Hydrologic Treatments Affect Gaseous Carbon Loss From Organic Soils, Twitchell Island, California, October 1995–December 1997

By Robin L. Miller, Lauren Hastings, and Roger Fujii

U.S. Geological Survey Water-Resources Investigations Report 00-4042

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CONVERSION FACTORS, ABBREVIATIONS, AND ACRONYMS

CONVERSION FACTORS

Multiply	By	To obtain
centimeter (cm)	0.3937	inch
gram per square meter (g/m^2)	.000204816	pound per square foot
gram per square meter per year ($g/m^2/yr$)	.000204816	pound per square foot per year
kilometer (km)	.6214	mile
liter (L)	.2642	gallon
meter (m)	3.281	foot
square meter (m^2)	.0002471	acre

Temperature in degrees Celsius (°C) may be converted to degrees Fahrenheit (°F) as follows:

$$°F = (1.8 \times °C) + 32$$

Sea level: In this report, "sea level" refers to the National Geodetic Vertical Datum of 1929 (NGVD of 1929)—a geodetic datum derived from a general adjustment of the first-order level nets of both the United States and Canada, formerly called Sea Level Datum of 1929.

ABBREVIATIONS

h, hour

mL, milliliter

ACRONYMS

ANOVA, analysis of variance
F, permanent shallow flooding
OW, open-water treatment
PVC, polyvinyl chloride
RF, reverse flooding treatment
SC, seasonal control treatment
USGS, U.S. Geological Survey

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EXECUTIVE SUMMARY

This report discusses the methods and results of a cooperative project with the California Department of Water Resources to study how establishing wetlands on Sacramento-San Joaquin Delta Islands affects carbon cycling, so that the relative potential of different wetland systems to mitigate subsidence in the region can be evaluated. Mitigation of subsidence of organic soils in the Sacramento-San Joaquin Delta, California, is important because the decrease in land surface elevation in the area increases risks of flooding that could endanger a primary source of drinking water for California.

Land surface elevations have been decreasing since the Delta's extensive freshwater tidal marshes were drained, levees were built, and the land was converted to agriculture during the period from the 1880s to the 1920s. Currently, land surface elevations are more than 6 meters below sea level in some areas. The decrease in land surface elevation has resulted in an increased hydraulic gradient across the levees, increasing the potential for levee breaks and flooding of islands. Catastrophic flooding of the islands could endanger an important source of drinking water in California because saltwater could be drawn into freshwater areas of the Delta where water is diverted to supply more than 22 million people.

Aerobic decomposition of the drained peat soils has been established as the primary cause of permanent loss of land mass in the Delta. Therefore, reversion of critical areas to wetland habitats has been proposed as an approach to limit further subsidence of organic soils. In order to assess this approach, four water-management treatments on Twitchell Island were compared from October 1995 through December 1997 to study their relative effects on gaseous carbon losses, which are a measure of decomposition processes and an indicator of subsidence of organic soils.

The hydrologic treatments were a seasonal control (SC), reverse flooding (RF), permanent shallow flooding (F), and open-water habitat (OW). The SC treatment was subjected to the region's Mediterranean climate and current Delta-island management conditions, where water was drained from the island soils by pumps to maintain water levels about 1 meter below the land surface. During the winter rainy season, these soils were usually saturated for a period of time. The RF treatment was established in the fall of 1995. This treatment was intentionally flooded from early dry season to midsummer and then was subjected to seasonal conditions from August until March. The F treatment was subjected to continuous shallow flooding of 35 centimeters or less throughout the year since 1993. The OW treatment, an excavated area, was continuously flooded to a depth that precludes the growth of emergent vegetation. Carbon gas emissions (in the forms of carbon dioxide and methane) from the soil were measured as indicators of carbon loss from each treatment. Carbon inputs in the form of higher plants were measured

with plant biomass harvests in two of the treatments, RF and F.

Hydrologic treatment significantly affected gaseous carbon losses. Gaseous carbon emissions indicated that a period of extended flooding during the warm dry season significantly decreased total gaseous carbon losses. Permanent flooding significantly decreases carbon dioxide emissions with a concomitant, though much lower, increase in methane emissions. Gaseous carbon emissions demonstrated pronounced seasonal variability. Highest carbon losses tended to occur in the summer. Carbon dioxide losses from the RF treatment rose dramatically following its midsummer drainage. On an annual basis, total gaseous carbon emissions from the shallow flooding treatments (RF and F), decreased to approximately one-quarter of the seasonal control values, while emissions from the more deeply flooded treatment (OW), which lacked abundant plant growth, were approximately one-eighth that of the seasonal control.

When plant carbon inputs are compared to gaseous carbon losses, the permanent shallow flooding treatment (F) is the best wetland treatment for mitigation of subsidence in the Delta because measured carbon inputs outweigh measured carbon losses. Estimated annual plant carbon inputs into this treatment were approximately three times those of the measured gaseous carbon losses. Plant carbon inputs into the RF treatment approximately equal gaseous carbon losses indicating that this treatment could potentially reduce further subsidence of the organic soil. Lowest gaseous carbon emissions were found in the OW treatment, but plant inputs into this system appeared to be equally low. All wetland treatments slowed gaseous carbon losses, compared to the seasonal control, demonstrating a reduction in decomposition with flooding.

It is difficult to make accurate predictions about treatment performance because there are limitations associated with the design and methods employed in this study. However, this study provides an initial assessment of the potential for the three wetland treatments studied, as well as background information for the development and implementation of further studies examining subsidence mitigation in the Sacramento-San Joaquin Delta, California.

ABSTRACT

Subsidence of organic soils in the Sacramento-San Joaquin Delta, California, has increased the potential for levee failure and flooding in the region. Because oxidation of the peat soils is a primary cause of subsidence, reversion of affected lands to wetlands has been proposed as a mitigation tool. To test this hypothesis, three 10 x 10 meter enclosures were built on Twitchell Island in the Delta and managed as different wetland habitats. Emissions of carbon dioxide and methane were measured in situ from October 1995 through December 1997, from the systems that developed under the different water-management treatments. Treatments included a seasonal control (SC) under current island management conditions; reverse flooding (RF), where the land is intentionally flooded from early dry season until midsummer; permanent shallow flooding (F); and a more deeply flooded, open-water (OW) treatment.

Hydrologic treatments affected microbial processes, plant community and temperature dynamics which, in turn, affected carbon cycling. Water-management treatments with a period of flooding significantly decreased gaseous carbon emissions compared to the seasonal control. Permanent flooding treatments showed significantly higher methane fluxes than treatments with some period of aerobic conditions. Shallow flooding treatments created conditions that support cattail [*Typha* species (spp.)] marshes, while deep flooding precluded emergent vegetation. Carbon inputs to the permanent shallow flooding treatment tended to be greater than the measured losses. This suggests that permanent shallow flooding has the greatest potential for managing subsidence of these soils by generating organic substrate more rapidly than is lost through decomposition. Carbon input estimates of plant biomass compared to measurements of gaseous carbon losses indicate the

potential for mitigation of subsidence through hydrologic management of the organic soils in the area.

INTRODUCTION

Organic soils of the Sacramento-San Joaquin Delta in California formed from marsh plants in the widespread wetlands of this inland Delta region over the last 10,000 years (Atwater, 1980). These soils have been subsiding since the Delta's extensive freshwater tidal marshes were drained, levees built, and the land converted to agriculture during the period from the 1880s to the 1920s. Land surface elevations in some areas are more than 6 meters (m) below sea level (Deverel and others, 1998). The decrease in land mass has resulted in an increased hydraulic gradient between channel water and island ground-water levels, increasing the potential for levee breaks and flooding of islands. Catastrophic flooding of the islands could endanger an important source of drinking water in California because saltwater could be drawn into freshwater areas of the Delta, where water is diverted to supply more than 22 million people.

Previous research in the region indicates that microbial oxidation of the drained peat soils is the primary mechanism causing present-day permanent subsidence (Deverel and Rojstacser, 1996). Because anaerobic conditions of wetland systems can slow decomposition of organic substrates, hydrologic management of the carbon rich Delta soils through establishment of permanent and seasonal wetlands can be an effective approach to mitigating further subsidence.

The objective of this study is to examine the effect of different hydrologic treatments on carbon cycling by measuring the gaseous carbon losses, as well as plant carbon inputs, from various wetland systems in order to evaluate the relative potential for subsidence mitigation. To achieve these objectives, a variety of wetland habitats were established for study on the southwestern end of Twitchell Island (fig. 1). The purpose of this report is to present the results of the study from October 1995, the inception of the reverse flooding (RF) treatment where soils are shallowly flooded from early spring through midsummer to simulate brood habitat for waterfowl, through December 1997. This wetland system was compared to previously established treatments of permanent shallow flooding [F; <35 centimeters (cm)], deeper permanent flooding (OW; 1 m), and the seasonal control (SC) site, which is artificially drained throughout the year. This work was done in cooperation with the California Department of Water Resources.

METHODS AND MATERIALS

Site Description and Study Design

The study site is on the southwestern end of Twitchell Island in the heart of the Sacramento-San Joaquin Delta of California, which has a Mediterranian climate (fig. 1). Twitchell Island is bordered on the south and east by the San Joaquin River, on the west by Threemile Slough, and on the north by Sevenmile Slough. The soil is classified as a Gazwell mucky clay, which has a poorly drained mineral horizon approximately 75 cm thick overlying an organic soil. Separate 10x10-m enclosures were constructed to contain the different wetland treatments.

Four treatments for the period October 1995 through December 1997 are compared. The seasonally managed control (SC) site is subject to current (2000) Delta island management conditions, where water is drained from the island soils by a network of drainage ditches and pumps to maintain water levels about 1 m below the land surface. During the winter rainy season, these soils often are saturated. The reverse flooding (RF) treatment is immediately east of the chambers for the SC treatment (fig. 2). The RF treatment is intentionally flooded to about 30 cm from early spring to midsummer to simulate brood habitat for waterfowl, and is subject to seasonal conditions from August until March. A permanently shallow flooded (F) treatment is a few meters east of the RF

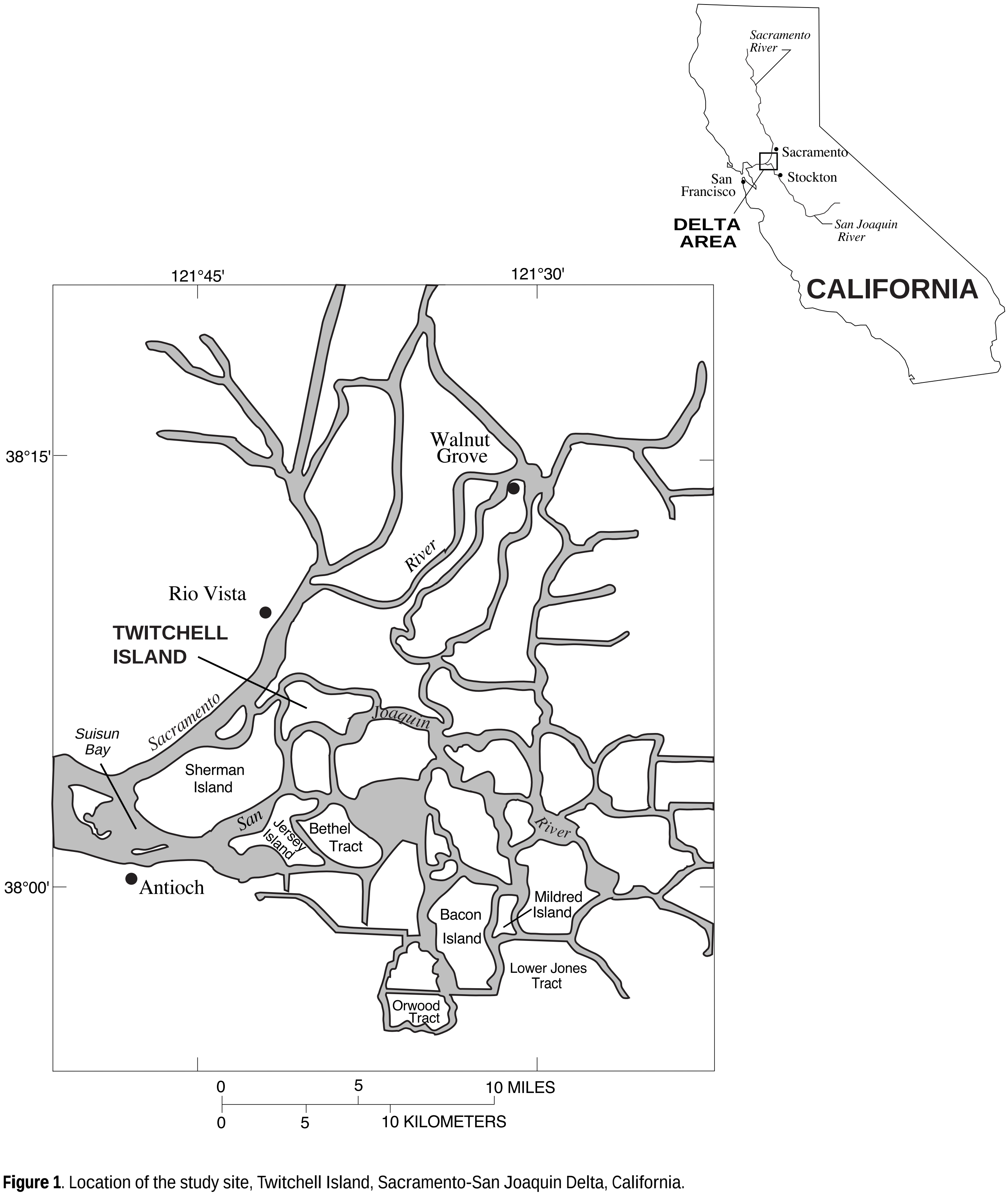

Figure 1. Location of the study site, Twitchell Island, Sacramento-San Joaquin Delta, California.

treatment. This site has been subject to continuous shallow flooding of about 35 cm since January 1993, and supports a dense *Typha* (cattails) stand. An excavated area to the east and north of these sites is continuously flooded (OW treatment) to a depth of about 1 m. This flooding precludes the growth of emergent vegetation, and is meant to simulate open-water habitat for water birds. The decision to implement the RF and the OW treatments was based on a recommendation from representatives of state and federal natural resource agencies and private organizations (California Department of Water Resources, California Department of Fish and Game, Delta Protection Commission, U.S. Geological Survey, U.S. Department of Agriculture–Natural Resources Conservation Service, California Waterfowl Association and Ducks Unlimited).

Chambers for measuring the flux of soil gas are permanently located in each of four treatment sites (fig. 2). At the OW treatment, three chambers, constructed to allow water to flow beneath them, are at the end of a floating pier. There is single thermocouple in the OW treatment placed 10 cm beneath the soil surface near the chambers. Boardwalks provide access to the chamber sites within the RF and the F treatments without disturbing the substrate. Two sets of thermocouples are placed 10 and 30 cm below the soil surface between the three permanent collars for the gas collection chambers installed in the SC, RF, and F treatments.

Measurement of Gaseous Carbon Emissions

Soil gas fluxes were determined using vented static chambers where the change in gas concentration over time is measured by collecting samples from within the sealed chamber at evenly spaced intervals over a short period (fig. 3). Chamber bases in the SC, RF, and F treatments are stainless steel, 29-cm-diameter collars. Chambers at all sites are constructed of PVC pipe with interlocking acrylic sleeves fitted with a septum-covered sampling port, a pressure equalization tube, and a thermocouple to measure the temperature inside the chamber during sampling. Chamber volume ranges from 4 to 70 liters (L), depending on water depth and chamber height. Gas samples from the chambers were collected in 5-mL glass syringes sealed with a Teflon valve. After collecting an initial air sample, the acrylic chamber tops were sealed with mylar. Headspace gas samples were collected by syringe through a septum-covered sampling port at regularly timed intervals during a 10–30 minute period. Between these monthly sampling periods, the mylar cover was removed and chambers left open to the atmosphere. In September 1997, fans were installed in chambers at the F and OW treatments to improve gas mixing in these large volume chambers. At the same time, removing PVC sleeves resulted in reduced chamber volumes at the SC and RF treatments by decreasing the height. This served to minimize gradients in gaseous carbon compounds within the chambers.

Chambers were sampled one-at-a-time over 2-day periods until August 1997, when sequential simultaneous sampling of chambers was initiated to minimize the effect of environmental variables on the measurement of gaseous emissions and to improve replication. Barometric pressure, soil and air temperatures, water depth, and chamber headspace volume were recorded for each sample. Unsaturated soil moisture was determined with a neutron soil moisture probe (model 4300, Troxler Electronic Laboratories, Inc.). Gas samples were analyzed for carbon dioxide with an MTI gas chromatograph (model M200) fitted with a HayeSep A column at 55ºC with a thermal conductivity detector, and for methane with a Hnu (model 301) gas chromatograph with a Poropak T column at 60ºC and flame ionization detector at 150ºC. Peak areas were determined by electronic integration and converted to moles by comparison to standard

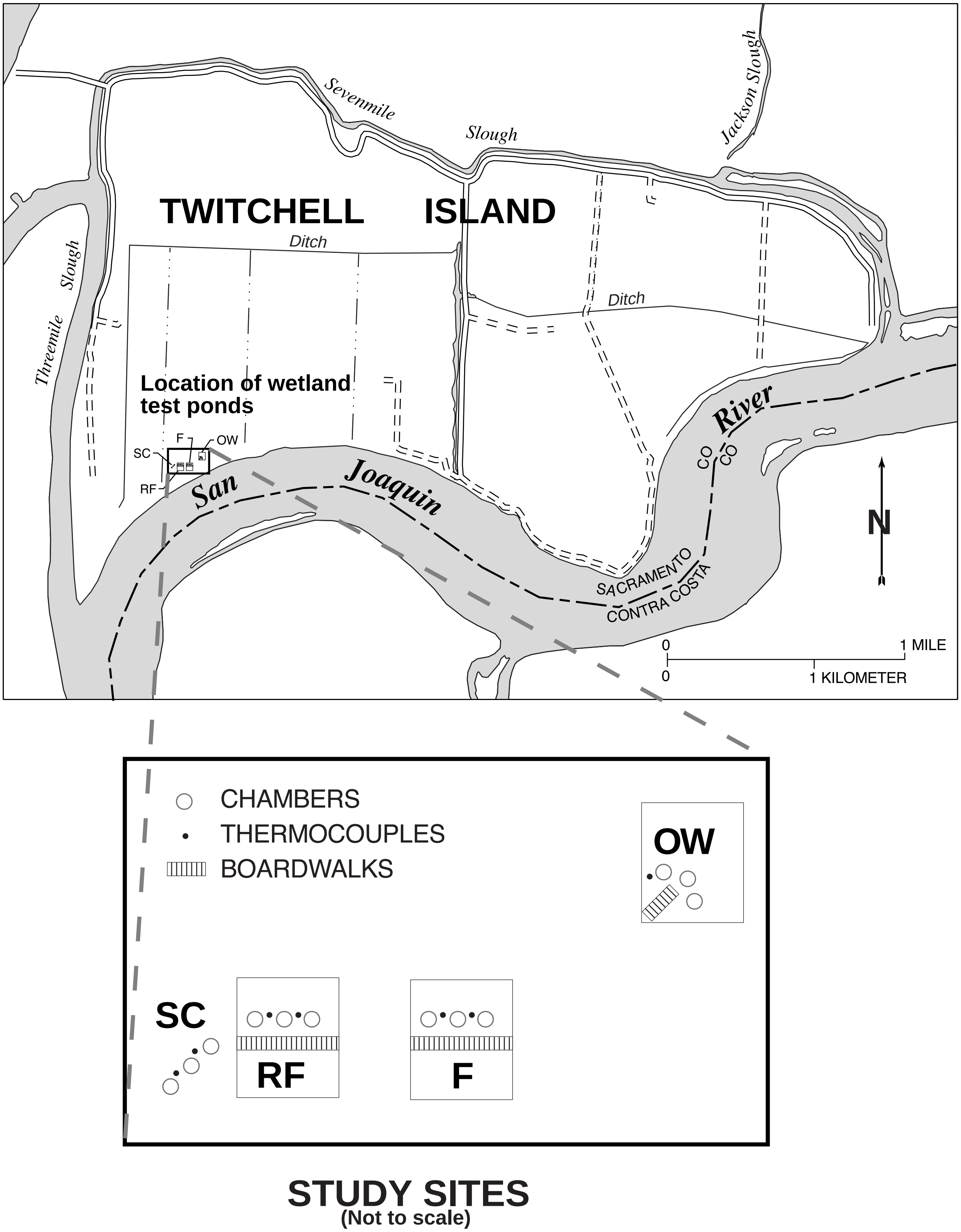

Figure 2. Enlargement of study site, Twitchell Island, Sacramento-San Joaquin Delta, California. F, permanent shallow flooding treatment; OW, open-water treatment; RF, reverse flooding treatment; SC, seasonal control.

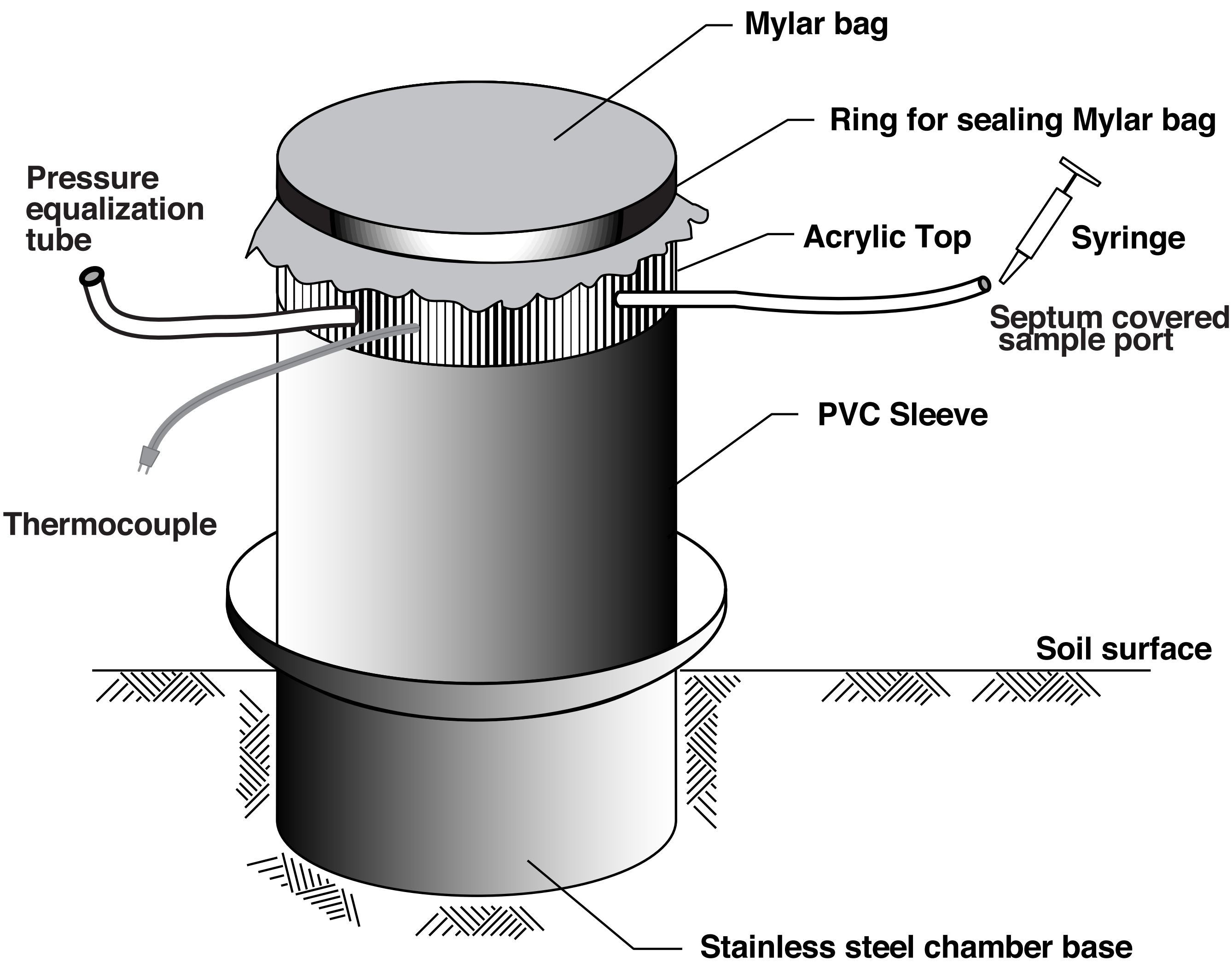

Figure 3. Vented static chamber with base used for collecting gaseous carbon emissions. (Not to scale)

gases (Intermountain Specialty Gas, Nampa, Idaho) of known concentrations. Gaseous carbon fluxes were determined by linear fits of chamber gas concentration against time. Total carbon emissions were established by adding carbon dioxide and methane flux rates from each sample.

Measurement of Plant Carbon Inputs

Wetland plants, particularly the clonal dominant *Typha*), have colonized the reverse flooding (RF) treatment and the continuous shallow flooding (F) treatment to differing extents. In order to assess additions of new carbon to these treatments, plant biomass was estimated with annual harvests. In October 1995, carbon inputs were estimated from a single harvest of all above-ground material in three 0.3716 square meter (m^2) plots in the RF and F treatments. In 1996, carbon inputs were measured by harvesting all above-ground plant material from six 0.3716 m^2 plots in the RF treatment in November and from the F treatment in November and December. Beginning in April 1997, living above-ground biomass was harvested monthly from five 0.1 m^2 plots on different transects established in the enclosures containing the RF and F treatments. From June until November 1997, root biomass was estimated monthly by collecting and washing viable roots from soil cores

(15 cm deep and 10 cm in diameter) from each harvested plot of the above-ground samplings. All collected plant material was dried for 48 hours at 65ºC and weighed to determine mean standing live biomass. Harvested plant biomass from 1995 was analyzed for carbon content with a Perkin Elmer CHNS/O analyzer (series II model 2400). In 1997, field-dried *Typha* was collected, ground, and analyzed for carbon and nitrogen because plant material often enters the system as senesced litter rather than living tissue.

Also in 1997, the turnover (cumulative plant growth that takes litter loss into account) of above-ground biomass was assessed in order to estimate annual carbon inputs from the dominant plant colonizer. This was accomplished by tagging several *Typha* spp. plants in the RF and F treatments and measuring leaf height and mortality through the growing season. Total leaf length over the season was divided by the mean standing live leaf height to estimate the turnover of each tagged plant. Mean standing live plant biomass, as determined in 1997, was multiplied by the plant turnover number to estimate the amount of new plant material introduced annually to the system (Davis, 1991).

Statistical Analysis

Statistical analyses were done using SAS (Statistical Analysis Systems Institute, 1990). Gaseous carbon flux data were rank transformed and analyzed using mixed model analysis of variance (ANOVA) procedures. Mixed model ANOVA procedures were used to assess plant biomass data from 1997 to account for the random effects of repeated measurements. T-tests were performed on plant biomass data from 1995 and 1996 to compare mean plant production values from the single harvest events. Turnover data from 1997 were assessed with ANOVA. The effects of temperature, moisture, and season on gaseous carbon emissions were tested through multiple regression analysis. The statistical significance of all tests was determined with an alpha level of 0.05.

RESULTS

Gaseous Carbon Emissions

Analysis of all gaseous carbon emissions showed a significant treatment effect (P=0.0001; fig. 4). Because total carbon emission estimates are affected differently by negative carbon fluxes and include a greater number of sample dates than either carbon dioxide or methane emission estimates alone, the total carbon flux is not simply the addition of carbon dioxide and methane emissions in each treatment, but is the mean of a different data set. The seasonal control (SC) treatment showed significantly higher total carbon emissions than all the other treatments (P=0.0001). On an annual basis, gaseous carbon emissions from the treatments with shallow flooding (RF and F) decreased to approximately one-quarter of the control treatment values, while carbon gas emissions from the more deeply flooded (OW) treatment were approximately one-eighth that of the control treatment. Total gaseous carbon losses from the reverse flooding (RF) treatment and the permanently shallow-flooded (F) treatment were not significantly different, but these treatments had significantly higher total carbon losses than the open-water (OW) treatment. Gaseous carbon emissions from the SC and RF treatments primarily were in the form of carbon dioxide. Methane comprised a large portion of the gaseous carbon losses in the permanently flooded treatments (F and OW). Methane emissions from these two treatments did not differ significantly from one another, but are significantly higher than those from the SC and RF treatments (P=0.0001). Total annual gaseous carbon emissions from these treatments, based on the measured monthly means from the autumn of 1995 to the end of 1997, are 2961±473 grams per square meters per year ($g/m^2/yr$) from the SC treatment; 788±158 $g/m^2/yr$ from the RF treatment; 692±105 $g/m^2/yr$ from the F treatment; and 342±123 $g/m^2/yr$ from the OW treatment.

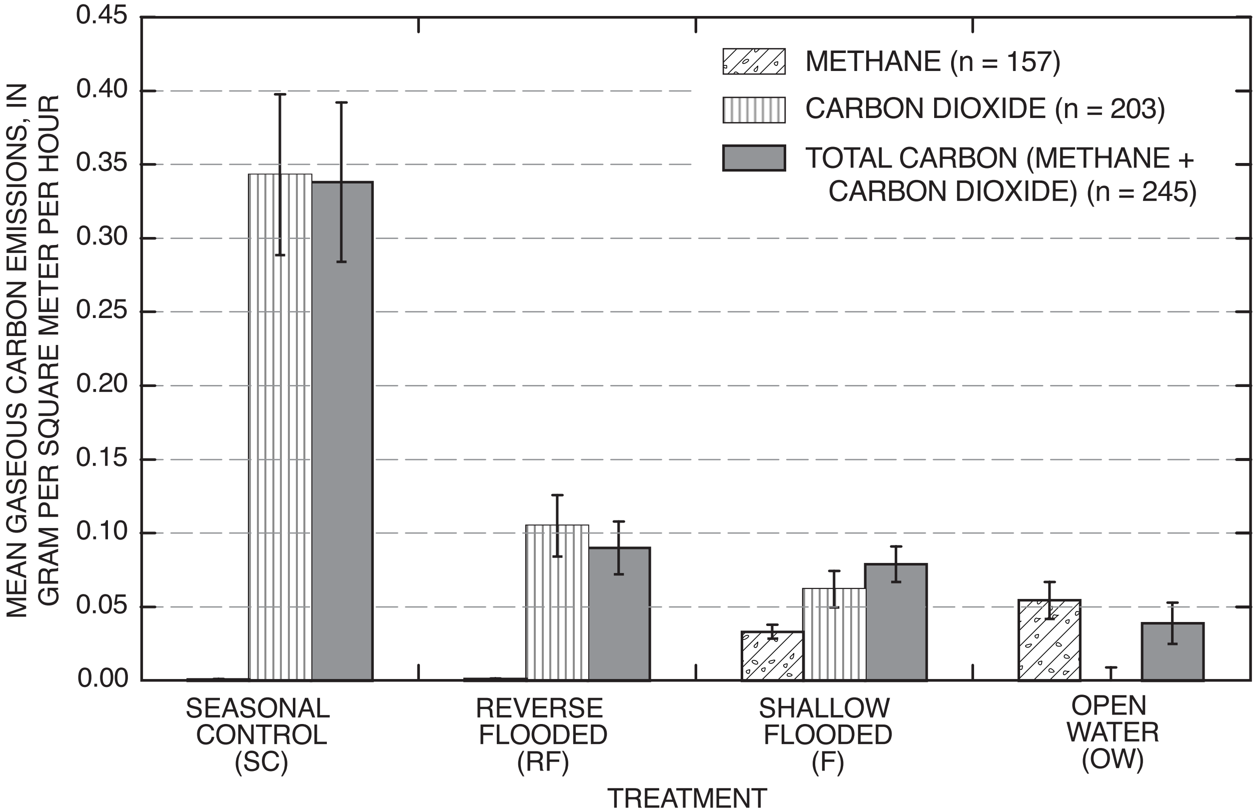

Figure 4. Mean gaseous carbon emissions from the four hydrologic treatments of organic soils on Twitchell Island, California, October 1995 to December 1997, as methane, carbon dioxide, and total gaseous carbon (carbon dioxide and methane). Vertical bars represent standard error.

Gaseous-carbon fluxes varied significantly both seasonally and spatially (P=0.0001 and P= 0.03, respectively). Monthly mean gaseous carbon flux measurements for total carbon, carbon dioxide and methane are depicted in figures 5–7, respectively. Highest carbon fluxes occurred in the summer. After being drained in midsummer, carbon dioxide fluxes from the RF treatment increased for the remainder of the warmer weather (fig. 6). High methane emissions from the OW treatment in peak summer months can indicate large losses through ebullition, rather than diffusion, through the water column (fig. 7).

Multiple regression of rates of gaseous carbon loss for all treatments versus soil and air temperatures, soil moisture, and water depth showed significant correlation for methane and carbon dioxide. Methane emissions were correlated positively to soil moisture and air temperature with a weaker relation to water-depth measurements (table 1). The best regression model for carbon dioxide fluxes included soil moisture and water depth, which were negatively correlated, and air temperature, which showed positive correlation (table 2). Soil temperature was not included in either final regression model because it showed no significant contribution to the models.

Hydrologic treatments can affect local soil and air temperatures. Temperature is important to the processes of microbial decomposition and physical gas laws. Statistical analyses of soil (P= 0.0001) and air temperatures (P= 0.0001) showed significant differences among the four treatments (fig. 8). Highest soil temperatures were recorded in the seasonal control (SC) and the open-water (OW) treatments. Highest air temperatures were recorded in the OW treatment. Seasonal trends in soil and air temperatures from the

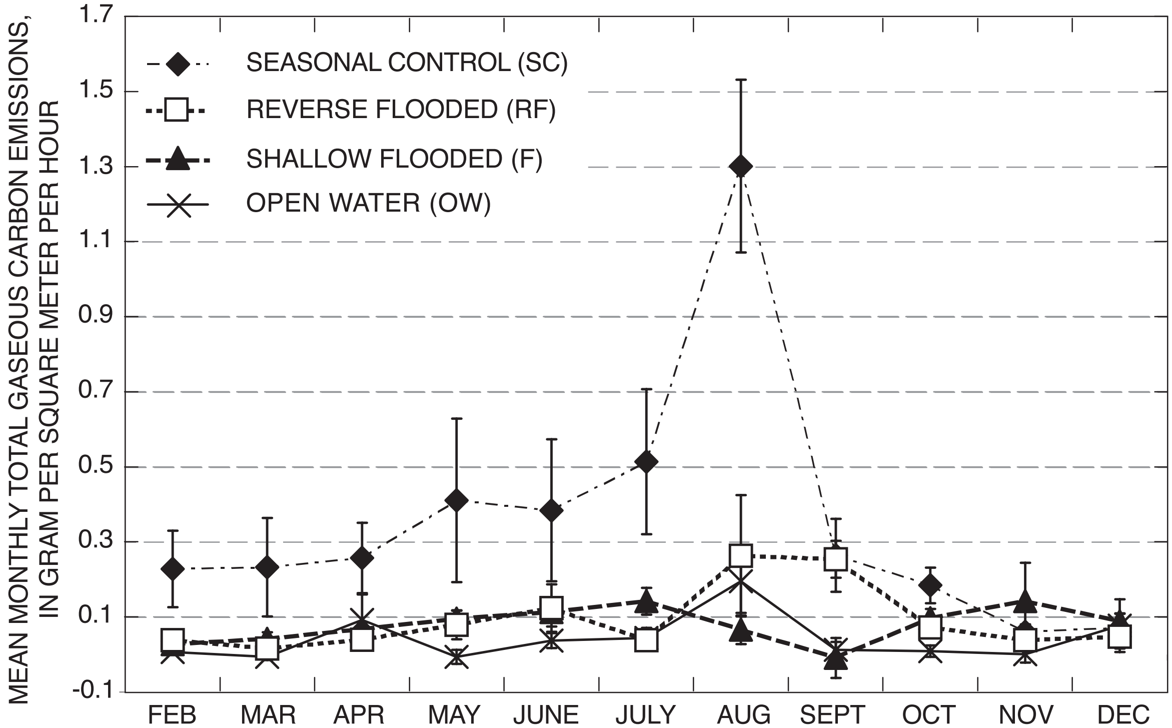

Figure 5. Mean monthly total gaseous carbon emissions from the four hydrologic treatments of organic soils on Twitchell Island, California, October 1995 to December 1997. Vertical bars represent standard error.

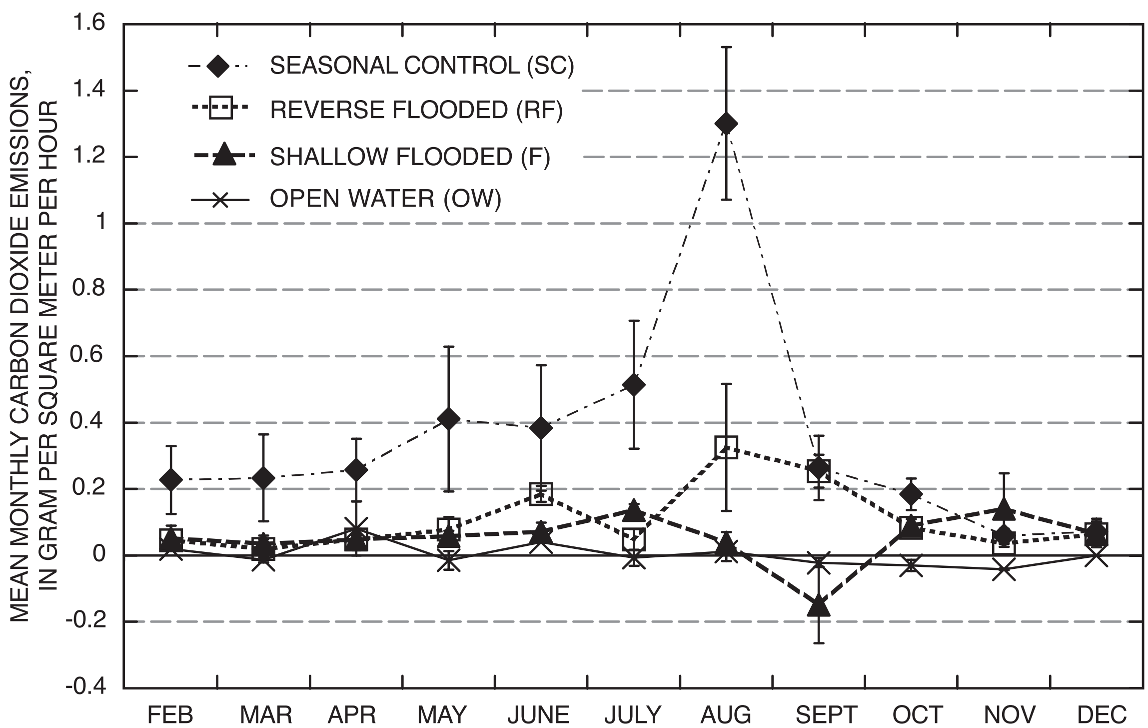

Figure 6. Mean monthly carbon dioxide emissions from the four hydrologic treatments of organic soils on Twitchell Island, California, October 1995 to December 1997. Vertical bars represent standard error.

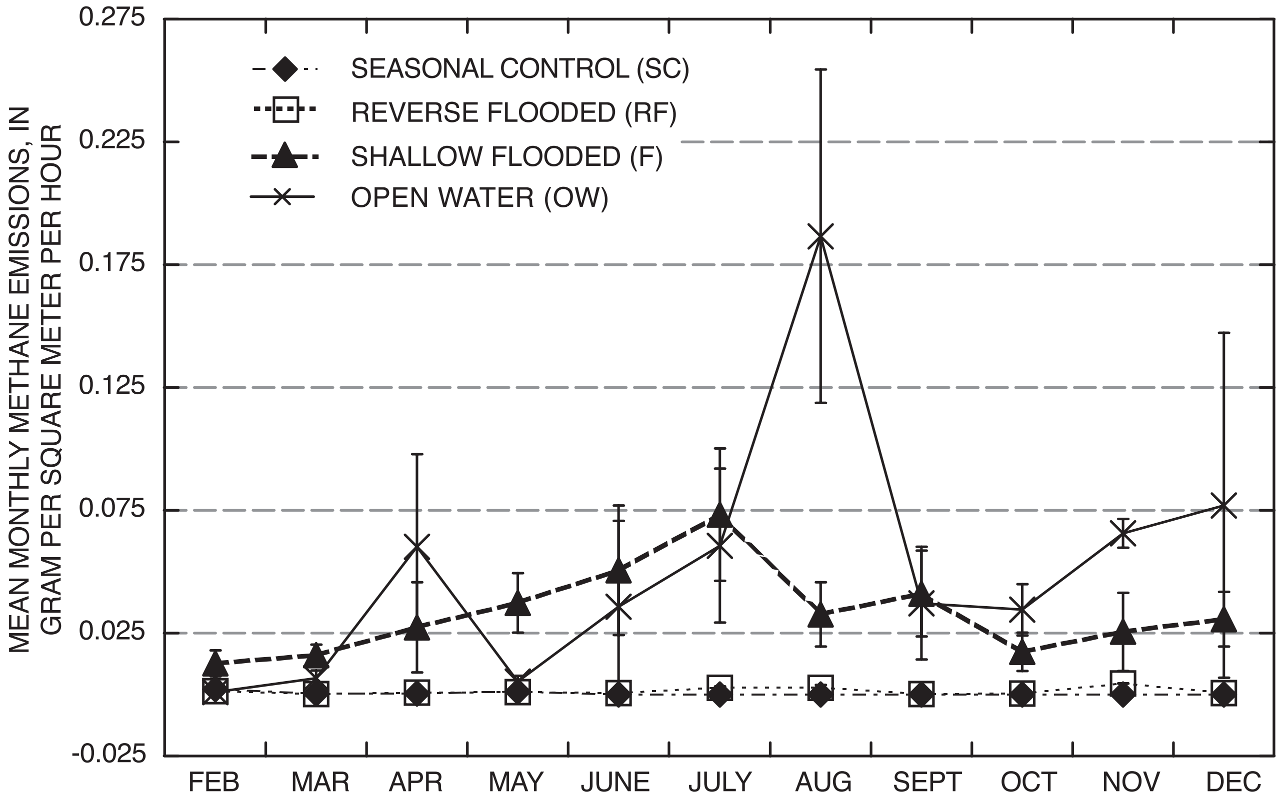

Figure 7. Mean monthly methane emissions from the four hydrologic treatments of organic soils on Twitchell Island, California, October 1995 to December 1997. Vertical bars represent standard error.

Table 1. Multiple regression model statistics for methane flux against soil and air temperature, soil moisture and water depth, including model significance and r^2, and significant explanatory variables with r^2

Parameter	Coefficient	P value	r^2 (n=114)
Overall regression model		0.0001	0.35
Soil moisture	0.0208	.0001	.28
Air temperature	.0591	.0077	.05
Water depth	.0152	.0879	.02
Intercept	-19.9957	.0001	

Table 2. Multiple regression model statistics for carbon dioxide flux against soil and air temperature, soil moisture and water depth, including model significance and r^2, and significant explanatory variables with r^2

Parameter	Coefficient	P value	r^2 (n=131)
Overall regression model		0.0001	0.36
Soil moisture	-0.0045	.0004	.32
Air temperature	.0345	.0185	.02
Water depth	-.1662	.043	.02
Intercept	-.3798	.0001	

treatments are shown in figures 9 and 10. Soil temperatures tended to be highest in the SC treatment from May to November, but were highest during cool weather months in the OW treatment. Soil temperatures were similar in the RF and F treatments, when both sites were shallowly flooded in the dry season, but increased in the RF treatment immediately following its drainage in midsummer. Air temperatures from within chambers during flux measurements were usually higher in the OW treatment than in all other treatments, particularly during the summer months.

Plant Carbon Inputs

Plant biomass was harvested only from the RF and F treatments, and autumn harvests from 1995 and 1996 did not show significant

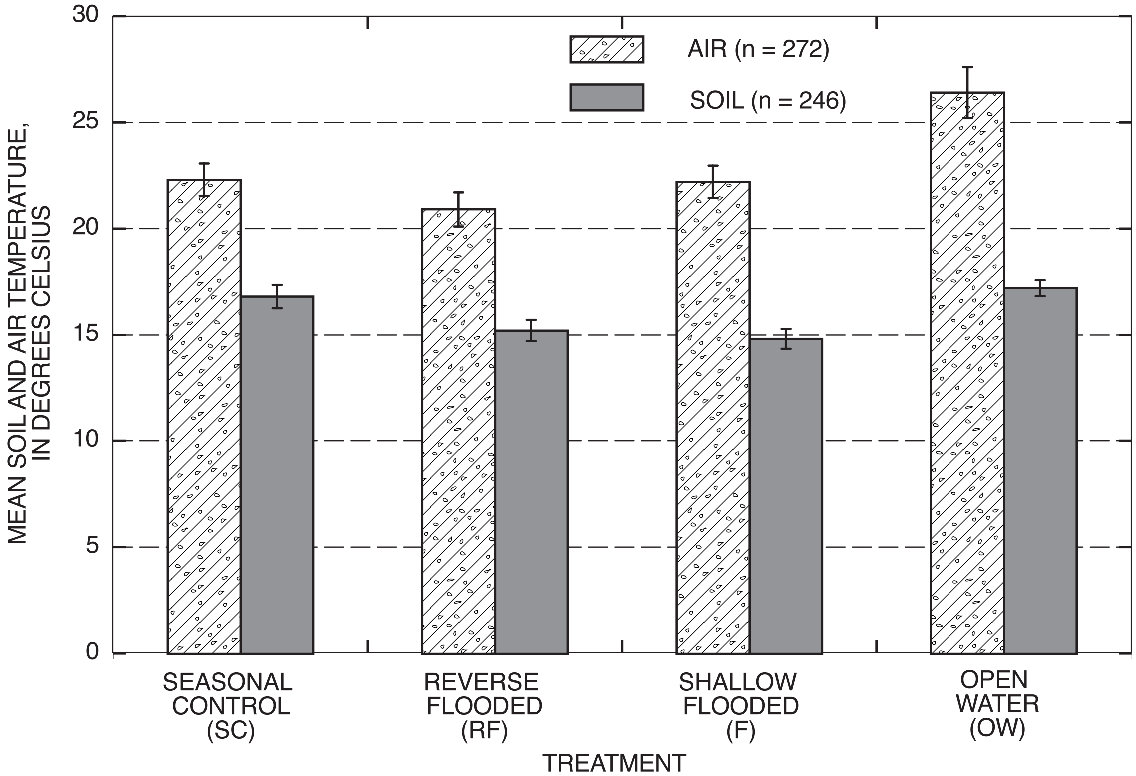

Figure 8. Mean soil and air temperatures of the four hydrologic treatments of organic soils on Twitchell Island, California, October 1995 to December 1997. Vertical bars represent standard error.

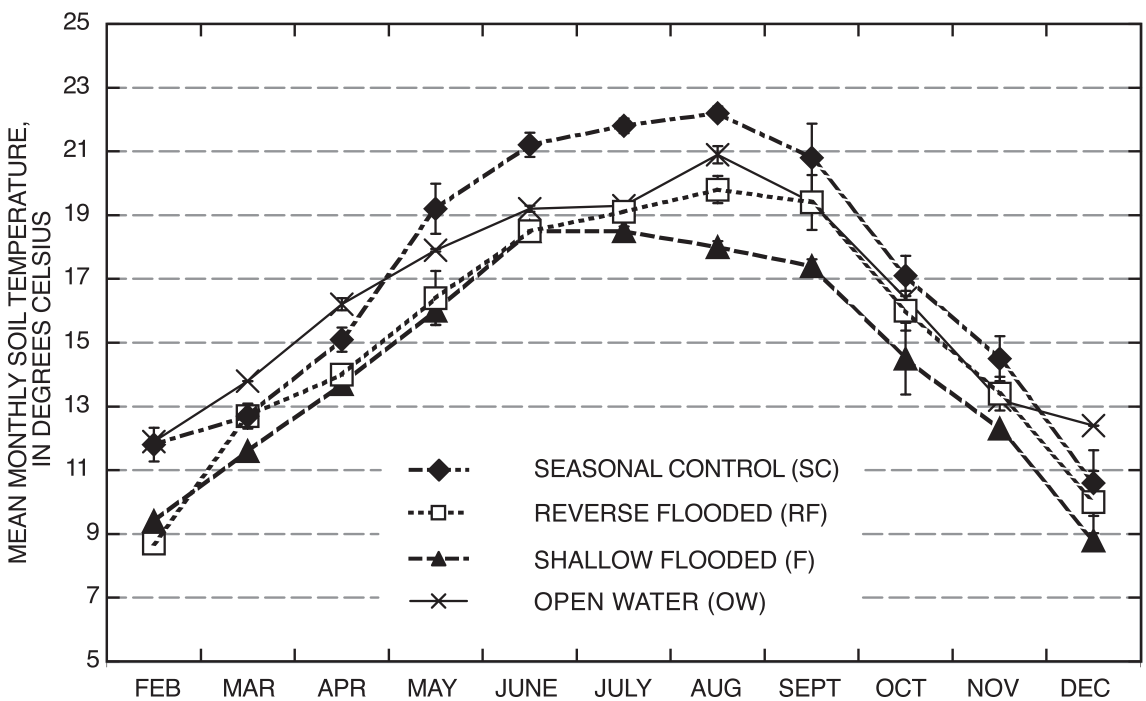

Figure 9. Mean monthly soil temperature, at 10-centimeter depth, of the four hydrologic treatments of organic soils on Twitchell Island, California, October 1995 to December 1997. Vertical bars represent standard error.

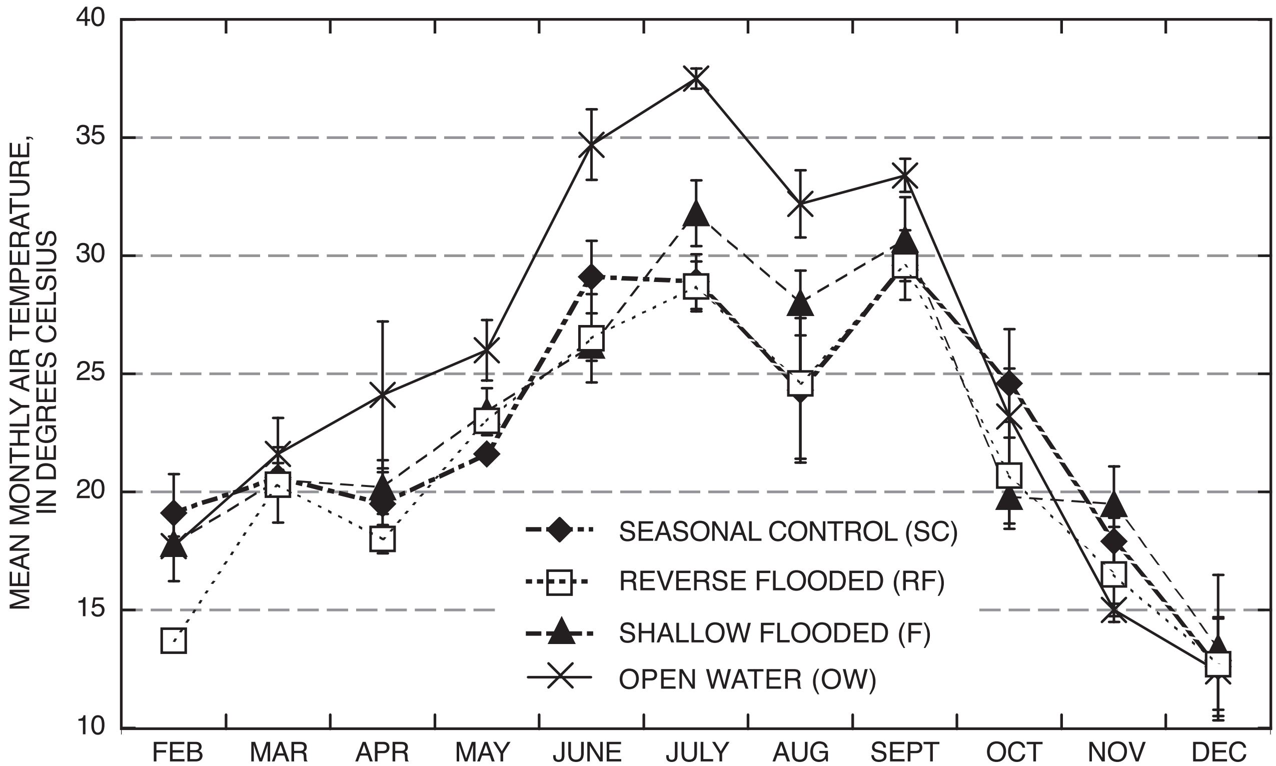

Figure 10. Mean monthly air temperature from chambers in the four hydrologic treatments of organic soils on Twitchell Island, California, October 1995 to December 1997. Vertical bars represent standard error.

differences. In 1995, mean above-ground biomass dry weights were 3,588 g/m^2±587 in the RF and 5,577±392 g/m^2 in the F treatments. Carbon content of the harvested material averaged 38±2 percent. Thus, plant carbon inputs in 1995 were approximately 1,360 g/m^2 in the RF and 2,120 g/m^2 in the F treatments. In 1996, 5,155±425 g/m^2 of dried plant material was harvested above ground from the RF and 5,365±651 g/m^2 from the F treatments. Using the 38 percent carbon content from 1995 harvests, plant carbon inputs were estimated to be 1,960 g/m^2 in the RF and 2,040 g/m^2 in the F treatments during 1996.

In 1997, biomass harvest data (table 3) showed that the F treatment had significantly larger standing live biomass than the RF treatment, with means of 2,873.7±533.4 g/m^2 and 950.6±181.2 g/m^2, respectively (P=0.0063). Also, the mean *Typha* density in the F treatment was significantly greater than in the RF treatment (P=0.0001). This reflects complete colonization of *Typha* in the F treatment, but only partial colonization in the RF treatment. The RF treatment, on the other hand, had significantly higher mean grass (*Paspalum distichum*) dry weight than the F treatment (P=0.02). As a result of these differences in species coverage of the ponds, the F treatment had significantly larger mean above- and below-ground standing live biomass dry weights than the RF treatment. Differences in root:shoot biomass allocation were not significantly different between these treatments. Standing live biomass also showed different seasonal trends in the RF treatment, compared to the F treatment (fig. 11). Above-ground biomass peaked in the RF treatment for only a short time in midsummer, prior to drainage. In the F treatment, above-ground plant biomass maintained peak production through most of the growing season. Grass made up a large portion of the biomass in the RF treatment during the season (fig. 12). The rapid decline in the mean standing live biomass in the RF treatment after the

Table 3. Mean biomass measurement from monthly harvests in 1997 for the reverse flooding (RF) and permanent shallow flooding (F) treatments and statistical comparisons. Above-ground samples were collected from April to November (n=5). Roots samples were collected from June to November (n=5)

[g/m^2, grams per square meter; m^2, square meter; n, number of samples]

Biomass measurements	Units	F Mean	F Standard error	RF Mean	RF Standard error	P value
Total dry weight	g/m^2	2873.7	533.4	950.6	181.2	0.0063
Above-ground dry weight	g/m^2	1651.9	296.8	619.1	119.3	.0099
Below-ground dry weight	g/m^2	1260.3	255.1	353	86.5	.0161
Grass dry weight	g/m^2	12.9	5.9	170.3	47.1	.0215
Typha density	m^2	43.6	5.2	14.1	2.5	.0001
Root:shoot		1.8	.6	5.3	4.1	.5754

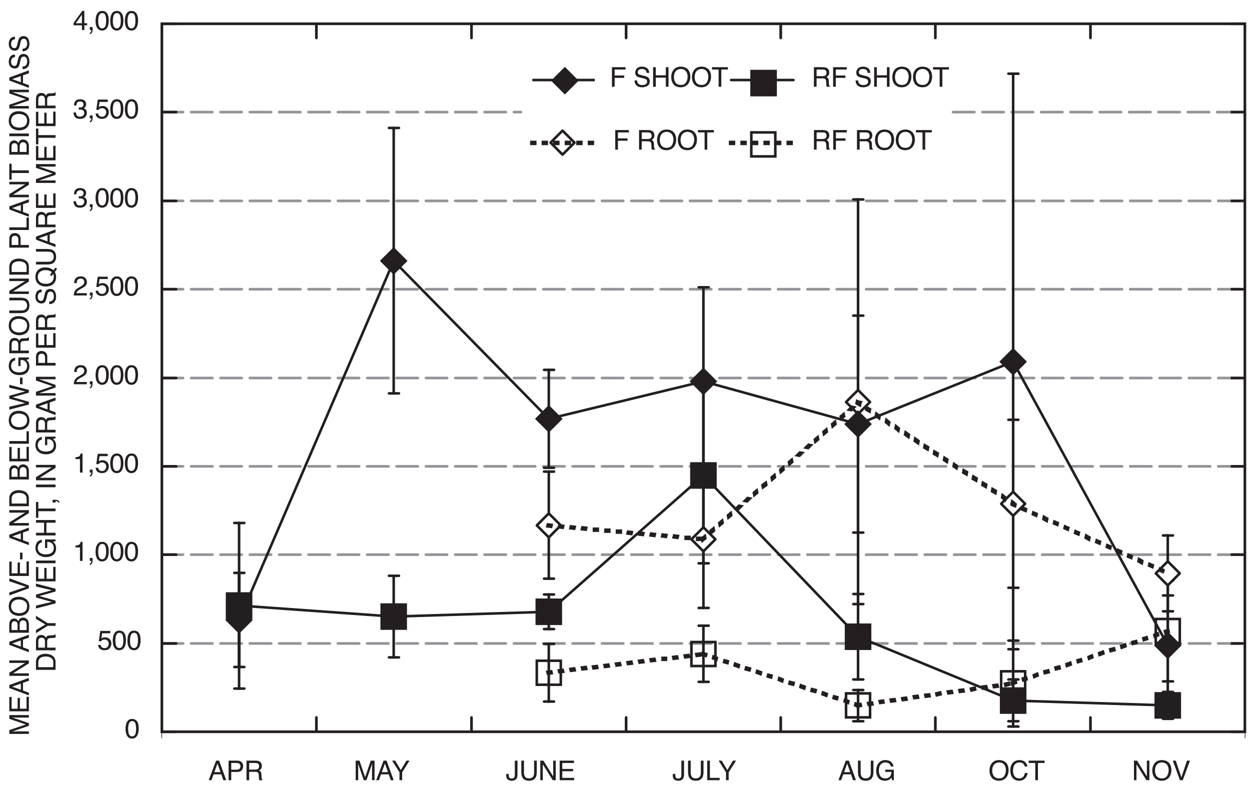

Figure 11. Mean above- and below-ground plant biomass for the reverse flooding (RF) and the permanent shallow flooding (F) treatments of organic soils on Twitchell Island, California, during the 1997 growing season. Root biomass was not measured in April and May. Vertical bars represent standard error.

site was drained in midsummer was, in large part, a result of the senescence of grasses.

In order to estimate total biomass inputs into the ponds, turnover of the primary colonizer, *Typha*, was measured in the RF and F treatments. In the RF treatment, only one species of *Typha*, *T. latifolia*, was present. In the F treatment, however, broad leaf (*T. latifolia*) and southern (*T. domingensis*) cattail were present. Annual turnover rates were measured for *T. latifolia* in both treatments and for both species in the F treatment. In order to establish turnover, total leaf height during the season and mean standing live leaf length were calculated. Mean standing live leaf length, as well as

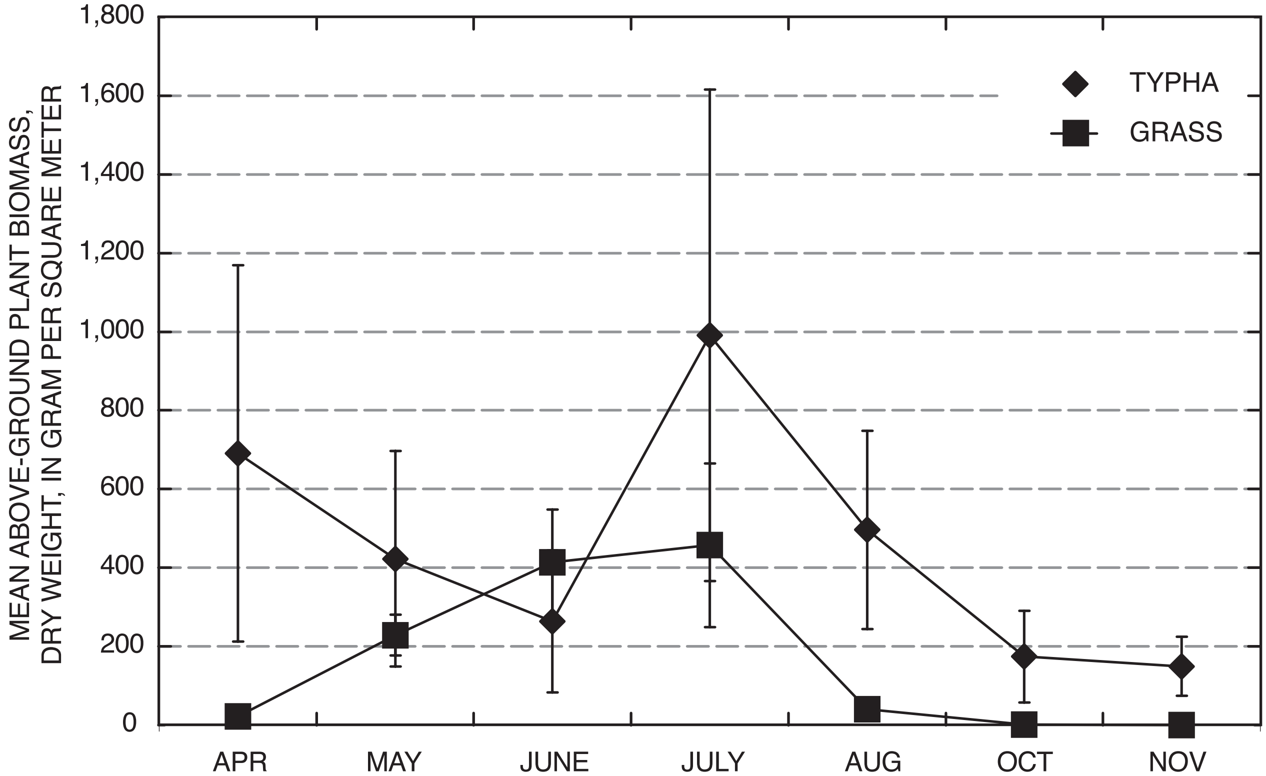

Figure 12. Mean above-ground plant biomass, depicted by plant type, from the reverse flooding treatment of organic soils on Twitchell Island, California, during the 1997 growing season. Vertical bars represent standard error.

total leaf length over the growing season, tended to be greater for *T. latifolia* than *T. domingensis* (fig. 13). Annual turnover of *T. latifolia* was 2.03±0.11 for plants in the RF and 1.85±0.07 in the F treatments, and did not differ significantly between treatments. The turnover of *T. domingensis* in the F treatment was 1.77±0.06, so differences in turnover between the *Typha* species in the F treatment were not significant. However, turnover of *T. latifolia* in the RF treatment was significantly greater than that of *T. domingensis* in the F treatment ($P<0.05$). *Typha* species differences in the F treatment also were apparent from the 1997 biomass harvests. *T. domingensis* tended to grow in more dense stands than *T. latifolia* (61.6±9.6 plants/m^2 and 31.6±3.8 plants/m^2, respectively; $P=0.0146$), but biomass production did not show significant differences between the two species on an area basis. Turnover for the dominant grass species in the RF treatment was not measured.

When the *Typha* turnover estimates were used to estimate annual plant biomass inputs into these treatments for 1997, plant growth in the F treatment contributed approximately 5,170 g/m^2, and the RF treatment received 1,930 g/m^2 from plant biomass contributions. This assumes that above- and below-ground plant tissues have similar rates of turnover. Carbon content of field dried *Typha* was 45±0.1 percent. Thus, annual carbon inputs from plant material were approximately 900 g/m^2 in the RF and 2,300 g/m^2 in the F treatments, based on harvest data from 1997. According to these estimates, the RF treatment had approximately equivalent carbon inputs and losses, while the F treatment had a net carbon gain.

DISCUSSION

The data in this study show effects of treatments on gaseous carbon losses from the systems.

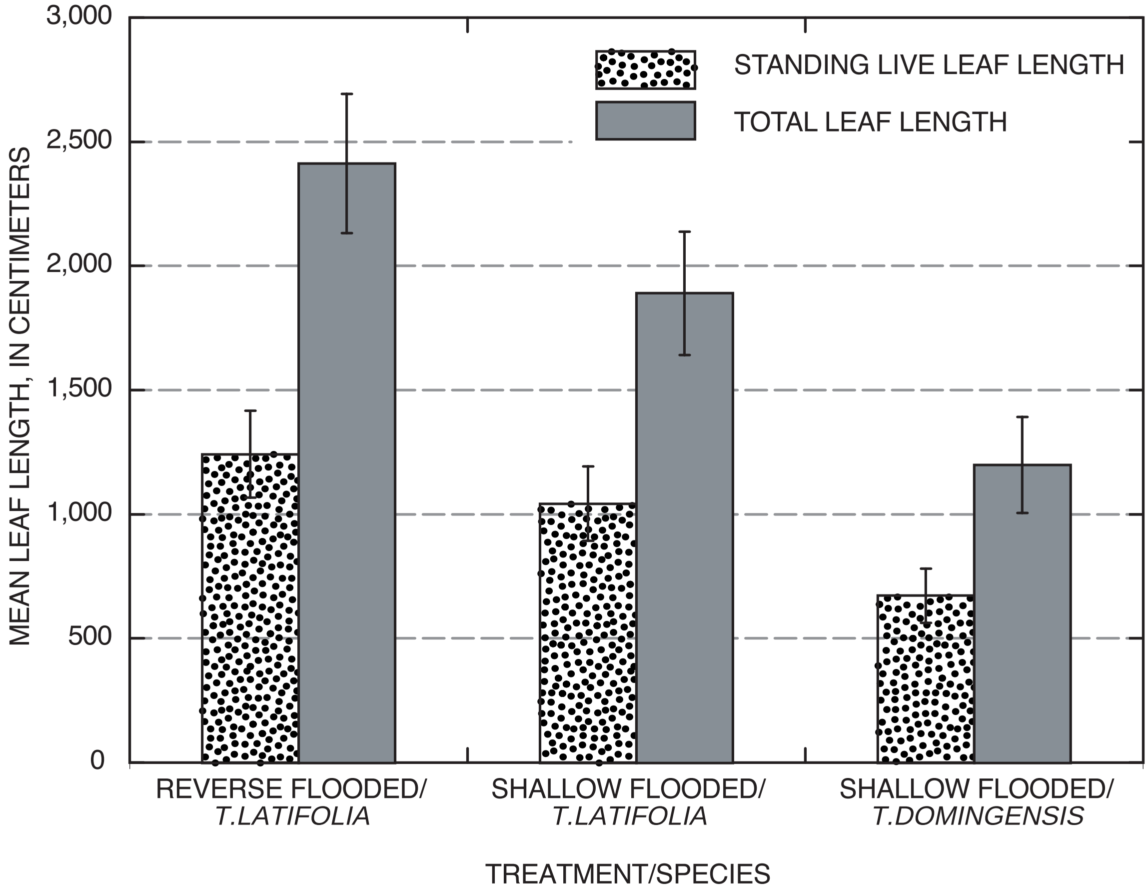

Figure 13. Mean total leaf length and mean standing live leaf length of *Typha spp.* in the reverse flooding (RF) and permanent shallow flooding (F) treatments of organic soils on Twitchell Island, California, 1997. Vertical bars represent standard error.

All treatments with a period of flooding as a component of their hydrologic management showed significantly lower total gaseous carbon emissions compared to the control site (SC) (fig. 4). Total gaseous carbon emissions from the treatments with shallow flooding (RF and F) decreased to approximately one-quarter of the control values, while emissions from the more deeply flooded treatment (OW), which lacked abundant plant growth, were approximately one-eighth that of the control site.

Furthermore, the period of flooding affected the predominant gaseous carbon compound that evolved from the treatments. Production of methane in this study was most strongly correlated to soil moisture, while carbon dioxide emissions demonstrated a negative correlation. Treatments with permanent flooding (F and OW) had significantly greater carbon losses in the form of methane than the SC and RF treatments (fig. 4). Nevertheless, a large fraction of methane produced in wetlands can be converted to carbon dioxide by methanotrophs (Boeckx and Van Cleemput, 1997), and the presence of roots in saturated soils allows methane oxidation in the rhizosphere (Watson and others, 1997). Thus, the potential methane oxidation mechanisms and plant respiration can help explain the higher carbon dioxide fluxes from the F and RF treatments, compared to the OW treatment (fig. 4). There also is evidence that land-use history can affect the methanogen population in peat soils (Chapman and others, 1996). This suggests that the alternating wet and dry periods in the RF treatment can restrict development of the anaerobic microbial community necessary to produce methane.

A strong seasonal trend in gaseous carbon emissions was evident in this study (figs. 5–7). Soil and air temperature are important components of the seasonal effect. However, the seasonal trend does not mimic temperature alone. Plant biomass production has been shown to correlate well to methane emissions from wetlands (Whiting and others, 1991). Thus, the seasonal effect on gaseous carbon emissions is likely the result of plant mediated additions of fresh carbon substrate into the systems as well as additional carbon dioxide from plant respiration. In this study, the F treatment tended to support higher inputs of plant biomass than the RF treatment, and to have higher methane fluxes; but the two treatments have similar gaseous carbon losses overall. This could be due to slower decomposition of newly added organic matter in the F treatment; or, decomposers in the RF treatment could be utilizing an additional, older source of carbon (the soil organic matter).

Carbon dioxide evolution from the RF treatment increases following drainage in midsummer (fig. 6). As the soils become oxidized, there is a surge in microbial activity and the end products of anaerobic decomposition can be further degraded by aerobic organisms. The increase in soil temperature in this treatment following summer drainage can be the result of water no longer acting as a temperature buffer. This increase in soil temperature can further enhance microbial activity.

Hydrologic treatment effects on the temperature parameters in this study are most likely the result of the relatively higher capacity of water to act as a temperature buffer, as well as the vegetative community structure supported by the treatment. Hydrology is a primary factor in determining plant community composition which, in turn, can affect the local environment. For example, the height of a *Typha* stand shades the water surface, thereby influencing local temperatures. Highest air temperatures occurred in the open-water (OW) treatment (fig. 7), where there was no shading and air movement was limited by the height of plant growth surrounding the site. Soil temperatures were lowest in treatments shaded by *Typha* and buffered by water. Highest soil temperatures were found in the treatment with the driest soil, the seasonal control (SC), except in winter, when the warmest temperatures of the four treatments were in the well-buffered OW treatment (fig. 9).

Annual biomass input estimates from the F treatment appeared to be more consistent during this study than inputs into the RF treatment, which also tended to be lower. The F treatment was established years before the reverse flooding (RF) treatment and was completely colonized by *Typha* by the beginning of this study period. In the RF treatment, however, plant species coverage has been changing through the course of this study. *Typha* has been spreading through the RF treatment since the reverse flooding was established, covering a little more than half the treatment by December 1997. This difference in plant species coverage of the treatments can help explain annual differences in plant biomass inputs between the treatments. Also, annual biomass in 1997 might be underestimated in the RF treatment because turnover was measured only for *Typha* and the large proportion of grass in this treatment could have a higher rate of turnover. As a result of the use of different sampling methods in each year of this study, plant biomass data cannot be compared statistically between the different years.

The difference in carbon content of plant material in 1995 and 1997 is partly a result of analyzing different plants at different life stages. The material harvested in 1995 (38±2 percent carbon) consisted of live, dead, and partially decomposed tissue from a variety of plant types. The material collected in 1997 (45±0.1 percent carbon) consisted solely of field-dried *Typha* collected at the end of the growing season. This allows for natural translocation of nutrients to the roots with senescence, and probably is more representative of the state in which most *Typha* litter enters these systems. However, it is not representative of all the plant species found in these treatments, particularly the RF treatment, where grass co-dominates.

An accumulation of plant litter clearly is evident in the F treatment. A pronounced litter layer is not found in the RF treatment. However, it is difficult to compare the treatments on this basis because the RF treatment is much younger than the F treatment, and so has had less time to develop. Also, the difference in species composition in the treatments could affect differences in litter accumulation between the treatments. Litter quality can be an important factor in decomposition (Updegraff and others, 1995; Szumigalski and Bayley, 1996), and the grass in the RF treatment, which makes up a large part of its biomass production, is likely subject to more rapid decomposition than *Typha*. Plant litter also can affect pH which, in turn, can affect decomposition rates (Valentine and others, 1994), as can the nitrogen and phosphorus fertility of the system (Amador and Jones, 1995; Paludan and Blicher-Mathiesen, 1996).

Comparison of measured carbon inputs and losses in this study show that hydrologic management of organic soils in the Sacramento-San Joaquin Delta can effectively slow gaseous carbon losses. The OW treatment had the lowest gaseous carbon losses, but also had negligible contribution from root respiration and the lowest carbon inputs. Gaseous carbon losses from the RF and F treatments were similar to one another, but plant carbon inputs into the F treatment during this study period (October 1995-December 1995) tended to be greater than in the RF treatment. Estimated carbon inputs and losses to the RF treatment were about equal, which indicates that this water-management system might be an effective strategy to mitigate further subsidence of the substrate. Prolonging the period of flooding to later in the summer could further reduce carbon losses from the RF treatment. Estimated plant carbon inputs to the F treatment were approximately three times measured gaseous carbon losses on an annual basis. This suggests that permanent shallow flooding shows the greatest potential for managing subsidence of these peat soils by generating organic substrate more rapidly than is lost through decomposition.

STUDY LIMITATIONS

Potential sources of error in this study include limitations of the experimental design and sampling procedure, as well as the inherent variability of environmental measurements. Temperature and moisture affect processes involved in soil gas emissions (Crill and others, 1988; Dunfield and others, 1993; and Boeckx and Van Cleemput, 1997), so rates of gaseous carbon flux show diurnal as well as seasonal responses (Bridgham and Richardson, 1992; and Benstead and Lloyd, 1996). Heterogeneity of environmental and soil conditions, which can affect microclimate conditions, soil-gas diffusion, carbon availability and microbial activity, causes spatial and temporal variability in soil gas flux measurements (Moore and others, 1990). In this study, environmental variation, which is not accounted for by the experimental design, results from constraints to treatment replication in the field. Furthermore, discrete sample collection on different days and at different times during the day among the treatments exacerbates variability in the carbon flux measurements within and between treatments as a result of diurnal changes in temperature. Diurnal flux rates often have maximums 3 or 4 times greater than their minimum, but maximums 50 times greater than the daily minimum also have been reported (Holzapfel-Pschorn and Seiler, 1986; Chanton and others, 1989; and Silvola and others, 1996). Also, gaseous flux data collection is subject to limitations of the method itself. For example, chamber placement can alter conditions by disturbing the substrate, impeding natural air movement over the diffusion interface, decreasing the diffusion gradient, and affecting soil and air temperatures within the chamber (Healy and others, 1996). During this field study, gradients of gaseous carbon compounds were detected by sampling at different heights in the taller chambers, which necessitated the installation of fans during sampling or reduction of chamber height.

Due to the low solubility of methane in water, methane emissions often occur as ebullition, or bubbles, arising when the partial pressure of methane in sediments exceeds the pressure exerted by overlying water. Ebullition can be triggered by changes in hydrostatic pressure over the substrate, such as decreasing water depth (Windsor and others, 1992; Silvola and others, 1996). As higher temperatures decrease the solubility of gases in water, ebullition can increase in warm weather, especially in deeper water where plant growth is inhibited. In some systems it is estimated that 50–90 percent of methane transport to the atmosphere may occur through ebullition (Chanton and others, 1989). Because these episodic emissions of methane can be difficult to quantify accurately in short sampling periods, carbon loss from the OW treatment may be much greater or less than reported, especially during the summer months.

Vegetation is a primary pathway for methane from wetland soils to reach the atmosphere (Cicerone and Shetter, 1981; Sebacher and others, 1985). Methane transport through plants occurs either by diffusion or pressurized convection of gases. Wetland plants therefore can decrease diffusion gradients between flooded soils and the atmosphere, with concomitant decreases in the amount of methane lost through ebullition. It has been estimated that 50 percent of the methane evolved from a dense cattail pond was mediated through plant transport (Sebacher and others, 1985). In this study, plant mediated transport of methane was not adequately addressed, so methane emissions from the RF and F treatments may be greater than reported here. Transport of methane through plants can decrease the partial pressure gradient of methane in sediments, decreasing ebullition in vegetated systems during the growing season. Thus, in vegetated sediments potential error due to ebullition is ameliorated.

From the data collected during this study, it is apparent that hydrologic management of organic soils in the Sacramento-San Joaquin Delta can significantly slow gaseous carbon losses from the soil. Nevertheless, it is difficult to estimate accurately the balances of carbon inputs and losses in these treatments. First, carbon inputs to the treatments were measured only for the reverse flooding (RF) treatment and the permanent shallow flooding (F) treatment, and did not include carbon in the form of root exudates, which can contain more than 10 percent of a plant’s total fixed carbon (Marschner, 1995). Furthermore, small plants like algae and duckweed (*Lemna*) were not included in plant biomass production estimates. Also, losses of soluble carbon from the systems were not ascertained and flux measurements may have underestimated gaseous carbon losses. Further study can address some of these limitations.

SUMMARY

Mitigation of subsidence in the Sacramento-San Joaquin Delta, California, is important because the loss of organic soils increases risks of flooding and endangers a primary source of drinking water. Oxidation of the drained peat soils has been established as the primary cause of present-day permanent subsidence in the Delta (Deverel and Rojstaczer, 1996). Four water-management treatments on Twitchell Island were compared from October 1995 through December 1997 to assess their relative effects on gaseous carbon losses, which are associated with subsidence.

The hydrologic treatments were seasonal control (SC), reverse flooding (RF), permanent shallow flooding (F), and open-water (OW). The seasonal control (SC) site is subject to the region’s Mediterranean climate and current Delta island management conditions, where water is drained from the island soils by pumps to maintain water levels about 1 meter below the land surface. During the winter rainy season, these soils are usually saturated for at least part of the time. The reverse flooding (RF) hydrologic treatment was established in the fall of 1995. This treatment is intentionally flooded from early dry season to mid-summer, and is subject to seasonal conditions from August until March. The F treatment has been subject to continuous shallow flooding of 35 centimeters or less throughout the year since 1993. The open-water (OW) treatment, an excavated area to the northeast of the other sites, is continuously flooded to a depth that precludes the growth of emergent vegetation. Carbon gas emissions from the soil were measured with vented static chambers. Carbon inputs in the form of higher plants were measured with destructive harvest methods in two of the treatments, RF and F.

Hydrologic treatment significantly affected gaseous carbon losses. Gaseous carbon flux data indicate that a period of extended flooding during the warm dry season significantly decreases total gaseous carbon losses, while permanent flooding significantly increases methane losses. Carbon emissions demonstrated pronounced seasonal variability. Highest carbon fluxes tended to occur in the summer. Carbon dioxide fluxes from the RF treatment rose dramatically following its midsummer drainage.

When plant carbon inputs are compared to gaseous carbon losses, the permanent shallow flooding (F) treatment is the best hydrologic management treatment for mitigation of subsidence in the Delta because measured inputs outweigh measured losses. Plant carbon inputs into the reverse flooding (RF) treatment approximately equal gaseous carbon losses, indicating that this treatment could potentially mitigate further subsidence of the organic soil substrate. Lowest gaseous carbon emissions were found in the open-water (OW) treatment but plant inputs into this system seemed to be equally low. All hydrologic management treatments slowed gaseous carbon losses compared to the seasonal control. It is difficult to make accurate predictions about treatment performance because there are limitations associated with the design and methods employed in this study. However, this study provides an initial assessment of the potential for the three water-management treatments studied and background information for the development and implementation of further studies examining subsidence mitigation in the Sacramento-San Joaquin Delta, California.

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