Fire and organic soil consumption in Florida wetlands

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Abstract

Fire in organic soils can sometimes generate large quantities of smoke, and as a result fire in wetlands is frequently suppressed. As part of a larger study of the effect of fire in wetlands, we measured changes in the depth of organic soil in six wetland and four upland communities in central Florida. About half our study locations were subject to fire (some were prescribed; others were wildfires) between 1996 and 2002. The presence of fire itself did not predict changes in soil depth, since fires vary considerably between plant communities and occur under differing conditions. However, analyses of changes in soil depth on burned sites show that the conditions under which the fires occur, as well the community in which it occurs, do predict the changes. These changes were best predicted using daily KBDI as calculated within a few km of the burn site; at larger distances, predictive power was lost. Using measurements of depth-to-water-table from a system of piezometers provided an even finer spatial scale of relevant measurements, but on a coarser (monthly) temporal scale, and these were also unable to predict soil loss. Using localized information – which might even include qualitative assessments of wetland soil hydration – can aid fire and land managers to allow fire into wetlands.

Non-technical summary: Loss of organic soil in wetlands fires is an important source of smoke and can damage some wetlands. We show that soil consumption varies considerably among wetland types, and is predictable by using localized estimates of soil hydration. Using such information, managers can permit fire into wetlands where indicated.

Introduction

Most wetlands (other than tidal wetlands) are hydrated seasonally, and are subject to fire during the dry season. In regions with marked fire seasons, wetlands may burn quite frequently – even annually. As a result, fire in wetlands is a widespread phenomenon, albeit a little-studied one.

Fire return intervals in freshwater wetlands in the presettlement southeastern United States varied considerably but were frequently quite high. Frost (1995) reviewed evidence on fire frequency from a large number of study sites throughout the southeast, and found that many freshwater wetland communities dominated by grasses and forbs had fire return intervals of 1-6 years. Return intervals of 7-25 years were typical of communities dominated by wax myrtle *(Myrica cerifera),* red maple *(Acer rubrum),* and sawgrass *(Cladium jamaicense).* Sites dominated by many other species – including tupelo *(Nyssa biflora)*, baldcypress *(Taxodium distichum)*, and Atlantic white cedar (*Chamaecyparis thyoides)* had fire return intervals ranging from 25 to over 300 years.

Fires in the southeastern US occur frequently, due to the strongly seasonal rainfall, high productivity, and the high frequency of lightning strikes. In the southeastern coastal plain, wetlands and uplands are interdigitated in complex patterns, so that many fires inevitably affect both wetland and upland areas. Prescribed fires often include wetland areas, and many other prescriptions must take into account the proximity of wetland habitats. Some soils from southeastern wetlands can ignite and sustain fire at relatively high moisture contents (Frandsen 1997).

Seasonal flooding or saturation of wetland soils typically leads to an accumulation of a partially decomposed organic layer over the mineral soil substrate, commonly referred to as "peat" or "muck" deposits. The degree of organic layer accumulation in wetlands is highly variable, but in some wetland situations this surface organic layer, diagnostic of histosols, can be several meters thick. When dry, the surface organic layer can ignite easily and is readily combustible. Organic soil fires are common in the southeastern US; between 1981 and 2004, for example, an average of 59.3 (± 7.8 standard error; range= 10-180) fires per year were reported in Florida for which organic soil was the primary fuel (Florida Division of Forestry, unpublished data).

Fire in organic soils is important to those concerned with fire, land management, and conservation, for at least three general reasons. First, organic soil fires sometimes generate significant quantities of smoke, with its attendant risks to public safety and health. As a result, fire prescriptions often take great pains to minimize the risk of fire in wetlands, and fire managers sometimes take pains to prevent wildfires from entering wetlands. Second, consumption of organic soil may be a contributing factor in changes in some plant communities. For example, fires in bayheads (which typically have deep organic soil layers) are thought to not only kill many trees, but also to consume much soil, often resulting in replacement of the bayhead community (Wade et al. 1980). Third, organic soil consumption is likely to affect hydrology. Often this will be by speeding runoff and percolation through the soil, but depending on the elevations involved, loss of the moisture-holding organic soils can also lead to increased periods of inundation.

We speculate that fires consuming significant quantities of organic soil may be increasingly likely in the Southeast, for two reasons. First, widespread fire suppression to protect spreading human settlements means that wetland fires (when they occur) are likely to be more severe. Second, wetland ditching for development and groundwater pumping for drinking water and irrigation have lowered water tables in many areas. As a result, there are many areas in which organic soils are no longer hydrated, increasing the chance of wildfire, and especially of smoldering fire (Wade et al. 1985, Bacchus 1995). There is a need to understand the consequences of fire in southeastern wetlands, including negative impacts of organic soil fires and the effects of fire on wetland populations, communities, and ecosystems.

Nevertheless, land managers, fire agencies, and ecologists generally lack information on the effects of fire in wetlands (Brennan 1995). Such data are often difficult to obtain. Land and fire managers first need to act to control fires and minimize risk to lives and buildings, before obtaining data. As a result, both application-oriented managers as well as academic researchers have little knowledge of the consequences and correlates of wetland fires.

An extensive monitoring network at The Nature Conservancy's Disney Wilderness Preserve (DWP) near Kissimmee, FL, presented to us an unusual opportunity to study wetlands fires. The monitoring network was initially established to assess restoration progress of the wetlands as mitigation for impacts at Walt Disney World and Greater Orlando Aviation Authority. It consists of a series of transects in a large number of wetlands of different types, sizes, and histories. Along each transect there is a system of piezometers (shallow groundwater wells) to document water table conditions. Organic soil depth measurements were repeatedly taken along transects. Prescribed and wild fires burned approximately 50% of the installed sampling network during 1996-2002. Using pre-fire organic soil depth measurements taken mainly in 1996, we were able to compare organic soil depths before and after these fires as well as in the absence of fires at the locations that did not burn. We also examined the effect of fire in many wetland and upland communities and under differing hydrological and weather conditions. Because the dataset involves both piezometer readings and records of Keetch-Byram Drought

Index (KBDI) on several spatial scales, we are also able to examine the relative utility of these as predictors of soil consumption during fire events.

The Keetch-Bryam Drought Index (Keetch and Byram 1968) was developed to quantify the effect of drought on the drying of the top 8 inches of soil, including the organic layer. KBDI is defined on an arbitrary scale, such that fully saturated soils are given a value of 0, and completely dried soils are given a value of 800. Intermediate values represent relative deficits of water – for example, a KBDI value of 300 means that 3 inches of precipitation are needed to bring the 8 inches of soil to a completely saturated state. As a rough guide, KBDI values below 200 typically mean that little or no soil is burned, values from 200-300 mean little soil consumption and particularly patchy fires, values from 300-500 mean that fires are likely to consume most of the litter and upper layers of organic soil, and values over 500 mean that fires are likely to consume most of the organic soil (Melton 1989).

We focus on several important questions relating to the generality and predictability of surface organic soil loss (which will often be a predictor of smoke generation):

1. Can fire in general be assumed to lead to substantial soil loss in wetlands? If so, it might suggest that policies aimed at excluding fire from wetlands might often be advisable. On the other hand, if soil loss varies with the type of wetland and/or the conditions under which the fire occurs, it would point to the need for more nuanced policies. We predicted a priori that statistical models using the simple presence/absence of fire would be poor predictors of soil loss; we also predicted that models classifying fires as prescribed vs. wildfires would also have poor predictive power. The

alternative hypothesis was that there would be a substantial improvement in the predictive power of models if they included information about the conditions under which fires occurred.

2. If the loss of soil varies with wetland type or fire conditions, which of these lead to large, and which to small, soil loss? We expected to find a pattern of increased soil loss with increasing drought, as measured by KBDI or depth-to-water-table. However, we also expected that the soil loss would vary considerably among plant communities, even for similar measurements of KBDI or hydrology, as some communities (like bayheads) usually occur on deep organic soils and others (like cypress swamps) usually occur on sandy soils.
3. If the loss of soil varies with wetland type or fire conditions, which of several predictors (KBDI measured daily on several increasingly coarse spatial scales, or depth-to-water-table measured monthly on a fine spatial scale) is the most useful? It seems clear that there are potential tradeoffs here.

Methods

Data collection

104 transects and their associated piezometers were established in 24 wetland complexes (hydrologic unit, HU), using the following criteria. We established transects in each major wetland within each HU, and additional transects as necessary to document restoration in representative examples of minor wetlands. Each transect crossed the transition from wetland to upland habitat and was approximately perpendicular to the

edge of the wetland, where compatible with other criteria. Each transect passed through as many major habitat types as possible and, where feasible, was oriented so that it could be extended into additional lower or higher elevation habitat types. Transects were spaced around the wetland to measure conditions in different portions of the site, particularly in areas where wetland restoration was anticipated. Since the ability to detect plant community shifts along the topographic gradient was a major purpose of these transects, we avoided sites where continuity of the gradient had been significantly affected by human activities. This was primarily a problem associated with improved pastures and logging activities. Once the general area for a transect was determined based on the above criteria, the starting point was randomly selected and the transect laid out along a compass line to avoid bias in where it passed through each habitat. Once a transect had been marked, the major plant communities present along it were identified on the basis of dominant species composition and structural characteristics, and piezometer sites were located centrally in representative examples of each major community type. The upper and lower piezometers along each transect were located at least 20m into continuous upland or wetland habitat, respectively.

Most transects were established in 1993 and 1994; in these, sample plots were systematically located in each plant community along each transect. We added a small number of transects and extended several of the established transects in subsequent years, and for these we used a stratified random design to locate plots within communities.

We sampled organic soil depths using a 1.5m soil probe and extension rods at points every 10 meters along all 104 transects. Measurements were accurate to 0.05m; this level of precision may limit our ability to discern fine-scale pattern in the results. Additional 1.5 meter sections were added to the rod when deeper organic soils were encountered. We inserted the rod until it stopped moving smoothly, indicating the transition from the organic soil to the firmer sand substrate. We took these measurements adjacent to the cleared paths to avoid sampling disturbed substrates.

Soil measurements were taken in two phases. Phase 1 baseline measurements were taken in May-August 1996 and in June 2000; we measured organic soil depth and correlated variables at 2800 locations on 104 transects. A subset of the original locations were remeasured (Phase 2) in May-November 2002. Phase 2 measurements were conducted on 19 burned, 15 unburned, and 12 burned/unburned combination transects for a total of 783 samples (402 burned, 381 unburned) locations.

There was a minor change in sampling method between the two measurement periods. During Phase 1, up to three measurements were taken at each location; if the first two were identical, that value was recorded. Otherwise the mean of the three values was recorded. During Phase 2, we recorded the mean of three measurements, rounded to the nearest 0.05 meters. As consistent methods were used during each phase, this change introduced no bias; moreover the observed variation between measurements at a location was always quite small.

We recorded the major plant community at each location. Wetland communities in this study include bay, cypress, floodplain, marsh, wet prairie, and flatwoods. Upland communities include dry prairie, hardwood, pasture, and scrubby. The shrub and slash pine communities occur on sites that are historically wetlands but have experienced fire suppression or have been drained.

Depth to the water table was measured with 448 piezometers and 44 continuous water level recorders. The actual number sampled at any given time is a subset of these totals. Most piezometers/recorders are located along transects, though some exist independent of the transect network. The piezometers on each transect are located in the middle of each community, providing a view of the ground water level along that transect. Water levels from the piezometers were measured monthly. Because of the small number of continuous recorders, we did not analyze those data for the present paper.

Weather data were collected by electronic sensors at 5 weather stations on the preserve. The Main weather station measures, among other things, precipitation and air temperature. Four additional stations (Far North, North, Central and South) measure precipitation. Precipitation totals were collected in 15-minute increments. Air temperature was sampled every minute, and hourly minimum, maximum and average air temperatures were recorded. We calculate daily and monthly values for each of these parameters at each location.

For each location, we have records on whether burns occurred, their dates, and whether they were wildfires or prescribed. These records were obtained from TNC's database of fires on the property. As part of its fire management program, TNC documents the extent of burns (both prescribed and wildfire) using a global positioning system and integrating this information into the Preserve's Geographic Information System database. The GIS fire layer was used to estimate burn extent along transects, and vegetation sampling data helped corroborate this information on transects that were sampled within a year of the fire event.

To examine the relationship between recent weather and soil loss to fire, and to examine the scale on which weather indicators may predict soil loss, we used KBDI measurements at several spatial scales. KBDI is a function of mean annual rainfall, and the record of daily precipitation and maximum temperature for a site. In the simplest case, one begins with saturated soil and estimates the remaining soil moisture content through an accounting procedure that estimates the loss due to evapotranspiration by using the daily records of precipitation and temperature. Clearly the utility of KBDI may then depend on how close a site is to the recording weather station, especially if rainstorms are particularly patchy. For the Disney Wilderness Preserve, weather data used for KBDI calculations are collected by a main weather station located at the north end of DWP, and four stations located at the far north, north, central, and south areas of DWP (see Figure x). For all stations the Main weather station maximum air temperature was used to calculate KBDI. Also, in the event of rain gauge or recorder failure, rainfall data from the nearest functional station was substituted to maintain a continuous KBDI record for each station. We used KBDI calculations from (a) the weather station closest to the soil measurement location; (b) the main weather station on the property; and (c) the county-wide KBDI provided by the Florida Division of Forestry. As DWP is located in two counties and many fires crossed the county lines, we took care to apply the correct county-wide data for each location.

Statistical methods

We analyzed the relationship between change in soil depth and several independent variables, including presence/absence of fire in the study interval, and indicators of hydrologic and weather conditions during fires.

We used mixed-model analysis of variance and other mixed generalized linear models (GLIM's) to analyze all data. Some data violate assumptions of GLIM's, as the residuals are not normally distributed. However, there are no practical alternatives: there are no known nonparametric models with the appropriate structure, and resampling techniques for data of this sort are prohibitive. The violation of normality simply means that p-values need to be interpreted cautiously; the models are still best-fits. For all analyses, we used Type III sums of squares, which can handle unbalanced data sets. We treated individual wetlands and transects nested within them as random effects, with all other terms as fixed effects. As individual points along a given transect are correlated, we analyzed models using the mean value of points within a community along a given transect, rather than using each point. This results in conservative tests. We use the corrected Akaike Information Criterion (AICc) to compare models for their utility (Burnham and Anderson 1998). Briefly, the uncorrected AIC involves a balance between the likelihood of the model (given the data) and the number of parameters used to fit it; AICc corrects for bias due to sample size.

We analyzed both the relative change in organic soil depth (change/original depth) and the absolute change. There were two reasons for doing so. First, since changes in soil depth varied greatly among communities even in the absence of fire, it was clear that these changes might well dominate the analyses of the GLIM's. Using relative change allows us to analyze all communities on the same scale and eliminates the potential for this particular artifact. Second, a change of depth (e.g., 0.2m) in organic soil pre- and post-burn could have significantly more impact in an upland community with relatively little organic soil to begin with than it does in a wetland community that naturally has more organic soil.

The data come from observational studies rather than planned, randomized experiments, and as a result the sample sizes are uneven among communities. We believe that the principal benefit of studies of this sort is to identify pattern and generate initial hypotheses for testing. For these reasons, as well as the conservative nature of the tests we applied, we use a significance level of $\alpha = 0.1$. In some exploratory studies of this kind investigators use even higher levels of α but in the present case few conclusions would change qualitatively by doing so.

Results

Most sites, including unburned sites, lost soil during the study period, likely as a result of oxidation and low levels of accretion of organic matter during the severe Florida drought of 1998-2001.

A heterogeneity of slopes model (using original soil depth as the covariate) showed that absolute change in soil depth differed significantly among communities, but not on burned vs. unburned sites, or on a fire x community interaction (Table 1). The original depth x community interaction is significant, and the AICc for this model is lower than for the corresponding analysis of covariance (i.e., the same model without any interaction terms involving original depth), so the heterogeneity of slopes model is preferable for this question. An ANOVA for the relative loss of soil has a highly significant term for community, and the community x burn interaction term has a *P*-value of 0.0776 (Table 1).

Restricting the analysis to sites that experienced fire, there was no discernible effect of fire type (prescribed or wildfire) on change in soil depth, whether measured in absolute or relative terms (Table 2). We analyzed general linear models with all permutations of predictor variables and their effect on both absolute and relative change in soil depth, and none showed a term for fire type that even remotely suggested the possibility that this is a useful predictor of soil loss. Soil loss (given fire) did depend significantly on community: absolute depth change was predicted by a depth x community interaction, and relative depth change was predicted by community.

Given that fire occurred at a location, which measurements of site moisture best predicted soil loss? We analyzed models involving the various combinations of original depth, community, and several indicators of drought. For $KBDI_{nearest}$, the KBDI measured at the nearest station, the ANCOVA model including only main effects of original depth, $KBDI_{nearest}$, and community was a far better fit than the model that also included a $KBDI_{nearest}$ x community interaction (Δ_{AIC} = -98). Original depth, community, and $KBDI_{nearest}$ were all significant predictors of absolute change in soil depth (Table 3). A similar conclusion can be drawn for relative change in soil depth: the model involving only main effects of community and $KBDI_{nearest}$ was a much better fit (Δ_{AIC} = -85) than one involving interaction terms, and both of the main effects were significant predictors of relative soil loss. Least-square means of the estimated soil loss, by community, are shown in Figure 1, as estimated from the model for absolute soil loss in Table 3.

Using $KBDI_{main}$ (the KBDI measured at the main station on the property) yields results somewhat different from those using $KBDI_{nearest}$. The best-fitting model (Δ_{AIC} = -78.8) for absolute change in soil depth was the model using original depth, community, and their interaction, plus $KBDI_{main}$ (Table 4). Original soil depth and its interaction with community are significant predictors of soil loss in this model. For relative soil depth change, the best-fitting model (Δ_{AIC} = -36.1) included only main effects of community and $KBDI_{nearest}$, and only community was a significant predictor.

Still different results come from an even coarser measure, $KBDI_{county}$, the KBDI measured for the entire county. The same models (Table 5) were best-fitting as in the case of $KBDI_{main}$ but now only original depth is a significant predictor of absolute soil loss and neither term is a significant predictor of relative soil loss. Our data set is not large enough to ask how the predictability of soil loss depends on distance to weather station or piezometer.

Analysis of piezometer readings (i.e., depth to water table) as predictors of absolute soil loss (Table 6) resulted in a best-fitting model ($\Delta_{AIC} = -8$) including original depth, community, and their interaction, plus depth to water table. Only original soil depth was a significant predictor of soil loss. A somewhat different result came from analysis of relative soil loss; the best-fitting model ($\Delta_{AIC} = -9.5$) included only community and depth to water table, and the latter was a significant predictor of relative soil loss.

Each set of models using different hydrologic or climate measurements involves a different set of data, and in particular the dataset for the depth-to-water-table models is substantially smaller than that for the others. To ask whether the apparent changes in the utility of various predictors with resolution of the hydrological data are real or artifacts of differing sample size, we reanalyzed the model including $KBDI_{nearest}$ to include only data points that were in the depth-to-water-table dataset. Results (not shown) were qualitatively unchanged; all three terms (original depth, community, and $KBDI_{nearest}$) were still significant predictors of the absolute loss of soil, and the latter two terms were both still significant predictors of relative soil loss.

Discussion

We predicted that presence or absence of fire itself would not have a significant effect on loss of soil, because fires occur under quite varied weather conditions and in quite different communities. The present results provide strong support for this prediction. There is no generalization one can make about the effect of fire on soils – the effect of fire depends on the conditions under which it occurs and on the community in which it occurs. This provides a strong argument against blanket policies excluding fire from wetlands in either prescribed burns or wildfires. Within the constraints imposed by the need to protect public health and safety, fire managers should strongly consider allowing fires into wetlands under many circumstances. In forthcoming papers we will examine the effects of fire (and fire conditions) on the plant communities themselves.

Comparison of the results using different hydrological predictors reveals a striking pattern: with improved spatial resolution in KBDI, we become increasingly able to predict soil loss. Original soil depth was always a significant predictor of absolute soil loss. However, we were able to detect significant effects of KBDI itself only when using $KBDI_{nearest}$.

Use of piezometers, with a very fine spatial resolution but coarser temporal resolution, does not appear to improve matters. The best-fitting model using depth-to-water-table is remarkably uninformative: the only significant predictor of organic soil loss is the amount of organic soil present! Using the same data set, we were able to evaluate the dependence of soil loss on $KBDI_{nearest}$ and community. Taken together, these results suggest that, at least for central Florida, daily hydrologic data on the scale of several km may be substantially more useful in predicting soil loss connected with fire events than data taken at coarser temporal or spatial scales.

We believe that this conclusion flows from the scale on which soil moisture content is likely to vary. At a given time, soil moisture content (at a given elevation) in central Florida is likely to vary on the scale of a few km because it depends heavily on two factors: the size of thunderstorms, the size of watersheds, and the scale on which soil types varies. Typically, thunderstorms in this region occur along paths a few km to (at most) a few tens of km wide. Watersheds are typically a few km across. Thus, information on the moisture content of soils on this scale – whether calculated as KBDI, or a direct measurement of the depth-to-water-table – should have much greater predictive power than information on a coarser spatial scale.

It seems clear that monthly measurements of depth-to-water-table are on too coarse a temporal scale to predict soil loss. Thunderstorms occur as brief, often intense events. During the hot months, loss to evapotranspiration can be great. As a result, monthly measurement of depth-to-water-table, although on a fine spatial scale, is not useful as a predictor of soil consumption.

We speculate that an extensive network of automatic well recorders would likely provide data sufficient to predict soil loss – and might even be a better predictor than $KBDI_{nearest}$. Clearly such a network would involve considerable cost, however. Table 8 provides estimates of the per-unit cost of the TNC monitoring network. The four weather stations have the highest per-unit cost. However, the 80 current piezometers collectively cost considerably more, and provide poorer prediction of soil loss. Had the network been established to predict soil loss (rather than to monitor hydrological conditions) it would be difficult not to conclude that the piezometers were a poor investment.

Having said this, it is clear that most fire managers will not have information on KBDI available on the scale of our $KBDI_{nearest}$. Typically, managers will have information on the scale of our $KBDI_{county}$, which was not a useful predictor of soil loss. We do not believe that this means that one cannot predict the consequences of fire in wetlands. In preparing prescribed burns, the fire managers at DWP normally visit every wetland in a burn unit and examine the soils. In effect, they are roughly estimating something related to KBDI in each wetland. Practices of this sort, in the hands of trained fire managers, are likely to be as useful as having fine-scale weather or soil moisture data, and are certainly much less costly.

Soil losses differ among wetland communities experiencing fire. Figure 1 suggests that soil losses were rather similar in five communities – bay heads, flatwoods, hardwood forests, slash pine, and wet prairies. On the other hand, soil losses were substantially smaller in the scrubby and shrub communities, and on average the cypress domes, marshes, and pastures gained soil during our study.

The estimated mean changes in soil depth (Figure 1) were on the order of the precision of our measurement. Thus we regard this study as establishing some "minimum" results: planned experiments using measurements on a finer scale may well find additional pattern that we are unable to discern with our current data. However, such added precision in measurement would not, however, reduce the statistical significance of pattern that we have succeeded in identifying.

Fire varies considerably among plant communities in its role and consequences; fire and plant ecologists have known this for some time. There has been considerable interest in predicting duff consumption and estimating ignition boundaries (Little et al. 1986, Reinhardt et al., 1991 Hungerford et al. 1996, Miyanishi and Johnson 2002), but this is the first study of which we are aware documenting differences in fire-caused organic soil consumption among plant communities. Our results provide quantitative evidence that burns in some wetland types consume substantial soil, while those in other wetland types do not.

Soil consumption is a key component of smoke production. Our study points to the need for planned experiments that can quantify how smoke production may vary with plant community and soil hydration. Our forthcoming papers will address the consequences of fires for plant communities themselves.

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Table 1. Significance tests for the best-fitting models of the effect of burning on change in depth of organic soil.

Effect	Num df	Den df	*F*	*P*
		Absolute change		
Depth	1	459	24.87	<0.0001
Burn (y/n)	1	457	0.00	0.9844
Depth x Burn	1	457	0.26	0.6099
Community	10	457	1.78	0.0619
Depth x Community	9	453	4.06	<0.0001
Community x Burn	7	455	0.42	0.8906
Depth x Community x Burn	7	454	0.84	0.5510
		Relative change		
Burn (y/n)	1	397	0.06	0.8081
Community	10	431	5.71	<0.0001
Burn x Community	8	430	1.79	0.0776

Table 2. Significance tests for the best-fitting models of the effect of fire type (prescribed or wild) on change in organic soil depth, given that fire occurred.

Effect	Num df	Den df	*F*	*P*
	Absolute change			
Depth	1	178	3.56	0.0607
Fire type	1	180	0.15	0.6966
Depth x Fire type	1	179	0.00	0.9976
Community	9	181	0.51	0.8654
Depth x Community	8	180	2.22	0.0283
Community x Fire type	4	176	0.41	0.7986
Depth x Community x Fire type	4	177	0.29	0.8860
	Relative change			
Fire type	1	166	0.32	0.5703
Community	8	171	3.05	0.0031
Fire type x Community	4	169	1.65	0.1629

Table 3. Significance tests for the best-fitting models of the effect of $KBDI_{nearest}$ on change in organic soil depth, given that fire occurred.

Effect	Num df	Den df	*F*	*P*
		Absolute change		
Depth	1	144	33.77	<0.0001
Community	9	178	2.11	0.0312
$KBDI_{nearest}$	1	46.5	10.15	0.0026
		Relative change		
Community	8	149	4.05	0.0002
$KBDI_{nearest}$	1	62.6	7.95	0.0064

Table 4. Significance tests for the best-fitting models of the effect of $KBDI_{main}$ on change in organic soil depth, given that fire occurred.

Effect	Num df	Den df	*F*	*P*
		Absolute change		
Depth	1	186	10.22	0.0016
Community	9	186	1.06	0.3967
Depth x community	8	190	4.24	0.0001
$KBDI_{main}$	1	62.4	1.10	0.2982
		Relative change		
Community	8	174	3.71	0.0005
$KBDI_{main}$	1	94.8	0.90	0.3445

Table 5. Significance tests for the best-fitting models of the effect of $KBDI_{county}$ on change in organic soil depth, given that fire occurred.

Effect	Num df	Den df	*F*	*P*
		Absolute change		
Depth	1	27.2	8.35	0.0075
Community	5	27.4	0.79	0.5644
Depth x community	5	28.2	1.42	0.2470
$KBDI_{county}$	1	6.3	0.25	0.6357
		Relative change		
Community	5	27.3	0.56	0.7278
$KBDI_{county}$	1	8.8	0.11	0.7523

Table 6. Significance tests for the best-fitting models of the effect of depth-to-water-table on change in organic soil depth, given that fire occurred.

Effect	Num df	Den df	*F*	*P*
		Absolute change		
Depth	1	19.5	3.61	0.0722
Community	5	11.3	0.25	0.9308
Depth x community	5	16.3	1.56	0.7262
Water table	1	5.9	0.57	0.4801
		Relative change		
Community	4	24	1.61	0.2042
Water table	1	24	3.53	0.0723

Table 7. Comparison of models using $KBDI_{nearest}$ and depth to water table at nearest piezometer to predict soil loss. *P* value is for the effect of KBDI or piezometer reading.

	KBDI		**Peizometer**	
	AIC	*P*	AIC	*P*
Absolute	-115.1	0.0106	-119.9	0.1501
Relative	52.6	0.0074	46.7	0.0425

Table 8. Estimated installation costs and annual costs per unit for manual piezometers, automatic well recorders, and weather stations. Annual costs include maintenance, data acquisition, data processing, and associated management costs.

	Installation	Annual costs
Piezometers	$183	$113
Automatic recorders	$1500	$130
Weather stations	$2000	$200

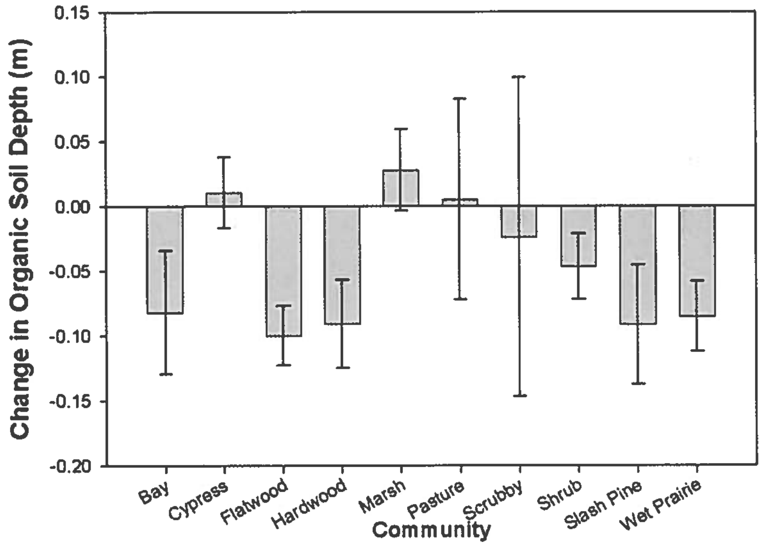

Figure 1. Least-square means estimates of change in organic soil depth (m) ± 1 SE. Estimated from the model for absolute change in soil depth, using $KBDI_{nearest}$. Changes were significantly different from zero for Bay, Flatwood, Hardwood, Shrub, Slash Pine, and Wet Prairie communities.