DIVISION S-3—SOIL BIOLOGY & BIOCHEMISTRY

Flush of Carbon Dioxide Following Rewetting of Dried Soil Relates to Active Organic Pools

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ABSTRACT

Soil quality assessment could become more standardized with the development of a simple, rapid, and reliable method for quantifying potential soil biological activity. We evaluated the flush of CO_2 following rewetting of dried soil under standard laboratory conditions as a method to estimate an active organic matter fraction. The flush of CO_2 following rewetting of dried soil (3 d incubation at ≈50% water-filled pore space and 25°C) was assessed for 20 soil series containing a wide range of organic C (20 ± 13 g kg^{-1}) from Alberta–British Columbia, Maine, Texas, and Georgia. This flush of CO_2 explained 97% of the variability in cumulative C mineralization during 24 d [$y = 12 + 3.3(x)$; $n = 471$], 86% of the variability in soil microbial biomass C [$y = 337 + 2.4(x)$; $n = 399$], and 67% of the variability in net N mineralization during 24 d [$y = 18 + 0.10(x) - 0.00002(x)^2$; $n = 327$]. Accounting for geographical differences in mean annual temperature and precipitation, which could affect soil organic matter quality, further improved relationships between the flush of CO_2 and active, passive, and total C and N pools. Measuring the flush of CO_2 following rewetting of dried soil may have value for routine soil testing of biological soil quality because it (i) is an incubation procedure patterned after natural occurrences in most soils, (ii) exhibits strong overall relationships with active organic pools, (iii) shows relatively minor changes in relationships with active organic pools that may be due to climatic variables, (iv) has a simple setup with minimal equipment requirements, and (v) has rapid analysis time.

Active fractions of soil organic matter are important to the plant-available nutrient supply, decomposition of natural and synthetic organic amendments, and manipulation of soil structure as a result of microbial biomass and activity. Assessments of biological soil quality must estimate these important biogeochemical functions of soils.

Numerous physical, chemical, and biological indicators have been proposed for soil quality assessment (Doran and Parkin, 1994). A dilemma faced by those making soil quality assessments is to define the optimum set of indicators that provide the most information with minimal duplication. Doran and Parkin (1994) stated that good soil quality indicators will (i) encompass ecosystem processes and relate to process-oriented modeling, (ii) integrate soil physical, chemical, and biological properties and processes, (iii) be accessible to many users and applicable to field conditions, (iv) be sensitive to variations in management and climate, and (v) where possible, be components of existing soil databases. Similarly, Holloway and Stork (1991) suggested that ecological indicators (i) show a prompt and accurate response to perturbation, (ii) reflect some aspect of ecosystem function, (iii) be readily and economically accessible, and (iv) be universal in distribution yet show individual specificity to temporal or spatial patterns.

Many proposed indicators meet one or more of these criteria, but few meet them all. This means that many indicators would be needed to cover all of the criteria and provide some overlap for verification of their validity.

Assessment of soil quality will become more meaningful and useful to land managers if they are not overwhelmed with the multitude of soil properties that have been suggested as important. Quantifying a few key indicators linked to other mechanistically important soil biogeochemical functions could provide a meaningful surrogate system for land managers, rather than measuring all actual soil properties or functions, which would be laborious and expensive. A simplified, surrogate system could lead to greater adoption of, and appreciation for, soil quality assessment.

Release of potentially mineralizable nutrients, decomposition, and biophysical manipulation of soil structure are generally functions of the soil microbial biomass and its activity. Substrates for microbial biomass and its activity depend upon plant production and other organic inputs. Therefore, two of the key functions of soil [i.e., providing a medium for plant production and maintaining environmental quality by decomposing various amendments (Doran and Parkin, 1994)] are linked to microbial biomass and its activity through substrate availability. Indeed, previously proposed biological soil properties important for soil quality assessment included potentially mineralizable C and N, microbial biomass C and N, and their proportions of total organic C (Doran and Parkin, 1994).

Methodology for determining the traditional suite of soil biological properties can be laborious and lengthy and can require expensive analytical equipment. Alternative methodology is needed that expresses several biological properties in a simplified manner. Recently,

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Published in Soil Sci. Soc. Am. J. 64:613–623 (2000).

Abbreviations: CMIN, carbon mineralization; NMIN, net nitrogen mineralization; POC, particulate organic carbon; Pr, mean annual precipitation; Pr/PET, mean annual precipitation/potential evapotranspiration; SMBC, soil microbial biomass carbon; SOC, soil organic carbon.

several simple alternatives to traditional microbial biomass methods have been examined, including chloroform-fumigation extraction (Vance et al., 1987), direct-chloroform extraction (Gregorich et al., 1990), dehydration and extraction (Sikora et al., 1994), and hot-water-soluble extraction (Sparling et al., 1998). Additionally, simpler alternatives to lengthy aerobic incubations for potentially mineralizable N have been proposed, including extraction of NH_4–N following autoclaving (Keeney, 1982) and extraction of NH_4–N with hot KCl (Jalil et al., 1996). Some of these methods have not necessarily been tested with a wide range of soils and tend to require expensive analytical equipment. Further, all extraction methods are chemical-based and, therefore, may extract a variable or unknown fraction of a relatively small labile organic pool. Extraction efficiency may be altered by variations in soil physical and chemical properties, including texture, structural integrity, organic matter, and pH (Badalucco et al., 1997; Anderson and Joergensen, 1997).

A more direct expression of potential microbial activity is through incubation, rather than chemical extraction. Incubations allow naturally occurring interactions among chemical, physical, and biological components of the soil to guide the analytical result obtained. Incubation-based methods have traditionally been too lengthy (i.e., 14–210 d) for routine soil testing (Keeney, 1982). Recently, it was reported that the flush of CO_2 during the first day following rewetting of dried soil was related to both soil microbial biomass C and potentially mineralizable C and N in eight soils from Texas (Franzluebbers et al., 1996a). Hence, the flush of CO_2 following rewetting of dried soil may have the potential to indicate nutrient cycling and decomposition capacity (Marumoto et al., 1982; Sparling et al., 1995), amount and quality of substrates available (Sorensen, 1974; Sparling and Ross, 1988), and size of the microbial biomass pool (Haider et al., 1991). If the flush of CO_2 following rewetting of dried soil could be related to microbial biomass and mineralizable C and N for different soils under different environments, it would be a useful indicator of several biologically active components of soil quality and could move practical assessment of soil quality toward reality.

Our objectives were to (i) develop relationships between the flush of CO_2 following rewetting of dried soil and active (i.e., soil microbial biomass C and potentially mineralizable C and N), passive (i.e., particulate organic C), and total organic C in soils from diverse ecological regions and (ii) test the sensitivity of the flush of CO_2 to land-management variations compared with other standard estimates of soil biological properties.

MATERIALS AND METHODS

Soils were collected from various depth increments to a maximum of 0.3 m from several long-term management sites in Alberta–British Columbia, Maine, Texas, and Georgia during April through June of 1992 to 1997 before planting of row crops or summer forage growth (October 1997 following crop growth in Maine) (Table 1). Management effects and further description of experimental setup can be found in Franzluebbers and Arshad (1996a, 1996b, 1997a, 1997b) for samples collected in Alberta–British Columbia, in Haney (1997) and Schomberg and Jones (1999) for samples collected in Texas, and in Franzluebbers et al. (1999a, 1999b) for samples collected in Georgia.

The four regions we selected could be characterized relatively as cold–dry [Alberta–British Columbia; 2°C mean annual temperature, 0.5-m mean annual precipitation (Pr), 0.9 mean annual precipitation/potential evapotranspiration (Pr/PET)], cold–wet (Maine; 7°C, 1.1-m Pr, 1.9 Pr/PET), hot–dry (Texas; weighted mean 18°C, 0.6-m Pr, 0.6 Pr/PET), and hot–wet (Georgia; 17°C, 1.3-m Pr, 1.4 Pr/PET).

Carbon mineralization (CMIN) was determined from 15 to 120 g subsamples of soil under the following set of standard conditions: Different amounts of soil were used to obtain more similar amounts of CO_2 evolved from different soil depths. Soil was oven-dried (55°C, 48 h) and gently crushed to pass a 4.75-mm screen. Duplicate soil subsamples were moistened to 50% water-filled pore space (i.e., soil lightly packed in graduated bottles and water added to fill 50% of the available pore space, assuming a particle density of 2.65 Mg m^{-3}) and incubated at 25°C ± 1°C in 1-L canning jars containing a vial with 10 mL of 1.0 *M* NaOH to absorb CO_2 and a vial with 10 mL of water to maintain humidity. Alkali traps were replaced at 3 and 10 d and removed at 24 d. The quantity of CO_2–C evolved was determined by titration of NaOH with 1.0 *M* HCl (Anderson, 1982). At 10 d, one of the two subsamples was removed, fumigated with chloroform for 24 h, and incubated separately under the same conditions to determine the flush of CO_2–C representing soil microbial biomass C (SMBC) using an efficiency factor of 0.41 (Voroney and Paul, 1984). Determination of SMBC following rewetting of dried soil and 3 to 10 d of pre-incubation has been shown to yield estimates equivalent to those from field-moist soil (Franzluebbers et al., 1996a; Franzluebbers, 1999b). Deviations from this standard protocol were using air-dried soil, sieving to pass a 5.6-mm screen, and adjusting water content to ≈ −33 kPa for soils in Alberta–British Columbia; sieving to pass a 6-mm screen, air-drying, and removing alkali traps at 3, 10, and 25 d for the Pullman SCL(1) soils in Texas; and oven-drying at 40°C, sieving to pass a 5-mm screen, adjusting water content to ≈ −33 kPa, and removing alkali traps at 1, 3, 7, 17, and 24 d for the remaining soils in Texas. Variation in sieve size from 4.7 to 6 mm should not have significantly affected results (Franzluebbers, 1999b). Water-filled pore space of 50% corresponds to −7 to −22 kPa for many of the soils collected in Georgia (Franzluebbers, 1999a). Maximum microbial activity for soils in Georgia occurred within a range of matric potentials from −1 to −160 kPa (Franzluebbers, 1999a). Increasing drying temperature from air-dried to 55°C may have increased the flush of CO_2 during the first 3 d following rewetting by 40 ± 4 mg kg^{-1} soil, which was of the same magnitude of increase in extractable C with higher drying temperature (i.e., shift from 40 to 60°C) (R. Haney, unpublished data, 1998).

Net N mineralization (NMIN) was determined from changes in inorganic N (NO_3–N + NO_2–N + NH_4–N) concentration between 0 and 24 d of incubation in 2 *M* KCl extracts using Cd reduction and salicylate–nitroprusside autoanalyzer techniques (Bundy and Meisinger, 1994). Soil at 0 and 24 d was oven-dried (55°C, 48 h), sieved to pass a 2-mm screen, and a 10-g subsample was shaken with 20 mL of 2 *M* KCl for 30 min. Deviations from this standard protocol were drying at 60°C, taking a 7-g subsample, and shaking with 28 mL of 2 *M* KCl for soils in Texas and analyzing NH_4–N using a citrate buffer autoanalyzer technique for soils in Alberta–British Columbia. We did not expect any systematic errors due to these variations in protocol.

Soil organic C and N were determined either by dry combustion for soils in Maine and Georgia (pH < 7) or dichromate

Table 1. Soil and environmental characteristics of samples and sites used to develop relationships among active soil C and N pools.

Location	USDA soil classification	Mean annual pH	Mean annual Tem	Mean annual Precip	Land management
			°C	mm	
Dawson Creek, BC (55° N, 120° W)	Donnelly silt loam (sandy-skeletal, mixed, Typic Eutrocryepts)	5.5	1	504	Barley (*Hordeum vulgare* L.) under conventional and no tillage
Rolla, BC (55° N, 120° W)	Donnelly sandy loam (sandy-skeletal, mixed, Typic Eutrocryepts)	6.6	1	504	Wheat (*Triticum aestivum* L.), canola (*Brassica campestris* L.), barley under conventional and no tillage
Rycroft, AB (55° N, 118° W)	Falher clay (fine, montmorillonitic, frigid, Typic Natriboralfs)	5.7	2	449	Barley, wheat, pea (*Pisum sativum* L.) under conventional and no tillage
Beaverlodge, AB (55° N, 119° W)	Hythe clay loam (fine, montmorillonitic, frigid Mollic Cryoboralfs)	6.7	2	452	Barley, canola, pea under conventional and no tillage
Newport, ME (45° N, 69° W)	Bangor silt loam (coarse-loamy, mixed, frigid Typid Haplorthods)	5.4	7	1070	Grass sod and potato (*Solanum tuberosum* L.) cropping
Stillwater, ME (45° N, 68° W)	Nicholville very fine sandy loam (coarse-silty, isotic, frigid Aquic Haplorthods) and Lamoine-silt loam (fine, illitic, nonacid, frigid Aeric Epiaquepts)	6.1	7	1070	Wheat, clover (*Trifolium pratense* L.), dry bean (*Phaseolus vulgaris* L.), maize (*Zea mays* L.) cropping
Lubbock, TX (33° N, 101° W)	Acuff loam (fine-loamy, mixed, superactive, thermic Aridic Paleustolls)	7.4	16	457	Sorghum [*Sorghum bicolor* (L.) Moench] with 0 to 220 kg N ha^{-1} and 0 to 132 kg P ha^{-1}
Amarillo, TX (35° N, 101° W)	Pullman clay loam (fine, mixed, superactive, thermic Torrertic Paleustolls)	6.0	17	425	Wheat, sorghum under conventional and no tillage with 0 to 55 kg N ha^{-1}
Stephenville, TX (32° N, 98° W)	Windthorst fine sandy loam (fine, mixed, thermic Udic Paleustalfs)	6.5	18	750	Bermudagrass [*Cynodon dactylon* (L.) Pers.] hay receiving 0 to 450 kg N ha^{-1} dairy manure
College station, TX (30° N, 96° W)	Weswood silty clay loam (fine-silty, mixed superactive, thermic Udifluventic Usto-chrepts	8.2	20	978	Sorghum, wheat, soybean [*Glycine max* (L.) Merr.] under conventional and no tillage
Overton, TX (32° N, 94° W)	Bowie fine sandy loam (fine-loamy, siliceous, semiactive, thermic Plinthic Paleudults	5.9	20	1050	Bermudagrass hay receiving 0 to 450 kg N ha^{-1} as poultry manure
Corpus Christi, TX (27° N, 97° W)	Victoria clay (fine, smectitic, hyperthermic Udic Pellusterts)	8.0	22	765	Sorghum, maize under conventional and reduced tillage with 0 to 60 kg N ha^{-1}
Farmington, GA (33° N, 83° W)	Appling, Cecil, Madison, Pacolet, Wedowee loamy sand, sandy loam, loam, sandy clay loam (fine, kaolinitic, thermic Typic Kan-hapludults), Grover loam, sandy loam, sandy clay loam (fine-loamy, micaceous, thermic Typic Hapludults) Louisa sandy clay loam (loamy, micaceous, thermic, shallow Ruptic-Ultic Dystrudepts)	6.0	17	1250	Bermudagrass pasture receiving N via inorganic, clover (*Tri folium incarnatum* L.), poultry litter sources under low and high cattle-grazing pressures
Watkinsville, GA (34° N, 83° W)	Cecil sandy loam (fine, kaolinitic, thermic Typic Kanhapludults)	6.0	17	1250	Cotton (*Gossypium hirsutum* L.), rye (*Secale cereale* L.) under no tillage with various frequencies and intensities of tillage and tall fescue (*Festuca arundinacea* Schreb.) with and without endophyte (*Neotyphodium coeno-phialum*)

Table 2. Regression of C mineralization during 0–24 d ($CMIN_{0\text{-}24\,d}$), soil microbial biomass C (SMBC), particulate organic C (POC), and soil organic C (SOC) on C mineralization during 0–3 d ($CMIN_{0\text{-}3\,d}$), using the linear equation ($Y = \beta_0 + \beta_1 \cdot CMIN_{0\text{-}3\,d}$) for several data sets in Alberta–British Columbia, Maine, Texas, and Georgia.

Conditions	*n*	$CMIN_{0\text{-}3\,d}$ Mean ± SD	$CMIN_{0\text{-}24\,d}$			SMBC			POC			SOC		
			β_0	β_1	r^2	β_0	β_1	r^2	β_0	β_1	r^2	β_0	β_1	r^2
Alberta/British Columbia; cropland; conventional and no tillage; 0- to 50-mm, 50- to 125-mm, 125- to 200-mm depths														
Donnelly L	24	108 ± 83	−86	5.11	0.97	317	3.62	0.90	4.3	0.032	0.83	14.0	0.038	0.61
Donnelly SiL	18	146 ± 90	−84	4.91	0.98	427	3.18	0.87	2.5	0.060	0.78	10.6	0.097	0.72
Hythe CL	24	162 ± 88	−143	4.93	0.93	389	3.57	0.84	4.5	0.053	0.74	17.3	0.096	0.65
Falher C	24	107 ± 63	−108	5.29	0.97	444	4.47	0.91	1.0	0.055	0.78	24.7	0.108	0.64
Alberta/British Columbia; cropland, conventional and no tillage; 0- to 50-mm, 50- to 125-mm, 125- to 200-mm depths														
1- to 5.6-mm WSA†	90	188 ± 107	−25	3.49	0.91	10	2.44	0.91	ND‡	ND	ND	14.0	0.082	0.65
0.25- to 1-mm WSA	90	155 ± 69	16	2.95	0.93	72	2.30	0.79	ND	ND	ND	14.0	0.120	0.72
0.05- to 0.25-mm WSA	90	116 ± 50	23	3.04	0.93	46	3.03	0.84	ND	ND	ND	3.9	0.193	0.72
Crushed 1- to 5.6-mm WSA	90	220 ± 154	95	2.88	0.95	ND	ND	ND	ND	ND	ND	ND	ND	ND
Crushed 0.25 to 1-mm WSA	90	179 ± 99	101	2.68	0.92	ND	ND	ND	ND	ND	ND	ND	ND	ND
Particulate organic fraction	90	33 ± 17	−58	6.78	0.64	ND	ND	ND	ND	ND	ND	0.5	0.277	0.84
Maine; cropland and grass sod; 0- to 50-mm, 50- to 100-mm, 100- to 200-mm depths														
Bangor SiL, Nicholville SL, Lamoine SiL	48	179 ± 81	16	2.36	0.96	132	1.90	0.60	ND	ND	ND	19.1	0.034	0.36
Texas; cropland and hayland; 0- to 100-mm, 0- to 75-mm, 0- to 20-mm, 20- to 40-mm, 40- to 80-mm depths														
Acuff L	9	76 ± 12	−23	5.15	0.68	115	4.47	0.48	ND	ND	ND	7.5	−0.016	0.02
Bowie SL	8	139 ± 12	175	3.36	0.64	671	1.86	0.07	ND	ND	ND	18.3	−0.016	0.03
Orelia SCL–Victoria C	6	71 ± 16	−95	4.84	0.97	143	7.53	0.91	ND	ND	ND	2.4	0.089	0.84
Pullman CL(1)	90	84 ± 52	91	3.12	0.92	378	3.29	0.68	ND	ND	ND	8.9	0.008	0.09
Pullman CL(2)	26	91 ± 40	−62	4.44	0.71	195	6.04	0.71	ND	ND	ND	7.5	0.020	0.28
Weswood SiCL	18	76 ± 41	41	3.55	0.95	475	5.75	0.57	ND	ND	ND	8.6	0.029	0.13
Windthorst SL	20	149 ± 54	108	3.51	0.81	569	3.24	0.51	ND	ND	ND	9.8	0.051	0.75
Georgia; bermudagrass pasture; 0- to 20-mm, 20- to 40-mm, 40-mm to 60-mm depths														
Appling LS, SL, L	36	325 ± 245	−1	3.02	0.86	427	1.93	0.86	2.5	0.023	0.77	6.8	0.049	0.82
Cecil LS, SL, L, SCL	162	355 ± 276	−95	3.22	0.94	453	1.99	0.79	1.9	0.024	0.68	5.8	0.048	0.88
Grover SL, L, SCL	84	374 ± 287	−45	2.99	0.94	466	1.68	0.77	2.8	0.021	0.66	6.5	0.047	0.83
Louisa SCL	6	360 ± 341	−165	3.66	0.98	490	1.82	0.90	1.0	0.024	0.89	4.8	0.045	0.99
Madison LS, SL, SCL	336	380 ± 312	−51	3.10	0.95	462	1.83	0.79	2.5	0.022	0.70	6.4	0.048	0.86
Pacolet LS, SL, SCL	96	384 ± 336	−15	2.98	0.93	482	1.79	0.82	2.9	0.021	0.72	6.9	0.043	0.87
Wedowee SL	12	436 ± 381	−41	2.98	0.94	551	1.76	0.91	3.7	0.013	0.56	9.4	0.031	0.76
Unclassified sites	222	319 ± 261	−15	2.91	0.93	533	1.39	0.71	3.2	0.025	0.79	7.1	0.056	0.98
Georgia; tall fescue pasture; 0- to 25-mm, 25- to 75-mm, 75- to 150-mm, 150- to 300-mm depths														
15-yr, 112 kg N ha^{-1}	80	440 ± 634	81	2.90	0.96	415	2.53	0.90	2.2	0.016	0.93	8.8	0.030	0.91
15-yr, 336 kg N ha^{-1}	80	425 ± 642	11	3.15	0.97	403	2.97	0.70	2.4	0.019	0.76	10.5	0.029	0.78
8-yr, 336 kg N ha^{-1}	72	584 ± 936	22	2.59	0.98	138	2.37	0.95	0.6	0.015	0.95	5.7	0.021	0.88
Georgia; cropland; tillage type and frequencies; 0- to 25-mm, 25- to 75-mm, 75- to 150-mm depths														
Cecil SL	90	304 ± 203	−120	3.08	0.99	136	2.31	0.94	−1.0	0.018	0.85	3.9	0.027	0.84

† WSA is water-stable aggregate fraction.
‡ ND is not determined.

oxidation with heating to 100°C for 1 h and Kjeldahl digestion for soils in Alberta–British Columbia and Texas. Particulate organic-matter fractions (≥0.05-mm diam.) were dispersed and collected according to the procedures described in Franzluebbers and Arshad (1997b) for soils in Alberta–British Columbia and according to the procedures described in Franzluebbers et al. (1999a) for soils in Georgia. Carbon concentration of the particulate organic fraction was determined according to the methods described for soil organic C.

Except for the data presented in Table 2, which were analyses on individual replications, all values were means of 3 to 13 replications per treatment per depth. Relationships between the flush of CO_2 during 3 d following rewetting of dried soil ($CMIN_{0\text{-}3\,d}$) and other organic-matter pools were evaluated based on slopes and coefficients of determination (r^2) using the general linear models procedure of SAS (SAS Institute, 1990). In the analyses of geographical differences, only $CMIN_{0\text{-}3\,d}$ values <500 mg kg^{-1} were used because this was the upper limit for all regions except Georgia.

RESULTS AND DISCUSSION

Following an initial flush of microbial activity that was most dominant during the first 3 d following rewetting of dried soil, CMIN gradually declined to a basal soil respiration rate (Fig. 1). The basal soil respiration rate differed among soils and was related to the level of CO_2 flush. The five soils in Fig. 1 were selected from the bermudagrass [*Cynodon dactylon* (L.) Pers.] pasture data set in Georgia. If CMIN rates among soils were not to interact with time, as observed in Fig. 1, then the initial flush of CO_2 could reflect longer-term potential CMIN. Further, if the C/N ratio of the mineralizable fraction remained stable among soils during the course of incubation, then the initial flush of CO_2 could reflect long-term NMIN (Franzluebbers et al., 1996a). Measurement of CMIN would be advantageous because in the short term (i.e., 0–3 d), low levels of mineralization would be more discernable with CMIN since ≈8 to 12 times more C than N is mineralized. In addition, NMIN must be calculated from initial and final concentrations, which adds to overall variability by involving two sources of experimental error.

The flush of CO_2 evolved during 3 d following rewetting of dried soil was highly related to that evolved during 1 d (Fig. 2). The strong relationship between $CMIN_{0\text{-}1\,d}$ and $CMIN_{0\text{-}3\,d}$ observed among these four soils collected in Texas suggests that results obtained with either protocol could be extrapolated to estimate active organic pools. For several soils in Texas, the flush of CO_2 during 1 d following rewetting of dried soil was highly related to $CMIN_{0\text{-}21\,d}$, SMBC, and $NMIN_{0\text{-}21\,d}$ (Franzluebbers et al., 1996a). Nevertheless, we chose to measure the flush of CO_2 following rewetting of dried soil during 3 d in this study, rather than 1 d, as a reasonable compromise between the following concerns: (i) to achieve high precision with a larger amount of CO_2 released (i.e., ≈2.5-fold greater) and (ii) to keep the incubation time short to be useful in commercial laboratories.

Relationship Between the Flush of Carbon Dioxide and Potential Carbon Mineralization

Within several data sets having a wide range in $CMIN_{0\text{-}3\,d}$, the relationship between $CMIN_{0\text{-}24\,d}$ and

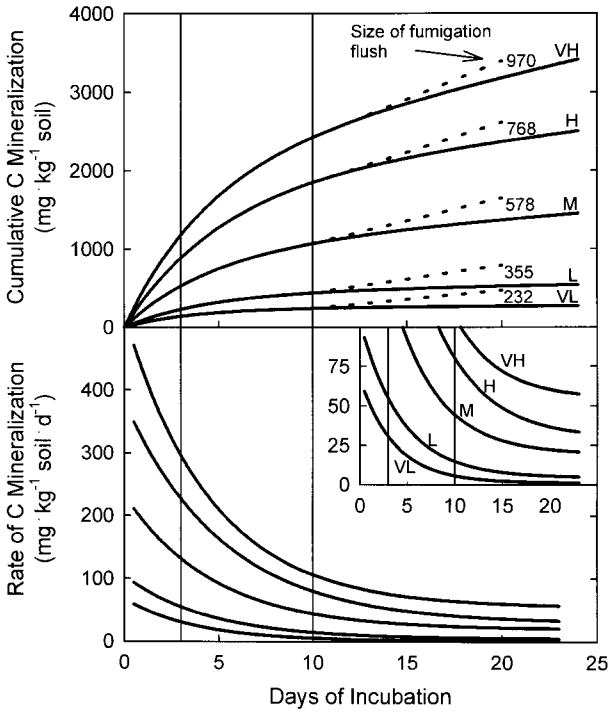

Fig. 1. Typical responses of cumulative C mineralization and rate of C mineralization following rewetting of dried soil in soils with various quantities of available C (VL is very low, L is low, M is medium, H is high, and VH is very high). Basal-soil respiration is achieved at ≈10 d. Dashed lines indicate the C mineralized due to fumigation at 10 d. The inset magnifies the rate of C mineralization with time.

$CMIN_{0\text{-}3\,d}$ was extremely strong (Table 2). Coefficients of determination relating $CMIN_{0\text{-}3\,d}$ to $CMIN_{0\text{-}24\,d}$ were greater than 0.8 in most instances, except for the particulate organic fraction of soils from Alberta–British Columbia and the Acuff L, Bowie SL, and Pullman CL(2) soils from Texas. Low r^2 values in those data sets may have been partly due to the low ranges of $CMIN_{0\text{-}3\,d}$ among observations (Table 2, column 3). Table 2 represents soils from four very diverse regions, different soil textures within a region, and different soil fractions within soils in Alberta–British Columbia (i.e., whole aggregates, crushed aggregates, and particulate organic fraction). Within data sets, coefficients of determination

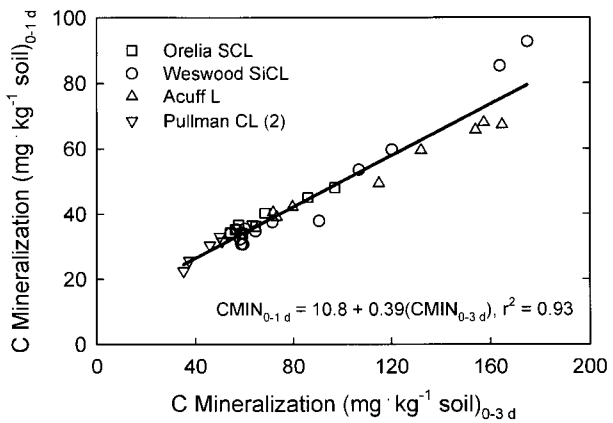

Fig. 2. Relationship of C mineralization during 0–3 d with C mineralization during 0–1 d in four soils from Texas.

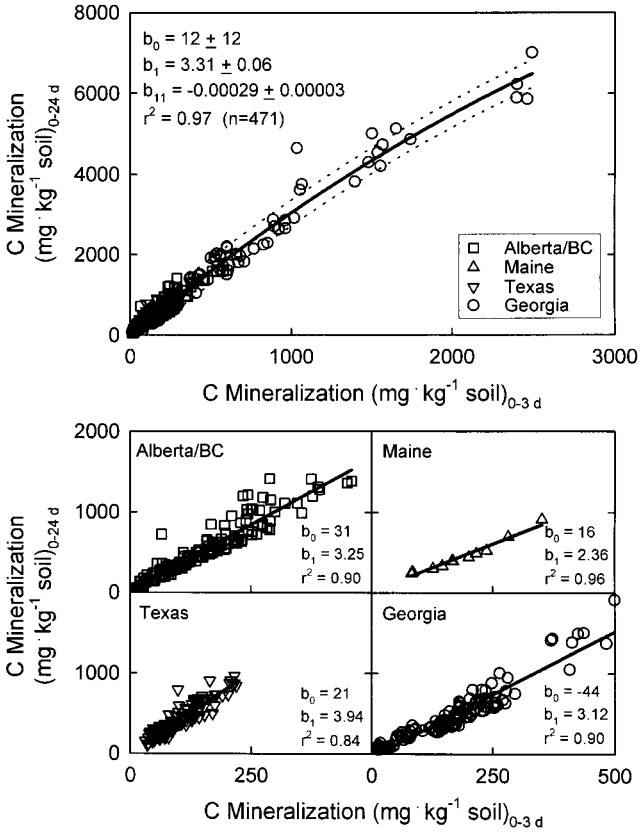

Fig. 3. Relationships of C mineralization during 0–3 d with C mineralization during 0–24 d in soils from Alberta–British Columbia, Maine, Texas, and Georgia. Lower panels are magnifications of the 0 to 500 mg kg^{-1} range in $CMIN_{0\text{-}3\,d}$ for each of the four regions.

relating $CMIN_{0\text{-}3\,d}$ to $CMIN_{0\text{-}24\,d}$ were 0.90 ± 0.10 (n = 30), indicating a very close association between these variables, regardless of type and portion of soil collected.

We chose $CMIN_{0\text{-}24\,d}$ as an estimate of long-term CMIN to represent an active pool of organic matter. We realize that long-term CMIN might have been better estimated in several-month-long incubations. However in preliminary experiments, we observed that $CMIN_{0\text{-}24\,d}$ and $CMIN_{0\text{-}100\,d}$ were highly related (r^2 = 0.94, n = 8) in soils from Oklahoma and Texas (R. Haney, unpublished data, 1998). Although $CMIN_{0\text{-}24\,d}$ is not completely independent from $CMIN_{0\text{-}3\,d}$, making the correlation between these two variables is obvious since cumulative CMIN would normally be estimated from 0 to 24 d and not from 3 to 24 d. However, the relationship between $CMIN_{3\text{-}24\,d}$ and $CMIN_{0\text{-}3\,d}$ was strong (r^2 = 0.90, n = 471) and the quantity of $CMIN_{3\text{-}24\,d}$ to that of $CMIN_{0\text{-}3\,d}$ was 2.5 ± 1.0 times greater, which support our approach of relating $CMIN_{0\text{-}3\,d}$ to $CMIN_{0\text{-}24\,d}$ without compromising statistical rigor.

Slopes of $CMIN_{0\text{-}24\,d}$ on $CMIN_{0\text{-}3\,d}$ for whole soils from Alberta–British Columbia were intermediate between those from particulate organic fractions and water-stable aggregate fractions (Table 2). A greater slope of $CMIN_{0\text{-}24\,d}$ on $CMIN_{0\text{-}3\,d}$ in the particulate organic fraction than in water-stable aggregate fractions indicates that the former fraction produced a lower flush of CO_2. This result suggests that the particulate organic fraction is a comparatively passive, or less active, pool of organic matter than C in whole soil or in macroaggregates, which supports the conclusions of Cambardella and Elliott (1992). Both aggregate and particulate organic fractions were washed and dried (60°C, 24 h) during preparation. Separation of particles in the particulate organic fractionation procedure may have removed SMBC and other labile components normally protected within aggregates. Because the lighter portions of particulate organic matter are more readily mineralized than heavier portions (Hassink, 1995), the quality of the particulate organic fraction as a whole may be reflecting properties of two extremes (i.e., labile microbial byproducts and more stable humic matter).

Within a region, slopes of $CMIN_{0\text{-}24\,d}$ on $CMIN_{0\text{-}3\,d}$ were very similar, despite large differences in soil texture and management in data sets from Alberta–British Columbia and contrasting soil series, type of vegetation, and management in data sets from Georgia (Table 2). Among regions, the slope of $CMIN_{0\text{-}24\,d}$ on $CMIN_{0\text{-}3\,d}$ was lowest in Maine, intermediate in Alberta–British Columbia and Georgia, and highest in Texas (Fig. 3). The slope of $CMIN_{0\text{-}24\,d}$ on $CMIN_{0\text{-}3\,d}$ was negatively correlated with Pr/PET ($r = -0.96$, $n = 4$), suggesting that greater effective moisture stress may reduce the size of the flush of CO_2 relative to longer-term CMIN. Intense and infrequent rainfall events in semiarid climates may cause pulses of C availability to soil microorganisms, which could favor selective decomposition of readily mineralizable fractions, leaving behind more resistant fractions. Despite these regional differences, when all data were pooled, a very strong relationship of $CMIN_{0\text{-}24\,d}$ on $CMIN_{0\text{-}3\,d}$ was observed spanning a large range in $CMIN_{0\text{-}3\,d}$ (Fig. 3).

As a rapid test, $CMIN_{0\text{-}3\,d}$ was a good predictor of longer-term C mineralization (i.e., $CMIN_{0\text{-}24\,d}$). Predicted values of $CMIN_{0\text{-}24\,d}$ using the pooled relationship in Fig. 3 were within ±25% of actual values 73% of the time. Ninety-six percent of the predictions were within ±50% of actual values. The deviation from actual values was 27% or less for 75% of the predictions.

Sensitivity of the relationship between $CMIN_{0\text{-}24\,d}$ and $CMIN_{0\text{-}3\,d}$ on soil pretreatment was evident when comparing the intact and crushed aggregate fractions in Alberta–British Columbia (Table 2). Crushing of both of the macroaggregate fractions (i.e., 0.25–1 and 1–5.6 mm) to <0.25 mm reduced the slope of $CMIN_{0\text{-}24\,d}$ on $CMIN_{0\text{-}3\,d}$ by 9 and 18%, respectively, indicating a release of C due to crushing that was susceptible to mineralization during 3 d. The slope of $CMIN_{0\text{-}24\,d}$ on $CMIN_{0\text{-}3\,d}$ was also reduced by 9% in soils sieved to <2 mm and by 20% in soils sieved to <0.5 mm, compared with intact soil cores (Franzluebbers, 1999b). Despite changes in slopes with changes in soil preparation, strong relationships between $CMIN_{0\text{-}24\,d}$ and $CMIN_{0\text{-}3\,d}$ always occurred (Table 3; Franzluebbers, 1999b). Therefore, as long as methodology is standardized, comparisons of biological soil quality among soils appear valid.

Relationship Between the Flush of Carbon Dioxide and Soil Microbial Biomass Carbon

Relationships between SMBC and $CMIN_{0\text{-}3\,d}$ were nearly as strong as between $CMIN_{0\text{-}24\,d}$ and $CMIN_{0\text{-}3\,d}$

Table 3. Relationship of soil C and N pools with $CMIN_{0\text{-}3\,d}$† as affected by regional differences. Data limited to $CMIN_{0\text{-}3\,d} < 500$ mg kg^{-1}; $n = 284$ for $CMIN_{0\text{-}24\,d}$‡, SMBC§, $NMIN_{0\text{-}24\,d}$¶, SOC#, and $n = 155$ for POC††.

Source of variation	$CMIN_{0\text{-}24\,d}$	SMBC	$NMIN_{0\text{-}24\,d}$	POC	SOC#
			Percentage of variability explained		
$CMIN_{0\text{-}3\,d}$	79.9***	38.1***	41.1***	36.6***	26.4***
AB + TX (dry) vs. ME + GA (wet)	8.9***	23.4***	0.2	ND‡‡	4.4***
AB + ME (cold) vs. TX + GA (hot)	0.0	2.7***	1.2**	ND	29.3***
AB + GA vs. ME + TX	0.5***	0.2	8.0***	37.3***	2.0***
			Regression coefficients		
$ß_0$	−26.3	306.6	9.6	0.9	7.2
$ß_1$ (Alberta–BC)	4.6	4.2	0.09	0.07	0.14
$ß_1$ (Maine)	2.6	1.1	0.19	ND	0.09
$ß_1$ (Texas)	4.3	4.8	0.22	ND	0.04
$ß_1$ (Georgia)	3.0	2.3	0.14	0.2	0.03

** and *** are $P \leq 0.01$ and $P \leq 0.001$, respectively.
† $CMIN_{0\text{-}3\,d}$ is the flush of CO_2–C evolved after rewetting dried soil during 0–3 d incubation (mg kg^{-1}).
‡ $CMIN_{0\text{-}24\,d}$ is the cumulative C mineralization during 0–24 d of incubation (mg kg^{-1}).
§ SMBC is soil microbial biomass C (mg kg^{-1}).
¶ $NMIN_{0\text{-}24\,d}$ is net N mineralization during 0–24 d of incubation (mg kg^{-1}).
# SOC is soil organic C (g kg^{-1} soil).
†† POC is particulate organic C (g kg^{-1}).
‡‡ ND is not determined.

for many of the data sets taken from the four diverse geographical regions (Table 2). In general, slopes of SMBC on $CMIN_{0\text{-}3\,d}$ varied less within than across geographical regions. Similar to the slope of $CMIN_{0\text{-}24\,d}$ on $CMIN_{0\text{-}3\,d}$, the slope of SMBC on $CMIN_{0\text{-}3\,d}$ was lowest in Maine, intermediate in Alberta–British Columbia and Georgia, and highest in Texas (Fig. 4). The fact that slopes were greater in Texas than all other regions is intriguing. Soils in Texas with higher clay content (e.g., Orelia SCL–Victoria C, Pullman CL, and Weswood SiCL) tended to have greater slopes than soils with lower clay content (e.g., Acuff L, Bowie SL, and Windthorst SL). Perhaps the combination of predominantly montmorillonitic clay and an overall drier climate in Texas resulted in little available C for the flush of CO_2–C following rewetting of dried soil, despite a larger pool of microbial biomass, which may have been protected by clay and/or clay-induced macroaggregation mechanisms. The flush of CO_2 during 3 d following the rewetting of dried soil relative to total organic C also decreased with the increasing clay content of soils in Georgia (Franzluebbers, 1999b).

Despite these regional differences, pooling data from all regions resulted in a strong relationship between SMBC and $CMIN_{0\text{-}3\,d}$, with a combined coefficient of determination of 0.86 (Fig. 4). As a rapid test, $CMIN_{0\text{-}3\,d}$ was a good predictor of SMBC, although less so than as a predictor of $CMIN_{0\text{-}24\,d}$. Predicted values of SMBC using the pooled relationship in Fig. 4 were within ±25% of actual values 46% of the time. Seventy-seven percent of the predictions were within ±50% of actual values. Deviation from actual values was 47% or less for 75% of the predictions.

Relationship Between the Flush of Carbon Dioxide and Net Nitrogen Mineralization

Relationships of $NMIN_{0\text{-}24\,d}$ with $CMIN_{0\text{-}3\,d}$ were significant, but not particularly strong within a region ($r^2 \approx 0.5$), except for the Maine data (Fig. 5). Slopes of $NMIN_{0\text{-}24\,d}$ on $CMIN_{0\text{-}3\,d}$ formed two groups of similarity, where they were greater in Maine and Texas than in Alberta–British Columbia and Georgia. Pooling all data resulted in a fairly strong relationship between $NMIN_{0\text{-}24\,d}$ and $CMIN_{0\text{-}3\,d}$ across a very wide range of $CMIN_{0\text{-}3\,d}$ ($r^2 = 0.67$, $n = 327$). Curvature in the relation-

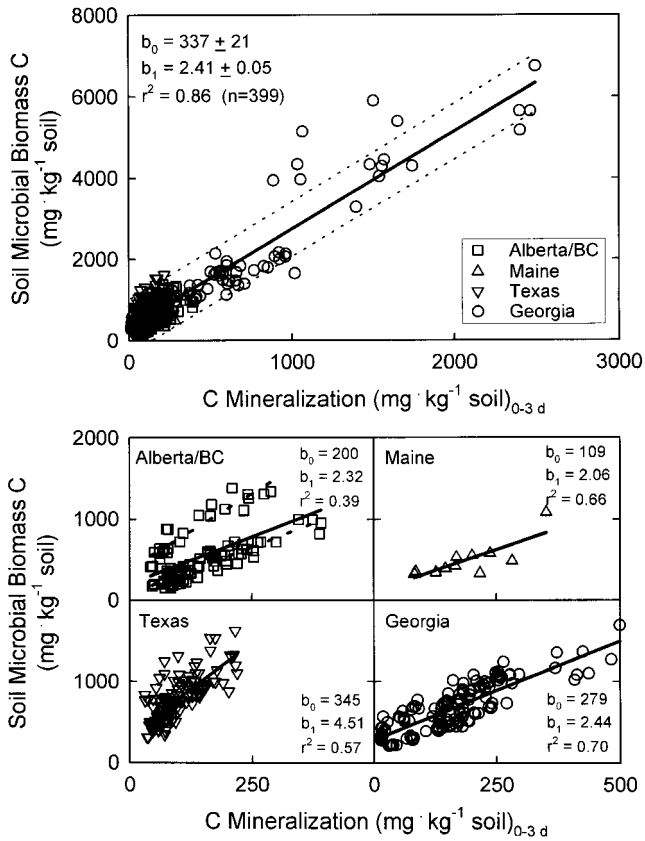

Fig. 4. Relationships of C mineralization during 0–3 d with soil microbial biomass C in soils from Alberta–British Columbia, Maine, Texas, and Georgia. Lower panels are magnifications of the 0 to 500 mg kg^{-1} range in $CMIN_{0\text{-}3\,d}$ for each of the four regions. Dashed lines in the Alberta–British Columbia subpanel indicate regression lines of whole soil (upper line, $r^2 = 0.87$) and aggregate fractions (lower line, $r^2 = 0.85$).

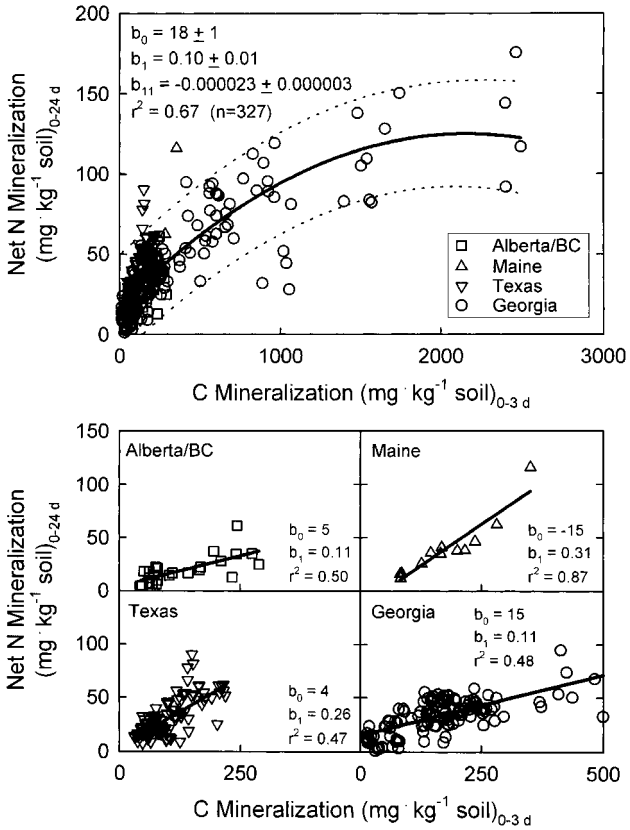

Fig. 5. Relationships of C mineralization during 0–3 d with net N mineralization during 0–24 d in soils from Alberta–British Columbia, Maine, Texas, and Georgia. Lower panels are magnifications of the 0 to 500 mg kg^{-1} range in $CMIN_{0\text{-}3\,d}$ for each of the four regions.

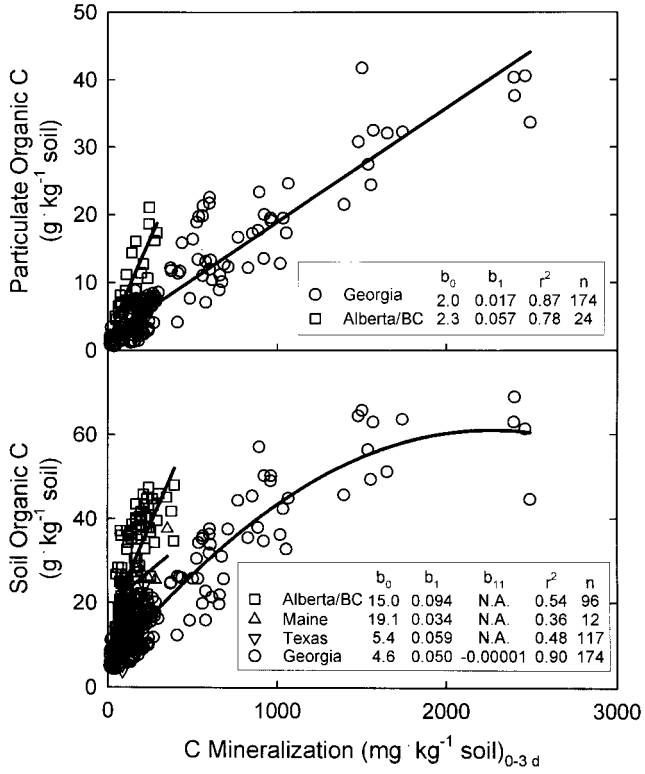

Fig. 6. Relationships of C mineralization during 0–3 d with particulate and total organic C in soils from Alberta–British Columbia, Maine, Texas, and Georgia.

ship at very high levels of $CMIN_{0\text{-}3\,d}$ was likely due to immobilization of N to meet the demands of the very active soil microbial population that developed during incubation. Although $NMIN_{0\text{-}24\,d}$ did not follow a linear relationship with $CMIN_{0\text{-}3\,d}$, potential N mineralization would likely increase with longer incubation once external nutrient demands of the active microbial population were reduced. Available data indicates that net N mineralization during the first 14 d following rewetting of dried soil is highly related ($r^2 = 0.80 \pm 0.05$) to net N mineralization during 168 to 210 d (Stanford and Smith, 1972; Smith et al., 1994; Jalil et al., 1996).

As a rapid test, $CMIN_{0\text{-}3\,d}$ was as good a predictor of $NMIN_{0\text{-}24\,d}$ as it was for SMBC. Predicted values of $NMIN_{0\text{-}24\,d}$ using the pooled relationship in Fig. 5 were within ±25% of actual values 45% of the time. Seventy-seven percent of the predictions were within ±50% of actual values. The deviation from actual values was 47% or less for 75% of the predictions.

Relationship Between the Flush of Carbon Dioxide and More Resistant Organic Carbon Pools

Relationships of POC and SOC with $CMIN_{0\text{-}3\,d}$ were weaker than those of $CMIN_{0\text{-}24\,d}$ and SMBC with $CMIN_{0\text{-}3\,d}$ in several data sets (Table 2). Soils from Alberta–British Columbia had greater slopes of both POC on $CMIN_{0\text{-}3\,d}$ and SOC on $CMIN_{0\text{-}3\,d}$ than soils from Georgia. The lower flush of activity following rewetting per unit of organic component in Alberta–British Columbia than in Georgia suggests that a greater proportion of biologically resistant (or intermediately available) C was present in a colder and drier climate. Pooling data resulted in significant, but more variable, relationships between POC and $CMIN_{0\text{-}3\,d}$ and between SOC and $CMIN_{0\text{-}3\,d}$ (Fig. 6). As an independent test of the strength of relationships, comparison of r^2 values among the 30 data sets in Table 2 followed this order: $CMIN_{0\text{-}24\,d}/CMIN_{0\text{-}3\,d} > SMBC/CMIN_{0\text{-}3\,d} > POC/CMIN_{0\text{-}3\,d} = SOC/CMIN_{0\text{-}3\,d}$ ($P \leq 0.01$; paired t-tests). General weakening of relationships with $CMIN_{0\text{-}3\,d}$ from active ($CMIN_{0\text{-}24\,d}$ and SMBC) to passive (POC) to total (SOC) organic-matter pools suggests that $CMIN_{0\text{-}3\,d}$ may be the most descriptive of the biologically active pools of soil organic matter.

Separation of Regional Differences in Relationships

Relationships between active soil C and N pools and the flush of CO_2 varied among regions and could be partly attributed to mean annual temperature and precipitation differences among regions (Table 3). Data analyzed in Table 3 were from whole soils for all regions, which excluded the aggregate and particulate fractions from Alberta–British Columbia. Further, only observations with $CMIN_{0\text{-}3\,d} < 500$ mg kg^{-1} soil were analyzed to eliminate any bias due to large values above this limit that occurred only in some highly C-enriched surface

soils from pastures in Georgia. The flush of CO_2 during 3 d alone was an excellent predictor of $CMIN_{0\text{-}24\,d}$, explaining 80% of the variation. An additional 9% of variation in $CMIN_{0\text{-}24\,d}$ was explained by a precipitation regime (Table 3), in which soils under wetter regimes had a greater flush of CO_2 relative to a particular level of $CMIN_{0\text{-}24\,d}$ (i.e., the slope of $CMIN_{0\text{-}24\,d}$ on $CMIN_{0\text{-}3\,d}$ was lower). At least a part of this difference in slope between precipitation regimes might be attributable to the fact that soils in Alberta–British Columbia and Texas were either air-dried or dried at 40°C, while soils in Maine and Georgia were dried at 55°C. Further work is needed to clarify how much of this difference was climatically related and how much was due to pretreatment of soil.

The relationship between SMBC and $CMIN_{0\text{-}3\,d}$ was greatly improved by accounting for precipitation differences among the regions (23% of variability), but less so by considering temperature differences (3% of variability) (Table 3). Less of the microbial biomass was represented in the flush of CO_2 following rewetting of dried soil sampled from drier as opposed to wetter precipitation regimes and also under hotter as opposed to colder temperature regimes. Again, further work is needed to determine the extent of climate vs. soil handling (drying temperature) on the observed mean annual precipitation effect. However, the effect of mean annual temperature was without complication and suggests less of the microbial biomass was expressed in the flush of CO_2 during 3 d in hotter than colder climates, regardless of precipitation regime.

A significant interaction occurred among regions separated by temperature and precipitation regimes in the relationship between $NMIN_{0\text{-}24\,d}$ and $CMIN_{0\text{-}3\,d}$ (Table 3). Soils from Alberta–British Columbia mineralized much less N per $CMIN_{0\text{-}3\,d}$ than from other regions, which appears to have been due to greater immobilization of N, perhaps as a result of a large pool of semi-decomposed, intermediately resistant organic matter that may have a great affinity for sequestering N. Ratios of $CMIN_{0\text{-}24\,d}/NMIN_{0\text{-}24\,d}$ averaged 23 from soils in Alberta–British Columbia, 8 from soils in Maine, 10 from soils in Texas, and 11 from soils in Georgia. Further research is needed to understand these differences among regions.

The relationship between SOC and $CMIN_{0\text{-}3\,d}$ was improved more by accounting for temperature differences among regions (29% of variability) than precipitation differences (4% of variability) (Table 3). The strong temperature dependence of $SOC/CMIN_{0\text{-}3\,d}$ contrasted with the strong precipitation dependence of $CMIN_{0\text{-}24\,d}/CMIN_{0\text{-}3\,d}$ and $SMBC/CMIN_{0\text{-}3\,d}$. Increasing temperature resulted in a lower ratio of $SOC/CMIN_{0\text{-}3\,d}$, suggesting that soils in warmer climates have a greater portion of SOC composed of rapidly mineralizable C. Powlson and Jenkinson (1976) also reported that tropical soils from Nigeria contained a greater labile fraction of SOC than did temperate soils from England.

The close relationship observed between the flush of CO_2 during the first 3 d following rewetting of dried soil and active pools of soil C and N probably reflects both (i) microbial population dynamics, including the death of microorganisms due to drying (Sorensen, 1974), the death of microorganisms due to osmotic shock following rewetting with water (Kieft et al., 1987), and a flush of growth from surviving microorganisms on lysed metabolites (Jenkinson, 1966), and (ii) part of the steady-state rate of C mineralization that reflects the quality of organic matter. Chemical and physical disturbances of soil organic matter have also been proposed as mechanisms for increasing the flush of CO_2 from drying and rewetting (van Gestel et al., 1991), although these are probably of smaller magnitude. As an extreme example, severe physical disturbance (i.e., grinding to a powder) exposes organic matter otherwise protected by macroaggregates and releases a rapidly mineralizable fraction of soil organic matter that may account for 0.1 to 0.7% of SOC (Beare et al., 1994; Franzluebbers and Arshad, 1997a). Breaking soil aggregates into increasingly smaller units (i.e., from intact cores to sieving <0.5 mm) resulted in increasingly greater flushes of CO_2 per unit of $CMIN_{0\text{-}24\,d}$, SMBC, and $NMIN_{0\text{-}24\,d}$ (Franzluebbers, 1999b). However, relationships between these active soil C and N pools and $CMIN_{0\text{-}3\,d}$ were equally strong at all handling pretreatments and suggested that as long as standardized laboratory techniques were used, the flush of CO_2 would be able to predict active soil biological properties across a wide range of soils within a region (Franzluebbers, 1999b).

Sensitivity of the Flush of Carbon Dioxide to Soil Management

If $CMIN_{0\text{-}3\,d}$ were the only soil biological property measured, then what would be the consequence for biological assessment of soil quality? Soil C and N data from Alberta–British Columbia and Georgia (Table 2) represent six experiments that were used to test the sensitivity of soil biological properties to changes in management. The flush of CO_2 during 3 d detected as many significant differences among management variables as $CMIN_{0\text{-}24\,d}$, SMBC, $NMIN_{0\text{-}24\,d}$, POC, and SOC. To further evaluate the sensitivity of the flush of CO_2 to management, we compared the probability of obtaining greater *F*-values among a priori orthogonal contrasts of management effects within soil depth increments, and then subjected *F*-values to analysis with pair-wise *t* tests. Sensitivity of soil properties to management varied among experiments. For example, in a cattle-grazing study in Georgia sampled in 1996 under bermudagrass (A. Franzluebbers, unpublished data, 1998), SMBC was more sensitive (*t* test $P \leq 0.1$, $n = 42$) to differences in forage management (hayed, unhayed, and low and high grazing pressure) and N fertilization (inorganic, clover + inorganic, and broiler litter) than all other indices of soil biological potential examined. The flush of CO_2 during 3 d was similar in sensitivity to $CMIN_{0\text{-}24\,d}$, but more sensitive than POC and SOC. In the same cattle-grazing study sampled in 1997, $CMIN_{0\text{-}3\,d}$ was as sensitive to management as $CMIN_{0\text{-}24\,d}$, SMBC, and SOC ($n = 42$), but was more sensitive than POC. In a tillage study under cereal cropping in Al-

berta–British Columbia (Franzluebbers and Arshad, 1996a, 1996b), $CMIN_{0\text{-}3\,d}$ was as sensitive to differences in tillage management (conventional and zero) as $CMIN_{0\text{-}24\,d}$, SMBC, $NMIN_{0\text{-}24\,d}$, POC, and SOC ($n = 12$). The flush of CO_2 during 3 d was as sensitive to management as all other soil properties in (i) a cattle-grazing study under tall fescue (*Festuca arundinacea* Schreb.) ($n = 16$) differentiated by endophyte infection (low and high) and fertilization regime (low and high) in Georgia (Franzluebbers et al., 1999b), (ii) a tillage study under cotton (*Gossypium hirsutum* L.) ($n = 19$) differentiated by tillage management (conventional and minimal) in Georgia (Franzluebbers et al., 1999a), and (iii) a tillage study evaluating water-stable aggregate fractions in Alberta–British Columbia ($n = 36$) differentiated by tillage management (conventional and zero) (Franzluebbers and Arshad, 1997a).

When the *F*-value comparisons among these six experiments were combined ($n = 166$, except $n = 130$ with POC), $CMIN_{0\text{-}3\,d}$, $CMIN_{0\text{-}24\,d}$, and SMBC were equally sensitive to management. Particulate organic C and SOC were also equally sensitive to management, although less sensitive than $CMIN_{0\text{-}3\,d}$, $CMIN_{0\text{-}24\,d}$, and SMBC. Coefficients of variation were similar among $CMIN_{0\text{-}3\,d}$ (23% ± 15%), $CMIN_{0\text{-}24\,d}$ (22% ± 11%), and SMBC (19% ± 10%) ($n = 29$).

The flush of CO_2 during 3 d following rewetting of dried soil was also sensitive to the increase in microbial activity associated with wheat (*Triticum aestivum* L.) root development in several cropping systems in Texas (Franzluebbers et al., 1995, 1996b). The quantity of $CMIN_{0\text{-}3\,d}$ increased during the wheat growing season until maximum vegetative growth (i.e., 20–30% greater at flowering than at planting), closely mimicking the temporal variation in basal-soil respiration and SMBC. In addition, immediately following sorghum [*Sorghum bicolor* (L.) Moench] or soybean [*Glycine max* (L.) Merr.] residue incorporation, $CMIN_{0\text{-}3\,d}$ was ≈20% greater than 30 to 60 d later when readily mineralizable substrates had disappeared. Similar to the measurement of SMBC, close attention should be given to the time of sampling for the flush of CO_2, to separate long-term from short-term variations. For assessment of long-term effects, sampling in winter or spring for summer crops, or as late as possible following residue addition, is recommended to minimize the influence of short-term effects caused by plant additions that contain transient, readily mineralizable components.

SUMMARY AND CONCLUSIONS

Strong relationships between $CMIN_{0\text{-}3\,d}$ and $CMIN_{0\text{-}24\,d}$, SMBC, $NMIN_{0\text{-}24\,d}$, POC, and SOC were observed in several data sets from Alberta–British Columbia, Maine, Texas, and Georgia. The relationship between SMBC and $CMIN_{0\text{-}3\,d}$ was strongly influenced by gross regional differences in mean annual temperature and precipitation. Therefore, as a predictive tool, $CMIN_{0\text{-}3\,d}$ may need to be adjusted, depending upon the macroclimatic region, to provide estimates of SMBC comparable across regions. Smaller regional influences occurred between $CMIN_{0\text{-}24\,d}$ and $CMIN_{0\text{-}3\,d}$ and between $NMIN_{0\text{-}24\,d}$ and $CMIN_{0\text{-}3\,d}$. The relationship between $CMIN_{0\text{-}3\,d}$ and $NMIN_{0\text{-}24\,d}$ was not as strong as between $CMIN_{0\text{-}3\,d}$ and other active organic pools, in part, perhaps, because of variable N immobilization among soils. As a response variable, $CMIN_{0\text{-}3\,d}$ was (i) as sensitive to tillage, forage, and fertilization management effects as $CMIN_{0\text{-}24\,d}$, SMBC, and $NMIN_{0\text{-}24\,d}$, (ii) slightly more sensitive than POC, and (iii) more sensitive than SOC. Although measuring a suite of soil biological properties would be advantageous if unlimited resources were available, our results indicate that measurement of $CMIN_{0\text{-}3\,d}$ could provide an indication of biological soil quality if resources and time were limited. Like other active organic-matter pools, the higher the flush of CO_2, the higher would be the biological soil quality. The flush of CO_2 following rewetting of dried soil arguably makes an excellent biological soil quality indicator because it (i) reflects soil microbial biomass and potential activity, (ii) shows a prompt and accurate response to management, (iii) integrates physical, chemical, and biological conditions of soil during incubation, (iv) appears to be broadly applicable across soil texture and management systems, with only minor modifications to relationships that may be due to climatic conditions, and (v) would be readily and economically accessible to a wide range of users. Measurement of $CMIN_{0\text{-}3\,d}$ might be most appropriate (i) as a soil-testing tool where time of analysis is a critical factor, (ii) in spatial assessments of soil quality that require sampling from many points, and (iii) in integrated natural-resource assessments, where a key indicator from each of the biological, chemical, and physical components of the soil is needed to avoid collecting excessive data.

ACKNOWLEDGMENTS

We appreciate the contributions of Mr. A. David Lovell, Mr. Steven W. Knapp, and Ms. Georgette Trusty in the field and laboratory.

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