# Linking soil and water quality in conservation agricultural systems

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## ABSTRACT

Conservation agricultural systems (i.e., conservation tillage, cover crops, and pastures) that have high surface-soil organic C are highly effective in improving surface soil properties and processes, thus reducing water runoff and soil erosion and improving water quality. Literature was reviewed to document the important linkage that surface soil organic matter has on soil and water quality. Soil organic matter is a key property that drives many important soil functions, e.g. supplying and cycling of nutrients; infiltrating, filtering, and storing water; sequestering C from the atmosphere; and decomposing organic matter and xenobiotics. Stratification of soil organic matter with depth under various conservation agricultural systems was shown to influence water runoff volume and quality in studies across small plots, fields, and water catchments. Soil organic matter stratification with depth buffers soil and water quality against "normal" perturbations in agricultural systems. Perturbations of concern still remain with excessively high nutrient applications from fertilizer and manure inputs that can cause leaching of nitrate to groundwater and runoff of dissolved P to surface water bodies. To meet the human nutritional needs of the rapidly expanding global population while sustaining our invaluable natural resources, a multidisciplinary approach is needed to develop, characterize, and implement alternative, highly productive management systems that also conserve soil and water resources for the future.


**Keywords:** conservation tillage; nitrogen; pasture management; phosphorus; soil organic matter; water runoff

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## INTRODUCTION

Soil and water resources are fundamental components of agriculture. Achieving a balance between agricultural production and conservation of natural resources is a necessary goal for development of sustainable agricultural systems. During the past century, agriculture in the USA has shifted focus several times from production-oriented to conservation-oriented approaches. The expansion of tillage-intensive agriculture throughout the Great Plains combined with the drought of the 1930s instigated a major conservation movement that placed emphasis on soil conservation. The chemical-intensive agriculture that developed to overcome the low food supply during World War II eventually resulted in increasing water pollution, placing greater emphasis on water quality protection. During the latter decades of the 20th century, a variety of factors led to the development of conservation-tillage systems, which conserve soil water, use less energy, and reduce soil erosion. Some of these factors included increasing competition for water between municipal-industrial-agricultural sectors, the energy crisis of the 1970s that magnified machinery operating expense, and on-going, wide-spread soil erosion that required abatement. Recently, high oil prices and the motivation to develop biofuels from agricultural products have caused another production-oriented stimulus. A counter balance to this biofuel production emphasis has been to raise awareness of the potential soil and water degradation that could ensue (Wilhelm et al., 2004; Simpson et al., 2008).

My objectives were to summarize relevant literature on soil and water quality and to demonstrate the strong linkage that surface soil organic matter has on these two natural resources. Data were summarized to illustrate the inherently strong influence that surface soil condition has on water runoff, soil erosion, and nutrient loss, irrespective of whether studies were conducted at the plot, field, or water catchment scales. Future research efforts to better characterize the linkage between soil and water quality are recommended so that highly productive systems that conserve soil and water resources can be developed and implemented immediately to meet the expanding needs of society.

### Soil Quality

Soil quality is indirectly linked to food production, food security, and environmental quality (e.g., water quality, global warming, and energy use in food production) through its influence on key soil functions. Unfortunately, moderate to severe degradation of soils (i.e., loss of soil biodiversity, poor soil tilth, and unbalanced elemental composition) has occurred with the adoption of modern, industrialized agricultural production systems. Environmental consequences of agricultural industrialization were not immediately apparent, but are now being recognized as a result of (1) fossil-fuel based mechanization that has led to soil erosion from vast areas of insufficiently covered soil surfaces, (2) agricultural enterprise specialization with concentration of animal feeding operations that expose nearby water resources to excessive fecal-borne pathogen and nutrient loads, and (3) liberal application of synthetic plant protection and nutrient amendments that threaten soil and water quality. Reports on the state of agricultural land in America suggest that soil sediment, nutrients, and organic matter have been lost at rates far exceeding a sustainable level, the result of which has had enormous direct and indirect consequences on the profitability, productivity, and environmental quality of agriculture (NRC, 1993; USDA-NRCS, 1996).

Scientific assessment of soil quality is needed to help evaluate the sustainability of agricultural systems. Soil quality is a complex subject, encompassing the many valuable services humans derive from soil, as well as the many ways soils impact terrestrial ecosystems. Different definitions of soil quality have been proposed, each reflecting a different perspective on the use and value of soils:

- potential utility of soils in landscapes resulting from the natural combination of soil chemical, physical, and biological attributes (Johnson et al., 1992);
- capability of soil to produce safe and nutritious crops in a sustained manner over the long-term, and to enhance human and animal health, without impairing the natural resource base or harming the environment (Parr et al., 1992);
- capacity of soil to function within ecosystem boundaries to sustain biological productivity, maintain environmental quality, and promote plant and animal health (Doran and Parkin, 1994);
- capacity of soil to function (Karlen et al., 1997); and
- how well soil does what we want it to do (Schjønning et al., 2003).

Most definitions link soil quality to some defined use of soil. The use-dependent nature of soil quality definitions has created some debate among scientists about the concept of soil quality and ignited questions of how soil quality should be subjected to rigorous scientific investigation (Sojka and Upchurch, 1999). Notwithstanding, the concept of soil quality is intended as a scientific tool for land managers to adaptively manage soil resources for sustainable future use (Andrews and Moorman, 2002). Curiously, the impetus to define and assess soil quality has occurred primarily from outside the scientific community, because of societal concerns for the health of the environment (Carter, 2002). Since soil quality emphasizes maintenance or improvement in the natural resource base, it has become an integral component of sustainable agriculture (Miller and Wali, 1995; Warkentin, 1995). Some key papers describing soil quality and its determination have been presented during the past two decades (Larson and Pierce, 1991; Haberern, 1992; Papendick and Parr, 1992; Doran and Parkin, 1994; Karlen et al., 1997; Doran et al., 1999; Wander et al., 2002; Andrews et al., 2004; Wienhold et al., 2006).

Soil quality assessment distinguishes between static and dynamic soil properties. Static soil properties reflect the inherent characteristics of a particular site, e.g. soil texture, mineralogy, and classification, all of which are influenced by geologic history and climatic conditions. In addition, topography, hydrology, and climate are factors that affect productivity and environmental quality of a site, somewhat independent of management. Static soil properties have been adequately characterized in North America with regional sampling approaches by the USDA's Natural Resources Conservation Service through the periodic National Resources Inventory (Brejda et al., 2000). Similar efforts have been conducted by Agriculture and Agri-Food Canada (MacDonald et al., 1995). Static soil properties provide the contextual background for how soil management practices might eventually alter dynamic soil properties.

Dynamic soil properties are those properties that can change value over relatively short time periods (e.g., months, years, and decades). Dynamic soil properties are at the leading edge of soil quality assessment, because they change quickly, and oftentimes dramatically, in response to management. They can indicate whether a farm uses agronomically and ecologically sustainable practices.

Changes in soil properties with time are a key component of dynamic soil quality assessment. Sustainable cropping systems will improve soil quality; often brought about through diverse crop rotations, minimal use of tillage for weed control and seedbed preparation, and addition of organic amendments like animal manures, crop residues, and compost. Management systems that cause a decline in soil quality indicators with time will lead to low soil quality; often induced by cropping systems with low residue production, intensive tillage, and near monoculture cultivation.

Two concepts are helpful is assessing changes in soil quality – the resistance of soils to degradation and the resiliency of soils to recover following a period of declining soil quality. Resistance of soil to degradation can be assessed by determining the extent of change in dynamic soil quality indicators (e.g., soil organic matter content, rate of water infiltration, soil biological activity, etc.), such as during a period of intensive tillage with little organic matter addition. Low resistance of a soil property to disturbance might induce a permanent and damaging change in soil functional capabilities. High resistance to disturbance is a positive attribute, reflected in strong functional capabilities that are supported by a range of management approaches. Resilience of soil is another desirable soil characteristic that can be assessed by determining how fast a dynamic soil quality indicator rebounds from a period of poor management.

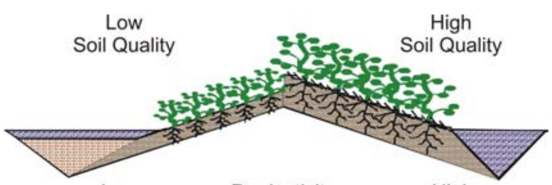
Soil quality indicators are often divided into three main classes: (1) chemical, (2) physical, and (3) biological. Within each of these classes, a variety of soil properties or processes can be selected to indicate soil functional capabilities. Currently, most commercial soil testing laboratories offer a variety of soil tests to determine soil physical and chemical properties, but few have tests for rapid and reliable determination of soil biological activity and condition. Particularly in organic agriculture, soil biological properties and processes are of great importance, since nutrients are derived from microbial decomposition of various fractions of organic matter. The structure and function of highly active soil microbial communities may also impart plant protection mechanisms to ward off diseases and create less stressful conditions for plant growth.

A minimum dataset approach for assessing soil quality should have the following characteristics (Doran and Parkin, 1994; Soil Quality Institute, 2008):

- easy to measure
- detect changes in soil function
- integrate soil physical, chemical, and biological properties and processes
- accessible to many users and applicable to field conditions
- sensitive to variations in management and climate
- encompass ecosystem processes and relate to process-oriented modeling
- where possible, be components of existing soil data bases

Achieving high soil quality requires that soil be able to perform several key ecosystem functions to an optimum capacity within the constraints of inherent soil characteristics and climatic conditions. Some key soil functions of interest in agriculture are:

- supplying and cycling nutrients for optimum plant growth;
- receiving rainfall and storing water for root utilization;
- filtering water passing through soil to protect groundwater quality;




| Low | Productivity | High |
|---|---|---|
| ? | Healthy Food Supply | Yes |
| Low | Environmental Protection | High |
| Low | Land Stewardship | High |
| Low | Soil Organic Matter | High |
| High | Erosion / Sedimentation | Low |
| Low | Soil Microbial Activity | High |
| Low | Biological Diversity | High |
| Poor | Water Quality | Clean |
| Poor | Air Quality | Clean |
| ? | Human Condition | Healthy |

Fig. 1. Diagrammatic representation of soil quality and its potential impacts on the environment and people. High-quality soil is able to produce abundant plant materials, which feed, clothe, and provide shelter to humans. Plant residues not consumed must be returned to the soil to feed soil organisms and provide the organic nutrients for creating a biologically active food web. High-quality soil protects the environment from degradation by reducing soil erosion and nutrient runoff (i.e., water quality protection) and by storing carbon in soil and reducing greenhouse gas emissions. Low-quality soil lacks sufficient organic matter to sustain productivity in the long term, leads to excessive soil erosion and poor water quality, has low soil biological activity and diversity, and could lead to an unhealthy food supply and human condition.

- storing soil organic C for nutrient accumulation and mitigating greenhouse gas emission; and
- decomposing organic matter and xenobiotics to avoid detrimental exposures to plants and the environment.

Soil organic matter – as a source of energy, substrate, and biological diversity – is one of the key attributes of soil quality that is vital to many of these soil functions. Stratification of soil organic matter with depth is common in many natural ecosystems, managed grasslands and forests, and conservation-tilled cropland (Franzluebbers et al., 2000; Blanco-Canqui et al., 2006; Jinbo et al., 2006). The soil surface is the vital interface that receives much of the fertilizer and pesticides applied to cropland and pastures, receives the intense impact of rainfall that can lead to surface sealing following disruption of surface aggregates, and partitions the flux of gases into and out of soil. Franzluebbers (2002a) described a soil quality evaluation protocol that related the degree of soil organic matter stratification to soil quality or soil ecosystem functioning through its conceptual relationship to erosion control, water infiltration, and conservation of nutrients (Fig. 1).

Stratification of soil organic matter with time occurs when soils remain undisturbed from tillage (e.g., with conservation tillage and pastures) and sufficient organic materials are supplied to the soil surface (e.g., with cover crops, sod rotations, and diversified cropping systems). On a Fluventic Ustochrept in Texas, increasing cropping intensity and management with conservation tillage resulted in greater stratification of different organic matter factions (Franzluebbers, 2002a). On reclaimed minesoils in Ohio, stratification ratio of soil organic C (0-15 cm / 15-30 cm depth) increased with time under pasture and forest management (Fig. 2). During pasture development in Georgia, stratification ratio of soil organic C (0-15 cm / 15-30 cm depth) increased from 2.4 at initiation to 3.0 $\pm$ 0.7 at the end of 5 years to 3.6 $\pm$ 0.6 at the end of 12 years (Franzluebbers and Stuedemann, 2005; 2008).

Soil organic matter is a critical component of soil quality. Accumulation of surface residues and organic matter at the soil surface are especially beneficial to soil quality, because of their positive effects on conserving water, preserving nutrients, and creating a suitable habitat for soil biological diversity. Conservation agricultural systems often lead to high surface soil organic matter, and therefore, high soil quality.

## Water Quality

The Clean Water Act of 1972 (http://www.epa.gov/r5water/cwa.htm) initiated a vigorous national effort to control water pollution in the USA. The first national assessment of the Water Resources Council (1968) helped to draw attention, not only to the scarcity of water in some regions, but also to the continued pollution of water from (1) waste discharge derived from domestic, industrial, and agricultural uses,

(2) sediment and other diffused wastes in runoff from urban, industrial, and agricultural lands, (3) logging, and (4) transportation system developments. Many agricultural research efforts ensued to gather fundamental knowledge about the way agriculture impacts water quality. Two notable large-scale research efforts have been the Management Systems Evaluation Areas program in Iowa, Minnesota, Missouri, Nebraska, and Ohio (Ward et al., 1994) and the Conservation Effects Assessment Project throughout the USA (Mausbach and Dedrick, 2004).

Water quality concerns from agriculture are primarily from sediment, nutrient, and pesticide runoff from cropland and fecal-borne pathogen and nutrient runoff from pastureland and livestock operations (Chesters and Schierow, 1985; Myers et al., 1985). Development and adoption of conservation-tillage systems on cropland have revealed a key fundamental linkage between soil and water quality. Surface residue cover and undisturbed soil are key factors in limiting soil and nutrient losses from cropland (Holt, 1979; Lindstrom et al., 1979), as well as from pastureland (Barnett et al., 1972; Jones et al., 1985; Harmel et al., 2004). Conservation tillage has often been shown to improve soil quality (Arshad et al., 1999; Chan and Hulugalle, 1999; Tebrügge and Düring, 1999; Hernanz et al., 2002; Cambardella et al., 2004; Liebig et al., 2004) and to reduce the quantity of nutrients in water runoff (Langdale et al., 1985; Sharpley et al., 1992). However, there is increasing evidence that, although surface residue and reduced soil disturbance effectively control sediment and total nutrient losses, enriched surface-soil nutrient concentration with surface application of fertilizers and animal manures can lead to greater loss of soluble nutrients (N and P), which have the potential to rapidly deteriorate water quality from algal blooms (Sharpley et al., 1992; Nichols et al., 1994).

The remainder of this paper will review how conservation agricultural systems (i.e., conservation tillage and pastures) lead to stratification of soil organic

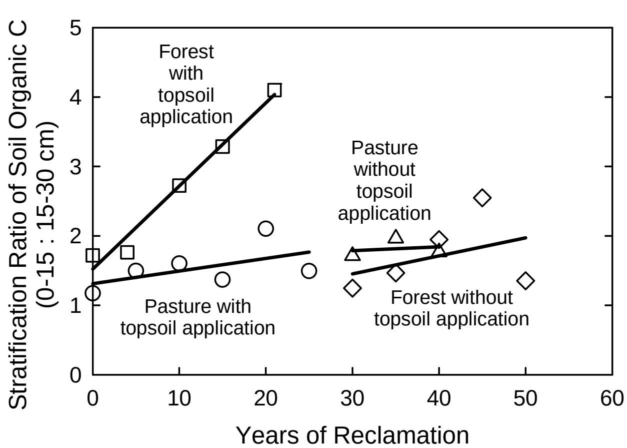



Fig. 2. Stratification ratio of soil organic C (0-15:15-30 cm) as affected by years following minespoil reclamation with pastrures and forest in Ohio. Data from Akala and Lal (2001).

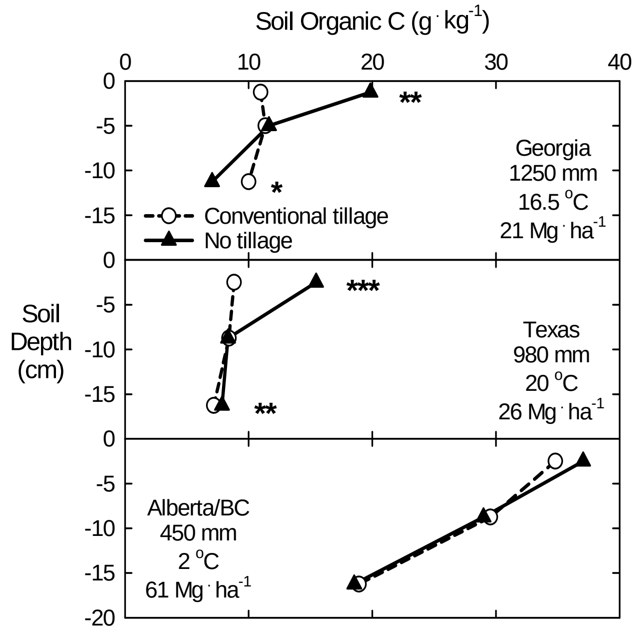



Fig. 3. Depth distribution of soil organic C under long-term, conventional- and no-tillage management systems in three different regions of North America. *, **, and *** next to symbols indicate a significant difference between means at $P \leq 0.1$, $P \leq 0.01$, and $P \leq 0.001$, respectively. Each location was characterized by mean annual precipitation (mm), mean annual temperature (˚C), and soil organic C content (Mg $ha^{-1}$). From Franzleubbers (2002a).

matter and affect water runoff volume and losses of sediment and nutrients.

## Stratification of Soil Organic Matter and Nutrient Fractions

Soil organic C under long-term conservation tillage systems is often more stratified with depth than under conventional, inversion tillage (CT) (Fig. 3). This difference develops as a result of crop residues left at the soil surface, where temperature and moisture fluctuations limit decomposition and result in accumulation of soil organic C.

Many soil organic matter fractions can become stratified with depth, including total, particulate organic, microbial biomass, and mineralizable C and N (Franzluebbers, 2002a). The degree of stratification of soil organic matter with conservation tillage depends upon (a) the inherent level of soil organic matter dictated by climatic conditions, (b) type and intensity of soil disturbance, (c) type of cropping system that determines the quantity and quality of organic C inputs, and (d) years of management. In an analysis of stratification ratios (soil organic C in the surface 5 cm divided by that at 12.5 to 20 cm depth) under no tillage (NT) in three different ecoregions, greater differences in the stratification of soil organic C between tillage systems occurred in hot-wet-low soil organic matter environments than in cold-dry-high soil organic matter environments (Fig. 3). Those soils with low inherent levels of organic

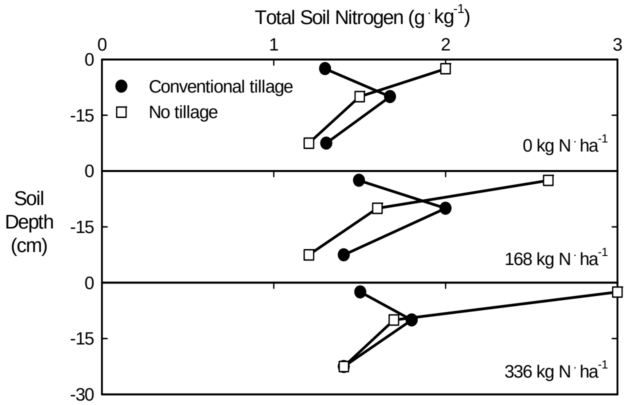



Fig. 4. Depth distribution of total soil N under conventional- and no-tillage management systems with different levels of N fertilization as $NH_4NO_3$ on a silt loam in Kentucky. Data from Ismail et al. (1994).

matter may be the most functionally improved with conservation tillage, despite modest or no change in total standing stock of soil organic C within the rooting zone. Alternatively, those soils with inherently high soil organic matter even under CT management would likely obtain relatively little additional soil functional benefit with adoption of conservation tillage, since inherent soil properties would be at a high level.

Stratification ratio of particulate organic C in a Typic Kanhapludult in Georgia was greatest in a cropping system with minimum soil disturbance and lowest in a cropping system with frequent disturbance (Franzluebbers, 2002a). Less intensive mixing of soil preserves crop residues and soil organic matter near the soil surface, where it has the most beneficial impact. Stratification of mineralizable C in a Fluventic Ustochrept in Texas increased with increasing cropping intensity under CT, but was always greater under NT than under CT, independent of cropping intensity (Franzluebbers, 2002a). More intensive cropping increases the quantity of residues produced, which can lead to higher soil organic matter.

Stratification ratio of soil organic C (0-2.5 cm / 12.5-20 cm depth) in an Aquic Hapludult in Maryland was 1.0 under plow tillage and increased with time under NT from 1.1 at 1 year, to 1.4 at 2 years, and to 1.5 at 3 years (McCarty et al., 1998). From a survey of farms in the southeastern USA, stratification ratio of soil organic C (0-5 cm / 12.5-20 cm depth) increased from an average of 1.4 under CT to a plateau of 2.8 within 10 years of adopting conservation tillage (Causarano et al., 2008). Stratification ratios of particulate organic C, microbial biomass C, and mineralizable C were also greater under conservation tillage than under CT, but ratios were even greater under pastures than under conservation tillage (Causarano et al., 2008). On a Typic Paleudalf in Kentucky, stratification ratio of soil organic C and extractable P increased with time under NT and values after 2 years of NT were always greater than under CT (Diaz-Zorita and Grove, 2002).

Stratification of total soil N at the end of 20 years of conservation-tillage management on a silt loam in Kentucky increased with increasing N fertilization (Fig. 4). Accumulation of total N at the soil surface was likely a function of greater crop production that contributed to surface residue accumulation and subsequent residue transformation into soil organic matter at the surface (Ismail et al., 1994). Total soil N was highly stratified with depth under 9 years of NT and ridge tillage on a Typic Calciustoll in Texas (Zibilske et al., 2002), as well as with 6 years of NT on an Alfisol, Andisol, and two Vertisols in Mexico (Salinas-Garcia et al., 2002). Extractable P followed similar depth distribution patterns as total N and soil organic C in both of these studies, in which values were stratified with depth under NT and uniformly distributed under CT. At the end of 23 years of NT management of a Pachic Argiustoll in Kansas, soil organic C was greater under NT than under CT within the surface 7.5 cm, but not below this depth (Guzman et al., 2006). Extractable P and K were greater under NT than under CT only at a depth of 0-2.5 cm, whereas extractable K, Ca, and Mg were lower under NT than under CT at a depth of 7.5-15 cm. On a Typic Hapludalf in Pennsylvania, soil organic matter and pH were greater under NT than under CT at a depth of 0-5 cm due to surface residue accumulation and 14.5 Mg $ha^{-1}$ of lime application during 25 years (Duiker and Beegle, 2006). Corresponding to changes in surface soil organic matter, extractable P, K, Ca, and CEC were also greater under NT than under CT at a depth of 0-5 cm. The low mobility of P in soil when surface applied appears to create an equally stratified distribution with depth as soil organic matter fractions.

During the 7$^{th}$ to 10$^{th}$ years of NT on a Mollic Cryoboralf in Alberta Canada, soil inorganic N was not different between tillage treatments, but soil microbial biomass N was greater at a depth of 0-10 cm under NT than under CT (Soon and Arshad, 2005). Level of soil microbial biomass N was positively correlated with barley and canola yield, indicating that turnover of labile N in surface soil organic matter contributed significantly to crop yield. On a Typic Natrustalf in Queensland Australia, soil organic C and total N at a depth of 0-10 cm were greater under NT than under CT at the end of 9 years of management (Thomas et al., 2007). Exchangeable K and extractable P were also greater under NT than under CT, but exchangeable Mg and Na and cation exchange capacity were lower, suggesting greater solute movement through surface soil under NT than under CT. During the first 4 years of NT management of a Calciortidic Haploxeralf in Spain, soil organic C and total N became greater with NT than with CT (Martin-Rueda et al., 2007). Extractable P, K, Fe, Mn, Cu, and Zn were also greater in the surface 30 cm of soil under NT than under CT, contributing to enhanced fertility and maintenance of yield. Similar enhancement of surface-soil nutrients was observed at the end of 8 years of NT on a Fluventic Ustochrept in Texas

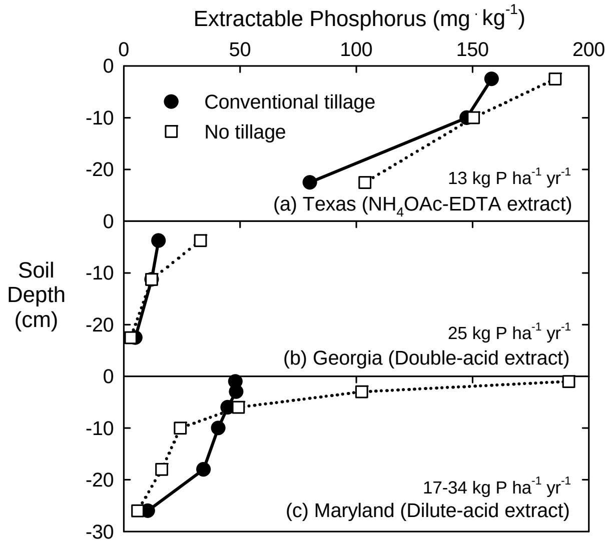



Fig. 5. Depth distribution of extractable soil P under conventional- and no-tillage management systems from (a) a silty clay loam (Fluventic Ustochrept) in Texas (Franzluebbers and Hons, 1996), (b) a sandy loam (Typic Kanhapludult) in Georgia (Hargrove et al., 1982), and (c) three silt loam soils (Hapludults) in Maryland (Weil et al., 1988). Method of P extraction is indicated in parentheses next to location.

(Franzluebbers and Hons, 1996) and at the end of 5 years of reduced tillage compared with CT in Texas (Wright et al., 2007). Greater stratification of soil nutrients under reduced tillage was associated with significantly greater cotton lint yield (Wright et al., 2007).

With the continuous application of P fertilizer (either inorganic or organic), stratification of soil P can occur in the soil profile, especially under conservation tillage. Different research organizations determine soil P with different extraction protocols, because of variations in management goals, soil mineralogy, and historical calibrations for various crops. Despite these differences, greater stratification of soil P under conservation tillage than under CT was observed at the end of 9 years in Texas (Fig. 5a), at the end of 5 years in a wheat/soybean double cropping system in Georgia (Fig. 5b), and at the end of 11 years in Maryland (Fig. 5c). At the end of 4 years of NT on a clay soil in Finland, organic C and water-extractable P were greater in the surface 5 cm of soil than under CT (Muukkonen et al., 2007). In this same report, but for a clay loam, no differences in C and P occurred between tillage systems. The accumulation of total and labile soil P at the soil surface under conservation tillage is currently viewed as a threat to water quality from dissolved nutrients in overland flow (Sharpley, 2003).

These aforementioned studies overwhelmingly support the idea that stratification of soil organic matter and its biochemical fractions occur with conservation agricultural systems. Because of this important spatial characteristic, a linkage to soil quality (nutrient cycling, hydrologic function, and organic matter decomposition)

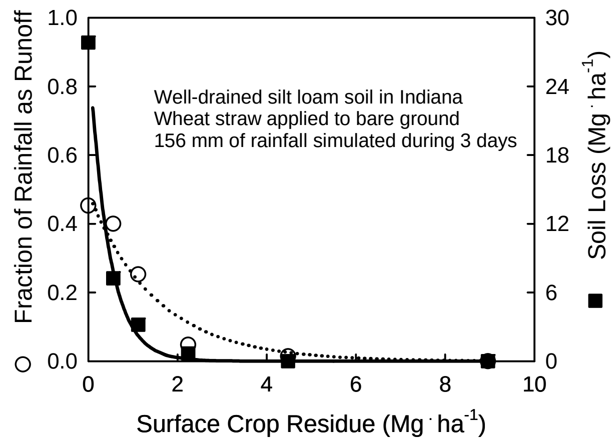



Fig. 6. Fraction of rainfall as runoff (dotted line) and soil loss via runoff (solid line) as a function of surface-placed crop residue mass immediately prior to rainfall simulation. Data from Mannering and Meyer (1963).

within the entire soil profile can be established. How stratification of organic matter in the soil profile affects water quality has not been clearly established. This linkage will be the focus of the next section.

## Water Runoff Volume and Quality

Water runoff volume and quality as affected by tillage management have been evaluated using a variety of approaches, including (1) small-plot rainfall simulations (<10 $m^2$) conducted over one to several hours, (2) instrumented field plots (<0.1 ha) with data collected over seasons or years, and (3) instrumented water catchments (>1 ha) integrated across landscape positions with data collected across a few years to decades. Each approach has advantages and disadvantages, the balance of which can help to interpret the effect of management on water runoff volume and

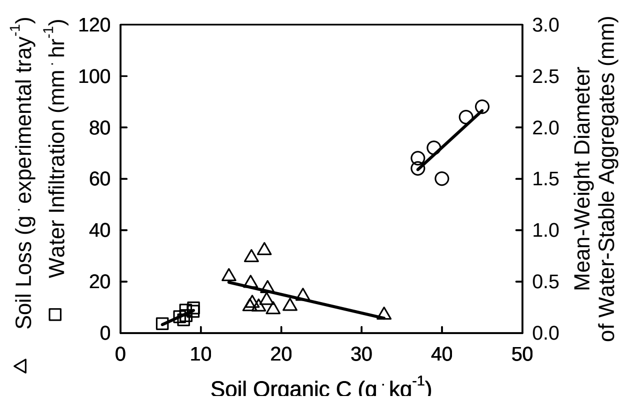



Fig. 7. Effect of soil organic C concentration on (a: Δ) soil loss from simulated rainfall on Typic Hapludalfs from Pennsylvania conducted in the laboratory (McDowell and Sharpley, 2003), (b: □) rate of water infiltration in Typic Kanhapludults in Georgia (Carreker et al., 1977), and (c: ○) mean-weight diameter of water-stable aggregates on Mollic Cryoboralfs from Alberta Canada (Arshad et al., 2004).

quality.

Water runoff volume and soil erosion have been related to a number of surface conditions and soil properties. Many changes in soil properties with adoption of conservation tillage can affect soil and water quality simultaneously. However, there are also a few issues that put soil and water quality at odds. Surface residue cover reduces surface sealing, thereby increasing water infiltration, and decreases rainfall and runoff energy so that particle detachment and transport are held in check (Fig. 6). Sufficient surface residue cover is often a prerequisite towards improving soil quality as a function of tillage. Surface soil roughness has also been shown to positively affect infiltration and reduce water runoff volume and soil loss, at least temporarily (Cogo et al., 1984), but which may eventually reduce soil quality. An increase in soil organic C with high surface residue cover and lack of inversion tillage is typically accompanied by an increase in water-stable aggregation (Bruce et al., 1995; Franzluebbers et al., 1999), especially nearest the soil surface, as well as by an increase in water infiltration (Fig. 7). Accumulation of soil organic C with concomitant increase in soil biological activity can improve soil aggregation and create macropores capable of channeling water quickly from the soil surface to the soil profile (Shipitalo et al., 2000). However, concern with rapid nutrient and pesticide transport through macropores to groundwater has been raised (Addiscott and Thomas, 2000). There is also concern with the highly stratified P distribution in pastures receiving high levels of animal manures (Sharpley, 2003) and the potential for water quality impairment as a result of dissolved P transport in overland flow (Nichols et al., 1994; Pierson et al., 2001; Pote et al., 2003).

Separation of the effects of surface residue cover from long-term surface soil condition has been investigated with mixed results. On a Typic Argiudoll managed with NT for 15 years in Illinois, removal of surface residue prior to rainfall simulation (70 mm $hr^{-1}$; 90 minute duration) resulted in a final water infiltration rate of 47 $\pm$ 8 mm $hr^{-1}$ compared with $\geq$70 mm $hr^{-1}$ with crop residue intact (Bradford and Huang, 1994). Tilling a small plot of this soil and then replacing crop residues on the surface prior to rainfall simulation resulted in infiltration rate of 64 $\pm$ 3 mm $hr^{-1}$, while a bare soil surface had an infiltration rate of 36 $\pm$ 3 mm $hr^{-1}$. On a Typic Kanhapludult managed with CT and NT for 5 years in Georgia, simulated rainfall (60 mm $hr^{-1}$) resulted in 18 $\pm$ 7% of applied water as runoff under CT with crop residues intact and 29 $\pm$ 6% under CT with crop residues removed (West et al., 1991). Under NT, applied water as runoff was 2% with crop residues intact and 7 $\pm$ 3% with crop residues removed. Surface soil properties (0- to 1.5-cm depth) were sufficiently improved under NT compared with CT that crop residue removal did not dramatically alter water runoff characteristics under NT. Soil organic C was 10 $\pm$ 3 g $kg^{-1}$ under CT and 23 $\pm$ 5 g $kg^{-1}$ under NT, while water-stable aggregation was 51 $\pm$ 6% under CT and 87 $\pm$ 1% under NT. Surface residue cover was 38 $\pm$ 15% under CT and 83 $\pm$ 5% under NT. Soil loss was 2.8 $\pm$ 1.4 Mg $ha^{-1}$ with residues and 4.5 $\pm$ 0.8 Mg $ha^{-1}$ without residues under CT and 0.4 $\pm$ 0.3 Mg $ha^{-1}$ with residues and 1.3 $\pm$ 0.6 Mg $ha^{-1}$ without residues under NT.

The presence of surface residues and high surface soil organic matter are a natural consequence of long-term conservation agricultural management, but the importance of each to water runoff control and water quality protection are not easily separated. Available data indicate that even under NT, temporary removal of surface residues can increase water runoff, probably due to less physical impedance of water moving over the soil surface. The relative importance of surface residues versus surface soil organic matter may be affected by the scale of investigation, whereby contact time of water at the soil surface may be variably controlled. The following sections separate water runoff data from the literature among (1) small-plot rainfall simulations, (2) field plots (<0.1 ha), and (3) water catchments (>1 ha).

Table 1. Mean loss of P in runoff from 1.35 $m^2$ plots during six rainfall simulation events (73 to 136 mm $hr^{-1}$) on a silt loam under conventional tillage (moldboard plow) and no tillage in a maize cropping system in Wisconsin. Data from Andraski et al. (1985).

| Tillage system | Soil organic C[a] | Extractable soil P[a] | Phosphorus loss: Total | Dissolved | Bioavailable |
|---|---|---|---|---|---|
| | Mg $ha^{-1}$ | mg $kg^{-1}$ | ------------ kg $ha^{-1}$ event$^{-1}$ ---------- | | |
| Conventional | 32.5 | 39 | 1.31 | 0.02 | 0.21 |
| No tillage | 38.3 | 62 | 0.18 | 0.01 | 0.03 |

[a] Soil properties at a depth of 0 to 2.5 cm.

### Small-Plot Rainfall Simulations

During 1-hr rainfall simulations (5.5 $m^2$) supplying 50 mm of precipitation, water runoff volume was not different between cropping systems managed for 9 years under CT (48 $\pm$ 14% of applied water as runoff) and under NT (52 $\pm$ 17% of applied water as runoff) on a Glossic Fragiudalf in Mississippi (Rhoton et al., 2002). On a Typic Hapludalf in Ohio with the same type of rainfall simulation, water runoff volume was 53 $\pm$ 9% under CT and 14% under NT (Rhoton et al., 2002). Soil loss in runoff was 2.2 $\pm$ 2.5 Mg $ha^{-1}$ under CT in Mississippi and 2.9 $\pm$ 0.8 Mg $ha^{-1}$ under CT in Ohio, but nil under NT at both locations. Stratification of soil organic C (0-3 cm / 7.6-15.2 cm) was inversely related to soil loss (r = -0.96), averaging 2.2 under CT and 3.6 under NT in Mississippi and 1.1 under CT and 3.1 under NT in Ohio. This study provided direct evidence of the

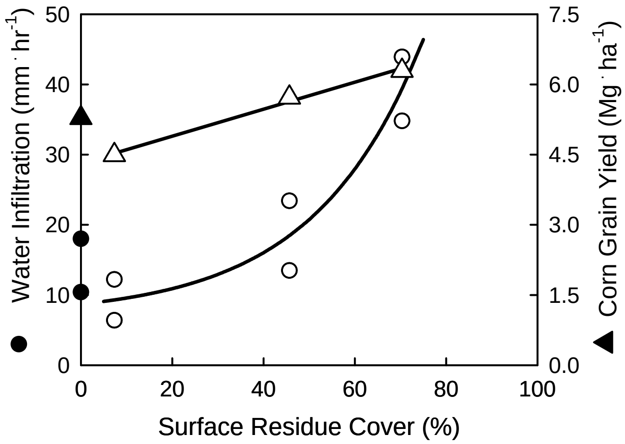



Fig. 8. Water infiltration (○) and corn grain yield (Δ) as a function of surface residue cover on Typic Fragiudalfs in Ohio. Closed symbols are under moldboard plowing and open symbols are under no tillage. Data from Triplett et al. (1968).

strong linkage between soil and water quality, as suggested by the negative relationship between stratification ratio of soil organic C and soil loss.

On a Typic Argiudoll in Wisconsin, mean soil P loss in runoff during 1-hr rainfall simulation events in summer and autumn was lower under NT than under CT (Table 1). Loss of P in runoff was statistically lower under NT than under CT in (a) 6 of 6 events for total P, (b) 2 of 6 events for dissolved P, and (c) 5 of 6 events for bioavailable P (Andraski et al., 1985). Despite extractable soil P was greater under NT than under CT near the soil surface, runoff loss of P fractions was lower under NT than under CT, and therefore, P loss under NT was mitigated by the presence of surface residue and high surface soil organic C.

At the end of 5 years of crop management on a Rhodic Paleudult in Alabama, soil loss in runoff during 2 hr of rainfall simulation on 1 m$^2$ plots in autumn and summer was 3.3 Mg ha$^{-1}$ under CT and 0.9 Mg ha$^{-1}$ under NT (Truman et al., 2003). Water runoff volume with NT (40% of applied as runoff) was not different from that with CT (45% of applied as runoff). Surface residue cover was 1.2 Mg ha$^{-1}$ under CT and 3.5 Mg ha$^{-1}$ under NT. Soil organic C at a depth of 0-6 cm was estimated as 7.2 Mg ha$^{-1}$ under CT and 8.4 Mg ha$^{-1}$ under NT. Both surface residue cover and soil organic C were negatively correlated with soil loss in runoff.

On a Typic Fragiudalf in Ohio, water infiltration during a rainfall simulation at the end of 3 years (1962-1964) of continuous corn was not different between CT and NT with normal residue production, but was greater under NT when twice the normal quantity of residue was placed on the surface (Fig. 8). Corn grain yield during these 3 years was also positively related to surface residue cover (Triplett et al., 1968). On similar silt loam soils in eastern and central Ohio, these authors reported a strong positive effect of NT versus CT on corn grain yield with increasing amount of surface residue cover.
On the same plots initiated in 1962 and maintained until 1979, water runoff volume during a 1-hr rainfall simulation (45 m$^2$; 63 mm hr$^{-1}$) was 75 $\pm$ 15% of water applied under CT and 50 $\pm$ 6% under NT, while soil loss was 9.6 $\pm$ 2.6 Mg ha$^{-1}$ yr$^{-1}$ under CT (6 $\pm$ 2% residue cover) and 0.7 $\pm$ 0.5 Mg ha$^{-1}$ yr$^{-1}$ under NT (61 $\pm$ 9% residue cover) (Van Doren et al., 1984). Stratification ratio of soil organic C (0-3 cm / 7.6-15.2 cm depth) was 1.1 under CT and 3.1 under NT in 1997 (Rhoton et al., 2002). On a similar soil in a corn-soybean rotation, water runoff volume was 68 $\pm$ 26% of water applied under CT and 81 $\pm$ 15% under NT, while soil loss was 2.5 $\pm$ 1.4 Mg ha$^{-1}$ yr$^{-1}$ under CT (34 $\pm$ 27% residue cover) and 1.4 $\pm$ 0.6 Mg ha$^{-1}$ yr$^{-1}$ under NT (73 $\pm$ 9% residue cover).

On Paleudults and Hapludults in Louisiana, bermudagrass pastures receiving different histories of poultry litter application were exposed to rainfall simulations (74 $\pm$ 8 mm hr$^{-1}$) on 3.8 m$^2$ plots during 3 events in 2 years (Gaston et al., 2003). Soil organic C stratification ratio (0-5 cm / 5-15 cm depth) in pastures varied from 2.7 to 5.6, but was unrelated to water runoff volume and nutrient loss in runoff (Table 2). The low levels of water runoff and nutrient loss were probably due to the high surface soil organic C and permanent rooting channels under perennial vegetation that fostered rapid water infiltration.

The effect of different sized rainfall simulators on sediment and P loss was compared on two soils in Pennsylvania (Typic Dystrudept and Typic Fragiudult) (Table 3). A small (2 m$^2$) simulator employed by the National Phosphorus Research Project and a larger simulator (33 m$^2$) employed by the Water Erosion Prediction Project used the same nozzles and water pressure to produce rainfall of comparable characteristics

Table 2. Surface soil conditions, water runoff, and loss of dissolved and total P from Hapludults in Louisiana. Data from Gaston et al. (2003).

| Field | Soil organic C (0-5 cm) (g kg$^{-1}$) | Stratification ratio of soil organic C (0-5:5-15 cm) | Bray-2 P (0-5 cm) (mg kg$^{-1}$) | Stratification ratio of Bray-2 P (0-5:5-15 cm) | Runoff as percentage of water applied (%) | Dissolved P in water runoff (kg ha$^{-1}$) | Total P in water runoff (kg ha$^{-1}$) |
|---|---|---|---|---|---|---|---|
| 1 | 7.9 | 3.7 | 124 | 4.6 | 24.0 | 0.17 | 0.18 |
| 2 | 9.9 | 3.4 | 286 | 3.2 | 6.4 | 0.08 | 0.08 |
| 3 | 15.9 | 2.7 | 751 | 2.7 | 4.6 | 0.17 | 0.17 |
| 4 | 18.0 | 5.6 | 1409 | 2.9 | 2.4 | 0.14 | 0.14 |

Table 3. Soil loss, dissolved P, and total P in water runoff on loam soils in Pennsylvania (Typic Dystrudepts and Typic Fragidults) as affected by size of rainfall simulator [2 $m^2$ used by National Phosphorus Research Project (NPRP) and 33 $m^2$ used by Water Erosion Prediction Project (WEPP)] and land use (grass, CT, conventional tillage, and NT, no tillage). Data from Sharpley and Kleinman (2003).

| Land use | Soil organic C (0-5 cm) (g $kg^{-1}$) | Mehlich-3 P (0-5 cm) (mg $kg^{-1}$) | Soil loss (kg $ha^{-1}$) | | Dissolved P loss (kg $ha^{-1}$) | | Total P loss (kg $ha^{-1}$) | |
|---|---|---|---|---|---|---|---|---|
| | | | NPRP | WEPP | NPRP | WEPP | NPRP | WEPP |
| CT | 13.7 | 320 | 946 | 767 | 0.06 | 0.02 | 0.70 | 0.52 |
| NT | 25.2 | 330 | 469 | 312 | 0.08 | 0.03 | 0.39 | 0.27 |
| Grass | 16.6 | 400 | 184 | 104 | 0.12 | 0.03 | 0.41 | 0.19 |

(75 mm $hr^{-1}$) (Sharpley and Kleinman, 2003). Both simulators produced similar effects on soil loss (grass < NT < tilled), dissolved P in runoff (few differences among land uses), and total P loss in runoff (grass = NT < tilled). The authors concluded that limited use of small plots and rainfall simulators could be used to assess the positive relationship between soil-test P and loss of dissolved P in runoff.

Lack of soil disturbance in grazed pastures and abundant application of animal manure to meet N requirements can lead to highly stratified depth distribution of soil P (Sharpley, 2003). To assess the water quality impacts of inverting surface-P-enriched soil to lower depths by plowing, rainfall simulations were conducted on a Typic Dystrochrept in Pennsylvania. Mehlich-3 P concentration of soil (0-5 cm) was initially 495 mg $kg^{-1}$ and was reduced to 136 mg $kg^{-1}$ after chisel-plowing to 25-cm depth and disking to prepare a new grass seedbed. Stratification ratio (0-5 cm / 5-20 cm depth) of soil organic C was 5.2 and for total P was 4.7. Plowing probably reduced soil organic C and surface residue cover, resulting in greatly enhanced loss of sediment bound P (data not shown), soil sediment, and dissolved P in runoff (Fig. 9). If the 30-week period after plowing were not to experience high-intensity rainfall, then the positive effect of plowing to reduce dissolved P could be viewed as beneficial. However, the threat of greatly enhanced loss of sediment-bound P in runoff with plowing may be of greater concern. Further work is needed to assess the soil and water quality impacts of various tillage and amendment practices to modify high surface-nutrient concentrations and their transport potential.

High surface-soil P concentration with repeated application of animal manure to pasture can be mitigated with respect to water runoff quality by mixing manure with alum [$Al_2(SO_4)_3 \cdot 14H_2O$] or ferrous sulfate ($FeSO_4 \cdot 7H_2O$) to chemically transform soluble P into insoluble forms (Shreve et al., 1995). Long-term application (8 years) of alum-treated poultry litter revealed no adverse effect of alum on tall fescue productivity and exchangeable Al in soil (Moore and Edwards, 2005). Alum-treated poultry litter had similar decomposition and N mineralization dynamics as untreated litter (Gilmour et al., 2004). Modification of the poultry diet with phytase (enzyme that cleaves stable P in grain for greater bioavailability and reduces the need for P supplementation in diet) is another strategy that could be used to reduce the P application load of poultry litter (Smith et al., 2004).

## Field Plots

On an Ultic Hapludalf in Virginia, mean runoff volume and losses of soil sediment, N, and P following corn harvest were lower under NT than under CT (Table 4). Management at the site was under wheat (CT) / soybean (NT) double-crop rotated with maize (NT) for a number of years. Although soil organic matter was not determined, it was expected to be stratified with depth based on management history. By inverting soil with plow tillage, nutrient concentrations in runoff were greater compared with undisturbed soil (Ross et al., 2001). Dissolved N ($NO_3$ + $NH_4$) in runoff from NT was approximately half of contents in runoff from CT.

Table 4. Mean runoff and loss of sediment, N, and P in runoff from 112 $m^2$ plots during three consecutive rainfall simulation events (1, 0.5, and 0.5 h events at 43 mm $h^{-1}$) on a loam under conventional tillage (moldboard plow) and no tillage with various nutrient amendments following maize harvest in Virginia. Data from Ross et al. (2001).

| Tillage / fertilizer | Runoff | Sediment | N | P |
|---|---|---|---|---|
| | mm | ------------------ kg $ha^{-1}$ ------------------ | | |
| Conventional + inorganic | 46 | 3558 | 10.3 | 4.1 |
| No tillage + inorganic | 10 | 18 | 0.5 | 0.3 |
| No tillage + poultry litter | 11 | 34 | 0.6 | 0.4 |
| No tillage/subsoil + inorganic | 11 | 5 | 0.5 | 0.3 |
| No tillage without fertilizer | 14 | 21 | 0.6 | 0.3 |

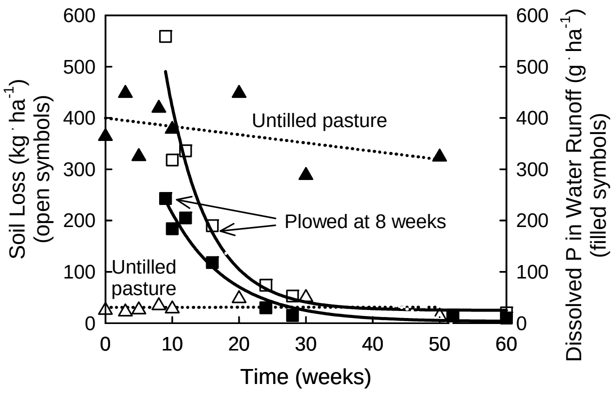



Fig. 9. Soil loss (Δ untilled pasture and □ plowed pasture) and dissolved P (▲ untilled pasture and ■ plowed pasture) in water runoff as a function of time since soil (Typic Dystrochrept) under pasture with long-term poultry litter application was mixed to create a more uniform distribution of soil P. Data from Sharpley (2003).

However, dissolved P in runoff from NT was 4 times greater than from CT. Phosphorus loss in runoff under CT was 95% associated with sediment, while P loss under NT was 77% associated with the dissolved fraction.

On a Plinthic Kandiudult in Georgia, water runoff volume from 0.15-ha plots during the first 5 years of a cotton-cotton-peanut rotation was 27 $\pm$ 13% of precipitation under CT and 15 $\pm$ 9% of precipitation under strip tillage (Bosch et al., 2005). Soil loss in runoff during the first 2 years was greater under CT (4.7 $\pm$ 2.3 Mg ha$^{-1}$ yr$^{-1}$) than under strip tillage (1.1 $\pm$ 0.4 Mg ha$^{-1}$ yr$^{-1}$) (Potter et al., 2004). Because more precipitation ran off of soil under CT, lower volume of water flowed through subsurface drains under CT (9 $\pm$ 5% of precipitation) than under strip tillage (16 $\pm$ 7% of precipitation) during the first 5 years. Crop yield was similar between tillage systems. Runoff loss of cotton defoliants was not different between tillage systems (10 $\pm$ 5% of chemical applied) during a rainfall simulation (50 mm hr$^{-1}$) that lasted for 1 hr and occurred 1 hr after defoliant application (Potter et al., 2003).

On a Typic Paleudalf in Kentucky, simulated rainfall (66 mm hr$^{-1}$ during 3 events totaling 2 hr during a 26-hr period) on 0.01-ha bordered plots resulted in 34% of water as runoff under moldboard plowing, 22% as runoff under chisel plowing, and 6% as runoff under NT (Seta et al., 1993). Soil loss was 15.5 Mg ha$^{-1}$ under moldboard plowing, 3.3 Mg ha$^{-1}$ under chisel plowing, and 0.3 Mg ha$^{-1}$ under NT. Similar trends with respect to tillage systems were observed for inorganic N in runoff (4.9, 2.4, and 1.0 kg ha$^{-1}$, respectively), $PO_4$-P in runoff (0.7, 0.4, and 0.3 kg ha$^{-1}$, respectively), and atrazine in runoff (41, 30, and 17 g ha$^{-1}$, respectively). Under natural rainfall during the first 4 years of this study, water runoff volume was 13% of precipitation under moldboard plowing and 5% of precipitation under the surface-residue conserving management systems of chisel plowing and NT (Blevins et al., 1990). Soil loss was 4.9, 0.2, and 0.1 Mg ha$^{-1}$ yr$^{-1}$ under moldboard plowing, chisel plowing, and NT, respectively, while loss of $NO_3$-N in runoff was 0.6, 0.4, and 0.5 kg ha$^{-1}$ yr$^{-1}$, respectively, loss of $PO_4$-P in runoff was 0.14, 0.04, and 0.07 kg ha$^{-1}$ yr$^{-1}$, respectively, and loss of atrazine in runoff was 8.3, 2.4, and 7.5 g ha$^{-1}$ yr$^{-1}$, respectively.

On a Typic Kanhapludult in Georgia, mean runoff concentration of $NO_3$-N across several naturally occurring runoff events throughout several years was relatively similar across tillage systems (Table 5). Runoff concentration of $PO_4$-P was greater under NT than under CT, with differences greater in the cropping system receiving poultry litter (Endale et al., 2004). Water runoff volume averaged 100 mm from CT and 56 mm from NT from April to November in Year 6 of the study (Endale et al., 2001). This study typifies other studies, in which runoff loss of dissolved N was lower with NT compared with CT and runoff loss of dissolved P was greater with NT.

Conversion of cultivated cropland to set-aside grassland in the United Kingdom resulted in reduced water runoff volume and greatly reduced soil erosion (Fullen, 1991). During 9 years of set-aside grassland, water runoff volume was 0.2% of precipitation and mean

Table 5. Mean runoff characteristics of N and P from 270 m$^2$ plots during two evaluation periods under naturally occurring rainfall events on a sandy loam under conventional tillage (CT, disk plow) and no tillage (NT) with inorganic and poultry litter applied to cotton/rye (Year 6; Endale et al., 2001) and maize/rye (Years 9-11; Endale et al., 2004) in Georgia.

| Condition | CT | NT |
|---|---|---|
| | Runoff P concentration (mg L$^{-1}$) – Year 6 | |
| Inorganic | 0.2 | 0.7 |
| Poultry litter | 0.3 | 1.4 |
| | Runoff P concentration (mg L$^{-1}$) – Years 9-11 | |
| Inorganic | 0.6 | 4.0 |
| Poultry litter | 1.5 | 6.8 |
| | Runoff $NO_3$-N concentration (mg L$^{-1}$) – Years 9-11 | |
| Inorganic | 1.7 | 1.5 |
| Poultry litter | 0.9 | 1.0 |

soil loss was 0.2 Mg $ha^{-1}$ $yr^{-1}$ (Fullen et al., 2006). Soil organic C in the surface 5 cm increased from 11.8 g $kg^{-1}$ at the time of cropland-grassland conversion to 18.0 g $kg^{-1}$ at the end of 10 years of grass set-aside. During a 15-year evaluation of permanent grassland (soil organic C of 26 g $kg^{-1}$), water runoff volume was 0.1% of precipitation. These data exemplify the positive benefit of grassland soil cover on improving water cycling and quality, as mitigated by an increase in surface soil organic matter concentration.

Many of the field plots reviewed in this section did not report surface soil organic matter concentration, but is expected that surface soil organic matter would be greater under NT than under CT. These missing data would have been valuable in assessing the relationship between stratification of soil organic C and water runoff and loss of sediment and nutrients.

**Water Catchments**

In a pair of water catchments (0.3 $\pm$ 0.1 ha) in Maryland, runoff volume, soil loss, dissolved N, total N, and total P were greater under CT than under NT (Angle et al., 1984). Only dissolved P was not different between tillage systems. Total runoff volume was low (2 $\pm$ 2% of precipitation) in this 3-year evaluation.

A long history (since the 1940s) of soil erosion and water runoff data have been collected from various water catchments (0.6 $\pm$ 0.1 ha) in Coshocton, Ohio (Shipitalo and Edwards, 1998). Soil loss was greatest during corn years (15.7 Mg $ha^{-1}$ $yr^{-1}$) of a corn-wheat-meadow-meadow rotation with moldboard plowing (Edwards and Owens, 1991). With continuous corn production during 4 years, water runoff volume was 16 $\pm$ 7% of precipitation under CT and <1% of precipitation under NT (Shipitalo and Edwards, 1998). Soil loss was 5.3 $\pm$ 4.4 Mg $ha^{-1}$ $yr^{-1}$ under CT and <0.1 Mg $ha^{-1}$ $yr^{-1}$ under NT. Continuation of the NT corn production system for a further 13 years produced nearly similar estimates of <1% of precipitation as water runoff and <0.1 Mg $ha^{-1}$ $yr^{-1}$ of soil loss.

On four water catchments with 7-11% slope under corn-soybean rotation in Ohio, water runoff volume during 10 years was 5 $\pm$ 5% of precipitation under chisel-plow tillage (a surface-residue conserving tillage practice) and 7 $\pm$ 4% of precipitation under NT (Edwards et al., 1993; Shipitalo et al., 1997). Soil loss during the first 6 years was 0.5 $\pm$ 0.8 Mg $ha^{-1}$ $yr^{-1}$ under chisel-plow tillage and 0.5 $\pm$ 0.9 Mg $ha^{-1}$ $yr^{-1}$ under NT (Edwards et al., 1993). Comparative soil loss from these water catchments during previous moldboard-plow tillage of a corn-wheat-meadow-meadow rotation was 2.6 $\pm$ 4.0 Mg $ha^{-1}$ $yr^{-1}$ (Edwards et al., 1993). Three major events (May-June) during the 6-year period caused 28% of total soil loss in runoff under chisel-plow tillage and 63% of soil loss under NT. Runoff loss of $NO_3$-N and sediment-bound N and P averaged 3.3, 1.3, and 0.4 kg $ha^{-1}$ $yr^{-1}$, respectively, and did not vary significantly between tillage systems (Owens and Edwards, 1993). Herbicide loss in runoff was low and did not vary significantly between tillage systems, averaging 0.31% with atrazine, 0.20% with linuron, 0.14% with metribuzin, and 0.05% with alachlor (% of chemical applied) (Shipitalo et al., 1997). Low concentration of herbicides in runoff in this water catchment study contrasted with much higher runoff losses under laboratory rainfall simulations, in which loss of atrazine was 3 to 14% of applied and loss of metolachlor was 4 to 10% of applied (Zhang et al., 1997). Concentration of alachlor and atrazine in water catchment runoff exceeded maximum contaminant levels only during runoff events that occurred within 2 months of application. Results from the Coshocton watershed station have illustrated the value of maintaining surface soil cover with conservation tillage systems to limit soil erosion and herbicide loss in runoff.

On 1.3- to 2.7-ha water catchments with Typic Kanhapludults in Georgia, water runoff volume during CT cultivation of various crops averaged 10-15% of precipitation (Langdale et al., 1985). Comparing different periods of CT and conservation tillage cropping on these watersheds, water runoff volume during 7 years of CT (55 events) was 9 $\pm$ 7% of precipitation and during 10 years of conservation tillage (20 events) was 2 $\pm$ 2% of precipitation. Dissolved P in runoff was not different between tillage systems (0.24 $\pm$ 0.11 kg $ha^{-1}$ $yr^{-1}$), but total P (sediment plus dissolved P) in runoff was greater under CT (1.9 kg $ha^{-1}$ $yr^{-1}$) than under conservation tillage (0.4 kg $ha^{-1}$ $yr^{-1}$). Stratification ratio of soil

Table 6. Water runoff volume, soil sediment, and nutrient losses from 1.6 to 4.8-ha water catchments managed under different land uses (CT, conventional tillage, NT, no tillage, and native or introduced grass) at two locations in Oklahoma from 1979 to 1990. Data from Sharpley and Smith (1994).

| Management | Watershed-years | Water runoff (mm $yr^{-1}$) | Soil sediment loss (Mg $ha^{-1}$ $yr^{-1}$) | Nitrate-N loss in runoff (kg $ha^{-1}$ $yr^{-1}$) | Total N loss in runoff (kg $ha^{-1}$ $yr^{-1}$) | Dissolved P loss in runoff (kg $ha^{-1}$ $yr^{-1}$) | Total P loss in runoff (kg $ha^{-1}$ $yr^{-1}$) |
|---|---|---|---|---|---|---|---|
| El Reno, Oklahoma (740 mm precipitation $yr^{-1}$) | | | | | | | |
| CT wheat | 29 | 84 | 3.68 | 2.19 | 15.01 | 0.21 | 2.17 |
| NT wheat | 7 | 158 | 0.37 | 2.01 | 9.24 | 0.98 | 1.55 |
| Grass | 20 | 80 | 0.04 | 0.09 | 1.60 | 0.08 | 0.15 |
| Woodward, Oklahoma (600 mm precipitation $yr^{-1}$) | | | | | | | |
| CT wheat | 11 | 39 | 9.35 | 0.44 | 14.91 | 0.16 | 3.53 |
| NT wheat | 5 | 65 | 0.88 | 0.86 | 5.14 | 0.49 | 1.30 |
| Grass | 22 | 16 | 0.16 | 0.13 | 0.71 | 0.06 | 0.14 |

Table 7. Mean runoff volume (% of rainfall), soil loss, and P from 2.8 $\pm$ 0.8-ha watersheds during 5 years under conventional tillage and no tillage at 3 locations in Oklahoma and Texas. Data from Sharpley et al. (1992).

| | | | P in runoff | | |
|---|---|---|---|---|---|
| Tillage | Runoff | Soil loss | Particulate | Bioavailable | Total |
| | % | Mg ha$^{-1}$ | ------------ kg ha$^{-1}$ yr$^{-1}$ ------------ | | |
| | | Bushland TX, Torrertic Paleustoll (540 mm rainfall) | | | |
| Stubble mulch | 5 | 0.9 | 0.5 | 0.1 | 0.5 |
| No tillage | 8 | 0.5 | 0.3 | 0.2 | 0.4 |
| | | Woodward OK, Typic Ustochrept (600 mm rainfall) | | | |
| Disk tillage | 17 | 39.6 | 14.4 | 0.9 | 14.9 |
| No tillage | 23 | 1.9 | 1.8 | 1.5 | 2.9 |
| | | El Reno OK, Udertic Paleustoll (740 mm rainfall) | | | |
| Plow tillage | 20 | 12.8 | 5.7 | 1.2 | 5.9 |
| No tillage | 24 | 0.4 | 0.5 | 1.4 | 1.7 |

organic C (0-6 cm / 12-20 cm depths) during several decades of conservation tillage at this location was 3.5 under NT (Franzluebbers et al., 2007) and probably would have been 1-2 under CT.

At two research locations in Oklahoma, runoff characteristics were evaluated among native grass, CT wheat, and NT wheat on 2.4 $\pm$ 1.2-ha water catchments (5 $\pm$ 3% slope) from 1979 to 1990 (Table 6). Water runoff volume was greater under NT than under CT at both locations, but soil loss and total N and P losses in runoff were lower under NT than under CT (Sharpley and Smith, 1994). Dissolved N and P were NT $\geq$ CT and both cropping systems produced greater loss of dissolved N and P than under native grass. Nitrate concentration in groundwater increased under NT cropping at both locations, seemingly as a result of greater water storage, lower wheat yield, and greater movement of nutrients through the soil profile than under CT.

From 10 water catchments (4.8 $\pm$ 2.7 ha) in Texas on Udic Haplusterts, loss of nutrients in runoff under CT cropland was greater than under undisturbed pastures, each receiving varying levels of poultry litter application (Harmel et al., 2004). Loss of total N in runoff was 22.5 $\pm$ 6.8 kg ha$^{-1}$ yr$^{-1}$ from cropland and 0.9 $\pm$ 1.4 kg ha$^{-1}$ yr$^{-1}$ from pasture. The fraction of total N loss as dissolved N was 0.72 $\pm$ 0.11 from cropland and 0.23 $\pm$ 0.30 from pasture. Loss of total P in runoff was 3.2 $\pm$ 1.0 kg ha$^{-1}$ yr$^{-1}$ from cropland and 0.4 $\pm$ 0.3 kg ha$^{-1}$ yr$^{-1}$ from pasture. The fraction of total P loss as dissolved P was 0.34 $\pm$ 0.13 from cropland and 0.70 $\pm$ 0.33 from pasture.

From paired watersheds in the Southern Plains USA, mean soil loss and total P in runoff were lower under NT than under CT (Table 7). Despite water runoff volume being NT $\geq$ CT, these results indicate that nutrient runoff loss was reduced with NT. However, runoff loss of bioavailable P tended to be greater with NT than with CT, suggesting that overland flow of water without sediment transport could carry a significant load of dissolved nutrients.

No-tillage management of a 2.7-ha cropped watershed for 24 years on a Typic Kanhapludult in Georgia reduced water runoff to 23 mm yr$^{-1}$ compared with 180 mm yr$^{-1}$ under previous management with CT (Endale et al., 2000). Soil loss was even more dramatically reduced with NT management (<0.1 vs. 23.3 Mg ha$^{-1}$ yr$^{-1}$). Stratification ratio of soil organic C (0-5 cm / 12.5-20 cm depths) was 3.3 at the end of 24 years of conservation tillage (Franzluebbers and Stuedemann, 2002) and was expected to be 1-2 under CT. Stratification ratio (0-6 cm / 12-20 cm depths) of particulate organic C was 9.0 and of soil microbial biomass C was 4.4 during several decades of conservation tillage (Franzluebbers et al., 2007). A greenhouse study to separate the short- and long-term effects of disturbance on soil hydraulic properties of this same soil revealed that previous doubling of soil organic C content in freshly tilled soil improved water infiltration by 27% (Franzluebbers, 2002b). However, water infiltration was 3.3 times greater in intact cores from long-term conservation tillage with a high degree of soil organic matter stratification compared with intact cores from a long-term conventionally tilled soil (but untilled during the previous 14 months) with a low degree of soil organic matter stratification. Thus, surface accumulation of soil organic C was more effective for water infiltration than uniform distribution of organic C in soil.

These water catchment studies illustrated a diversity of water runoff and water quality responses that depended on climatic region, soil type, and/or previous management conditions. However all studies suggested a strong reduction of soil loss and sediment-borne nutrient loss with conservation agricultural systems compared with CT. Further characterization of the changes in surface soil properties (i.e. soil organic C and N, water infiltration, aggregation, etc.) in response to conservation management would have been useful to develop a database to mechanistically defend the implied strong linkage between soil and water quality.

## CONCLUSIONS

Soil organic matter under conservation agricultural management becomes increasingly stratified with depth over time. This stratification can be viewed as an improvement in soil quality, because several key soil functions are enhanced, including water infiltration, conservation and cycling of nutrients, and sequestration of C from the atmosphere. Stratification of various soil organic C and N fractions with conservation tillage generally reduces water runoff volume and soil loss from agricultural fields. Perennial pastures often reduce water runoff volume and soil loss even further than with conservation-tillage cropland due to greater accumulation of surface soil organic matter. Total loss of nutrients is often lower with conservation tillage than with conventional tillage, because of a reduction in sediment-borne nutrients. Dissolved P in water runoff can be a threat to water quality with excessive nutrient applications from fertilizers and manures (possibly even under conservation management), although quantitative relationships of how dissolved P might directly affect water quality responses should be developed further. This review demonstrated that conservation tillage and perennial pastures can mitigate sediment and nutrient loss to the environment. Further studies are needed to validate the concept that stratification of soil organic matter can be effectively linked to water runoff and quality. Many detailed water runoff and nutrient transport studies have been conducted, but unfortunately, there has been a lack of detailed soil organic matter characterization by depth. The evidence gathered thus far suggests that strong linkages between soil and water quality aspects of conservation agricultural management do occur. A goal in the future should be to more fully characterize these linkages.

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