Changes in Soil Physical Properties Due to Organic Waste Applications: A Review[1]

R. KHALEEL, K. R. REDDY, AND M. R. OVERCASH[2]

ABSTRACT

Land application of organic wastes such as animal manure, municipal wastes, and sewage sludge could alter the soil physical properties. Repeated substantial applications of waste increase the soil organic matter percentage. The available data on effects of waste applications on soil physical properties such as bulk density, water holding capacity at both field capacity and wilting point, and saturated hydraulic conductivity were summarized. Based on data from 12 different sources, 21 soil types, 7 waste types, and 8 crop types, a linear regression analysis of observed increases in soil organic C as a result of waste applications on percent reduction in bulk density indicated a highly significant relationship (r^2 = 0.69). The results of an exponential multiple regression analysis of percentage sand and increase in organic C percentage on the percent increase in water holding capacity indicated that approximately 80% of the observed variations in percent increases in water holding capacity, at both field capacity and wilting point, could be attributed to variations in soil texture and soil organic C increases. The data on hydraulic conductivity as well as on infiltration rates are very limited and are not sufficient for quantitative analyses. The limitations of the available data were discussed in terms of identifying future research needs.**

Additional Index Words: **bulk density, organic matter, water holding capacity, hydraulic conductivity.**

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[2] Assistant Professor of Hydrology, New Mexico Inst. of Mining and Technol., Socorro, N. Mex.; Assistant Professor, AREC, Univ. of Florida, Sanford; Associate Professor, Dep. of Biol. and Agric. Eng., North Carolina State Univ., Raleigh; respectively.

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The soil has substantial capacity to treat and assimilate organic wastes. Applying wastes to land at agronomic rates for plant nutrient supply has been the traditional means of waste management. In recent years, there has been an increasing interest in disposal of wastes at rates far in excess of traditional agronomic rates. Application of wastes, either for plant nutrient supply or for disposal purposes, increases the C content of the soil. An increase in C content of the soil increases aggregation, decreases bulk density, increases water holding capacity, and hydraulic conductivity (e.g., Biswas and Khosla, 1971; Gupta et al., 1977; Kladivko and Nelson, 1979; Klute and Jacob, 1949; Mays et al., 1973; Salter and Haworth, 1961; Tiarks et al., 1974; Unger and Stewart, 1974; Volk and Ullery, 1973; Webber, 1978; Weil and Kroontje, 1979; Williams and Cooke, 1961). Furthermore, organic matter and soil aggregation are inversely related to runoff volumes and sediment loss (Wischmeier and Mannering, 1965; 1969). Several investigators, in monitoring runoff water quality from small plot-sized land application areas, reported less runoff volumes from these plots compared to control plots which received no wastes (Hensler et al., 1970; Long, 1979; McCaskey et al., 1971; Young, 1974; Young and Mutchler, 1976). The low runoff losses may be due to improved soil physical properties as a result of waste applications. Because of increased aggregation, less ero-

Table 1—Pertinent information on various experiments.

Location	Waste type	Percent C content of waste	Years of study	Soil type	Crops grown	Waste application rates metric tons $ha^{-1} yr^{-1}$	Depth of incorporation, cm	Method of incorporation	Properties studied	Remarks	Reference
Elk River, Minnesota	Anaerobic sludge	14.0	2	Hubbard coarse sand (Udorthentic Haploborolls) 90.1% sand	Vegetable crop	0, 112, 225, 450	15	Rototill	Organic carbon (C), bulk density (BD), field capacity (FC), wilting point (WP), hydraulic conductivity (HC)	One application each year, all measurements at the end of study period	Gupta et al. (1977)
Lafayette, Indiana	Anaerobic sludge	19.4	1	Celina silt loam (Aquic Hapludalfs), 22.5% sand	None	0, 22.4, 56, 89.6	15	Rototill	C,BD,FC,WP	91 × 152 cm² plots, single application, all measurements at the end of study period	Kladivko & Nelson (1979)
				Celina silt loam	None	0, 56	5	Disk	C,BD,FC,WP		
				Blount silt loam (Aeric Ochraqualf) 21.1% sand	None	0, 56	15	Rototill	C,BD,FC,WP		
				Tracy sandy loam (Ultic Hapludalf) 58% sand	None	0, 56	15	Rototill	C,BD,FC,WP		
Mead, Nebraska	Cattle feedlot	32.0	2	Sharpsburg silt loam (Typic Argiudoll)	Sorghum (*Sorghum bicolor* L.)	0, 90, 200, 415	20	Disk	C,BD,HC	5 × 19 m² plots, single application each year, all measurements at the end of 2 years	Tiarks et al. (1974)
Bushland, Texas	Cattle feedlot	32.0†	4	Pullman clay loam (Torrertic Paleustolls), 32% sand	Sorghum	0, 22, 67, 134, 268	20	Moldboard plowing	C,BD,FC,WP	Single application each year, all measurements at the end of 4 years	Unger & Stewart (1974)
Guelph, Ontario	Solid domestic waste	37.0	1	Guelph loam (Typic Hapludalph), 32% sand	Corn (*Zea mays* L.)	0, 188	30	--	C,BD,FC,WP	7.6 × 6.4 m² plots, single application, all measurements at the end of 5 years	Webber (1978)
Muscle Shoals, Alabama	Municipal compost	26.8–34.2	2	Sango silt loam, 27% sand	Sorghum	0, 23, 41, 82, 164	--	Fall applications plowed under, spring applications disked in	C,BD,FC	3.6 × 9 m² plots, single application each year, all measurements at the end of 2 years	Mays et al. (1973)
Wooster, Ohio	Beef cattle	31.6	4	Wooster silt loam (Typic Fragiudalf), 8.8% sand; Celina silt loam (Aquic Hapludalf), 11.3% sand; Hoytville silty clay (Mollic Ochraqualf), 16.3% sand	Corn	0, 49, 158, 316	20	–	C	60 cm diam. by 90 cm deep lysimeter plots, one-time applications, measurements at the end of 1, 2, 3, & 4 years	Haghiri et al. (1978)
Orange, Virginia	Poultry manure	36.7‡	5	Davidson clay loam (Rhodic Paleudult), 31% sand	Corn	0, 27, 56, 85, 110	10–20	Fall applications plowed under using moldboard plowing (20 cm); spring applications disked in (10 cm)	C,BD,FC,WP	5 × 6.7 m² plots, two applications (one-half in spring, one-half in fall) per year, all measurements at the end of 5 years, FC & WP data only at 0 and 110 metric tons $ha^{-1}yr^{-1}$ loading rates	Weil & Kroontje (1979)

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sion has also been reported (Kelling et al., 1977; Kladivko, 1977;[3] Young, 1974).

Changes in the soil-waste system and the relationship to surface runoff and accompanying erosion are influencing factors in control of nonpoint source pollution from land areas receiving organic wastes. Transport of potential pollutants such as N, P, and O-demanding compounds (BOD and COD) in runoff may occur due to waste organic matter and runoff of soluble nutrients originating from the waste. A unified understanding of changes in the soil-waste system due to organic waste applications is, therefore, essential from a surface water quality standpoint.

[3] E. J. Kladivko. 1977. Changes in erodibility and physical properties of three soils resulting from application of sewage sludge. Unpublished M.S. Thesis. Purdue Univ., West Lafayette, Ind.

Reviews describing changes in soil physical properties due to waste applications are available (Azevedo and Stout, 1974; Moore et al., 1977; Powers et al., 1975). However, documentation of soil property changes in these reviews has been limited to such qualitative aspects as net improvement or reduction in crop yields. The objective of this study is to summarize the available experimental data on effects of organic waste applications on soil physical properties. Attempts are also made to relate changes in soil physical properties with net observed increases in soil organic C as a result of waste applications. The soil physical properties considered are: bulk density (BD), water holding capacity (WHC) both at field capacity (approximately 1/3 bar) and wilting point (15-bar), and saturated hydraulic conductivity (HC). These properties are critical in determining the hy-

Table 1—Continued.

Location	Waste type	Percent C content of waste	Years of study	Soil type	Crops grown	Waste application rates metric tons $ha^{-1} yr^{-1}$	Depth of incorporation, cm	Method of incorporation	Properties studied	Remarks	Reference
Fort Collins, Colorado	Sewage sludge	--	1	Nunn clay loam (Aridic Argiustolls), 30.4% sand	Corn & wheat (*Triticum aestivum* L.)	0, 34, 68	--	--	BD,FC,WP	All measurements at the end of 1 year	Ohiri (1977)
	Beef cattle	--	1		Corn & wheat	0, 112, 448	--	--	BD,FC,WP		
Auburn, Alabama	Dairy cattle	47.0§	3	Norfolk sandy loam (Typic Paleudults)	Millet (*Bennisetum americanum* L.) & rye (*Secale cereale* L.)	0, 45	15	Rototill	C	0.04-ha plots, one-time spring applications each year, all measurements at the end of 3 years	Long et al. (1975)
Huntsville, Alabama	Dairy cattle	47.0§	3	Decatur silty clay loam (Rhodic Paleudults), 22% sand	Millet & rye	0, 22, 44, 89, 178, 267	15	Rototill	C	2.7 × 2.7 m² plots, one time spring applications each year, all measurements at the end of 3 years	Mugwira (1976)
Woodstown, Maryland	Anaerobic sewage sludge	23.5	1	Woodstown silt loam (Aquic Hapludult)	Corn	0, 40, 240	--	Rototill	C	6 × 6 m² plots, one-time application, measurements at the end of 1½ years	Epstein et al. (1976)
	Sludge compost	13.3				0, 40, 240			C		
Wellesbourne, Warwick, England	Farmyard manure	23.0¶	6	Sandy loam (81% sand)	--	0, 45	15–18	Shallow ploughing (15–18 cm) and rotavating (15–18 cm)	C,BD,FC,WP	Measurements are at a 7.6-cm depth, all measurements at the end of 6 years	Salter & Haworth (1961)
New Haven, Connecticut	New Haven digested sewage sludge	29.0	--	Cheshire loam	Vegetable crop	0, 41, 82#	--	--	C,BD,FC	Outdoor soil-frame experiments, one-time application, measurements at the end of 2 years	Lunt (1959)
	Torrington sludge	28.0	--	Cheshire loam	Vegetable crop	0, 65, 130#	--	--	C,BD,FC		
Boardman, Oregon	Shredded municipal waste	48.5	1	Sagehill sandy loam	Fescue & alfalfa	0, 125, 250, 500	8	Rototill	BD,FC,WP	6.1 × 36.6 m² plots, measurements at the end of 1 year	Volk & Ullery (1973)
Riverhead, New York	Horse manure	--	25	Sassafras silt loam, 27% sand	Vegetable crop	0, 22, 45, 90	--	--	C,BD,FC,WP	6.1 × 14.8 m² plots, one-time application each year, all treatments including control received fertilizer applications, all measurements at the end of 25 years	Klute and Jacob (1949)
Harpenden, Herts, England	Farmyard Manure	23.0¶	85	Rothamsted clay loam, 47% sand	Mangolds (*Beta vulgaris* L.)	0, 31	--	--	C,BD,AWC	Control treatment under fallow for many years, measurements at the end of 85 years	Williams & Cooke (1961)
			18	Woburn sandy loam, 88% sand	Vegetable crop	0, 67	--	--	C,BD,AWC	Control treatment received only residues of cereals (roots & stubble), measurements at the end of 18 years	
Nasirpur India	Farmyard manure	23.0¶	14	Loam, alluvial	Senji	0, 11.6	15	--	C,BD,FC,WP, HC	All measurements at the end of study period	Biswas & Khosla (1971)
Gurdaspur, India			18	Silt loam, alluvial	Wheat	0, 20.4	15	--	C,BD,FC,WP, HC		
Bhubaneswar, India			15	Loam, lateritic	Rice	0, 45 kg N/ha	15	--	C,BD,FC,WP, HC		
Sabour, India			30	Loam, alluvial	Wheat	0, 74.6	15	--	C,BD,FC,WP HC		
Ranchi, India			14	Silty clay loam, red	Maize	0, 45 kg N/ha	15	--	C,BD,FC,WP, HC		
Poona, India			19	Clay, black	Fallow	0, 45.8	15	--	C,BD,FC,WP, HC		
Braunschweig, W. Germany	Dairy manure	47.0††	50	--	--	0, 30.0	--	--	C	Measurements at the end of study period	Ruther & Ansorge (1959)

† Mathers and Stewart (1970).
‡ R. R. Weil, Agronomy Dept., Univ. of Maryland, personal communication.
§ F. L. Long, USDA-SEA, Auburn Univ., Auburn, AL personal communication.
¶ Salter and Schollenberger (1939).
# One-time applications.
†† Assumed.

drology of runoff phenomena, and therefore, in influencing nonpoint source contributions of pollutants like N, P, and O-demanding compounds in runoff water from areas receiving organic wastes.

NATURE OF AVAILABLE DATA

Table 1 contains the pertinent information on various experiments. The list includes studies which utilized solid wastes only. Liquid wastes generally have a lower organic C content compared to solid wastes;

Table 2—Soil physical properties at the end of indicated study periods and as affected by various waste applications (numbers in parentheses refer to those for control treatments).

Reference	Carbon application rate	Study period	Net increase in soil C	Bulk density	WHC at field capacity	WHC at wilting point	Available water capacity	Saturated hydraulic conductivity	Soil type
	metric tons C ha^{-1} yr^{-1}	years	%	g/cm^3	—	% by weight	—	cm/hour	
Kaldivko and	4.3	1	0.63	1.27 (1.35)	19.8 (18.0)	6.3 (5.8)	13.5 (12.2)		Celina sl
Nelson (1979)	10.9†		1.07	1.29	20.3	6.8	13.5		
	17.4		1.14	1.21	20.8	6.0	14.8		
	10.9‡		1.27	1.09 (1.36)	24.4 (17.8)	7.5 (5.5)	16.9 (12.4)		
	10.9		1.26	1.03 (1.18)	25.5 (21.5)	10.4 (9.3)	15.1 (12.2)		Blount sl
	10.9		0.56	1.13 (1.23)	16.9 (14.4)	6.4 (5.5)	10.5 (8.9)		Tracy sal
Gupta et al.	15.7	2	0.44	1.37 (1.43)	7.0 (5.3)	5.5 (3.8)	3.0 (2.6)	115.2(108)	Hubbard cs
(1977)	31.5		1.30	1.24	9.7	9.2	2.3	122.4	
	63.0		2.69	1.03	18.0	16.8	3.9	127.2	
Tiarks et al.	28.8	2	0.50	1.00 (1.02)				13.0 (4.0)	Sharpsburg sl
(1974)	64.0		1.05	1.00				11.5	
	132.8		2.55	0.85				24.0	
Unger and	7.0	4	0.42	1.33 (1.37)	28.6§(28.0)	18.9 (18.2)	9.7 (9.8)		Pullman cl
Stewart	21.4		0.69	1.28	29.2	18.7	10.5		
(1974)	42.9		0.80	1.20	30.3	19.5	10.8		
	85.8		0.68	1.12	32.3	19.3	13.0		
Webber (1978)	69.6	1	0.59	1.25 (1.38)	23.2 (21.7)	11.2 (10.9)	12.0 (10.8)		Guelph l
Mays et al.	6.2¶	2	0.21¶	1.31¶(1.37)					Sango sl
(1973)	11.7		0.25	1.27					
	23.5¶		0.63¶	1.24¶	13.0 (11.1)				
	46.8		1.56	1.12	15.3				
Weil and	9.9	5	1.57#	0.97††(1.11)					Davidson cl
Kroontje	20.6		1.90	0.94					
(1979)	31.2		3.47	0.80					
	40.4		3.69	0.78	38.5 (30.5)	26.0 (23.0)	12.5 (7.5)	51.0 (27.0)	
Salter and Haworth (1961)	10.4	6	0.36‡‡	1.65‡‡(1.82)	16.0‡‡(11.9)	6.2‡‡(5.6)	9.8 (6.3)		sl

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hence the positive effects of liquid wastes on soil physical properties are lower compared to solid wastes (DeTar, 1977; Hinesly et al., 1978; Hinrichs et al., 1973;[4] Wallingford et al., 1974). Also, all experiments reported in Table 1 were field studies. Compared to field experiments, less decomposition of waste material occurs under laboratory conditions so that a larger fraction of organic C is present to alter the soil physical properties (Bouyoucos, 1939; Hafez, 1974; Volk and Ullery, 1973). The location of the research is included in Table 1 so that the climatic region can be identified. Precipitation amounts and temperature are important variables that affect the mineralization of organic C present in the wastes.

Although most experiments reported in Table 1 are short-term studies (1- to 6-year study period), several researchers studied long-term effects of wastes on soil physical properties. Examples are: the 85-year study at Rothamsted Experiment Station, England (Williams and Cooke, 1961); the 50-year study at Braunschweig, West Germany (Rüther and Ansorge, 1959, cited by Tietjen and Hart, 1969); the 30-year study in India (Biswas and Khosla, 1971); and the 25-year study in New York (Klute and Jacob, 1949). All studies reported in Table 1 do not contain data on changes in all desired soil physical properties; C, BD, WHC, and HC. The long-term experiment in West Germany (Rüther and Ansorge, 1959) and several short-term studies (Epstein et al., 1976; Haghiri et al., 1978; Long et al., 1975; Mugwira, 1976) include data on changes in soil organic C only. Therefore, these studies could not be utilized to develop relationships between changes in soil physical properties and net increases in C in soils as a result of waste applications. Several other limitations of the available data are discussed later.

TOTAL ORGANIC CARBON

The effect of solid wastes on soil physical properties largely depends on the rate of decomposition of wastes and its contribution to soil organic C. Factors affecting the rate of decomposition include (i) chemical composition of the waste (i.e., C content, C/N ratio); (ii) soil temperature, (iii) soil moisture; (iv) method of waste application, i.e., surface-applied or soil-incorporated; and (v) rate of application.

Available data on net increases in soil organic C as a result of waste applications are summarized in Table 2. The data on short-term experiments (1- to 6-year study period) indicate increases in C, especially at high loading rates. The long-term experiments (18–85 years), which primarily utilized farmyard manure, also indicate C increases, but to a lesser degree (Biswas and Khosla, 1971; Klute and Jacob, 1949; Rüther and Ansorge, 1959; Williams and Cooke, 1961). However, the waste application rates for long-term experiments are based essentially on crop utilization rates; therefore, these rates are generally lower compared to those for disposal purposes. It is difficult to measure any significant changes in soil physical properties when rates are low and primarily based on agronomic considerations.

The data (Table 2) represent a broad range of climatic conditions which affect the mineralization of organic C present in the wastes. The climatic influences are particularly apparent from the studies of Haghiri et al. (1978) in Ohio and Unger and Stewart (1974) in Texas (Table 2). The data by Haghiri et al. (1978) indicate relatively high increases in C content, especially at higher waste loading rates, compared to those of Unger and Stewart (1974). Carbon increase at the Ohio site ranged from 0.05 to 2.4% at the end of 1 year for C application rates ranging from 15.5 to 99.9 metric tons of C/ha. The effect of soil type on organic C decom-

[4] D. G. Hinrichs, A. P. Mazurak, N. P. Swanson, and L. N. Mielke. 1973. Effect of effluent from beef feedlots on physical properties of soil. Agron. Abstr. p. 124.

Table 2—Continued.

Reference	Carbon application rate	Study period	Net increase in soil C	Bulk density	WHC at field capacity	WHC at wilting point	Available water capacity	Saturated hydraulic conductivity	Soil type
	metric tons C $ha^{-1} yr^{-1}$	years	%	g/cm^3	% by weight	% by weight	% by weight	cm/hour	
Haghiri et al. (1978)	15.5	1	0.05						Wooster sl
	49.9		0.78						
	99.9		1.81						
	15.5		0.19						Celina sl
	49.9		0.60						
	99.9		2.00						
	15.5		0.31						Hoytville scl
	49.9		1.33						
	99.9		2.36						
Lunt (1959)	11.9§§	2	0.28	1.09 (1.13)	25.1 (24.3)				Cheshire l
	23.8		0.57	1.07	25.8				
	18.2		0.57	1.05	27.2				
	36.4		1.12	0.99	29.9				
Long et al. (1975)	21.2	3	0.42						Norfolk sal
Mugwira (1976)	10.3	3	0.35						Decatur scl
	20.7		0.64						
	41.8		1.16						
	83.6		2.56						
	125.5		4.65						
Epstein et al. (1976)	9.4	1	0.20¶¶						Woodstown sl
	56.4		0.47						
Williams and Cooke (1961)	7.1	85	1.37	1.04 (1.19)			73.0 (50.0)		Rothamsted cl
	15.4	18	0.51	1.38 (1.45)			46.0 (40.0)		Woburn sal
Rüther and Ansorge (1959)	14.2##	50	0.28						
Klute and Jacob (1949)		25	0.15	1.24 (1.30)	18.0†††(17.1)	6.0 (5.7)	12.0 (11.4)		Sassafras sl
			0.56	1.19	17.7	5.7	12.0		
			1.21	1.09	20.1	6.8	13.3		
Biswas and Khosla (1971)	2.7	14	0.09	1.61 (1.67)	12.6 (12.4)	4.6 (4.6)	8.0 (7.8)	1.12 (0.79)	l
	4.7	18	0.03	1.63 (1.68)	18.2 (17.4)	4.6 (4.6)	13.6 (12.8)	0.46 (0.39)	sl
		15	0.17	1.31 (1.53)	14.2 (10.7)	5.8 (3.2)	8.4 (7.5)	2.16 (1.68)	l
	17.2	30	2.29	1.29 (1.47)	30.5 (21.3)	11.8 (5.6)	18.7 (15.7)	0.47 (0.43)	l
		14	0.11	1.30 (1.37)	25.2 (22.6)	12.6 (10.9)	12.6 (11.7)	0.50 (0.33)	scl
	10.5	19	0.10	1.18 (1.26)	45.9 (45.3)	27.1 (27.0)	18.8 (18.3)	0.10 (0.06)	c
		37	0.27	1.14 (1.21)	55.6 (54.6)	35.2 (34.8)	20.4 (19.8)	0.03 (0.01)	c

† Rototill method of incorporation.
‡ Disk method of incorporation.
§ WHC @ 0.2-bar.
¶ Average of 2 treatments with the same loading rate but receiving different annual N rates. Control treatment is also average of 2 treatments receiving different annual N rates.
# Organic C values for all loading rates estimated from BD measurements and the regression equation BD = 1.23 − 0.052 (OM) (Weil and Kroontje, 1979).
†† Numbers are means of observations at 3 different times in the year.
‡‡ Numbers are means of 2 treatments—shallow ploughing (15–18 cm) and rotavating (15–18 cm).
§§ All applications are one-time applications in metric tons of C/ha.
¶¶ Estimated using CEC data.
Applied every other year.
††† Based on moisture equivalent.

position is also apparent from this study which utilized three soils (Table 2). The decomposition of waste was higher in the Celina and Wooster soils than in the Hoytville soil. Also, at a low loading rate (15.5 metric tons of C/ha), a greater proportion of all the decomposition that took place occurred during the first year, while for higher loading rates (49.9 and 99.9 metric tons of C/ha) the proportions of the total decomposition which occurred during the same period were not as high.

The effect of method of waste incorporation on C accumulation in soil is evident from Kladivko and Nelson's (1979) data. These data yielded net increases in C contents of about 0.63–1.27% at the end of 1 year for C application rates ranging from only 4.3 to 17.4 metric tons of C/ha. Two methods of sludge incorporation were used; the sludge was either disked-in (depth of incorporation 5 cm) or incorporated by rototilling (depth of incorporation 15 cm). In both cases, the organic C content was reported only for the top 5 cm of the soil, which is considerably different from the 15- to 30-cm depths for other available data (Table 1). The disked-in sludge treatment resulted in the greatest increase of the organic C content because of the higher concentration of the sludge in the top 5 cm of the soil. However, due to uneven distribution during the rototilling procedure, there was also a much higher concentration of sludge for the rototilled plots in the top 5 cm of the soil than in the lower depths (Kladivko and Nelson, 1979).

Attempts to obtain any significant relationships between C application rates through waste applications and net observed C increases in soil were not successful because of the large differences in C increases among various experiments (Table 2). The experiments represent different study periods and nearly all studies report increases in C at the end of their study periods; therefore, the changes in C on a year-to-year basis during the period of an individual experiment is not known. The study of Gupta et al. (1977), which includes data on decomposition of organic C, indicates that decomposition is not linear with time. The increase in C in their study in the first year was 67%, which was reduced to 50% in the second year. This nonlinearity in decomposition further complicates the problem of expressing the data on net C increases in soil as a function of C application rates.

In summary, relationships between net C increases and C application rates are difficult to establish under differing experimental conditions of climate; soil, vegetation, and waste types; and methods of waste incorporation. However, as shown in the following sections, direct relationships can be established for changes in bulk density and water holding capacity as a function of net increases in soil C.

BULK DENSITY

Considerable information is available concerning the effects of organic matter, such as animal wastes or sewage sludge, on soil bulk density (Table 2). Both long-term studies (Biswas and Khosla, 1971; Klute and Jacob, 1949; Williams and Cooke, 1961) and short-term studies (Gupta et al., 1977; Kladivko and Nelson, 1979; Lunt, 1959; Mays et al., 1973; Salter and Haworth, 1961; Tiarks et al., 1974; Unger and Stewart, 1974; Webber, 1978; Weil and Kroontje, 1979) indicate a decrease in BD with waste applications.

The decrease in BD as a result of waste applications is due to a dilution effect resulting from the mixing of the added organic matter (OM) with the more dense mineral fraction of the soil (Powers et al., 1975). Among the various soil types, the effect of waste incorporation on BD appears to be more pronounced for coarse-textured soils (Table 2). The addition of 450 metric tons/ha per year of anaerobically digested sewage sludge for 2 consecutive years decreased the BD of Hubbard coarse sand by 28% (Gupta et al., 1977); the bulk density decreased linearly with addition of OM in the sewage sludge. The available data (Table 2) indicate that compared to other wastes, poultry wastes produced a maximum reduction in BD. At applications of 110 metric tons/ha per year for 5 years to a Davidson clay loam soil, the BD was reduced from 1.11 g/cm^3 to 0.78 g/cm^3, which is almost a 30% reduction (Weil and Kroontje, 1979).

Bulk density data of Table 2 were used to derive a linear regression equation between observed increases in soil organic C due to waste incorporation (independent variable) and the percent reduction in BD (dependent variable). The resulting equation given in Table 3 is based on data from 12 different sources which report changes in both soil organic C and BD as a result of waste applications; 21 soil types ranging in texture from clay loam to coarse sand; 7 waste types; and 8 crop types. Among the studies which reported the particle size distribution of soils, the sand percentage varied from 21 to 90%, silt from 8 to 67%, and clay from 4 to 30%. The BD for control soils (without waste) ranged from 1.02 to 1.82 g/cm^3. In spite of considerable variation in soil as well as in waste and crop types, a significant linear relationship is indicated between observed increases in soil organic carbon due to waste applications and percent reductions in bulk density (Table 3). The percent reduction in BD was used to partially offset the large differences in bulk density inherent with data from various soil types. The intercept and the slope of the regression equation (Table 3) were found to be significantly different from zero, using a standard t-test ($p \leq 0.01$). The experimental data for the BD equation in Table 3 came from studies that ranged from 1 to 85 years in length.

Table 3—Regression equations for changes in soil physical properties as a result of waste applications.

Property	Equation	r^2, R^2	No. of Observations n
Bulk density	$\Delta BD† = 3.99 + 6.62\,(\Delta C‡)$	0.69**	42
Water holding capacity at field capacity	$\Delta FC§ = \exp[1.09 + 2.141\,(\Delta C) - 0.4091\,(\Delta C)^2 - 0.0167\,(SAND)¶ + 0.00038\,(SAND)^2]$	0.81**	21
Water holding capacity at wilting point	$\Delta WP\# = \exp[1.115 + 2.248\,(\Delta C) - 0.442\,(\Delta C)^2 - 0.0443\,(SAND) + 0.0007\,(SAND)^2]$	0.79**	19

† ΔBD = [Waste-incorporated soil bulk density (BD) − control soil BD]/(control soil BD) × 100.
‡ ΔC = [Waste-incorporated soil organic carbon (C) − control (without waste) soil C].
§ ΔFC = % increase in water holding capacity at field capacity (FC) = (waste-incorporated soil FC − control soil FC)/(control soil FC) × 100.
¶ SAND = % sand present in soil.
# ΔWP = % increase in water holding capacity at wilting point (WP).

WATER HOLDING CAPACITY

Several researchers (Gupta et al., 1977; Kladivko and Nelson, 1979; Klute and Jacob, 1949; Mays et al., 1973; Unger and Stewart, 1974; Webber, 1978; Weil and Kroontje, 1979) have reported increased water holding capacity (WHC) on a weight basis at both field capacity and wilting point, with an increase in waste application rates. Water holding capacity of soils is controlled primarily by: (i) the number of pores and pore-size distribution of soils; and (ii) the specific surface area of soils. Because of increased aggregation, total pore space is increased (Kladivko and Nelson, 1979; Tiarks et al., 1974; Volk and Ullery, 1973; Williams and Cooke, 1961). Furthermore, as a result of decreased bulk density, the pore-size distribution is altered and the relative number of small pores increases, especially for coarse-textured soils (Volk and Ullery, 1973). Since the tension which causes a particular pore to drain is dependent on the effective diameter of the pore, greater tension is required to drain small pores compared to large pores.

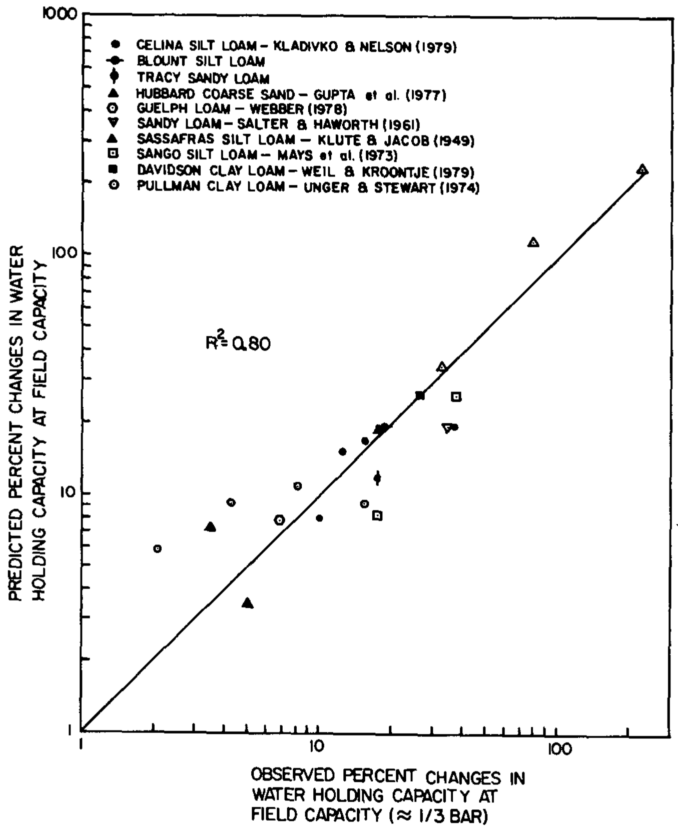

Fig. 1—Observed and predicted percent changes in water holding capacity (weight basis) at field capacity (various authors, various soils, various wastes).

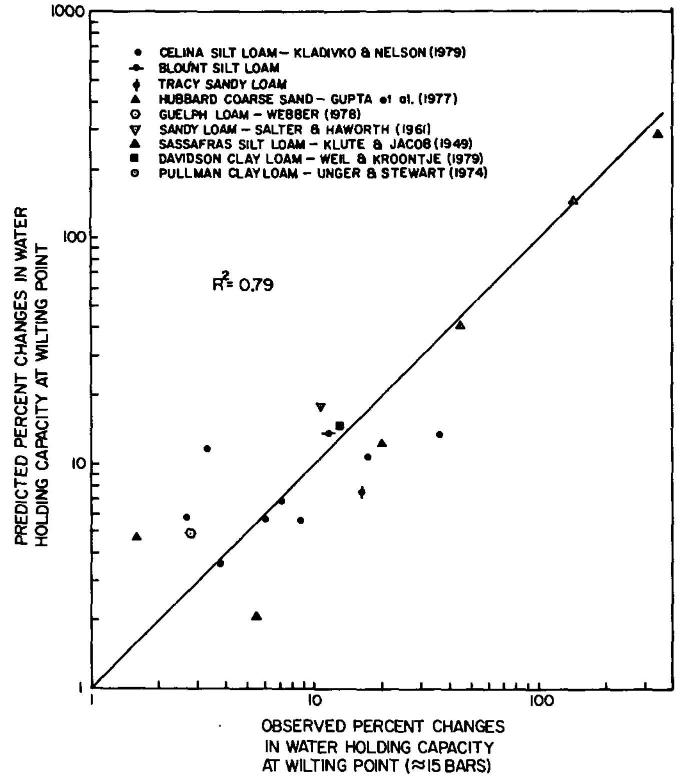

Fig. 2—Observed and predicted percent changes in water holding capacity (weight basis) at wilting point (various authors, various soils, various wastes).

The increased WHC at lower tensions such as those at field capacity is primarily the result of an increase in number of small pores.

At higher tensions close to wilting range, nearly all pores are filled with air and the moisture content is determined largely by the specific surface area and the thickness of water films on these surfaces. Sandy soils have much less surface area than clayey soils and, thus, retain much less water at higher tensions. However, with the addition of organic matter, specific surface area increases resulting in increased WHC at higher tensions (Gupta et al., 1977; Volk and Ullery, 1973).

The available data on changes in WHC (weight basis), at both field capacity and wilting point, as a result of waste applications are summarized in Table 2. Increases in WHC, both at field capacity and wilting point, are indicated for both fine-textured and coarse-textured soils. However, increases for coarse-textured soils (e.g., Gupta et al., 1977) are larger than those for fine-textured soils (e.g., Unger and Stewart, 1974). Such differential increases in WHC among soil textural classes are also evident from laboratory data of Bouyoucos (1939), who studied the effects of organic matter additions to soils having a broad range of textures.

In view of variations in textures among soils studied by various investigators, exponential multiple regression analyses were run of percentage sand (independent variable) and increase in organic C (independent variable) on the percent increase in WHC (dependent variable). The resulting regression equations are given in Table 3.

Approximately 80% of the observed variations in percent increases in WHC, at both field capacity and wilting point, can be explained by soil texture and variations in soil organic C increases. Using a standard F-test (Draper and Smith, 1966, p. 117), the intercept and the coefficients of the multiple regression equations (Table 3) were found to be significantly different from zero at 0.05 level of significance.

Not all available data on changes in WHC at field capacity and wilting point could be utilized in the analyses. Only those studies which reported data on particle size distribution (sand percentage), changes in organic C and changes in WHC, as a result of waste applications, were used. These references are listed in Fig. 1 and 2, which also indicate the fit between observed and predicted changes using the regression equations.

The effects of waste organic matter additions on water holding capacity can be summarized as:

1) Water holding capacity (weight basis) at both field capacity and witling point increases with waste organic matter additions, but increases vary with soil texture. The results of an exponential multiple regression analysis (Table 3) indicate that for fine-textured soils, increase in WHC at field capacity is greater than at the wilting point. For coarse-textured soils, the regression equations indicate that percentage sand present in the soils produces a larger increase in WHC at wilting point than at the field capacity.

2) If increases in organic C content cause an increase in moisture content at both the field capacity and the wilting point, the net result is that the amount of avail-

able water or available water capacity (AWC) may not be greatly affected, since AWC is defined as difference between moisture contents at field capacity and wilting point. Furthermore, an increase in soil organic C results in a decrease in soil bulk density. The decreased bulk density of the waste-incorporated soil tends to counterbalance any increased AWC on a weight basis, resulting in only small increases on a volume basis.

HYDRAULIC CONDUCTIVITY

Because of increased porosity with waste organic matter additions, the saturated hydraulic conductivity (HC) is expected to increase. The limited data available (Biswas and Khosla, 1971; Gupta et al., 1977; Tiarks, et al., 1974; Weil and Kroontje, 1979) on increases in HC as a result of waste applications are summarized in Table 2. The data indicate extreme variations in HC increases with waste applications. The saturated hydraulic conductivity for a coarse sand increased by only 18% over that of the control soil, for the 450 metric tons/ha per year sewage sludge treatment for 2 years (Gupta et al., 1977); whereas for a silt loam soil, the increase was up to 500% over control, for feedlot manure applied at 415 metric ton ha^{-1} yr^{-1} for 2 years (Tiarks et al., 1974). The HC data in individual studies were also extremely variable; Tiarks et al. (1974) found that after 2 years of manure application, the HC ranged from 0.2 to 52 cm/hour. Because of limited data on HC, unlike other soil properties, no quantitative analyses were done for the effect of organic C additions on increases in HC.

RESEARCH NEEDS

Standardization of data is needed so that research results from different locations can be compared. In reporting effects of organic waste applications on soil physical properties, data on characterization of the waste being studied (i.e., waste C content), soil particle size distribution (i.e., % sand, silt, and clay), climatic variables such as average annual precipitation and temperature of the location, should be included. In several cases, although data were available on changes in soil physical properties with waste application, these could not be utilized to develop WHC relationships (Table 3) since either no data were given on changes in soil organic C with waste applications (Ohiri, 1977)[5] or no information was available on soil texture (Biswas and Khosla, 1971; Lunt, 1959; Volk and Ullery, 1973).

Most of the data were collected after 1 or 2 years of waste application, with little or no information on the changes of various properties over time to that point. Limited data available on soil organic C accumulation over time suggest that organic C decomposition is not linear with time (Gupta et al., 1977; Haghiri et al., 1978). Mathers and Stewart (1970) found that under well-aerated conditions in the laboratory, almost 50% of the waste organic C evolved as CO_2 in 90 days. Under field conditions, the data of Haghiri et al. (1978) indicate that at low rates of application (49 metric tons/ha) the waste decomposes and becomes somewhat stabilized over a relatively short time, whereas at high rates the stability of the animal waste tends to occur over a longer period, which in their case was 3 years. Additional field studies are needed to verify these results under a broad range of climatic conditions, and soil and waste types.

Presently, no mathematical algorithms are available to describe and estimate the fate of added C in the soil-waste system. However, several C transformation models are available to describe the decomposition of plant residues (e.g., Hanna, 1975; Hunt, 1977; Jenkinson and Rayner, 1977; Paul and Van Veen, 1978). Similar models are needed to describe the decomposition of organic waste as a function of time. How do the kinetic rate constants for added C and native C decomposition vary as a function of time for different organic waste applications? How can one describe the dependence of these rate constants on soil moisture, soil temperature, and method of waste application?

This review indicated a relatively larger amount of information on changes in bulk density and water holding capacity due to waste organic matter additions, but very little on infiltration rate and hydraulic conductivity. Limited available data suggest that waste applications may improve both the initial infiltration rate (e.g., Cross and Fischbach, 1972) and the steady-state infiltration rate (e.g., Kladivko, 1977;[3] Mazurak et al., 1975; Smith et al., 1937; Swader and Stewart, 1972; Zwerman et al., 1970). However, negative effects of waste applications on infiltration rates have also been reported (Weil and Kroontje, 1979). At low to medium application rates, Manges et al. (1974) reported increases in steady-state infiltration rate; however at higher application rates, build up of Na^+ and K^+ was sufficient to decrease intake rates. Several other investigators have conducted studies showing that when wastes contain high concentrations of Na, soil aggregates become dispersed. This reduces the movement of water into the soil surface and through the soil matrix (Hinrichs et al., 1973;[4] Powers et al., 1975; Travis et al., 1971). Additional field studies are needed to evaluate the effects of waste applications on infiltration rates and hydraulic conductivity under a broad range of conditions.

Finally, limited data available so far have shown that if wastes are incorporated, sludge or manure organic matter may tend to stabilize soils against erosion by improving aggregation (Kelling et al., 1977; Kladivko, 1977;[3] Young, 1974). Consequently, there will be little loss of waste constituents in surface runoff. Furthermore, several investigators (e.g., Hensler et al., 1970; McCaskey et al., 1971; Young, 1974; Young and Mutchler, 1976) have reported less runoff volumes from waste-incorporated plots. The changes in the soil-waste system as a result of waste applications and the relationship to surface runoff and erosion-sediment transport should be further investigated.

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