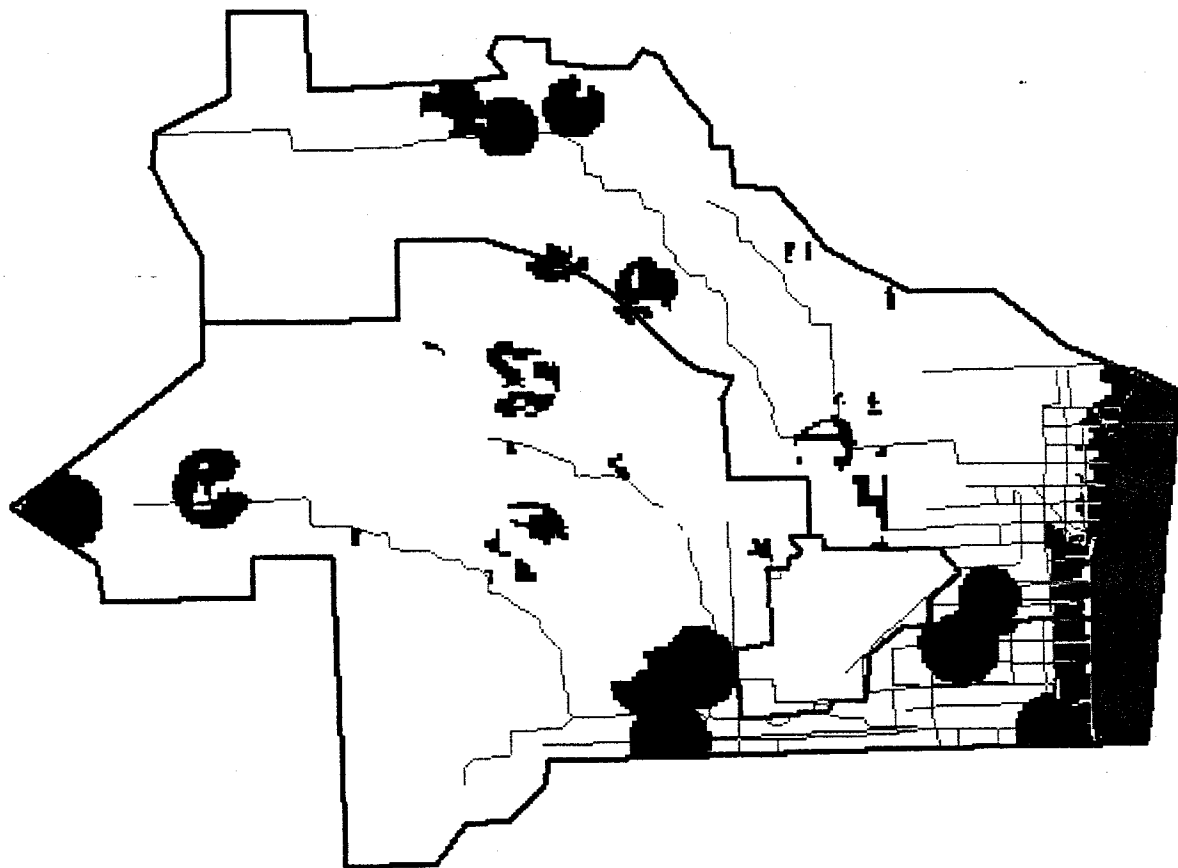


Figure 97. First land use scenario with WSTAs located randomly. Sub-basins are outlined in black, canals are indicated in grey, and Biscayne Bay is the large grey area at right.

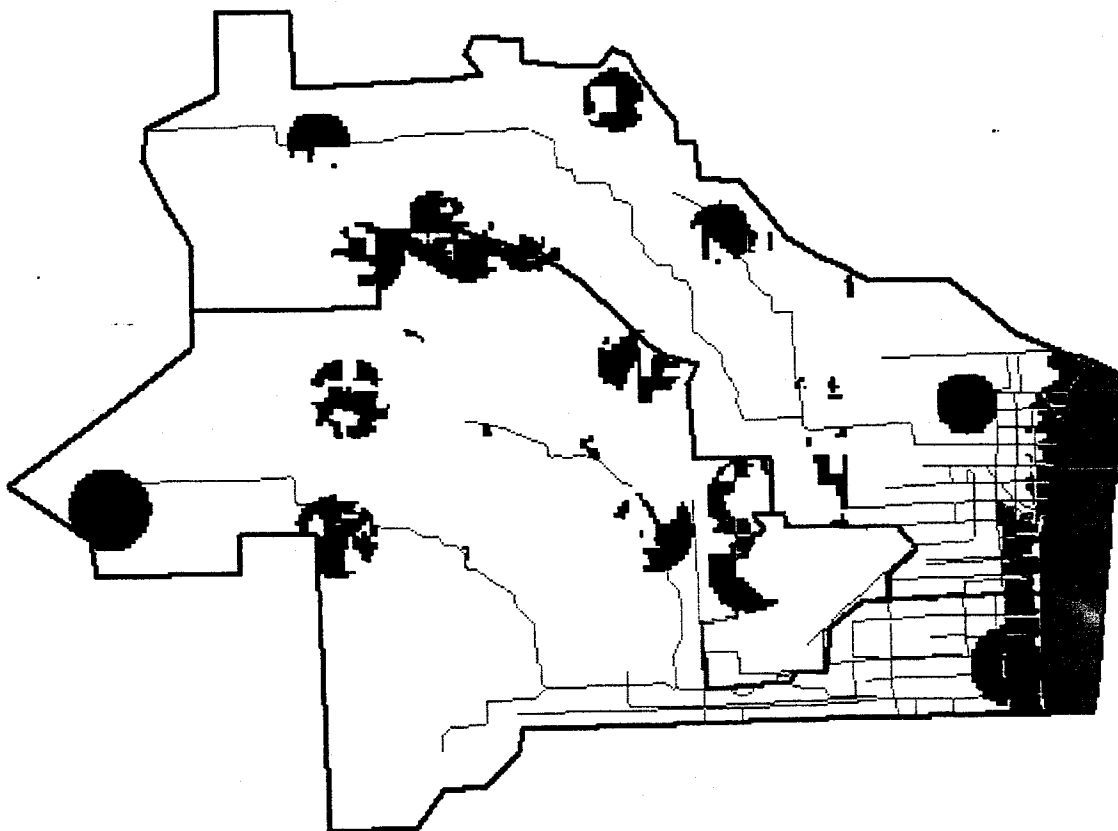


Figure 98. Second land use scenario with WSTAs located randomly. Sub-basins are outlined in black, canals are indicated in grey, and Biscayne Bay is the large grey area at right.

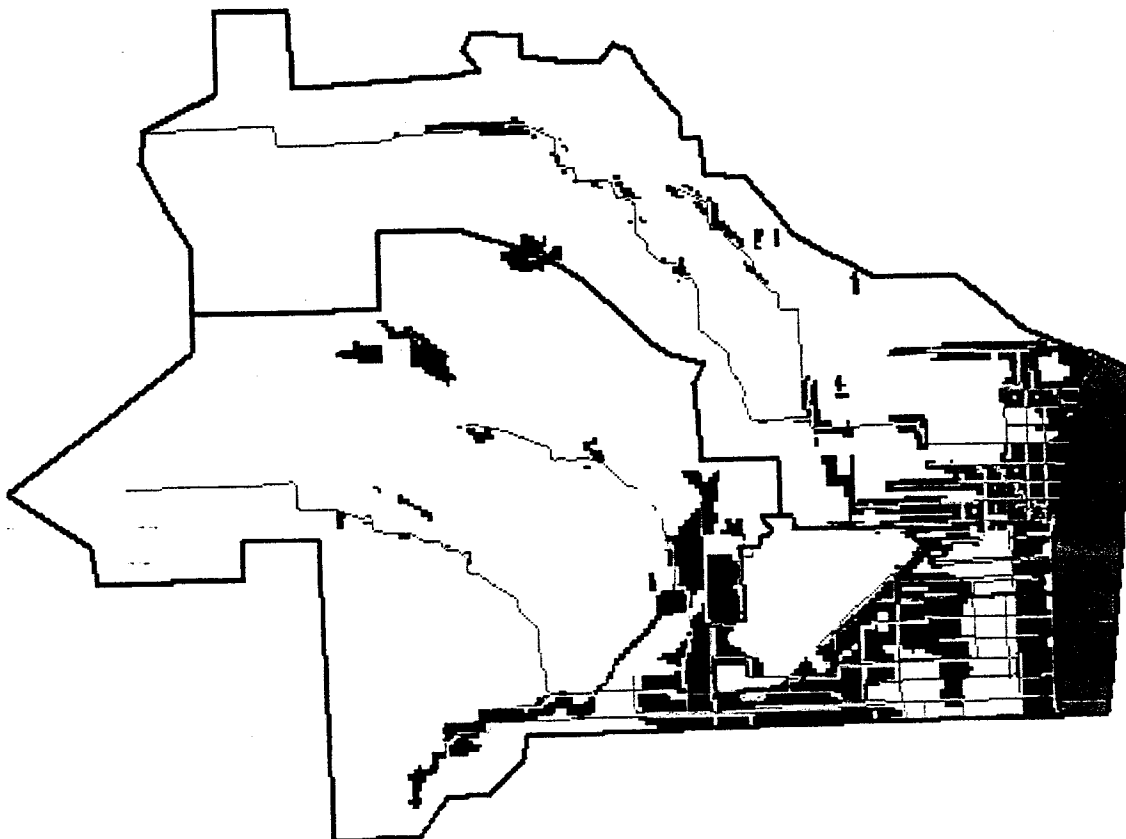


Figure 100. Land use scenario with WSTAs located on potential landscape sinks of TP. Sub-basins are outlined in black, canals are indicated in grey, and Biscayne Bay is the large grey area at right.

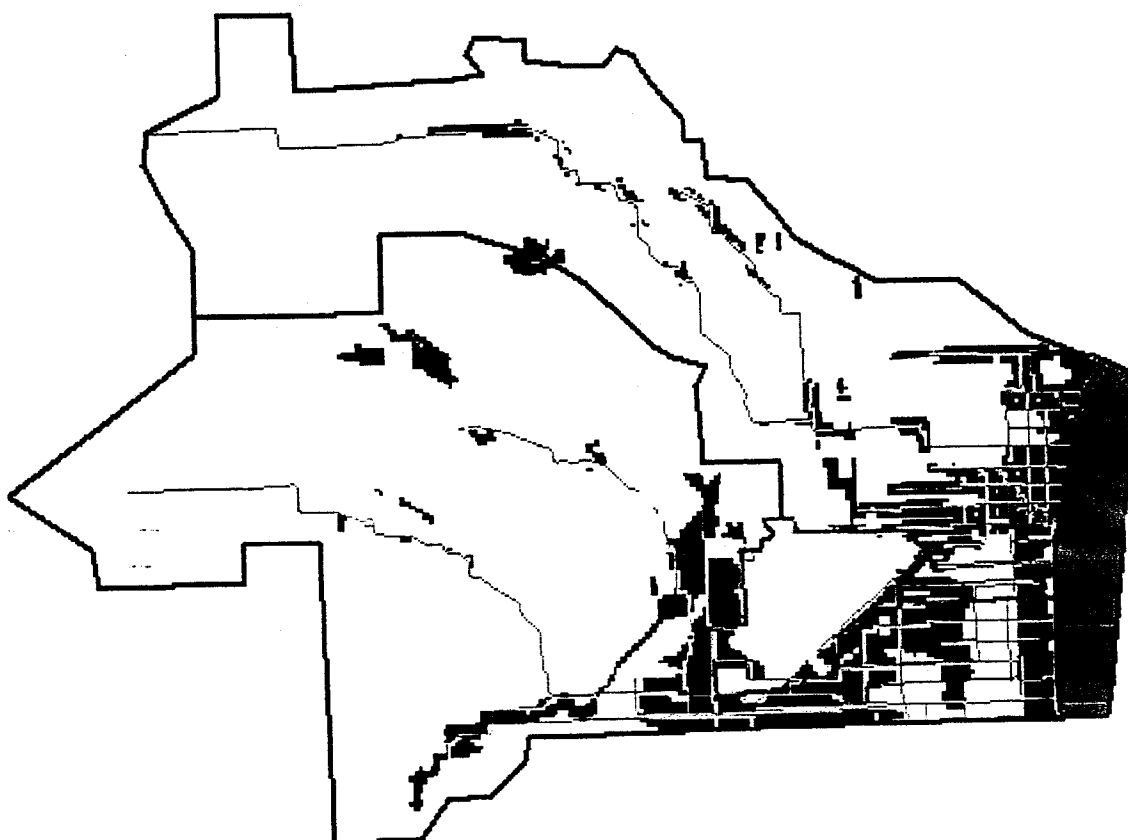


Figure 100. Land use scenario with WSTAs located on potential landscape sinks of TP. Sub-basins are outlined in black, canals are indicated in grey, and Biscayne Bay is the large grey area at right.

Another design alternative tested included a WSTA network configured in areas where hydrology was potentially favorable for wetlands, as determined based on the calibrated simulation run hydroperiod index results. The higher hydroperiod index landscape cells were included preferentially into the network (hereafter referred to as the "Favorable Hydrology" network) to produce the distribution shown in Figure 101. The resulting network included a large wetland near Biscayne Bay, some mid-size wetlands located along the canals, and many smaller wetlands spread throughout the study area.

The two WSTA networks testing effect of wetland distance from watershed discharge were created next. The WSTA network with wetlands that averaged 2000 meters from Biscayne Bay (hereafter referred to as the "2000 Meters" network) is shown in Figure 102. The network with wetlands located in a band that averaged 10000 meters from the Bay (hereafter referred to as the "10000 Meters" network) is shown in Figure 103. In both cases, many of the wetlands were not located along paths of significant surface flows.

A WSTA network was developed with a hierarchical size class distribution of wetlands in the watershed (hereafter referred to as the "SCD #1" network), and is shown in Figure 104. Many small wetlands were located throughout the watershed, medium wetlands were located along the canals, and the large wetlands were at the base of the watershed. The other network designed based upon the size class distribution (hereafter referred to as the "SCD #2" network) is shown in Figure 105. In the second size class distribution-organized network, smaller wetlands were eliminated in favor of medium and large wetlands, and it can be seen that wetlands occurred more in the lower reaches of the watershed than as in the first distribution.

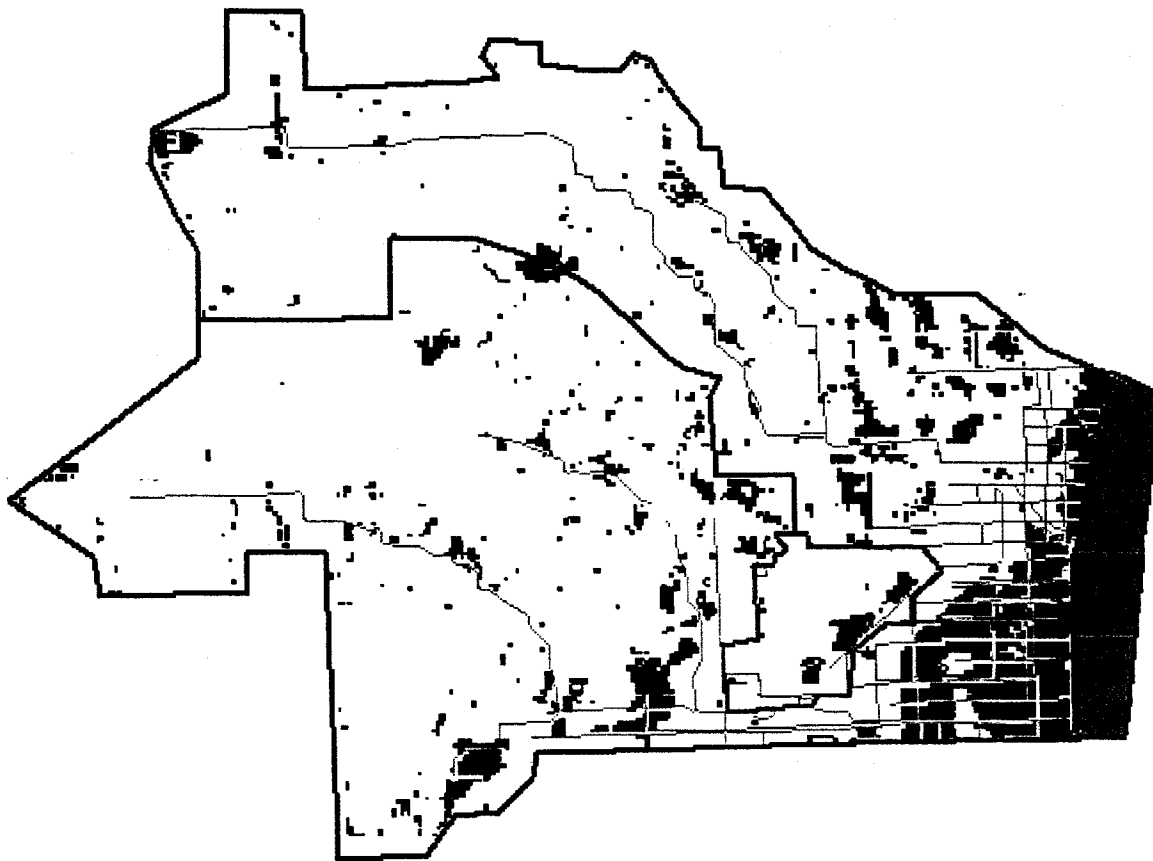


Figure 101. Land use scenario with WSTAs located on lands of hydrology potentially favorable for wetlands, based on results from the average rainfall year, 1995 land use simulation. Sub-basins are outlined in black, canals are indicated in grey, and Biscayne Bay is the large grey area at right.

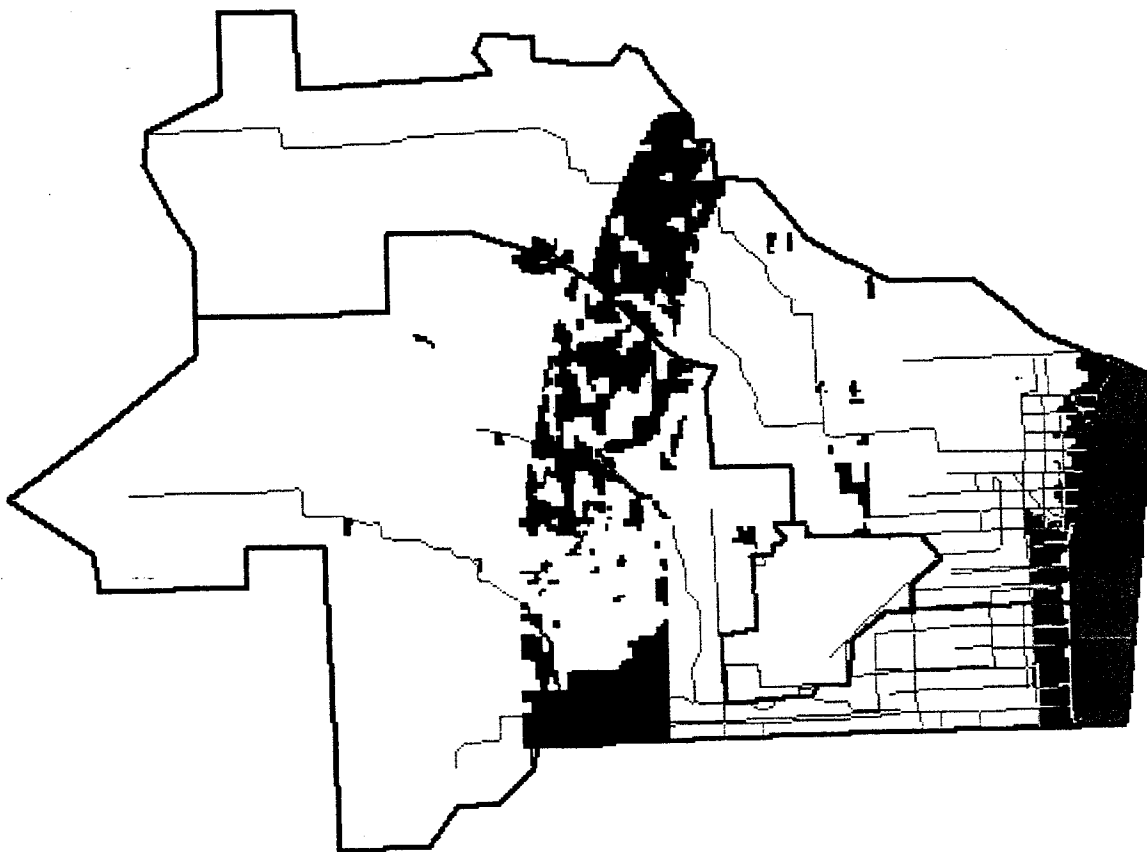


Figure 102. Land use scenario with WSTAs located an average 10000 meters from Biscayne Bay. Sub-basins are outlined in black, canals are indicated in grey, and Biscayne Bay is the large grey area at right.

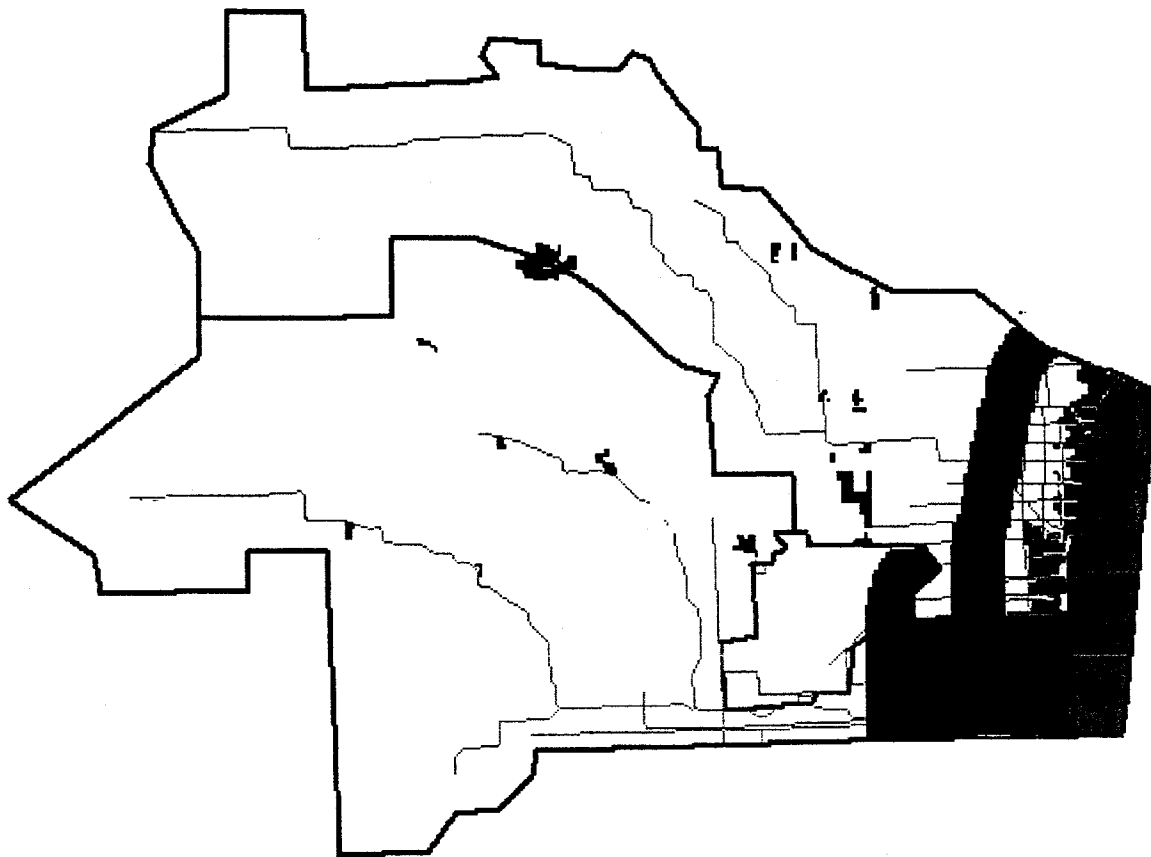


Figure 103. Land use scenario with WSTAs located an average 2000 meters from Biscayne Bay. Sub-basins are outlined in black, canals are indicated in grey, and Biscayne Bay is the large grey area at right.

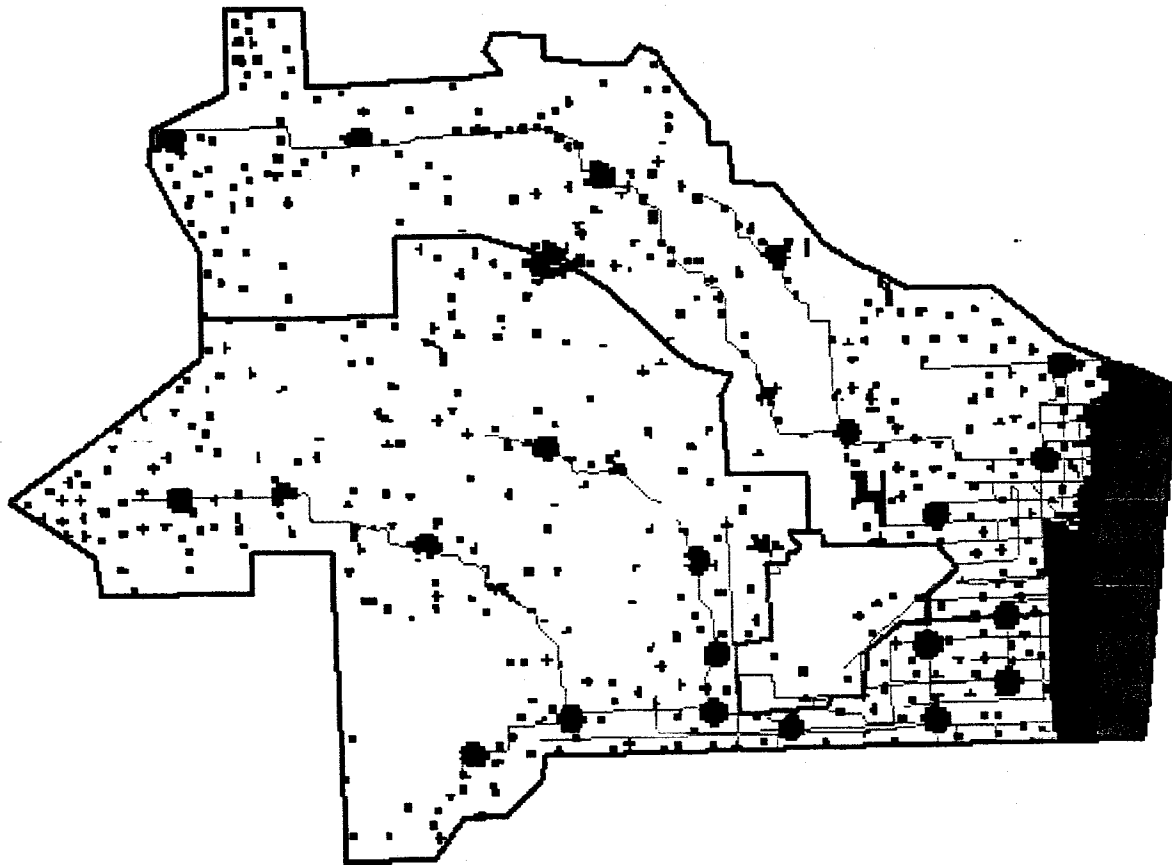


Figure 104. Land use scenario with WSTA network (SCD #1) developed to create a hierarchical size class distribution of wetlands within the watershed. Size class distribution shown is 70% small wetlands, 20% medium wetlands, & 10% large wetlands. Sub-basins are outlined in black, canals are indicated in grey, and Biscayne Bay is the large grey area at right.

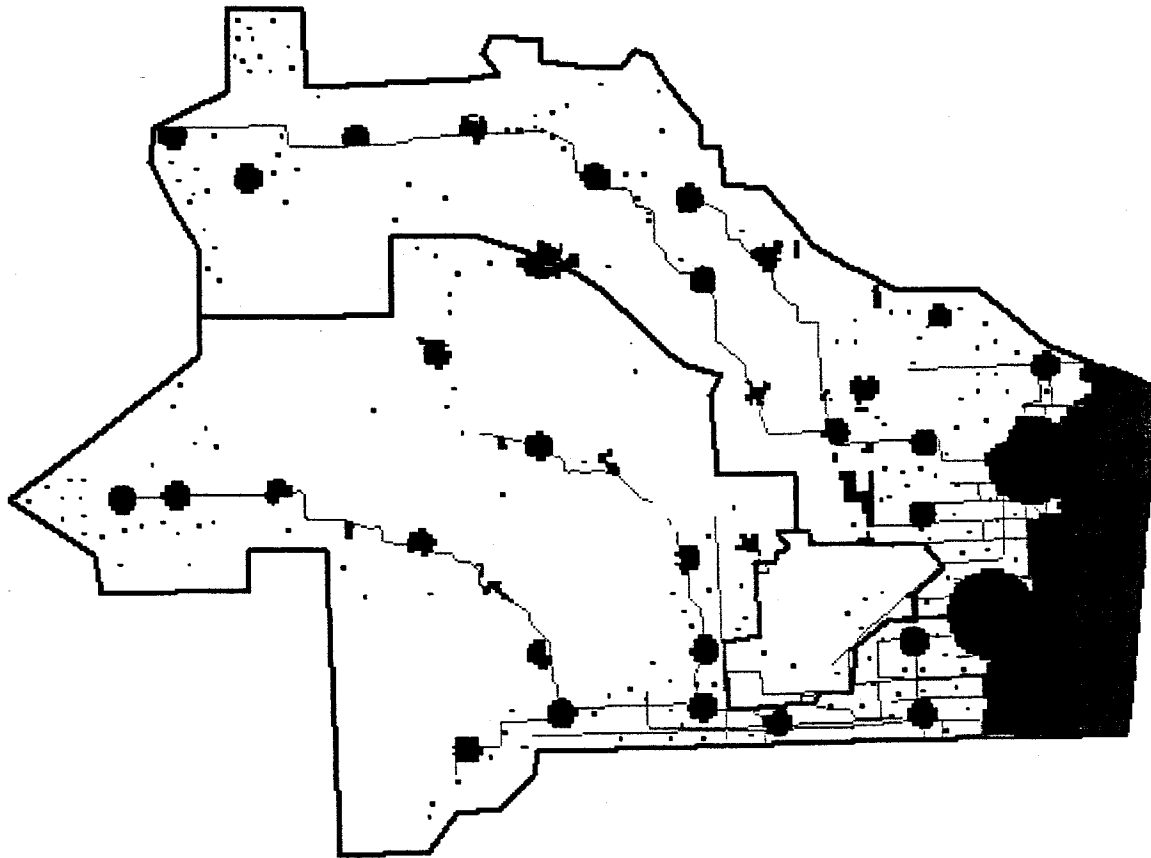


Figure 105. Land use scenario with WSTA network (SCD #2) developed to satisfy a size class distribution of wetlands within the watershed. Size class distribution shown is 5% small wetlands, 40% medium wetlands, & 50% large wetlands. Sub-basins are outlined in black, canals are indicated in grey, and Biscayne Bay is the large grey area at right.

Land Use Scenario Simulation Results

The watershed-scale hydrologic and water quality results from the simulation of the historic land cover scenario are included as Table 7. The simulated annual volume of water discharged from the watershed into Biscayne Bay was $3.45 \text{ E}+08 \text{ m}^3$, or 93.1% of the simulated 1995 land cover flow. Total annual TP and TSS flows through the modeled physical, chemical, and biological constituent processes were included into Table 7, along with calculated mass fractions of flow along each pathway with respect to total constituent outflow and to simulated flows from the 1995 land cover simulation. Comparison of the TP pathway fractions from the historic land cover scenario simulation indicated long-term biological uptake through plants was the primary (72.2% of total) simulated annual TP outflow. Simulated annual watershed TP discharge into Biscayne Bay was 2.3 % of total outflow at $6.59 \text{ E}+05 \text{ grams/year}$, and was 29.1% of the simulated 1995 oceanic TP discharge. For TSS, simulated historic watershed-scale pathway values for sedimentation and discharge into the ocean were nearly equal. Simulated annual discharge of TSS from the watershed into Biscayne Bay for the historic scenario was $3.83 \text{ E}+08 \text{ grams}$, which was 10.4% of the value from the 1995 conditions.

The hydroperiod index calculated from the historic scenario is included as Figure 106. The highest values were located to the east of the sand ridge, and in the west near the cells where the groundwater levels were explicitly controlled by the model. Intermediate index values occurred along the current locations of water management canals. An additional group of cells of intermediate values occurred in the south central parts of the analysis extent.

<u>TP Pathways</u>	<u>Grams/Year</u>	<u>% of Total Outflow Mass</u>	<u>% of 1995 Value</u>
Chemical Uptake	4.10E+06	14.2	6.7
Sedimentation	3.25E+06	11.3	10.7
Plant Uptake	2.08E+07	72.2	1122.6
Discharge to Ocean	6.59E+05	2.3	29.1

<u>TSS Pathways</u>	<u>Grams/Year</u>	<u>% of Total Outflow Mass</u>	<u>% of 1995 Value</u>
Sedimentation	3.97E+08	50.9	8.2
Discharge to Ocean	3.83E+08	49.1	10.4

<u>Water</u>	<u>m³/Year</u>	<u>% of Total Outflow Mass</u>	<u>% of 1995 Value</u>
Discharge to Ocean	3.45E+08	N/A	93.1

Table 7. Summary of simulated watershed-scale annual hydrologic and constituent flows from the historic land use scenario, and comparison to simulated 1995 values. Simulated historic flows are expressed as grams or m³, as appropriate, and as percentage of total outflows.

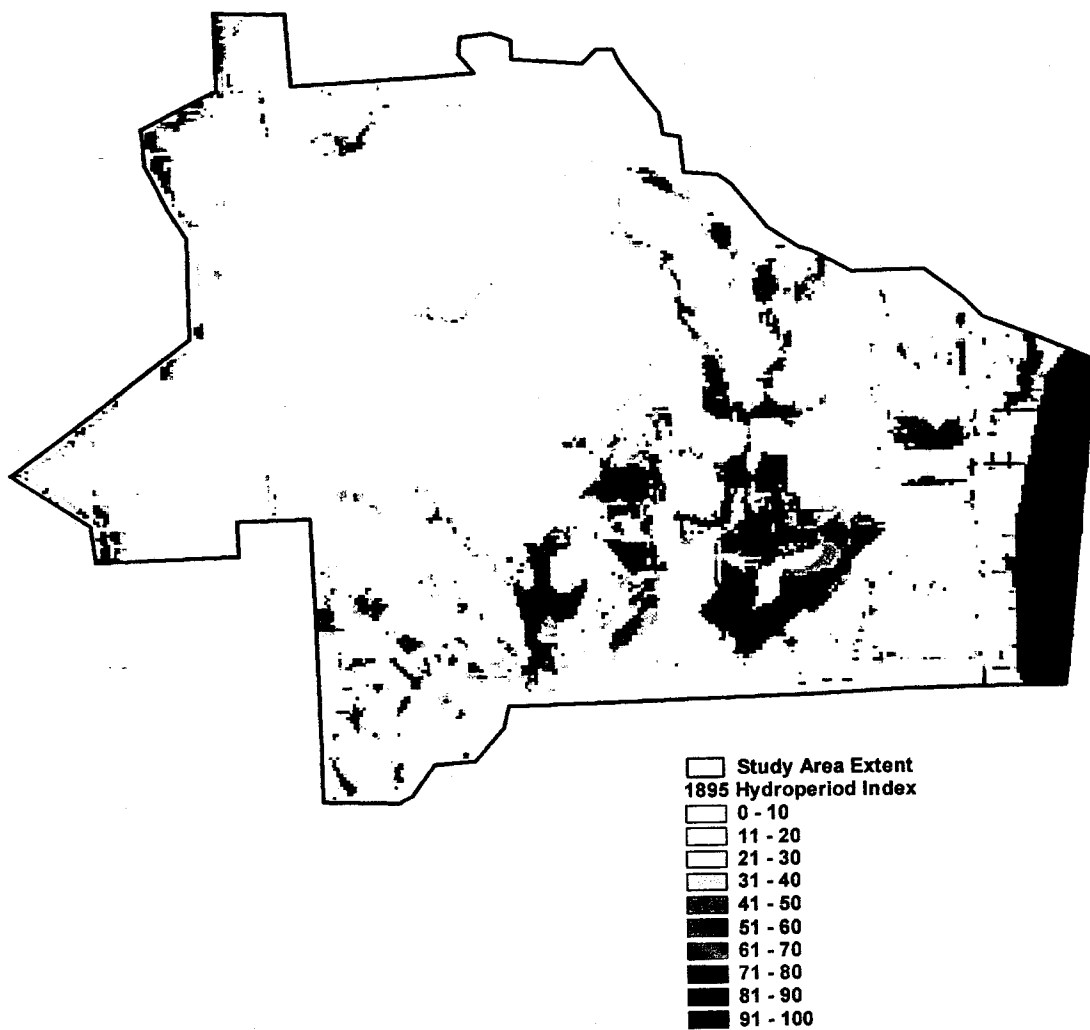


Figure 106. Hydroperiod index grid from simulation of historic land use scenario. Index values indicate percentage of year surface water storage occurred in a cell.

The watershed-scale hydrologic and water quality results from the simulation of the 2015 land use scenario are included in Table 8. Simulated annual watershed discharge of water into the ocean increased slightly from the simulated 1995 value to $3.72 \text{ E}+08 \text{ m}^3$. Simulated oceanic discharges of constituents increased to a greater degree, with annual TP discharge to the Bay increasing to $2.34 \text{ E}+06$ grams, 103.3% of the simulated 1995 flow. Simulated annual discharge of TSS to the ocean increased to $3.92 \text{ E}+09$ grams, which was 107.0% of the simulated 1995 value. The fractions of total constituent mass outflows via each pathway were essentially unchanged between the 1995 and 2015 land use conditions.

Table 9 includes watershed-scale hydrologic and water quality results from the simulation of the land use scenario with the Random #1 WSTA network. Included in the table results are comparisons, expressed as percentage values, between constituent pathway flow values from the WSTA network and the 2015 land use scenario simulations. The comparisons showed that simulated discharges of water, TP, and TSS decreased when the wetlands were incorporated. Simulated oceanic discharge of water was 98.8% of the 2015 value, at $3.68 \text{ E}+08 \text{ m}^3$. With the exception of plant uptake of TP, simulated annual flows along constituent pathways were all lower for the Random #1 network scenario than for the 2015 conditions. Simulated annual watershed discharge into Biscayne Bay was $1.20 \text{ E}+06$ and $2.17 \text{ E}+09$ grams for TP and TSS, respectively.

The hydrologic and water quality results from the Random #2 WSTA network simulation are shown in Table 10, along with the same comparisons to the 2015 condition results discussed above. The simulated annual oceanic discharge of water of $3.73 \text{ E}+08$

<u>TP Pathways</u>	<u>Grams/Year</u>	<u>% of Total Outflow Mass</u>	<u>% of 1995 Value</u>
Chemical Uptake	4.85E+07	63.4	78.7
Sedimentation	2.38E+07	31.1	77.9
Plant Uptake	1.86E+06	2.4	100.1
Discharge to Ocean	2.34E+06	3.1	103.3

<u>TSS Pathways</u>	<u>Grams/Year</u>	<u>% of Total Outflow Mass</u>	<u>% of 1995 Value</u>
Sedimentation	4.98E+09	55.9	102.5
Discharge to Ocean	3.92E+09	44.1	107.0

<u>Water</u>	<u>m³/Year</u>	<u>% of Total Outflow Mass</u>	<u>% of 1995 Value</u>
Discharge to Ocean	3.72E+08	N/A	100.5

Table 8. Summary of simulated watershed-scale annual hydrologic and constituent flows from the 2015 land use scenario, and comparison to simulated 1995 values. Simulated 2015 flows are expressed as grams or m³, as appropriate, and as percentage of total outflows.

<u>TP Pathways</u>	<u>Grams/Year</u>	<u>% of Total Outflow Mass</u>	<u>% of 2015 Value</u>
Chemical Uptake	3.28E+07	60.2	67.5
Sedimentation	1.72E+07	31.5	72.3
Discharge to Ocean	1.20E+06	2.2	51.4
Plant Uptake	3.30E+06	6.1	177.8

<u>TSS Pathways</u>	<u>Grams/Year</u>	<u>% of Total Outflow Mass</u>	<u>% of 2015 Value</u>
Sedimentation	3.21E+09	59.7	64.5
Discharge to Ocean	2.17E+09	40.3	55.3

<u>Water</u>	<u>m3/Year</u>	<u>% of Total Outflow Mass</u>	<u>% of 2015 Value</u>
Discharge to Ocean	3.68E+08	N /A	98.8

Table 9. Summary of simulated watershed-scale annual hydrologic and constituent flows from the Random #1 WSTA network land use scenario, and comparison to simulated 2015 values. Simulated flows are expressed as grams or m³, as appropriate, and as percentage of total outflows.

<u>TP Pathways</u>	<u>Grams/Year</u>	<u>% of Total Outflow Mass</u>	<u>% of 2015 Value</u>
Chemical Uptake	3.08E+07	56.5	63.5
Sedimentation	1.82E+07	33.4	76.6
Discharge to Ocean	1.58E+06	2.9	67.6
Plant Uptake	3.92E+06	7.2	210.9

<u>TSS Pathways</u>	<u>Grams/Year</u>	<u>% of Total Outflow Mass</u>	<u>% of 2015 Value</u>
Sedimentation	3.14E+09	58.3	63.0
Discharge to Ocean	2.25E+09	41.7	57.3

<u>Water</u>	<u>m3/Year</u>	<u>% of Total Outflow Mass</u>	<u>% of 2015 Value</u>
Discharge to Ocean	3.73E+08	N /A	100.1

Table 10. Summary of simulated watershed-scale annual hydrologic and constituent flows from the Random #2 WSTA network land use scenario, and comparison to simulated 2015 values. Simulated flows are expressed as grams or m³, as appropriate, and as percentage of total outflows.

m³ was nearly equal to the 2015 value. Among the significant changes in results as compared to other scenarios, simulated chemical uptake was reduced to 3.08 E+07 grams/year, and annual plant TP uptake was higher at 3.92 E+06 grams, 210.9% of the 2015 value. Annual simulated constituent discharges into Biscayne Bay were 1.58 E+06 grams for TP, and 2.25 E+09 grams for TSS.

The simulation conducted using the Source Proximity WSTA network generated the results summarized in Table 11. Simulated hydrologic discharge to the ocean was almost unchanged from the 2015 values, at 3.73 E+08 m³ for the year. The Source Proximity WSTA network generated annual simulated discharges to Biscayne Bay of 1.45 E+06 grams for TP, and 2.58 E+09 grams for TSS. Simulated constituent sedimentation was much lower for the Source Proximity wetlands for both TP and TSS. Simulated TP sedimentation was 1.69 E+07 grams/year, which was 71.1% of the 2015 result. Annual simulated sedimentation of TSS was 56.2% of the 2015 value, at 2.80 E+08 grams.

The watershed-scale hydrologic and water quality results from the simulation of the land use scenario with the Sink Co-Location WSTA network are shown in Table 12. Simulated watershed TP discharge was 1.32 E+06 grams for the simulation year, which was 56.3% of the 2015 value. Simulated annual chemical uptake of TP was 3.28 E+07 grams. Simulated annual TSS discharge into Biscayne Bay was 2.47 E+09 grams. Simulated sedimentation of both constituents were generally lower than for other scenarios, at 1.71 E+07 and 2.92 E+09 grams for TP and TSS, respectively.

Table 13 contains watershed-scale results from the simulation of the land use scenario with the Favorable Hydrology WSTA network. Simulated annual discharge

<u>TP Pathways</u>	<u>Grams/Year</u>	<u>% of Total Outflow Mass</u>	<u>% of 2015 Value</u>
Chemical Uptake	3.23E+07	59.3	66.5
Sedimentation	1.69E+07	31.0	71.1
Discharge to Ocean	1.45E+06	2.7	62.1
Plant Uptake	3.79E+06	7.0	203.8

<u>TSS Pathways</u>	<u>Grams/Year</u>	<u>% of Total Outflow Mass</u>	<u>% of 2015 Value</u>
Sedimentation	2.80E+09	52.0	56.2
Discharge to Ocean	2.58E+09	48.0	65.8

<u>Water</u>	<u>m3/Year</u>	<u>% of Total Outflow Mass</u>	<u>% of 2015 Value</u>
Discharge to Ocean	3.72E+08	N / A	99.9

Table 11. Summary of simulated watershed-scale annual hydrologic and constituent flows from the Source Proximity WSTA network land use scenario, and comparison to simulated 2015 values. Simulated flows are expressed as grams or m³, as appropriate, and as percentage of total outflows.

<u>TP Pathways</u>	<u>Grams/Year</u>	<u>% of Total Outflow Mass</u>	<u>% of 2015 Value</u>
Chemical Uptake	3.28E+07	60.3	67.6
Sedimentation	1.71E+07	31.5	72.1
Discharge to Ocean	1.32E+06	2.4	56.3
Plant Uptake	3.15E+06	5.8	169.6

<u>TSS Pathways</u>	<u>Grams/Year</u>	<u>% of Total Outflow Mass</u>	<u>% of 2015 Value</u>
Sedimentation	2.92E+09	54.1	58.5
Discharge to Ocean	2.47E+09	45.9	63.0

<u>Water</u>	<u>m3/Year</u>	<u>% of Total Outflow Mass</u>	<u>% of 2015 Value</u>
Discharge to Ocean	3.69E+08	N /A	99.0

Table 12. Summary of simulated watershed-scale annual hydrologic and constituent flows from the Sink Co-Location WSTA network land use scenario, and comparison to simulated 2015 values. Simulated flows are expressed as grams or m³, as appropriate, and as percentage of total outflows.

<u>TP Pathways</u>	<u>Grams/Year</u>	<u>% of Total Outflow Mass</u>	<u>% of 2015 Value</u>
Chemical Uptake	3.29E+07	60.3	67.7
Sedimentation	1.70E+07	31.2	71.4
Discharge to Ocean	1.18E+06	2.2	50.3
Plant Uptake	3.48E+06	6.4	187.5

<u>TSS Pathways</u>	<u>Grams/Year</u>	<u>% of Total Outflow Mass</u>	<u>% of 2015 Value</u>
Sedimentation	3.66E+09	68.0	73.5
Discharge to Ocean	1.72E+09	32.0	44.0

<u>Water</u>	<u>m3/Year</u>	<u>% of Total Outflow Mass</u>	<u>% of 2015 Value</u>
Discharge to Ocean	3.67E+08	N / A	98.7

Table 13. Summary of simulated watershed-scale annual hydrologic and constituent flows from the Favorable Hydrology WSTA network land use scenario, and comparison to simulated 2015 values. Simulated flows are expressed as grams or m³, as appropriate, and as percentage of total outflows.

values of water and constituents were lower with this network. Annual discharge of water into Biscayne Bay was $3.67 \text{ E}+08 \text{ m}^3$ for the year, which was lower than the simulated flow for both the 1995 and 2015 land use conditions. Simulated annual TP discharge to Biscayne Bay was $1.18 \text{ E}+06$ grams, which was 50.3% of the simulated 2015 flow. Simulated annual discharge of TSS to the ocean was 44.0% of the simulated 2015 value, at $1.72 \text{ E}+09$ grams.

The hydrologic and water quality flow and comparison results from the 2000 Meters WSTA network simulation are shown in Table 14. The simulated annual oceanic discharge of water of $3.63 \text{ E}+08 \text{ m}^3$ was 97.6% of the 2015 value. Annual simulated constituent discharges into Biscayne Bay were $1.27 \text{ E}+06$ grams for TP, and $2.08 \text{ E}+09$ for TSS.

Table 15 contains watershed-scale results from the simulation of the land use scenario with the 10000 Meters WSTA network. Simulated annual hydrologic discharge into Biscayne Bay was closer to the 2015 value than was the 2000 Meters result, at $3.69 \text{ E}+08 \text{ m}^3$ for the year. All constituent flows were closer to the 2015 values than were the 2000 Meters network simulation results. The 10000 Meters WSTA network generated annual simulated discharges to Biscayne Bay of $1.37 \text{ E}+06$ grams for TP, and $2.41 \text{ E}+09$ grams for TSS.

Simulation results where the WSTA network had the first size class distribution (many small wetlands) are shown in Table 16. A slight reduction between the 2015 and SCD #1 land use scenarios in annual oceanic discharge of water occurred to $3.67 \text{ E}+08 \text{ m}^3$. Constituent loads were also lower than 2015 results, especially for TSS. Simulated annual TP discharge to Biscayne Bay was $1.24 \text{ E}+06$ grams, which was 53.1% of the

<u>TP Pathways</u>	<u>Grams/Year</u>	<u>% of Total Outflow Mass</u>	<u>% of 2015 Value</u>
Chemical Uptake	3.23E+07	59.3	66.5
Sedimentation	1.74E+07	31.9	73.1
Discharge to Ocean	1.27E+06	2.3	54.1
Plant Uptake	3.54E+06	6.5	190.5

<u>TSS Pathways</u>	<u>Grams/Year</u>	<u>% of Total Outflow Mass</u>	<u>% of 2015 Value</u>
Sedimentation	3.31E+09	61.4	66.3
Discharge to Ocean	2.08E+09	38.6	53.0

<u>Water</u>	<u>m3/Year</u>	<u>% of Total Outflow Mass</u>	<u>% of 2015 Value</u>
Discharge to Ocean	3.63E+08	N /A	97.6

Table 14. Summary of simulated watershed-scale annual hydrologic and constituent flows from the 2000 Meters WSTA network land use scenario, and comparison to simulated 2015 values. Simulated flows are expressed as grams or m³, as appropriate, and as percentage of total outflows.

<u>TP Pathways</u>	<u>Grams/Year</u>	<u>% of Total Outflow Mass</u>	<u>% of 2015 Value</u>
Chemical Uptake	3.26E+07	59.9	67.2
Sedimentation	1.71E+07	31.4	71.9
Discharge to Ocean	1.37E+06	2.5	58.7
Plant Uptake	3.33E+06	6.1	179.4

<u>TSS Pathways</u>	<u>Grams/Year</u>	<u>% of Total Outflow Mass</u>	<u>% of 2015 Value</u>
Sedimentation	2.98E+09	55.3	59.8
Discharge to Ocean	2.41E+09	44.7	61.3

<u>Water</u>	<u>m3/Year</u>	<u>% of Total Outflow Mass</u>	<u>% of 2015 Value</u>
Discharge to Ocean	3.69E+08	N /A	99.0

Table 15. Summary of simulated watershed-scale annual hydrologic and constituent flows from the 10000 Meters WSTA network land use scenario, and comparison to simulated 2015 values. Simulated flows are expressed as grams or m³, as appropriate, and as percentage of total outflows.

<u>TP Pathways</u>	<u>Grams/Year</u>	<u>% of Total Outflow Mass</u>	<u>% of 2015 Value</u>
Chemical Uptake	3.25E+07	59.7	66.9
Sedimentation	1.72E+07	31.6	72.3
Discharge to Ocean	1.24E+06	2.3	53.1
Plant Uptake	3.53E+06	6.5	190.1

<u>TSS Pathways</u>	<u>Grams/Year</u>	<u>% of Total Outflow Mass</u>	<u>% of 2015 Value</u>
Sedimentation	3.72E+09	69.2	74.7
Discharge to Ocean	1.66E+09	30.8	42.3

<u>Water</u>	<u>m3/Year</u>	<u>% of Total Outflow Mass</u>	<u>% of 2015 Value</u>
Discharge to Ocean	3.67E+08	N /A	98.7

Table 16. Summary of simulated watershed-scale annual hydrologic and constituent flows from the SCD #1 WSTA network land use scenario, and comparison to simulated 2015 values. Simulated flows are expressed as grams or m³, as appropriate, and as percentage of total outflows.

simulated 2015 flow. Simulated annual discharge of TSS to Biscayne Bay was 42.3% of the 2015 value, at 1.66 E+09 grams.

The simulation conducted using the WSTA network with a size class distribution with more medium and large wetlands generated the results summarized in Table 17. As compared with SCD #1 results, the SCD #2 land use scenario resulted in higher annual chemical uptake and sedimentation of TP, while plant TP uptake and oceanic discharge of both constituents were lower. Simulated annual watershed discharge to Biscayne Bay was 1.09 E+07 and 1.42 E+09 grams for TP and TSS, respectively.

Comparisons of Collections of Land Use Scenarios

The land use scenarios were collected into two groups, and then comparisons were conducted for each group. Results from simulations run with the three urbanization condition (historic, 1995, and 2015) land use scenarios were collected into the first group, and the comparison results are shown as Table 18. Simulated discharges of water, TP, and TSS increased as urbanization was increased from pre-development to current, and then from current to projected future levels. Simulated annual plant uptake of TP was considerably higher in the historic scenario. The dominant simulated TP pathway was plant uptake in the historic scenario, but switched to chemical uptake for the 1995 and 2015 land use scenarios.

The comparisons of watershed-scale WSTA network simulation results consisted of rankings of annual flows for each network, and are included as Tables 19-25. Table 19 displays network rankings of the simulated annual flows of chemical TP uptake. Networks with the highest simulated annual chemical uptake flows were Favorable Hydrology, Sink-Co-Location, SCD #2, and Random #1. The network with the lowest

<u>TP Pathways</u>	<u>Grams/Year</u>	<u>% of Total Outflow Mass</u>	<u>% of 2015 Value</u>
Chemical Uptake	3.28E+07	60.2	67.5
Sedimentation	1.74E+07	32.1	73.4
Discharge to Ocean	1.09E+06	2.0	46.6
Plant Uptake	3.12E+06	5.7	168.0

<u>TSS Pathways</u>	<u>Grams/Year</u>	<u>% of Total Outflow Mass</u>	<u>% of 2015 Value</u>
Sedimentation	3.96E+09	73.6	79.5
Discharge to Ocean	1.42E+09	26.4	36.1

<u>Water</u>	<u>m3/Year</u>	<u>% of Total Outflow Mass</u>	<u>% of 2015 Value</u>
Discharge to Ocean	3.66E+08	N /A	98.2

Table 17. Summary of simulated watershed-scale annual hydrologic and constituent flows from the SCD #2 WSTA network land use scenario, and comparison to simulated 2015 values. Simulated flows are expressed as grams or m³, as appropriate, and as percentage of total outflows.

<u>TP Pathways</u>	<u>Historic</u>	<u>1995</u>	<u>2015</u>
Chemical Uptake	4.10E+06 (6.7)	6.17E+07	4.85E+07 (78.7)
Sedimentation	3.25E+06 (10.7)	3.05E+07	2.38E+07 (77.9)
Plant Uptake	2.08E+07 (1122.6)	1.86E+06	1.86E+06 (100.1)
Discharge to Ocean	6.59E+05 (29.1)	2.26E+06	2.34E+06 (103.3)
<u>TSS Pathways</u>			
Sedimentation	3.97E+08 (8.2)	4.86E+09	4.98E+09 (102.5)
Discharge to Ocean	3.83E+08 (10.4)	3.67E+09	3.92E+09 (107.0)
<u>Water</u>			
Discharge to Ocean	3.45E+08 (93.1)	3.70E+08	3.72E+08 (100.5)

Table 18. Table comparing simulated watershed-scale annual hydrologic and constituent flows from the three urbanization land use scenarios. Values in parenthesis indicate the percentage of simulated 1995 condition values. Simulated flows are expressed as grams of constituent or m³ of water, as appropriate.

<u>WSTA Network</u>	<u>Chemical TP Uptake (grams)</u>
Favorable Hydrology	3.29E+07
Sink Co-Location	3.28E+07
SCD #2	3.28E+07
Random #1	3.28E+07
10000 Meters	3.26E+07
SCD #1	3.25E+07
2000 Meters	3.23E+07
Source Proximity	3.23E+07
Random #2	3.08E+07

Table 19. Table ranking simulated annual watershed chemical TP uptake flows for the WSTA network land use scenarios. Simulated flows are expressed as grams of TP.

simulated annual flow, Random #2, also had the largest difference between it and the other network flows ($1.50 \text{ E}+06$ grams TP/year).

Table 20 shows the network rankings and simulated flow values for annual TP sedimentation. Most values were generally similar. The exception was for the simulated annual flow result for the Random #2 network, which was $8.00 \text{ E}+05$ grams/year higher than for the next highest network (SCD #2). The Favorable Hydrology and Source Proximity networks generated the lowest simulated annual TP sedimentation flows.

The Random #2 network also resulted in the highest simulated annual flow, at $3.92 \text{ E}+06$ grams, of plant TP uptake, as seen in Table 21. Simulated annual plant TP uptake was lowest for the Sink Co-Location and SCD #2 WSTA networks.

Table 22 includes the network rankings of simulated annual watershed discharge of TP into Biscayne Bay. The network generating the highest simulated annual discharge into Biscayne Bay was the Random #2 network, principally because of the reduced chemical TP uptake flow. Lowest simulated annual TP discharge values were achieved by the SCD #2 and Favorable Hydrology WSTA networks, at $1.09 \text{ E}+06$ and $1.18 \text{ E}+06$ grams, respectively. The 2000 Meters network simulated oceanic TP discharge was $1.00 \text{ E}+05$ grams lower than the 10000 Meters network value. Simulated annual TP flow to Biscayne Bay was second highest for the Source Proximity WSTA network, at $1.45 \text{ E}+06$ grams.

Network rankings of flow values of simulated annual TSS sedimentation are displayed in Table 23. The top two networks were SCD #2 and SCD #1. Lowest simulated annual TSS sedimentation flows were generated from the Source Proximity and Sink-Co-Location WSTA networks. Since the only other watershed-scale outflow of

<u>WSTA Network</u>	<u>TP Sedimentation (grams)</u>
Random #2	1.82E+07
SCD #2	1.74E+07
2000 Meters	1.74E+07
SCD #1	1.72E+07
Random #1	1.72E+07
Sink Co-Location	1.71E+07
10000 Meters	1.71E+07
Favorable Hydrology	1.70E+07
Source Proximity	1.69E+07

Table 20. Table ranking simulated annual watershed TP sedimentation flows for the WSTA network land use scenarios. Simulated flows are expressed as grams of TP.

<u>WSTA Network</u>	<u>Plant TP Uptake (grams)</u>
Random #2	3.92E+06
Source Proximity	3.79E+06
2000 Meters	3.54E+06
SCD #1	3.53E+06
Favorable Hydrology	3.48E+06
10000 Meters	3.33E+06
Random #1	3.30E+06
Sink Co-Location	3.15E+06
SCD #2	3.12E+06

Table 21. Table ranking simulated annual watershed plant TP uptake flows for the WSTA network land use scenarios. Simulated flows are expressed as grams of TP.

<u>WSTA Network</u>	<u>TP Discharge To Ocean (grams)</u>
Random #2	1.58E+06
Source Proximity	1.45E+06
10000 Meters	1.37E+06
Sink Co-Location	1.32E+06
2000 Meters	1.27E+06
SCD #1	1.24E+06
Random #1	1.20E+06
Favorable Hydrology	1.18E+06
SCD #2	1.09E+06

Table 22. Table ranking simulated annual watershed flows of TP discharged into Biscayne Bay for the WSTA network land use scenarios. Simulated flows are expressed as grams of TP.

<u>WSTA Network</u>	<u>TSS Sedimentation (grams)</u>
SCD #2	3.96E+09
SCD #1	3.72E+09
Favorable Hydrology	3.66E+09
2000 Meters	3.31E+09
Random #1	3.21E+09
Random #2	3.14E+09
10000 Meters	2.98E+09
Sink Co-Location	2.92E+09
Source Proximity	2.80E+09

Table 23. Table ranking simulated annual watershed TSS sedimentation flows for the WSTA network land use scenarios. Simulated flows are expressed as grams of TSS.

TSS was as discharge into Biscayne Bay, the rankings in Table 23 were reversed for annual watershed oceanic TSS discharge, as seen in Table 24.

Table 25 displays the network rankings of simulated annual hydrologic discharges into Biscayne Bay. Most networks generated very similar annual flows, specifically between $3.66 \text{ E}+08$ and $3.69 \text{ E}+08 \text{ m}^3$. Simulated annual hydrologic discharge was lowest from the simulation of 2000 Meters network. The highest simulated flow was from the Random #2 network simulation, which at $3.73 \text{ E}+08$ was also higher than the simulated oceanic water discharges from the three urbanization scenarios.

<u>WSTA Network</u>	<u>TSS Discharged To Ocean (grams)</u>
Source Proximity	2.58E+09
Sink Co-Location	2.47E+09
10000 Meters	2.41E+09
Random #2	2.25E+09
Random #1	2.17E+09
2000 Meters	2.08E+09
Favorable Hydrology	1.72E+09
SCD #1	1.66E+09
SCD #2	1.42E+09

Table 24. Table ranking simulated annual watershed flows of TSS discharged into Biscayne Bay for the WSTA network land use scenarios. Simulated flows are expressed as grams of TSS.

<u>WSTA Network</u>	<u>Water Discharged To Ocean (m³)</u>
Random #2	3.73E+08
Source Proximity	3.72E+08
10000 Meters	3.69E+08
Sink Co-Location	3.69E+08
Random #1	3.68E+08
SCD #1	3.67E+08
Favorable Hydrology	3.67E+08
SCD #2	3.66E+08
2000 Meters	3.63E+08

Table 25. Table ranking simulated annual watershed flows of water discharged into Biscayne Bay for the WSTA network land use scenarios. Simulated flows are expressed as m³ of water.

DISCUSSION

General Importance of the Watershed Model

Currently, watershed science can be described as somewhat incomplete in certain important areas. The verification of many theoretical ideas about the effects of watershed-scale landscape structure has been impeded by the practical difficulties of conducting important types of real world watershed analyses. Two significant difficulties to testing these hypotheses exist. First, the environmental variables including soils, climate patterns, land cover, geologic structure, and material cycles usually vary in character, complexity, and spatial distribution from one watershed to the next. Due to such landscape diversity, cross-watershed comparisons are confounded by the unlikelihood that an individual variable differentiating watersheds can be isolated and tested. The second difficulty is the impracticality of physically testing multiple large-scale land use scenarios within a watershed. Each land use scenario would require extensive rearrangement of the landscape, an event that usually occurs on short, research project-like time scales only because of natural disaster or war.

Modeling is a particularly valuable tool for circumventing such obstacles retarding the development and verification of scientific theories and appropriate management approaches for watersheds. The dynamic spatial model created here was unique because it was an initial attempt to preserve the resolution of the detailed spatial input data for a heterogeneous landscape within a model simulating hydrologic time dynamics. This integrated dynamic spatial modeling approach allowed the model to simulate hydrologic processes, and the systems governed by such processes, in a manner

not possible by current hydrologic models. Ultimately, the finished model revealed interesting spatial water quantity and quality trends between historic, current, and future land cover scenarios. It also indicated differences in effects of spatial stormwater wetland network configuration on water quality, which is useful for developing regional management plans.

Development of Spatial Lateral Flow Relationships

The principle challenges faced during model development centered around the need to define new types of lateral flow-path relationships. The modeling environment was primarily composed of areas of unconfined, non-channelized flow, which prohibited use of mathematical equations employed by more traditional models. Various algorithms involving functions provided by ArcView Spatial Analyst were tried until satisfactory results were obtained. Downstream DRAINing and the calculation of hydrologic slope within each cell were two of the most important functions involved in simulation of lateral flows.

DRAIN Process

At the heart of the lateral flow algorithms was the ArcView Spatial Analyst command "FlowAccumulation" (many GIS applications have an analogous function). Use of "FlowAccumulation", a function clearly designed specifically for static spatial modeling, as shown previously, required development of the mass-balance error correction to the DRAIN process. The correction weakened the ability to tweak lateral flow magnitudes according to landscape attribute-based factors alone, because it added two non-attribute factors to the lateral flow calculation. The first added factor was the ratio of runoff-out to runoff-in (or in the case of groundwater, the ratio of lateral seepage-

in and -out) within each cell. Such a factor isn't as much a concern to real water molecules as much as forces such as gravity (slope) and friction (impervious surface, permeability), but became very influential to the calculation of simulated flows.

The second added factor dealt with the actual size of the sub-watershed over which these ratios were calculated. In addition to maintaining mass-balance, the ratio had the effect of evening out flows between low and high flow areas (i.e. relocating some water from the lower to the upper watershed). Initially, the ratios were calculated for the study area as a whole. It quickly became apparent that this method translocated considerable quantities of water across the boundaries of hydrologically separate regions. The solution developed to partition the study area into sub-watersheds and then calculating the ratios within each area involved required setting a minimum size (number of cells) for what the model considered a distinct watershed.

The setting of the threshold watershed size was particularly important to the hydrologic simulation results within individual cells, as indicated by the surface water process sensitivity analysis. Larger threshold watershed sizes allowed more water to move greater distances across the analysis grid, but induced increased model instability. The most dramatic manifestations of the instability included cells with geyser-like hydrology to considerable re-distribution of water from canals to adjacent upland cells. A more common type of model instability, as shown by the surface water process sensitivity analysis time-histories (well illustrated at S-194 and S-20F), was an increase in both the day-to-day and seasonal fluctuation in surface water storage values. However, the sensitivity analysis also revealed that the watershed size threshold setting did not have much impact on overall watershed-scale hydrology. Specifically, simulated

total watershed hydrologic storage and hydrologic discharge into the ocean were almost unchanged.

It would be preferable to simulate lateral flows simply on the basis of physical landscape attributes and hydraulic gradients, as is done by classical hydrodynamic equations. Such calculations would be possible if GIS was provided with additional functions that allocated a specific quantity of water into one, rather than all, downstream cell(s). The distance of flow from upland source cell, down a flow-path, and into an individual downstream cell could be based upon the physical characteristics of the flow-path. The dynamic model has been useful in demonstrating the importance of developing such functionality into future GIS application software.

Calculation of Hydrologic Slope

Lateral flows were significantly influenced by the method of calculating hydrologic slope. Initially, the "Slope" function provided by ArcView Spatial Analyst was utilized. The problem with using ArcView "Slope", as well as the slope functions provided by most other commercial GIS software, is that the slopes are calculated solely on the basis of elevation gradients between the target cell and immediately adjacent cells. Although this can suffice with static models, in the dynamic spatial model it was inadequate. To illustrate the inadequacy, consider the cells where water was delivered from L-31 W into C-102 and C-103. For purposes of avoiding model instability, delivered water was added to several cells along the leg of each canal near L-31 W. As a result, before the runoff-out processes there would be a series of cells resembling the hydrologic equivalent of a geologic plateau. When the ArcView "Slope" function was executed, the easternmost cell at the edge of the plateau would have its slope calculated

compared to the rest of the canal, which was relatively much lower. As would be expected, the resulting runoff-out from this cell would be high, and the delivered water would spread out through the rest of the canal. However, ArcView "Slope" for the rest of the cells within the plateau would be calculated compared to other cells that also had the delivered water, and would have a comparably lower slope value. As a result, runoff-out from these cells would be lower, and the delivered water would not distribute through the canal system.

In addition, ArcView "Slope" simply takes whatever the maximum slope is between a cell and any of its neighbors. For hydrologic modeling purposes, the important slope is slope to the next downstream cell, rather than simply the maximum slope to any neighbor cell. Such a slope function would be another useful development project for GIS software developers.

The dynamic spatial model employed a type of downstream slope function by comparing a cell's hydrologic elevation value to the lowest hydrologic elevation found within the local sub-watershed. For canal cells, the cell elevation was compared to the minimum canal stage within each canal segment. The calculation of slopes on this more zonal basis deviates from the model's otherwise cell-based approach. As will be discussed shortly, the modification was done in the interest of speeding up model response time to changes in hydraulic gradients. The increase in spatial area over which slope was calculated allowed hydrologic interactions to occur at distances greater than 76 meters (the cell size of the model) for a simulation time-step of one day.

Characteristics of Simulated Hydrology

The Rainfall Relationship

Rainfall, as the principal input of new water into the simulated analysis extent, was closely tied to overall simulated hydrology. When rainfall levels were moderate, the model simulated hydrology about as occurred in reality. The calibrated model did not simulate as well when rainfall became either very low or very high. Very low water levels, even below sea level, occurred during the simulated dry seasons of the low and high rainfall year simulations. The very high rainfall period at the onset of the average rainfall year simulation generated canal stages well above observed stages, and differences between simulated and observed stages were at their greatest.

Response Time

The single-cell time-histories revealed characteristics about model response to changes in water level gradients. The time that was required for water levels to decline after sudden rainfall event-generated increases was longer in the wet season than in the dry season. As indicated by the calibration simulation error metrics, the simulated dry season response times to water level changes were usually very similar to observed responses at most points. By contrast, canal water levels did not decline as rapidly in the simulated stages as compared to observed values after the larger wet season water level increases. The most obvious examples, as illustrated by many of the canal stage time-histories, of increased response times were associated with the water level rise that occurred at the onset of the wet season between simulation dates 160 and 190. Similarly, sub-surface water tables exhibited a time lag for the simulated values to decline to the observed levels after the initial wet season increase.

Changes in calibration coefficients that reduced the response time to gradient changes also compromised model stability. The model instability was created because of

a spatial phenomenon generated by the runoff process. Under certain hydrologic conditions (such as high flow periods), individual cells along flow paths finished a simulation day with relatively higher water levels than adjacent upstream or downstream cells. Cells with these water level spikes short-circuited flows further downstream until the runoff process would eliminate the transient spike on the next simulation day. As calibration coefficients were tuned to increase lateral flows, the frequency and magnitude of such spikes increased. A good example of such an instance was provided by the comparison of the three time-histories of G-757A groundwater elevations obtained during the ground water process sensitivity analysis (Figure 85). In certain cases, the instability created by these disruptions was even enough to create significant water level oscillations in a neighborhood of cells that would last many days.

As a counterintuitive result, the effect of trying to increase lateral runoff sometimes had the opposite effect of slowing down overall broad-scale lateral flows. The time-history of canal stages at S-165 from the calibration condition simulation (seen in Figure 27) provided a clear example of this phenomenon. Day-to-day water level fluctuations were greater because of increased lateral flow volumes. However, the time it took for water levels to generally drop off during the drier period at the end of the simulation year was longer.

A probable solution for reducing response times without decreasing stability would be to shorten the time step to 8 or 12 hours. The tradeoff of increasing the temporal resolution is that simulation time rises proportionally. In light of the substantial time that would be required, reducing the simulation time-step was not considered feasible for this project.

The zonal approach to calculating slope was a successful method for reducing response time to water level fluctuations in a stable manner. In addition to circumventing the plateau effect described previously, the number and interference of spike cells along flow paths was reduced. As a result, higher runoff-out flows were achievable, and water levels declined faster.

Dynamic Flow Direction

The model was designed to simulate Variable Source Area hydrology, and this relied upon the dynamic flow direction method. To see if the model actually simulated the flow direction dynamically, flow direction grids from individual simulation dates throughout the year were frequently checked. No two flow direction grids were ever found to be alike, although most were generally similar, with most cells flowing to the south, southeast, or east toward Biscayne Bay. Cells which showed the most frequent changes included low spots, urban lands, and cells near the canal gates. The dynamic flow directions in low spots are characteristic of wetland hydrology. Furthermore, the dynamic flow directions eliminate some concerns about landscape sinks in traditional GIS hydrologic modeling.

The other interesting aspect of the dynamic flow directions was the change in the character of the groundwater flow direction grids as the simulation year progressed. At the beginning of the year, flows were almost exclusively toward the coast. During mid-summer, highest water tables were in the middle of the analysis extent coincident with the sandy ridge. At this time, the sub-surface flow direction grid indicated water moving in all directions, including westward away from the sand ridge.

Simulation of Sheetflow

Sheetflow was an important characteristic of the historic study area hydrology. In addition, future medium and large wetlands constructed in the study area will probably include sheetflow as an important component of the hydrologic design. The model's ability to reconstruct sheetflow was difficult to succinctly assess, but important to understand. The behavior of the ArcView hydrologic modeling commands lends some insight into the potential for the generation of sheetflow by the model. The combination of "FlowDirection" and "FlowAccumulation" routes water from one cell into only one other cell. Even if two cells adjacent to an upstream cell have the same elevation, only one of the cells will receive inflow according to the algorithms used by the "FlowDirection" function. As a result, the simulated flows travel along linear flow paths, rather than as the broad fronts of spreading water found in sheetflows.

However, after several time-steps, the dynamic flow directions had the effect of spreading the flows across the landscape. Hydroperiod index grids, in particular the historic land use scenario simulation index grid, frequently contained neighborhoods of twenty or more grid cells with intermediate index values. Closer scrutiny of runoff-out and runoff-in flows in these grid cells revealed that lateral flows were indeed spreading out across several cells over time. In conclusion, then, the model did not simulate sheetflow as a daily process, but did so over a period of several days. Reducing the time-step would allow the dynamics of flow directions to offset the GIS function limitations and enhance simulation of sheetflow hydrology.

Of course, another method for simulating sheetflows would be to modify the GIS functions to allow water to be distributed downslope to more than one adjacent cell where elevations are the same.

Water Quality Simulation Result Characteristics

The definitive characteristic of water quality results was a strong relationship between simulated constituent concentrations and rainfall. The relationship was illustrated by the increase in constituent concentrations generally observed after the onset of the wet season, as well as by the differences between simulated concentrations during alternate rainfall years. Since the major constituent source was rainfall-derived EMC loading, the characteristic was expected.

The ground water process sensitivity analysis results reflected the importance of simulated TP loads originating in soil water that discharged from adjacent cells into canals. As the calibration coefficient was adjusted to allow greater lateral ground water flow, the exchange of water between the surficial aquifer and the canal network increased. Since TP loads from groundwater emerging at the surface were simulated in proportion to these flows, total watershed TP storage and discharge into Biscayne Bay increased. In contrast, since no simulated load of TSS was associated with these waters, total watershed TSS storage and ocean discharge remained essentially unchanged.

Dominant Constituent Pathways

A major difference was exhibited in dominant pathway between the simulated TP and TSS processes. Specifically, the dominant TP pathway was through chemical uptake, whereas TSS dynamics were controlled more by sedimentation. The table of watershed-scale results from the calibrated conditions (Table 3) clearly indicated this difference in simulated fate for loads of each constituent. The tighter relationship that existed between simulated sedimentation and TSS results was also reflected by the water quality process sensitivity analysis, which tested model sensitivity to the constituent

sedimentation process. Total watershed storage and ocean discharge sensitivity analysis results varied more for TSS than for TP.

Discharge To Ocean

As with constituent concentrations, total watershed discharges into Biscayne Bay followed rainfall patterns. The discharges for the alternate rainfall years generally increased along with greater rainfall. The exception was for TSS, which during the high rainfall year conditions was less than for average rainfall year TSS ocean discharge. Although wet season TSS oceanic discharge during the high rainfall conditions was greater than average rainfall year wet season discharge, it was not enough to overcome the reduction in dry season TSS discharge experienced during the high rainfall year. This suggests that simulated TSS oceanic discharges were particularly sensitive to lower rainfall conditions at both annual and seasonal time scales.

The effect of the spatial distribution of land use/land cover on both magnitude and temporal flux of constituents into the Bay was displayed by the conspicuous difference in the daily ocean discharge patterns between TP and TSS. TP ocean discharge exhibited marked day-to-day fluctuations, whereas the time-history of TSS discharge was smooth, indicative of slower response. Lands with high TP EMC values occur throughout the study area, including areas near the Bay. By contrast, the lands with the highest TSS EMC values are generally the urban and transportation areas, which are primarily confined to the center of the study area. The shorter distances between the highest TP sources areas and the ocean resulted in less opportunity for landscape deposition and removal processes to buffer the sudden simulated constituent loading from rainfall events.

Interestingly, increased watershed discharge of constituents into the ocean coincided with greater model instability. In the case of both hydrologic process sensitivity analyses, total watershed hydrologic storages and discharges into the Bay were nearly equal for all settings. However, the hydrologic sensitivity analysis simulation runs conducted with settings that generated more instability also generated greater oceanic constituent discharge. The explanation lies in the fact that model instability resulted in more water washing laterally back and forth (as runoff) across the landscape, which increased EMC-derived constituent loading.

Discussion of Significant Model Parameters & Assumptions

Operation of Canal Gates

The observed canal stage time-histories offer evidence that the operation of water management canal gates is the primary control on the hydrology in the study area. For example, two sudden declines in 1992 S-21A and S-20F stages around simulation dates 170 and 230 indicate the water management reaction to flooding dangers (the latter date in anticipation of Hurricane Andrew), when the canal gates were opened. By contrast, gates are closed during most of the dry season, and water level declines were more associated with slower hydrologic processes such as evapotranspiration losses.

The model operated the canal gates located at various points along C-102 and C-103 according to the current published regulatory delivery schedule for each gate. A review of the observed canal stages, actual flows over the canals, and the delivery schedule revealed that the gates are not operated according to this schedule at all times. The other considerations including tropical cyclone threats and water levels in other parts of south Florida influenced the actual operation of the gates in ways that were not

simulated by the model. It may then be concluded that some of the discrepancies between the simulated and observed values were due to these divergences between the published schedule and actual operations.

A test simulation run performed to specifically hold gates open for short periods did not significantly change simulated canal stages, although annual oceanic discharges increased slightly. This suggests that water management operations were not as important to simulated hydrology as they are to actual hydrology.

Continuous Vertical Soil Column

The model assumed that the entire soil column was a homogenous extension of the surface soil type. The sandy Atlantic coastal ridge, in actuality usually only a few feet deep, was extended to the bottom of the simulated soil column (sometimes greater than 35 feet in depth). For the model, sand and sandy loam replaced the limestone substrate that actually sits underneath the ridge, which changed the attributes of the sub-surface. The central portions of the sand ridge were comprised of soils of high permeability and low water holding capacity. Potentially, simulated lateral seepage may have been greater than the actual flows, since groundwater seepage calculations were based partly on permeability. However, the high porosity of the sub-surface limestone suggests that if flows were overestimated, the error should not have been very large.

Significantly, the low water holding capacity may have influenced the accuracy of the groundwater table calculations. Water holding capacity, the central landscape attribute used in the water table calculation, is higher in the sub-surface limestone than in the sandy ridge, and the replacement of this limestone with sand for the simulation reduced the water holding capacity in these areas. For a given volume of water, a lower

water capacity resulted in an increase in the simulated water table value (i.e. the water was squeezed upward in the lower capacity environment). Consequently, the model would have inflated the groundwater elevation variations in the vicinity of the sand ridge. Comparison of observed and simulated values at the groundwater calibration point, located along the western portion of the ridge, seems to corroborate this line of reasoning.

Although partitioning the sub-surface into distinct layers may have added more accuracy to the soil water values output from the simulation model, the complexity added to the modeling process almost certainly would have outweighed the benefits.

Simulation of shallow groundwater was only included to improve simulated surface water quantity and quality, and was not intended as a primary focus such as in a surficial aquifer model. It is felt that the groundwater simulation model did a sufficient job for the purposes, and the lessons learned here may be useful for future endeavors, which could include either simpler or more complex models.

Continuity of Landscape

An important facet concerning modeled representation of the landscape involves the edges of areas with distinct soil types. The primary digital soil coverage was composed of hard boundaries between soils classified as sands, sandy loams, gravels, marls, and limestones. The attribute values (calcium carbonate percent, water holding capacity, permeability) vary as much as three orders of magnitude between some of the soil types. When attribute grids were created, the grids contained cells adjacent to one another with considerably different values. The more extreme values in the cells of grids of simulated groundwater elevations and lateral groundwater seepage volumes frequently occurred along these boundaries. It may be safely reasoned that, in some places, the

changes in soil properties in the study area are not so sudden as was modeled. The effect of sharp edges instead of smooth transitions in attribute grids may be of some importance for dynamic spatial modeling.

One method of minimizing the effect of sudden edges was unconsciously employed in the model. Originally conceived for tuning lateral flows, calibration coefficients for threshold soil property values effectively minimized differences in attribute grid values. Satisfactory model performance was achieved by reducing the range of values within each soil property grid from three orders to one order of magnitude.

Boundary Conditions

Boundary conditions in a region where sub-surface water freely flows across surface watershed boundaries are an important consideration. The model sensitivity to boundary conditions is best illustrated by considering the hydroperiod index grid from the historic land use condition. Most index values in the western study area were zero to forty, probably low when the fact that this was the historic Everglades is taken into account. The index values make sense when one considers that the significant historic surface and soil water flows from the north and northwest were not included in the model. Simulated water levels in the western study area were therefore depressed from those of reality. The model assumption that no sub-surface flows crossed watershed boundaries was not as hydrologically significant in other parts of the study area as along the west and northwest boundaries.

Seasonality of Event Mean Concentrations

The Event Mean Concentrations (EMCs) used to generate constituent loads from upland areas during runoff events were modeled as constant values throughout the year. In reality, EMCs probably exhibit distinct seasonal patterns due to several influences. The influences include time between rainfall events, differences in storm type (quick/intense versus slow/gentle rainfall), temperature effects on chemical cycling and ecosystem kinetics, and seasonal changes in human activities on uplands. In addition, the seasonal patterns certainly vary for different land use types. For example, EMCs for agricultural lands are highly dependent upon timing of fertilizer applications, which would be a non-factor for lands classified as beaches. Seasonal south Florida EMC values are currently not available in literature, and no government agencies were able to provide such reports. Use of constant EMCs influenced the accuracy of the magnitude and spatial distribution of simulated constituent flows and storages, although without knowing the actual seasonal EMC values it is difficult to say just how much.

General Assessment of Model Calibration

Since the model was used to study alternate land use scenarios, the model's accuracy when simulating land use conditions other than those used during initial calibration was relevant. As was demonstrated by the land use validation analysis, the model seemed to perform similarly when the amount of urbanized lands was greater. In addition, the strongest performance, as gauged by the magnitude and duration of differences between observed and simulated values, in single-point model results occurred at S-179, the structure located closest to the most urbanized lands. These comparisons lend some confidence in the simulation results of scenarios conducted with

current or increased levels of urbanization, which includes the land use scenarios with WSTA networks.

Since this model simulated hydrology and water quality with a unique type of space/time complexity, the calibration method itself raised an additional consideration about model accuracy. The calibration method, which was to tune model coefficients to reduce differences between simulated and observed values at a few individual grid cells, was sometimes complicated when tunings improved results at one point but made them worse at another. In addition, single-cell approaches proceed without regard for watershed-scale effects. Usually, the patterns at single-cells were still seen at the watershed-scale as well, but occasionally there were exceptions (compare Figures 81 and 83). More commonly, coefficient settings that were of great importance to single-point results had minimal effects at the watershed-scale, as seen in the hydrologic responses from the surface water process sensitivity analysis.

Discussion of Land Use Scenario Simulations

Simulated Degrees of Urbanization

It is safe to conclude that the historic land use scenario simulation did not model pre-development hydrology accurately throughout the study area. Simulated hydrology in some regions, specifically areas of transverse glades where simulated water flows were broadly spread out, was characteristic of pre-development hydrology. But the low hydroperiod index values in the western half of the study area, historically part of the primary Everglades ecosystem, indicate cells where standing water was not present during much of the simulation year. The low index values to the east of the sandy ridge, where there were once wet prairies, marshes, and mangrove forests, also indicate many

areas where prolonged inundation was not simulated. Although some regions had broad groups of cells with intermediate index values, indicative of wetland hydrology, they were not as expansive as the original wetland systems. A likely explanation is that the DEM used to simulate the historic condition possessed residual errors along the canal system. Evidence for this can be seen in the hydroperiod index grid, where higher values indicating higher inundation frequency occurred in many cells of the modern canal network, and cells immediately adjacent had the lower index values suggestive of drier hydrology.

The three urbanization condition simulation results suggest that plant uptake was the primary pre-development TP pathway, but that chemical uptake is now dominant, and will remain so for the near future. Furthermore, 1995 and 2015 scenario results suggest that very little long-term TP removal currently is being done by plants. In effect, it follows from model results that the drainage and conversion of the historic system, where most TP cycled biologically through wetland plants, has changed the phosphorus cycle into a chemical process that occurs primarily within the canals. Although the model did not simulate specific species of phosphorus, the combination of the simulated pathway shift and differences between these biological and chemical phosphorus cycles indicate another implication for the phosphorus being discharged into the estuary. Specifically, the implication is that much less of the phosphorus now being exported to Biscayne Bay occurs as organic phosphorus, and instead is as mineral phosphorus that is easily consumed by phytoplankton.

WSTA Networks

A characteristic common to all simulated WSTA networks was that the annual loads of constituents discharged into Biscayne Bay were significantly reduced from both current and projected future levels. The WSTA networks reduced simulated annual watershed discharges of TP between 14.8% and 41.3% from the simulated 2015 loads. Similarly, reductions in WSTA network simulated annual TSS discharges into Biscayne Bay ranged between 34.2% and 63.9% from simulated 2015 values. The WSTA network that achieved the least reductions in TP discharged to the ocean was the Random #2 network. The Source Proximity WSTA network achieved the smallest reductions in simulated TSS load discharged. The WSTA network that achieved the greatest reductions in both TP and TSS discharged to Biscayne Bay was the second wetland size class distribution WSTA network.

Simulated WSTA network oceanic discharges were lower partly because constituent loads were reduced. The loads decreased due to three reasons; 1) EMC values were reduced where urban lands were converted into wetlands, 2) daily watershed runoff volumes declined as the new wetland cells increased surface water storage capacity, and 3) since more cells stored surface water, fewer landscape cells used EMC values (i.e. new loads instead of pre-existing stored constituents) in daily calculations of constituent load-out.

The correlation between hydrologic discharge and constituent discharge, which was evident in the rainfall validation, also appeared in the WSTA network results. Although the differences were small (Table 25), the general pattern was that when WSTA network discharge of water into the ocean was low, constituent discharge was also low. For example, the Random #2, Source Proximity, and 10000 Meters WSTA networks had

the highest discharges of water and both constituents. In contrast, the SCD #2 and Favorable Hydrology WSTA networks consistently had low simulated oceanic discharges.

Comparisons of specific WSTA results indicate several conclusions about the effects of landscape distributions of wetlands on watershed flows. An item of considerable significance dealt with whether or not networks arranged around theoretical design parameters did or did not perform, as gauged by reductions in constituent loads, as well as randomly arranged wetland networks. When the average discharge values of the two random WSTA networks are compared to the results from the simulations of the other wetland configurations, it seems clear that the intentional arrangement of wetlands according to design parameters generally results in better treatment of stormwater runoff. The averaged random WSTA network annual watershed discharge of TP was $1.39 \text{ E}+06$ grams, which was only exceeded by the simulated discharge from the Source Proximity WSTA. The average random WSTA simulated oceanic TSS discharge of $2.21 \text{ E}+09$ grams was exceeded by the Source Proximity, 10000 Meters, and Sink Co-Location WSTA networks, and higher than the other four WSTA networks.

The network that discharged the most water and constituents (after averaging the random networks) was the Source Proximity WSTA network. The simulated performance of this network may be of some interest for managers since it is the management approach most widely practiced today. In effect, this is the spatial wetland network that results when the management plan is simply to require that any new development have on-site storage of storm event runoff. Although some constituent load reductions were achieved, annual hydrologic discharge actually increased slightly.

Simulation of the land use scenario contained a network of wetlands located on lands where soils had high soil carbonate achieved one of the highest simulated chemical TP uptake flows of all the networks. However, simulated sedimentation flows of both constituents were relatively low. Consequently, the simulated TP and TSS loads discharged into Biscayne Bay were higher than the discharges for most of the other WSTA networks.

The simulation results suggest that distance to watershed discharge is a design variable of some influence on overall discharge. Simulated watershed hydrologic and constituent discharge into Biscayne Bay was markedly lower from the 2000 Meters WSTA network as compared to the 10000 Meters WSTA network. These results are explained because the area contributing loads downstream of the treatment sites was greater when treatment wetlands were located farther from the ocean.

Simulation results to test effects of wetland size class distribution exhibited differences that both illustrated an important characteristic of the water quality model and revealed an example of how wetland size classes influence constituent dynamics. Simulated discharges of water, TP, and TSS into Biscayne Bay were considerably lower from the simulation where the size class of wetlands in the WSTA network had higher amounts of wetland area as medium and large wetlands, (as compared to SCD #1, which had more smaller wetlands). Furthermore, both TP and TSS annual sedimentation and annual chemical TP uptake were higher when the network had more area in medium and large wetlands. By contrast, when more area was in smaller wetlands, simulated annual plant TP uptake was greater.

The Favorable Hydrology WSTA network also achieved higher treatment than most other networks. This network had a similar wetland size class and spatial distribution to that of the SCD #2 network, so the better treatment is consistent. The siting and construction of wetlands in locations where they are more likely to achieve wetland hydrology is an additional attraction of this network.

These results were similar, although not identical, to the findings in the size class modeling study by Cohen (1999). In that study, smaller wetlands were concluded to be the major systems for treating sediments, medium wetlands were primarily responsible for phosphorus treatment, and large wetlands buffered significant hydrologic pulses. The major difference between the results of the two models was that smaller wetlands decreased sedimentation in this study.

The method this water quality model used to simulate phosphorus was to calculate chemical uptake and sedimentation flows as functions of cell constituent storage, whereas plant TP uptake rate was considered to be constant, independent of the storage value. That the constituent pathway calculations were not consistent in their use of cell storage explains differences in simulated flows, especially when placed in the context of how water and constituents are converged in lower parts of watersheds. The lower total constituent storage per unit area found in upper reaches of watersheds increases as water carries TP and TSS downstream into wetlands, streams, and lakes. The convergence continues until after the water and constituents reach the sea, at which point dilution occurs. When the landscape converged constituents in the model in the medium and larger wetlands located further downstream from the smaller wetlands, the storages within each cell rose dramatically.

As a result, simulated chemical uptake and sedimentation increased, but plant uptake did not. Therefore, plant uptake was a minor component of the TP dynamics in the medium and larger wetlands of the WSTA networks. WSTA networks with more wetland area in the upper watershed, such as Random #2 and SCD #1, had relatively high simulated plant uptake compared to the other pathways. But since simulated plant uptake flows were so much lower in magnitude compared to the other pathways, the overall WSTA network performance did not benefit much.

Since the constituent storages were much lower in the historic land use scenario, simulated sedimentation and chemical uptake were also lower for the reason just mentioned. Consequently, simulated plant uptake was proportionally high on the pre-development landscape. By extension, the higher loads, and attached increases in sedimentation and chemical uptake, of the modern landscape mean that no WSTA networks will achieve the proportionally high plant uptake of pre-development days.

Conclusions

- New types of GIS functions will greatly assist in the development of new types of dynamic spatial models.
- Rainfall input data is a major variable affecting simulation model results and validity.
- Cognizance of scale affects the entire scope of watershed modeling projects, from the calibration of models to assessment of simulation results. A focus on single-point results that ignores watershed-scale behavior can sometimes be ineffective and/or counterproductive.
- Model instability was easy to generate using a dynamic spatial modeling approach in a low-relief landscape, and limited the ability to tune coefficients to some degree.
- The water management system is currently the dominant control on study area hydrology.
- The water quality model was highly dependent upon the EMC values used to generate constituent loads.
- Lower hydrologic flows resulted in lower simulated constituent loads.
- Simulation of dynamic flow directions resulted in the ability to comparatively evaluate different spatial configurations of wetlands within the watershed.
- The land use change in the study area has increased oceanic constituent loads approximately one order of magnitude, and has slightly increased hydrologic flows.
- As a result of human development of the landscape, the primary pathway for removing phosphorus from the study area has changed from long-term plant uptake to chemical uptake.

- Adding wetlands as 10% of the landscape reduced simulated discharges of TP and TSS by a minimum of 14.8% and 41.3%, respectively.
- Watershed networks where stormwater treatment wetlands were located randomly did not perform as well as did most other arrangements.
- The current policy of requiring developers to build wetlands on-site reduces constituent loads, but not as much as WSTA networks where wetlands are located throughout other parts of the watershed.
- Wetlands located nearer to the ocean will treat more constituents than will wetlands further from the coast.
- Incorporation of medium and larger wetlands into the regional wetland network will achieve greater reductions in overall discharges of constituents into Biscayne Bay. The mechanism for removal was identified as the fact that larger wetlands take greater advantage of landscape hydrologic and constituent convergence.
- Constructing wetlands in locations where the landscape is potentially favorable for development of wetland hydrology reduced loads considerably, and offers the attraction of higher probability of project success.

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