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## The inorganic nutrient cost of building soil carbon

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“ Inorganic nutrients are a hidden cost for sequestering carbon in soil organic matter. ”

Alan E Richardson<sup>\*1</sup>, Clive A Kirkby<sup>1</sup>, Samiran Banerjee<sup>1</sup>, & John A Kirkegaard<sup>1</sup>



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There is widespread interest in increasing soil carbon content in many of the world's agroecosystems. Whilst soils are the largest sink for terrestrial carbon, the capacity of soils to 'store' stabilized carbon in the longer term is compromised substantially in many managed systems due to practices that either accelerate the loss of soil organic matter (SOM) or limit its formation and retention. Declines of some 50–75% of the original total SOM content of soils in agricultural systems (i.e., ~60 Mg C ha<sup>-1</sup>) are common [1]. Rebuilding the 'lost' soil carbon is a high priority for agricultural policymakers and practitioners not only as an effective means to mitigate against climate change but also to reinstate benefits to plant health and ecosystem services derived from SOM, including soil physical structure, fertility–nutrients and water, and beneficial root–soil–microbiome interactions [2].

Crop residues, such as wheat and rice stubbles or maize stover, are a global resource with enormous potential for contributing to soil carbon stocks. For example, in Australia the annual production of cereal crops alone generates around 40–50 million tonnes of carbon-rich crop residue (stubble) each year that could potentially be used more effectively to generate

SOM; further to its current use for soil conservation purposes in minimum-till systems, as low-quality feed supplements, or in preference to the widely discouraged practice of stubble burning. However, numerous studies along with our own observations have indicated that soil carbon content does not necessarily increase as expected in response to repeated addition of often large quantities of crop residues to soil [3]. On a long-term experiment subject to continuous cropping and different residue management regimes (>20 years, located at Harden, NSW, Australia), we found no significant change in soil carbon in field plots subject to 'minimum till' or 'conventional cultivation' with surface burial of residues, either with retention of the stubble or its removal through burning. This occurred irrespective of the organic carbon content of the 'agricultural soils' at Harden being significantly less than in an adjacent and non-cultivated 'native' grass landsoil (i.e., ~0.85 cf 2.2 mg C g<sup>-1</sup> soil, respectively [4]). Most recently, we showed that the dynamics of stubble decomposition and capacity to generate 'new' soil carbon from carbon-rich crop residues in the Harden soil, and a range of other Australian soils, is mediated to a large extent by the availability of inorganic nutrients, including nitrogen,

CSIRO Sustainable Agriculture Flagship, CSIRO Plant Industry, PO Box 1600, Canberra, ACT, 2601, Australia

<sup>\*</sup>Author for correspondence: Tel.: +61 262465189; Fax: +61 262 465000; E-mail: alan.richardson@csiro.au

phosphorus and sulphur [5,6]. As such, we suggest that management of inorganic nutrients is an important consideration for strategies to 'build' soil carbon from carbon-rich crop residues.

Compared to SOM, crop residues have a very wide stoichiometric ratio of nutrients to carbon and this has important implications for the efficiency with which carbon is converted from crop residues into 'soil carbon'. For instance, wheat stubbles typically contain ~17, 2 and 3 parts of nitrogen, phosphorus and sulphur per 1000 units of carbon, as compared to ~90, 19 and 14 parts per 1000 units in the 'fine fraction' of SOM (commonly referred to as soil humus [7]). As shown by Kirkby et al., this tight coupling of nutrients in SOM is a feature of soils throughout the world irrespective of soil type, geographical location or management history [7]. The stoichiometry of carbon, nitrogen, phosphorus and sulphur in SOM is also more similar (in comparison to crop residues) to the nutrient ratios found in soil micro-organisms (i.e., 250, 49, 26 and 103, 11, 9 parts nitrogen, phosphorus and sulphur per 1000 units of carbon in bacteria and fungi, respectively [7]), which appear to be tightly controlled and relatively consistent across different soils and ecosystems [8,9]. This is also consistent with the notion that SOM is derived predominantly from microbial detritus, rather than via direct deposition of recalcitrant plant materials [10,11]. Biosequestration of carbon through microbial deposition has similarly been reported to be a driving mechanism for long-term carbon accumulation in marine and sediment systems [12]. Conceptually, this indicates that accumulation and dynamics of SOM is mediated by microbial processes and provides evidence that the nutrient requirements of the microbial biomass is a major controller of soil carbon transformations, including the generation of new SOM, mineralization of antecedent SOM, and thus overall net sequestration of carbon in soil or its loss through respiration.

We have recently demonstrated the importance of nutrients in mediating carbon sequestration in soil using laboratory incubations, whereby the humification efficiency of wheat straw (i.e., net conversion of residue carbon to soil carbon) was increased by two- to eightfold (from 4.5–7.5% to 12.2–42.6%) across a range of soils (with varying clay contents) when inorganic nutrients (nitrogen, phosphorus, sulphur) were added to account for the stoichiometric requirements of SOM [5]. Increased humification efficiency in response to nutrient addition was associated with increased microbial biomass in soil and mechanistically was shown to be a result of both an increase in the generation of new SOM and reduction in the loss of 'existing' or 'old' SOM across the various soils [6]. This is indicative of a positive 'priming effect', whereby in the absence of

sufficient nutrients, increased mineralization of existing SOM occurs in order to meet the nutritional requirements of the microbial biomass [13,14]. As a consequence, the addition of carbon-rich residues can lead to a net decrease in soil carbon content [6,15], which is consistent with observations under field conditions where repeated addition of crop residues is often ineffective in increasing soil carbon content [3].

Soil micro-organisms require carbon for growth and respiration and thus the input and utilization of plant materials (fresh organic matter, including shoot and root residues, rhizodeposition, etc.) and microbial carbon-use efficiency are key factors that drive the potential for carbon sequestration in soil. Previous soil biogeochemical studies have highlighted the importance of fresh organic matter 'quality' (e.g., high-quality and nitrogen-rich green manure crops or legume residues, compared to cereal stubbles) in determining rates of decomposition and whether net mineralization or immobilization of nitrogen occurs, along with the magnitude of the priming effect associated with mineralization of SOM [13,14,16]. Typically, high nutrient availability minimizes the priming effect and promotes the sequestration of carbon into new SOM [6,15,17]. Given that phosphorus and sulphur are similarly required for the formation and function of microbial biomass, the concentration and availability of these nutrients will further influence residue decomposition and associated SOM mineralization [9]. In addition to the importance of nutrient stoichiometry, the magnitude of priming effects associated with SOM mineralization depends on soil environmental conditions, microbial growth rates and community composition. Bacteria and fungi interact differentially with organic substrates with fungi generally having lower nutritional requirement (based on their wider nutrient stoichiometry), higher growth efficiency and biomass production than bacteria, and thus have greater potential to sequester more carbon than bacterial-dominated systems [17,18]. Furthermore, the succession of decomposer communities and interplay between r- and K-strategists (i.e., based on micro-organism growth rates and substrate preferences; [14]) will further affect community composition, microbial biomass and potential to sequester carbon. Additional research is needed to understand how microbiome interactions contribute to soil carbon dynamics. In particular, there is a need to investigate whether nutrient mobilization from SOM can be decoupled from carbon mineralization in soil, and thus benefits of increased availability of nutrients for plant nutrition be achieved, without concurrent loss in SOM as has appeared to have occurred historically [2].

Carbon-rich crop residues are significant environmental resources that provide a feasible opportunity for

building soil carbon and SOM content. How this might be achieved both economically and agronomically is a significant future challenge. The agronomic imperatives of surface-retained residues for soil protection and water conservation as well as a nutrient-use efficiency paradigm judged by the capture of applied nutrients by crops in the short term are both at odds with the mixing of carbon-rich residues into the soil with applied nutrients. The management of crop residues to increase carbon sequestration must also be compatible with sustainable farming practices and be a viable option for producers and production systems. Further experiments and modelling studies are needed to confirm the benefits and practicality of using nutrients to manage crop residues under field conditions to maximize opportunities for sequestering carbon and agronomic innovation required to seek strategies to achieve these multiple goals. This will include better understanding of the timing, form and amounts of additional inorganic nutrients required to maximize carbon sequestration, along with appropriate ways to manage and handle stubble loads (e.g., left standing, bashed and surface retained, mulched, cultivated, etc.) and to apply the nutrients in farming systems. An assessment of current trade-offs between the ecosystem service benefits that stubble currently provides, and those related to soil-carbon sequestration, is required.

The long-term sequestration of carbon from crop residues into SOM will require the simultaneous 'sequestration' of significant amounts of inorganic nutrients in order to meet the stoichiometric requirements of the soil microbial biomass and newly generated SOM. For example, based on nutrient ratios (as indicated above), the generation of an additional 1 t of soil carbon per hectare will require some 73, 17 and 11 kg ha<sup>-1</sup> of nitrogen, phosphorus and sulphur, respectively (i.e., difference in units per thousand for wheat stubble compared to SOM). These are significant amounts of nutrients that must be 'accounted' for and are comparable to the current annual fertilizer requirements of many agricultural crops. This raises the question as to whether 'higher' nutrient inputs will therefore be required to maintain and increase crop yield and simultaneously maintain or

generate new and stabilized SOM in the long term. It also presents a possible paradigm shift in thinking about fertilizing 'the system' rather than the current approach of fertilizing 'the crop', and thus how nutrient availability within soil systems can then be best managed and synchronized to meet crop demand, but without increased risk to the environment. This includes consideration that increased accumulation and turnover of SOM may also increase the potential for greater losses of nitrogen (and other nutrients) from mismanaged systems via leaching and/or nitrous oxide emissions through microbially mediated processes. The acidification of soil that followed the widespread introduction of legume-based clover pastures in Australia, due to the mismatch in the flush of autumn mineralization of nitrogen from the legume residues, and the slow-growing annual crops in winter highlights such risk.

In conclusion, and based on differences in nutrient stoichiometry, we highlight the significant 'hidden nutrient cost' in the long-term sequestering of carbon in SOM through the use of carbon-rich crop residues. The 'cost' of these nutrients should therefore ideally be considered in programs aimed at increasing soil carbon content in terrestrial ecosystems and, where possible, be accounted for in various carbon-trading schemes that are being developed across the globe.

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