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1  **Landmark Papers: No. 7**

2

3  **Smith, K.A., Ball, T., Conen, F., Dobbie, K.E., Massheder, J. & Rey, A.. 2003. Exchange**

4  **of greenhouse gases between soil and atmosphere: interactions of soil physical factors and**

5  **biological processes. European Journal of Soil Science,** 54**, 779–791.**

6

7  Commentary on the impact of Smith et al. (2003): by G. Guggenberger, B. Ludwig & M. Menon

8

9

10  **Introduction**

11  Smith et al. (2003) published their review on the interactions of soil physical factors and

12  biological processes controlling the exchange of greenhouse gases between soil and atmosphere

13  at a time when global change was already considered to be one of the most important challenges

14  of mankind (IPCC, 2001). In the Climate Change 2014 Synthesis Report (IPCC, 2014) a global

15  warming of $0.7°C$ between 1951 and 2010 was reported and further warming and long-lasting

16  changes in all components of the climate systems forecasted. Smith K.A. et al. (2003) and later

17  Smith P. et al. (2008) emphasized that about one third of $CH_4$ and two thirds of $N_2O$ emitted

18  globally to the atmosphere per year derive from soil processes, while soil is considered a small

19  $CO_2$ sink, which may change with increasing warming (Crowther et al., 2016). This is reason

20  enough to analyse the processes that lead to this net emission of gases to the atmosphere. While

21  biological processes produce or consume these greenhouse gases, the size of the fluxes is

22  strongly controlled by soil physical factors. However, the controlling factors on the interaction

23  between the controlling physical factors and biological processes in the exchange of greenhouse

24  gases between the soil and atmosphere had not been widely considered. Keith Smith and his

25  co-authors were pioneering in this field (e.g. Smith, 1980; Ball et al., 1997a, Ball et al., 1997b;

26 Conen et al., 2000; Dobbie & Smith, 2001), which finally led to the review of Smith et al.

27 (2003).

28

29 Controlling factors for $CO_2$ emissions

30 Smith et al. (2003) summarized that the release of $CO_2$ by aerobic respiration can be described

31 by a non-linear function of temperature over a wide range of water contents. The link between

32 microbial processes and physical factors, in addition to availability of substrate and chemical

33 factors (e.g. soil pH), is of substantial importance because of the direct and indirect effects of

34 physical factors on the production of $CO_2$ by microorganisms and roots. The non-linear

35 response of $CO_2$ as a function of temperature has been confirmed in several recent studies (e.g.

36 Schaufler et al., 2010). The factors affected by water content that were discussed by Smith et

37 al. (2003) are also now well established; water is important for gas diffusivity (Ball, 2013) and

38 substrate supply to soil microorganisms (Schindlbacher et al., 2004). Notably, Schaufler et al.

39 (2010) reported that maximum $CO_2$ emissions from European soils under different land uses

40 occur at intermediate soil moisture, which accords well with the summarizing synthesis by

41 Smith et al. (2003).

42     Smith et al. (2003) reported a marked scatter of $Q_{10}$ values for $CO_2$ emissions and pointed

43 out the need for standardization and accurate interpretation of temperature responses of the

44 soil's $CO_2$ emissions at greater depths. They emphasized that for accurate determinations and

45 interpretations of $Q_{10}$ values, diurnal temperature changes, thermal conductivities and thermal

46 diffusivities of the soil need to be considered in greater detail in future studies. In fact, a later

47 study by Pavelka et al. (2007) also addressed this important issue and recommended

48 measurement of soil temperature at a very shallow soil depth to determine useable values of

49 $Q_{10}$, and suggested a procedure to standardize $Q_{10}$ values for soil temperatures measured at

50 different depths.

51

52 Controlling factors for N$_2$O emissions

53 For N$_2$O, Smith et al. (2003) focused on the important microbiological processes of nitrification

54 of ammonium and denitrification of nitrate in soil, and the governing processes for the

55 respective rates. In particular, they elucidated soil conditions, e.g. structure, wetness, O$_2$ content

56 of pores and soil depth, being responsible for the release of N$_2$O to the atmosphere or further

57 reduction to N$_2$. Nitrate ammonification and nitrifier denitrification as additional processes

58 leading to the formation of N$_2$O have been discussed since in greater detail by Baggs & Phillipot

59 (2010) and Smith (2017). The merit of the review by Smith et al. (2003) lies again in the

60 important emphasis of the link between microbial processes and physical factors in addition to

61 other factors, such as substrate availability and chemical factors such as soil pH (e.g. Weslien

62 et al., 2009). This link is crucial for an understanding and prognosis of N$_2$O emissions.

63   Smith et al. (2003) emphasized that the anaerobic volume is affected by increases in the

64 water-filled pore space (WFPS), where an increase in WFPS may also result in an exponential

65 increase in N$_2$O emissions. There is still some controversy about which physical soil property

66 is most useful for estimating N$_2$O emissions; for example the ratio of gas diffusivity within the

67 soil to that in free air, the degree of aggregation and compaction, matric potential, WFPS and

68 volumetric water content (for a discussion see Ball, 2013 and Smith, 2017). Smith et al. (2003)

69 indicated that N$_2$O emissions also increase markedly with temperature. They attributed this to

70 increases in the anaerobic volume fraction. An increase in temperature results in an increase in

71 the size of the anaerobic zones because of increased respiration, which causes larger gradients

72 in O$_2$. In addition, increased temperatures are also likely to lead to increased rates of

73 denitrification per unit anaerobic volume. Both increases then favour a dramatic increase in

74 N$_2$O emissions. In fact, the concept of anaerobic zones is a key feature of the process-based

75 DNDC (denitrification-decomposition) model, for which there are several versions for different

76 land uses. This model has a kinetic scheme for the anaerobic volumetric fraction (an 'anaerobic

77 balloon') that is implemented to calculate the anaerobic fraction of soil in a given soil layer in

78 relation to $O_2$ diffusion and the respiratory activity of soil micro-organisms and roots (for a

79 summary of the DNDC model see Gilhespy et al., 2014). Overall, there is no doubt that soil

80 temperature and soil moisture are important for explaining much of the temporal variation in

81 $N_2O$ emissions within a site (e.g. Pilegaard et al. 2006).

82

83 Controlling factors for $CH_4$ emissions

84 For $CH_4$ production and transport, Smith et al. (2003) reported that ebullition and diffusion

85 through the aerenchyma of rice and plants in natural wetlands contribute substantially to the

86 emission of $CH_4$ and that the proportion of the emissions taking place by each pathway varies

87 seasonally.. The oxidation of atmospheric $CH_4$ to $CO_2$ is controlled by gas diffusivity, whereas

88 the effect of temperature is small (Smith et al., 2003). Ball (2013) suggested that the control of

89 gas diffusivity on the oxidation of $CH_4$ might not hold for all sites and that the effect of pH,

90 moisture, temperature, and nitrogen and type of organic matter and content might be

91 pronounced. The role of nitrogen as a regulatory factor of $CH_4$ oxidation has been addressed in

92 detail by Bodelier & Laanbroek (2004), who discussed the inhibiting role of additions of

93 nitrogenous fertilizer. The effect of WFPS on $CH_4$ oxidation may be seen as a hump-shaped

94 function where the optimum oxidation occurs at 20–50% WFPS. At smaller water contents,

95 desiccation stress and at larger water contents diffusion limitation might be inhibiting $CH_4$

96 oxidation (Dunfield, 2007). Thus, moist, well-aerated soil favours $CH_4$ oxidation and $CO_2$

97 exchange (Ball, 2013).

98

99 Soil structure, microbial communities and greenhouse gas emissions

100 Smith et al. (2003) emphasized that although the greenhouse gases are produced by microbial

101 processes, the size of their fluxes between soil and atmosphere depends largely on soil physical

102 factors. The transport of gases within the soil and the gas exchange between soil and atmosphere

103 is a function of gas diffusivity, which depends on the air-filled porosity or, inversely, with the

WFPS. Most soils develop a three-dimensional architecture with pedogenesis, which is characterized by the aggregate size distribution. The distribution of aggregates largely controls almost every process in soil. This refers to the air-filled porosity or WFPS at a given matric potential (Ball, 2013) as well as to the distribution of microbial populations in soil (Nunan et al., 2003). Therefore, soil structure controls the habitat of the actors involved in the production of greenhouse gases and determines the diffusion of $O_2$ and dissolved organic matter (DOM) to fuel aerobic microbes. Consequently, inter- and intra-aggregate pore space needs to be considered. Sey et al. (2008) compared the greenhouse gas emissions from various aggregate size classes (<0.25 mm, 0.25–2 mm and 2–6 mm) and from 2-mm sieved bulk soil at different WFPS (20, 40, 80 and 80%). They found that denitrification was responsible for 95% of $N_2O$ emissions in microaggregates, whereas nitrification was responsible for 97–99% of $N_2O$ production in macroaggregates. This inferred that diffusion of $O_2$ was largely inhibited in microaggregates when the WFPS was 80%, whereas macroaggregates maintained aerobic conditions.

The interrelations between soil structure and greenhouse gas emissions can be readily investigated when the natural soil structure and size distribution of aggregates are disrupted due to external forces (e.g. compaction), which in turn can alter the pore size distribution and hydraulic properties (Menon et al., 2015). Beare et al. (2009) showed that the production of $N_2O$ was 67 times greater in compacted than uncompacted soil at field moisture contents, and they demonstrated the effect of soil moisture on emissions of $N_2O$ and $CO_2$. Deurer et al. (2012) reported enhanced carbon sequestration under the wheel tracks, probably because of reduced microbial decomposition of organic matter. Bessou et al. (2010) also found that compacted soil had smaller emissions of $CO_2$, but at the same time larger $N_2O$ emissions by inducing anoxic conditions favourable for denitrification activity.

Experiments with compacted soil also help to elucidate the relation between microbial communities and greenhouse gas emissions depending on soil physical factors. So Nadian et

130    al. (1998) reported a significant decline in vesicular-arbuscular mycorrhizal fungi biomass at

131    higher bulk density, and Peacock et al. (2001) found a significant reduction in microbial

132    biomass for heavy traffic treatments. Schnurr-Pütz et al. (2006) observed that fungi, in

133    particular, are negatively affected by soil compaction, whereas denitrifiers and methanogens

134    appear to be more prominent. From that, the links between soil physical properties and

135    greenhouse gas emissions can be conceptualized as in Fig. 1.

136

137    New developments in linking soil physical factors to biological processes

138    In their landmark paper on the interactions of soil physical factors and biological processes,

139    Smith et al. (2003) focus on gas diffusivity, which affects soil aeration and the capacity of the

140    soil microbial community to produce or consume $CO_2$, $N_2O$ and $CH_4$. The concept of hotspots

141    and hot moments (Kuzyakov & Blagodatskaya, 2015) adds the supply of the organic substrates,

142    which is also linked partly to soil physical factors. Transport of the labile OM sources to the

143    microbial community occurs largely through biotic activities such as the release of root

144    exudates (Jones et al. 2004) and the detritus of soil animals (Schrader et al. 2007), but also as

145    DOM leached from the O and A horizons (Qualls & Haines 1992).  Translocation of DOM to

146    the subsoil depends strongly on the flow paths in soil and on soil structure and precipitation

147    events (Leinemann et al. 2016). Because DOM is mainly translocated in the inter-aggregate

148    pore space of the soil, it is retained on aggregate surfaces, which are enriched in OM (Amelung

149    et al. 2002), thus creating a hotspot. At the same time, the inter-aggregate pore space usually

150    enables good aeration, leading to the release of $CO_2$ with microbial decomposition of the

151    substrate. In otherwise aerobic soil, strong microenvironments may exist that are important

152    sources of $N_2O$ and $CH_4$ (Keiluweit et al. 2016). Hotspots of denitrification and methanogenesis

153    in the intra-aggregate pore space results from slow diffusion of $O_2$, whereas in the rhizosphere

154    this is caused by the inflow of very available OM from root exudation (Henry et al. 2008). This,

155    once again, emphasizes the complex interplay of soil physical factors and biological processes

156  in the production of greenhouse gases in soil and their exchange between soil and atmosphere

157  (Smith et al. 2003).

158

159  Methodological progress

160  The landmark paper of Smith et al. (2003) on these interactions also triggered substantially the

161  methodological development with respect to the visualization of pores of different size, to

162  measurement of microbial activity and the resulting $O_2$ and $CO_2$ partial pressures at small

163  scales, and the development of physical and biophysical models. In the last decade much

164  progress in the understanding of soil structure and the associated pore-space architecture has

165  been? gained by X-ray computed tomography (CT), which enables an in-situ and real-time 3-

166  D mapping at scales of a few microns. Measured properties include porosity, pore-size

167  distribution, tortuosity and topology (Naveed et al. 2013; Vogel et al. 2010). Peth et al. (2014)

168  showed that synchrotron-based X-ray CT in combination with osmium staining is not only

169  suitable for describing soil structure, but also for identifying the location of organic matter in

170  soil, e.g. in the intra-aggregate pore space. Neutron radiography emerged as a useful method to

171  map the water distribution within soil and its temporal changes (Oswald et al. 2008; Carminati

172  et al. 2010), whereas the 2-D distribution of oxygen concentration can be analysed by

173  fluorescence imaging with planar optodes (Blossfeld et al. 2011). Rudolph-Mohr et al. (2017)

174  emphasized the great potential of combining neutron radiography with fluorescence imaging

175  to investigate the effect of different soil moisture conditions on the oxygen patterns in soil. Such

176  analyses may provide important input parameters for geometry-based mechanistic models.

177      Keith Smith also pioneered modelling of microbial respiration and denitrification at the

178  aggregate scale by systematically incorporating factors such as oxygen supply and nitrogen

179  concentration (Smith, 1980). Ebrahimi & Or (2015, 2016) have built on that and developed a

180  3-D pore-scale model that simulates the aerobic and anaerobic microbial communities within

181  aggregates together with rates of production of $N_2O$ and $CO_2$ along the aggregate radius. This

182    model considers substrate and oxygen diffusion processes and is integrated with individual cell-

183    based models that link soil physical processes with microbial community dynamics. Ebrahimi

184    & Or (2016) upscaled this modelling framework to quantify depth-resolved rates of production

185    of $CO_2$ and $N_2O$ depending on small-scale environmental conditions. In a very recent model,

186    this approach was used to quantify methane production in thawing permafrost soil, based on

187    the microbial activity dynamics in pore networks with? consideration of transport dynamics and

188    physiological aspects of the cells (Ebrahimi & Orr, 2017).

189

190    **Conclusions**

191    Smith and co-authors expressed hope that their review would demonstrate the key roles played

192    by soil physical factors in controlling the biological processes responsible for the exchange of

193    greenhouse gases between soil and atmosphere. Indeed, the authors convincingly built a bridge

194    between soil physics and soil biology. From this landmark publication and some other

195    manuscripts, soil biophysics has developed as an emerging field within the soil sciences. Inter-

196    and intra-aggregate pore architecture is decisive in the control of the availability of $O_2$ and

197    organic substrates to microorganisms. It is thus of utmost importance not only for the

198    production of the different greenhouse gases, but also for organic matter stabilization and biotic

199    redox processes associated with mineral weathering and mineral transformation. The effect of

200    biota on soil physical factors has also received increasing interest recently. This concerns, for

201    example, the formation of aggregates by living and dead organic agents, which affects soil

202    structure and associated pore architecture, or the rhizosphere, where water uptake by the roots

203    strongly modifies the WFPS. Novel instrumental and modelling approaches will allow an

204    understanding of the multiple interactions between soil physical and biotic processes in soil in

205    relation to soil functioning and ecosystem services. This is only possible by crossing the

206    boundaries in soils science, which is what this landmark paper emphasized.

207

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330

331

332

**Figure**

335    Figure 1. Conceptual model on the link between soil physical properties and greenhouse gas

336    emission depending on soil compaction (Menon and Blaud, unpublished). Soil compaction

337    leads to changes in soil structure (e.g. porosity), which will affect the flow of air and water,

338    and thereby create a more anaerobic environment in soil. This may lead to a shift in the

339    relative abundance and functions of the microbial population, shown here as effects on the

340    C and N cycles. Abundance of nitrifiers and aerobic degraders are given by dashed lines and

341    abundance of denitrifiers and methanogens are given by solid lines.

342

343

344