1 **Nonlinear turnover rates of soil carbon following cultivation of native grasslands and**

2 **subsequent afforestation of croplands**

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Abstract

29   Land use conversions can strongly impact soil organic matter (SOM) storage, which

30   creates paramount opportunities for sequestering atmospheric carbon into the soil. It is known

31   that land uses such as annual cropping and afforestation can decrease and increase SOM,

32   respectively; however, the rates of these changes over time remain elusive. This study focused on

33   extracting the kinetics (k) of turnover rates that describe these long-term changes in soil C

34   storage and also quantifying the sources of soil C. We used topsoil organic carbon density and

35   δ^{13}C isotopic composition data from multiple chronosequences and paired sites in Russia and

36   United States. Reconstruction of soil C storage trajectory over 250 years following conversion

37   from native grassland to continual annual cropland revealed a C depletion rate of 0.010 years^{-1}

38   (first-order k rate constant), which translates into a mean residence time (MRT) of 100 years

39   ($R^2 \geq 0.90$). Conversely, soil C accretion was observed over 70 years following afforestation of

40   annual croplands at a much faster k rate of 0.055 years^{-1}. The corresponding MRT was only 18

41   years ($R^2 = 0.997$) after a lag phase of 5 years. Over these 23 years of afforestation, trees

42   contributed 14 Mg C Ha^{-1} to soil C accrual in the 0 to 15 cm depth increment. This tree-C

43   contribution reached 22 Mg C Ha^{-1} at 70 years after tree planting. Over these 70 years of

44   afforestation, the proportion of tree-C to whole soil C increased to reach a sizeable 79%.

45   Furthermore, assuming steady state of soil C in the adjacent croplands, we also estimated that

46   45% of the prairie-C existent at time of tree planting was still present in the afforested soils 70

47   years later. As intrinsic of k modelling, the derived turnover rates that represent soil C changes

48   over time are nonlinear. Soil C changes were much more dynamic during the first decades

49   following a land use conversion than afterwards when the new land use system approached

50   equilibrium. Collectively, results substantiated that C sequestration in afforested lands is a

51   suitable means to proactively mitigate escalating climate change within a typical person's

52   lifetime, as indicated by MRTs of few decades.

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1 Introduction

The global effects of escalating climate change are in part driven by land use choices. Indeed, implementing certain land use changes can worsen or in other cases mitigate emissions of greenhouse gases from terrestrial ecosystems to the atmosphere (Paustian et al., 1992; Post and Kwon, 2000; Thilakarathna and Hernandez-Ramirez, 2021). Essentially, land use options that unintentionally accelerate biological oxidation of soil organic matter (SOM) contribute over time to atmospheric carbon dioxide concentrations (Sauer et al., 2007; Laganiere et al., 2010; Li et al., 2018), providing a portion of the radiative forcing that has been causing part of the global warming effect over the last decades (Guo and Gifford, 2002; Parry et al. 2007). Relative to contrasting types of land use systems, annual croplands commonly showed SOM depletion and marked reductions in soil C storage, in particular compared with their natural ecosystem counterparts (Chendev et al., 2015b; Hebb et al., 2017; Kiani et al., 2017).

Contrary to the potentially detrimental effects of annual cropping on SOM accumulation and overall soil quality (Guenette and Hernandez-Ramirez, 2018; Laganiere et al., 2010; Kiani et al., 2020), tree planting offer multiple environmental services and societal benefits (Hernandez-Ramirez et al., 2012; Sauer et al., 2012; Zhang et al., 2020). For instance, removing C from the atmosphere is a paramount contribution by trees (Guo and Gifford, 2002; Li et al., 2012; Li et al., 2018). In effect, soil C accrual (Paul et al., 2002; Dhillon and Van Rees, 2017; Khaleel et al., 2020) and stabilization (Hernandez-Ramirez et al., 2011; Wang et al., 2016; Quesada et al., 2020) beneath trees have been recognized as an effective means for sequestering atmospheric C. In addition to soil accruals, diverse microbial communities can flourish beneath mature trees (Kiani et al., 2017). Additional functions by tree vegetation include improving air quality, enhanced microclimate, and erosion control (Sauer et al., 2007; Hernandez-Ramirez et al., 2012;

79   Chendev et al., 2015b). Establishing agroforestry practices such as shelterbelt systems within

80   annual croplands can provide a balance between continual food production and tree benefits with

81   only a fraction of the landscape occupied by trees (Amadi et al., 2016; Dhillon and Van Rees,

82   2017).

83   Sequestering C in soils is governed by the balance of inputs of plant C with

84   decomposition and stabilization processes (Hernandez-Ramirez et al., 2009; Kiani et al., 2017; Li

85   et al., 2018). This overall functioning of C-related biology and cycling in soils can be described

86   as the turnover of soil C. Collectively, SOM mineralization, gains and losses, and net accrual can

87   be numerically integrated into C turnover rates (Richter et al., 1999; Hernandez-Ramirez et al.,

88   2011; Xiong et al., 2020). For instance, dynamic rates of net depletion of SOM pools caused by

89   continual cropping or tree contributions to soil C accretion and cycling under afforestation can

90   both be captured as C turnover rates (Guo and Gifford, 2002; Hu et al. 2013). Nevertheless, since

91   using linear rates to describe changes in soil C often leads to poor estimates of C inventories and

92   sequestration, soil C accrual rates need to be derived as nonlinear rates to accurately predict the

93   trajectory of soil C changes with time following land use conversions (Post and Kwon, 2000;

94   Garten 2002). Moreover, the direction and net rates of SOM accrual as a response to land use

95   changes need to be assessed in the long term (i.e., ranging from decadal to centurial scales)

96   (Paustian et al., 1992; Hernandez-Ramirez et al., 2011). This new knowledge will inform how

97   lasting these effects of land management options on soil C storage are, enabling predictions of

98   future soil C sequestration (Richter et al., 1999; Guo and Gifford, 2002). Our study endeavors to

99   address and fill these knowledge gaps.

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101   Testing accrual rates of SOM is still lacking in the literature. Previous studies have

102   evaluated soil C turnover rates as a function of changes in land management only over one or a

103   few decades (Jastrow et al., 1996; Hernandez-Ramirez et al., 2011; Mary et al., 2020); however,

104   the underlying assumption of asymptotic behavior in the rate of soil C change has rarely been

105   verified over longer periods such as over centuries. Likewise, earlier studies examining dynamics

106   of soil carbon in continuous annual croplands have suggested typical MRTs of 117 yr (Huggins

107   et al., 1998) and 57 yr (Collins et al., 1999); however, it is still unclear how long-term land use

108   changes from native grasslands to annual croplands and from annual croplands into afforestation

109   can impact the turnover rates of soil C over centuries.

110   In this study, we compiled soil C storage data from several field sites comparing land use

111   systems in Russia and United States (Chendev et al., 2015a, 2015b) in conjunction with

112   published (Hernandez-Ramirez et al., 2011) and newly-available soil ^{13}C isotope data. Based on

113   these data assemblage, we now focused on evaluating the long-term turnover rates of soil C as a

114   function of land use changes from native grasslands to annual croplands and subsequent

115   converting annual croplands into afforestation. We aimed at extracting turnover rates of soil C

116   depletion or accretion, which can enable future predictions of soil C storage depending on land

117   use systems. Also, our study quantified and documented the contributions of tree biomass-C to

118   soil C that was newly-accrued following afforestation. We further examined the stage and net

119   losses of the C that existent in the soil under annual croplands prior to tree planting.

120

121   **2 Materials and Methods**

122   **2.1 Chronosequences from native grasslands to annual croplands in Russia**

123   Three long-term chronosequences of land use conversion (i.e., a range of different

124 duration of cultivation) were used for extracting turnover parameters. These land use

125 chronosequences were situated in Belgorod oblast, Russia within the districts of Prokhorovskiy

126 (50°57′ N, 36°44′ E), Gubkinskiy (51°03′ N, 37°22′ E) and Ivnyanskiy (51°06′ N, 36°24′ E) as

127 previously described by Chendev et al. (2015a) (Fig. 1). Following chronosequence methods as

128 described by Laganiere et al. (2010), each chronosequence had four or five age-sites

129 encompassing a native grassland site that represented the time zero of conversion from grassland

130 to annual cropland. These native grasslands were undisturbed steppe dominated by plant species

131 with C3 photosynthetic pathway. The ages of the cropland sites were established through

132 historical records and geographic approaches described by Chendev et al. (2012, 2015b).

133 Additional information about the study sites is available at Chendev et al. (2015a), while the

134 focus in our study remains on developing models and extracting parameters of C turnover rates.

135 Typical crops species included cereals, sunflower (*Helianthus annuus*) and beet (*Beta vulgaris*)

136 managed under conventional tillage operations. Within the study region, soils were classified as

137 loamy Chernozems (Russian Soil Classification System), annual precipitation ranged between

138 480 and 580 mm and air temperature between 5.3 and 5.8 °C (Chendev et al., 2015a).

139   Composite soil samples (3 subsamples per sample) with at least 12 sampling locations

140 per age-site were collected using the core method in 10 cm depth increments to 1 meter depth.

141 Field moist soil samples were passed through 8- and 2-mm sieves, air dried, and ground with a

142 roller mill (Bailey Manufacturing Inc., Norwalk, IA) to create a fine powder consistency.

143 Identifiable plant materials were removed prior to grinding (Hernandez-Ramirez et al., 2011;

144 Chendev et al., 2015b).

145   Soil organic C mass density for the 0 to 30 cm depth was calculated as the sum of

146   products of organic C concentration (Hernandez-Ramirez et al., 2009), bulk density and soil

147   layer thickness, with units of Mg C ha^{-1}.

148   First-order kinetic modelling follows:

149    $C_{(t)} = C_e + (C_o - C_e)\ e^{-kt}$ [1]

150   where C_e is soil C storage at the oldest cropland site within each chronosequence which was

151   assumed to be at new dynamic equilibrium (i.e., C inputs = C outputs), C_o is soil C storage at the

152   native grassland site which was assumed to be the initial time of land use conversion from native

153   grassland to annual croplands (time zero), k is the fitted first-order kinetic rate constant (yr^{-1}),

154   which is equivalent to C turnover rate or net C mineralization (in the case of net C decreases),

155   and t stands for time (yr). In the case of increases in soil C over time, turnover rates become

156   equivalent to accretion rates. It is possible to model the soil C storage for each year, and hence,

157   the difference between consecutive years provides an estimation of the annual net C change (Mg

158   C ha^{-1} yr^{-1}). First-order kinetic modelling (Eq. [1]) assumes: (i) balance between C inputs and C

159   outputs and (ii) steady state conditions (i.e., $\delta C/\delta t = 0$) (Jastrow et al., 1996; Follett et al., 1997;

160   Hernandez-Ramirez et al., 2011). At the various study sites, the terrain slopes ranged up to 2%,

161   with the exception of the Huron site that had 3% slope. Hence, the general topography in our

162   study sites was classified as flat. As most sites are considered semi-arid, water erosion is

163   assumed minimal; likewise, enough vegetative cover limits wind erosion. Given the dominant

164   flat topography and low rainfall amounts, we also assumed negligible C removals or additions

165   due to erosion or deposition.

166   Mean residence time (MRT) of organic C in the soil was calculated as reciprocal of k.

167   Concomitantly, half-life of organic C in the soil was calculated as follows:

168   Half-life $= \ln(2)/k$ [2]

169   Note that under equilibrium, C $output_e$ is also equivalent to C $input_e$, and they correspond to the

170   annual C that enters and exits the soil C pool, respectively.

171   The performance of the derived first-order kinetic modelling was evaluated with the

172   normalized root mean square error (RMSEn) (Guenette and Hernandez-Ramirez, 2018; Kiani et

173   al., 2020), coefficient of determination (R^2) as well as a leave-one-out cross-validation of

174   predicted versus measured C (n= 6). Within the cross-validation, we tested the regression

175   coefficient (β_1) of a linear regression established for predicted vs. measured C against the 1:1

176   line.

177   **2.2 Comparison of adjacent paired sites in Russia: native grasslands, annual croplands and**

178   **shelterbelts (trees)**

179   In addition to the 3 abovementioned chronosequences, 3 additional sites were studied in

180   Russia: Streletskaya Steppe situated within Kursk oblast (51°32′ N, 36°05′ E), Yamskaya Steppe

181   in Belgorod oblast (51°11′ N, 37°37′ E), and Kamennaya Steppe in Voronezh oblast (51°02′ N,

182   40°44′ E) (Fig. 1). Soils at all these paired sites were classified as loamy Chernozems. Following

183   field methods as described by Laganiere et al. (2010), each site encompassed adjacent locations

184   representing three land uses: native grassland, annual croplands and broadleaf shelterbelts, as

185   described by Chendev et al. (2015b). Soil sample collections were conducted similar as

186   described above. Briefly, composite soil samples (3 subsamples per sample) were collected from

187   the native grassland (n= 6 composite soil samples), annual croplands (24) and shelterbelts (18) in

188   summer 2012. At time of soil sample collection, the ages of the annual croplands in Streletskaya

189   and Yamskaya were at least 140 years, and at least 145 years of age in Kamennaya. In all 3 sites,

190   the shelterbelts had been planted 55 years prior to soil sample collection. Tree species in the

191 shelterbelts include silver birch (*Betula verrucosa*), Manitoba maple (*Acer negundo*), and

192 English oak (*Quercus robur*). Long-term mean annual precipitations at Streletskaya, Yamskaya

193 and Kamennaya correspond to 580, 530 and 480 mm yr^{-1}, respectively.

194   It is noted that although trees at the Streletskaya, Yamskaya and Kamennaya sites were

195 planted 55 years prior to soil sample collection, for first-order kinetics modelling purposes, the

196 tree-C contributions to soil C accrual were accounted beginning from 50 years prior to soil

197 sample collection. This assumption is based on a literature review by Paul et al. (2002) who

198 suggested a lag phase of 5 years for tree-C contributions to effective start contributing to net

199 storage of soil C. Moreover, because of the uncertainty of how close the afforested soils were to

200 steady state and equilibrium of soil C storage, we evaluated two scenarios of first-order kinetics

201 modelling using Eq. [1]. We assessed trajectory 'A' under the premise that full steady state has

202 been reached at time of soil sample collection, and also trajectory 'B' where we assumed that the

203 C storage in these afforested soils had asymptotically reached 95% of the theoretical equilibrium

204 or ceiling capacity. We reported both trajectories and their associated C accretion rates (k).

205   The rationale for implementing the 95% scenario (trajectory 'B') arises from the

206 uncertainty of whether full C equilibrium has been reached. As the final phases in an asymptotic

207 trajectory are incremental, we undertook the evaluation of a narrow but identifiable deviation

208 below full C equilibrium. The choice of 95% represents that an analytical precision for

209 quantifying soil carbon can typically be found within 5%. In other words, based on a principle of

210 detection limit in soil C measurements, a 95% can be considered a minimal deviation from full

211 equilibrium (100%) that is already discernable, but still related to the general variability of the

212 quantification method and associated results. Hence, we took the freedom to assess this plausible

213 range at and below full theoretical equilibrium, with the soil C storage having reached 100% or

214   95% of the ceiling capacity, respectively. From a broader perspective, this alternative 95%

215   scenario also explores and represents the prospect that the soils beneath the shelterbelt could still

216   be incrementally accruing C even 55 years after tree planting.

217   **2.3 Pairwise comparisons in United States: annual croplands and afforestation**

218   Three field sites were studied within the Northern Great Plains of United States near the

219   cities of Huron (South Dakota, 44°15′ N, 98° 15′ W), Norfolk (Nebraska, 42° 03′ N, 97° 22′ W)

220   and Mead (Nebraska, 41°9′ N, 96°29′ W) (Fig. 1). While the focus remained on investigating the

221   turnover rates of soil C as a function of land use changes, it is noted that pedogenic ages of the

222   soils sampled in United States were relatively shorter than the soils studied in Russian sites. This

223   is because of the differences in geological times of parent materials exposure on the ground

224   surface between geographic regions since the sites in United States experienced the last

225   glaciation (i.e., Wisconsin glaciation).

226   The 3 sites in United States encompassed afforested areas and adjacent annual croplands,

227   co-located in paired sites as Laganiere et al. (2010). The native vegetation at the sites had been

228   tallgrass prairie (e.g., big bluestem *Andropogon gerardii* Vitman), which had been converted into

229   annual croplands, and trees were subsequently planted in areas of the croplands. Afforestation

230   took the forms of shelterbelts in Norfolk and Mead, and a forest plantation in Huron. The Huron

231   site also had an adjacent field with a representative undisturbed native prairie, which was also

232   sampled as a reference. In contrast to the long-term croplands in Norfolk and Mead, the cropland

233   at Huron had only 21 years since conversion from native grassland at time of soil sample

234   collection. Field sample collections were conducted in 2004 in Mead, and in 2012 in both

235   Norfolk and Huron. The trees had been planted 19, 35 and 70 years prior to soil sample

236   collections in Huron, Mead and Norfolk, respectively. Tree species included green ash (*Fraxinus*

237 *pennsylvanica* Marshall), red cedar (*Juniperus virginiana* L.), and oak (*Quercus macrocarpa*) in

238 Huron; red cedar, scotch pine (*Pinus sylvestris* L.), and cottonwood (*Populus deltoides* Bartram)

239 in Mead; Siberian elm (*Ulmus pumila*), red mulberry (*Morus rubra*), and cottonwood in Norfolk.

240 Annual croplands were managed under conventional farming practices. Annual crop species at

241 the study sites included wheat (*Triticum aestivum* L.), corn (Zea mays L.), soybean [Glycine max

242 (L.) Merr.], and sorghum (*Sorghum bicolor* L. Moench). An alfalfa (*Medicago sativa* L.) forage

243 field adjacent to the shelterbelt in Norfolk was also sampled. Long-term mean annual

244 precipitation in Huron, Norfolk and Mead were 582, 696 and 747 mm yr^{-1}, respectively. Long-

245 term mean annual air temperature in Huron, Norfolk and Mead were 7.7, 9.6, 9.9 °C,

246 respectively. Overall, study sites had soil pH near neutral and textures between loamy sand to

247 silty clay loam (Table 1)

248 Field methods of soil sample collections had previously been described in related reports

249 by Chendev et al. (2015a) for the Huron and Norfolk sites as well as by Sauer et al. (2007) for

250 Mead. Briefly, spatial grid patterns were established with composite samples (n= 4) collected

251 from each sampling location. Total grid sampling locations were 118 at Mead, 48 at Huron, and

252 42 at Norfolk. Plant tissue samples of the dominant species were also collected from each study

253 site.

254 Organic C concentration and δ^{13}C isotopic composition were determined in all soil and

255 plant samples via the dry combustion method using a Fison NA 15000 Elemental Analyzer

256 (ThermoQuest Corp., Austin, TX) interfaced to an isotope-ratio mass spectrometer Delta V

257 Advantage (Thermo Fisher Scientific, Waltham, MA). Pee Dee Belemnite was used as standard

258 and analytical precision of δ^{13}C measurements was 0.06‰. The δ^{13}C isotopic ratio was

259 expressed as:

260     $\delta^{13}C$ (‰) = [($^{13}C/^{12}C$ sample) / ($^{13}C/^{12}C$ standard) - 1] × 1000   [3]

261   When integrating multiple soil layers of a profile, averages of $\delta^{13}C$ were weighted by the

262   soil C mass density at the corresponding soil layers.

263   Prior to land use conversion to croplands, the native grasslands in United States were

264   undisturbed and dominated by plant species with C4 photosynthetic pathway, with certain mixed

265   presence of C3 species. Based on this legacy contribution of prairie vegetation to soil C over the

266   Holocene, approaches based on stable isotope signatures became feasible in the 3 sites in United

267   States. Furthermore, the ability to use a C stable isotope approach to partition the current soil C

268   into two specific C pools (i.e., remaining prairie-C and new tree-C) requires a constraining

269   assumption that much of the plant residues added yearly over the annual cropping stages

270   decomposes during the following growing season (Gregorich et al., 2017). Therefore, this

271   premise entails that most of C in crop residues enters the soil to become lost back to the

272   atmosphere within a year, and hence, having near-negligible contributions to changes in both net

273   C accrual and $\delta^{13}C$ isotopic composition in the soil. This method enabled us to examine the

274   sources of soil C and also derive the turnover rates of these soil C sources. This primarily applies

275   because trees are C3 species. This approach assumed that the differences in ^{13}C isotopic

276   signatures between C4-C3 mixed (native grassland) and C3 (trees) overrides any potential

277   differential effect of C isotopic fractionations during SOM decomposition of C3 vs. C4

278   substrates, between aboveground and below ground plant materials (roots vs. litter), or because

279   of SOM interactions with soil mineral surfaces (Martin et al., 1990; Hernandez-Ramirez et al.,

280   2011). Assuming mass conservation, the measured soil C storages were allocated into two

281   sources: (i) new tree-C and (ii) remaining prairie-C (native soil) as follows:

282   Tree-C + Prairie-C = 1   [4]

283   Tree-C = (δ^{13}C afforested soil - δ^{13}C native soil)/(δ^{13}C tree - δ^{13}C native soil)   [5]

284   It was inferred that all soil C different from the C identified as new 'Tree-C' was

285   preexisting soil C attributable to remaining 'Prairie-C'. Likewise, we assumed that the 'δ^{13}C

286   native soil' were represented reasonably well by the δ^{13}C measured in soil samples taken from

287   the annually-cropped fields adjacent to the afforested soils. Although they were adjacent, the

288   sampling locations providing the 'δ^{13}C native soil' were sufficiently distant from afforested areas

289   to preclude influence of trees on soil δ^{13}C. The 'δ^{13}C native soil' were -17.3‰ in Huron, -17.0‰

290   in Mead, and -17.5‰ in Norfolk, which are noted to be consistent with each other as these sites

291   share a common natural history of tallgrass prairie native vegetation. These 'δ^{13}C native soil' are

292   also consistent with earlier measurements in prairie soils by Follett et al. (1997) and Hernandez-

293   Ramirez et al. (2011). Furthermore, the δ^{13}C measured in tree samples averaged -27.6‰ in

294   Huron, -26.6‰ in Mead, and -27.9‰ in Norfolk as typical isotopic compositions of C3 plant

295   species. At the Norfolk site, tissue samples of alfalfa canopy yielded -27.4‰. Also, in Norfolk,

296   'δ^{13}C afforested soil' included all data from the soil samples taken within 10 m north and 10 m

297   south from the center of the shelterbelt. In Huron, 'δ^{13}C afforested soil' sample included all data

298   from soil samples taken at least 17 m away from the edge between the forest plantation and the

299   adjacent annually-cropped field. In Mead, as earlier presented by Hernandez-Ramirez et al.

300   (2011), 'δ^{13}C afforested soil' sample included all data from soil samples collected between the

301   existing two tree rows.

302   The mass densities of soil C derived from new tree-C and remaining prairie-C were

303   calculated by multiplying whole soil C storage beneath the trees with the corresponding fractions

304   expressed in Eq. [4].

305   When assessing first-order kinetics modelling (Eq. [1]) of soil C beneath trees, we

306   assumed that the remaining prairie-C in afforested soils at time of soil sample collection was at

307   steady state and had also reached new equilibria in the case of the long-term annual croplands at

308   Norfolk and Mead. In the specific case of Huron, because the afforested soil had experienced

309   only two years of annual cropping prior to tree planting, we assumed that the native grassland

310   prior to land use conversion to annual cropping was at equilibrium and steady state.

311   As abovementioned, the shift from C4-dominated to C3 vegetation in the case of the

312   afforested soils in United States enabled us to methodologically apportion the sources of soil C

313   and to identify these direct contributions from trees to increasing soil C storage. In the case of the

314   Russian sites, soil ^{13}C isotope composition does not resolve these C sources because of the lack

315   of shift between C4 and C3 vegetation in the natural history of these landscapes.

316   Relationships between allocations of soil C sources (% tree-C, and % prairie-C) and time

317   since tree planting (years) were examined through linear regression analyses. Likewise, a similar

318   linear regression was developed for the proportions of remaining prairie-C in the afforested soils

319   relative to whole C present in the adjacent cropland soils. We used SigmaStat Version 4.0

320   software (Systat Software, San Jose, CA) and an α critical level of 0.05. Where error terms (±)

321   are presented, they correspond to the standard errors of the means.

322

323   **3 Results**

324   **3.1 Soil C after conversion of grasslands to cropland and then shelterbelt: Russian cases**

325   Long-term cultivation of native grasslands decreased soil organic C storage in a nonlinear

326   fashion (Fig. 2). Within each of the 3 available land use chronosequences (i.e., each

327   encompassing a range of different duration of cropping since land use conversion), the declining

328   trajectory of soil C was represented reasonably well by first-order kinetic modelling. The

329   RMSEn were all lower than 4% and the R^2 were greater than 90%, which supports the suitable

330   performance of k models (Fig. 2A-C). Likewise, cross-validation results of cropland soils within

331   the age range from 10 to 200 years further indicated the high accuracy of k predictions when

332   compared with the 1:1 agreement line. This was based on a non-significant t-test with $\beta_1=1$ as

333   null hypothesis (Fig. 2D).

334   Within the soil layer of 0 to 30 cm depth in the chronosequences in Belgorod, Russia,

335   turnover rates (k) of soil C ranged from 0.0091 to 0.0183 yr^{-1} in Gubkinskiy and Prokhorovskiy,

336   respectively. Over the entire time spans of the 3 chronosequences (up to 250 years old), net soil

337   C losses were in the relatively narrow range from 31.2 Mg C ha^{-1} in Prokhorovskiy (Fig. 2A) to

338   36.9 Mg C ha^{-1} in Gubkinskiy (Fig. 2B). Focusing on these losses of the preexisting soil C, the

339   estimated lapses for half of these C losses to take place was between 38 and 76 years after the

340   time of land use conversion from native grassland to annual cropland (Fig. 2A and Fig. 2B,

341   respectively). It is noted that new dynamic equilibria were assumed to have taken place in the

342   oldest cropland soil within each chronosequence (i.e., > 200 yrs) as part of first-order kinetics

343   modelling. Furthermore, when examining the assumption of steady state (i.e., $\delta C/\delta t = 0$), soil C

344   trajectories at both Prokhorovskiy and Ivnyanskiy sites showed reasonable approximations to

345   this premise, with relatively low annual C losses occurring towards the end of these

346   chronosequences. On the other hand, the Gubkinskiy site still exhibited vigorous C losses at the

347   end of this chronosequence, challenging the steady state assumption at this site. In further details,

348   the trajectory at the Prokhorovskiy site showed estimated C losses of only 9.4 Kg C ha^{-1} yr^{-1}

349   during the last year of this chronosequence (Fig. 2A), which can be considered negligible and in

350   clear agreement with a steady state condition. Conversely, the last year of the Gubkinskiy

351   chronosequence still showed a C loss of 34.5 Kg C ha^{-1} yr^{-1}. It is noteworthy that these annual C

352   outputs typically take the form of soil respiratory losses (CO_2-C) that are resultant from

353   microbial mineralization of existing SOM. These estimations of net changes at near steady state

354   do not account for the CO_2 derived from the decomposition of recently-added plant residues, but

355   just the net change in SOM-C (Fig. 2A). For comparison purposes, the k-modelled trajectory of

356   the Prokhorovskiy chronosequence had estimated C losses of 566 Kg C ha^{-1} yr^{-1} during the very

357   first year after conversion from native grassland to annual cropland.

358   Based on the assessed pairwise comparisons, afforestation in the form of shelterbelts

359   replenished soil C storage after long-term annual cropping had led to decreasing soil C compared

360   with adjacent native grasslands (Table 2). Of the substantial soil C storage that had been depleted

361   over time during annual cropping (i.e., -18.9 ± 5.3 Mg C ha^{-1}), afforestation replenished on

362   average 81% of these cropping-induced C losses (Table 2).

363   Using the insights gained from both chronosequences (Fig. 2) and pair site comparisons

364   (Table 2), we undertook the reconstruction of soil C storage progression in the 0 to 30 cm soil

365   layer since the land use conversion from native grassland to annual cropland and subsequently

366   into shelterbelt (Fig. 3). After normalizing all cropland-chronosequence data (i.e., zero to one;

367   dimensionless), turnover rates (k) and first-order kinetic models of soil C storage were estimated

368   (Fig. 3). This long-term k model of soil C depletion in cropped soil had a reassuring coefficient

369   of determination (R^2) of 90% and a very low RMSEn of only 3.34%, which collectively indicates

370   the high precision of the k model. Over 250 years of cropland chronosequence, the C turnover

371   rate was quantified as 0.010 ± 0.004 years^{-1}, which is equivalent to a MRT of 100 years and a

372   half-life of soil C of 69 years. This first-order trajectory of soil C depletion in croplands

373   indicated that 28.9% of the initial soil C under native grassland was gradually lost − i.e., very

374   likely to the atmosphere − over 250 years of annual cropping (i.e., from 1 to 0.711, Fig. 3). In

375   further details, during the first year of cropping, we estimated that 0.310% of the pre-existing C

376   was lost from the soil. Conversely, after 250 years, during the last year of the cropland-

377   chronosequence trajectory, soil C losses were only 0.026% of the initial soil C – this is one order

378   of magnitude lower than calculated for the first year of cropping. This deceleration in SOM

379   mineralization while approaching a new equilibrium and at near steady state was captured

380   reasonably well by first-order kinetics. Based on the soil C initially present under native

381   grassland soils (3-chronosequence mean= 125.3 Mg C ha^{-1}, Fig. 2), these values of 0.31% and

382   0.026% were equivalent to C outputs of 392 and 33 Kg C ha^{-1} yr^{-1}, respectively.

383   We projected two potential trajectories (A and B) of how afforested soils can restore soil

384   C storage in cropland soils over five decades (Fig. 3). After replenishing 81% of the C lost

385   during long-term annual cropping, the shelterbelts had 94.5% of the initial C of the native

386   grassland (i.e., $0.81 \times 0.289 + 0.711 = 0.945$; Table 2). Trajectory A was estimated on the basis

387   that afforested soils fully reached a new steady state and equilibrium of soil C storage with first-

388   order modelling. This trajectory showed a steep increase in soil C storage over the first decade of

389   tree planting. In fact, the C accretion rate for trajectory A was 0.119 years^{-1}, which suggests a

390   potential for high soil C accretion under fast C cycling with afforestation. Because the soils

391   beneath the shelterbelt can still be actively accruing C even 55 years after tree planting, we also

392   developed trajectory B, which targets a scenario where soil C storage reached 95% of a

393   theoretical equilibrium (Fig. 3). For this trajectory, the resultant C accretion rate was 0.0334

394   years^{-1}, which corresponds to a modelled MRT of 30 years. When focusing on these progressive

395   gains of new soil C under trajectory B, the estimated time for half of this soil C portion to enter

396   the soil was 21 years. Over the two last decades of this progression, the soil C accretion starts to

397   gradually become asymptotic. During the first year of trajectory B, the net accrual of soil C was

398   equivalent to 0.88% of the soil C initially present in the native grasslands (Fig. 3). Conversely,

399   during the last year of trajectory B, soil C accretion corresponded to only 0.17%. Based on the

400   soil C initially present under native grassland soils (3-chronosequence mean= 125.3 Mg C ha^{-1},

401   Fig. 2), these 0.88% and 0.017% values were equivalent to a sizable 1.10 and 0.21 Mg C ha^{-1} yr^{-}

402   ^{1}, respectively. It is noted that in each of the two afforestation trajectories of soil C accretion (A

403   and B), the annual contributions of afforestation to net accrual of soil C began from the soil C

404   storage estimated by the k-modelled cropland trajectory 50 years prior to soil sample collection

405   (i.e., 0.728, Fig. 3). This is because we had assumed a lag phase of 5 years as noted above, and

406   tree planting was 55 years prior to soil sample collection.

407   Based on the assembled k models of cropland-C turnover and simultaneous tree-C

408   accretion (Fig. 3), of the whole soil C measured beneath the trees at the time of soil sample

409   collection (Table 2), 25% was estimated to be derived directly from tree-C contributions [i.e.,

410   $(0.945 - 0.711) / 0.945$]. This can indicate that although tree-C contributions were substantial,

411   the majority of the C stored in these steppe soils still originated from the initial native grassland

412   before land use conversion to annual croplands.

413   **3.2 Sources and turnover of soil C in afforested croplands: United States cases**

414   Larger accumulation of soil C was consistently found beneath trees relative to the

415   adjacent annual croplands in all 3 study sites within the Northern Great Plains of the United

416   States. At the shelterbelts in Norfolk (Fig. 4E) and Mead (data not shown) as well as the forest

417   plantation in Huron (Fig. 4F), the stable isotope approach followed by mass balance effectively

418   allocated and quantified the tree-derived soil C, in particular within the 0 to 15 cm soil depth

419   increment (Table 3).

420   At the Norfolk site, soils collected from the 0 to 15 cm depth increment beneath the trees

421   (i.e., within 10 m distance from the center of the shelterbelt) resulted in more than double of the

422   C mass density found in the annually-cropped topsoils that were located farthest from the trees

423   (28 vs. 13 Mg C ha^{-1}; Table 3, Fig. 4A). Concurrently, when comparing the same surface layer

424   and spatial sampling locations, soil δ^{13}C sharply shifted from a considerably depleted -25.7 $\pm$ 0.1

425   ‰ beneath the trees to -17.5 $\pm$ 0.1 ‰ in the cropped soils north from the shelterbelt (Fig. 4C). As

426   a result, a significant 79% of the soil C storage measured beneath the trees at the time of sample

427   collection was attributed to tree-C contributions (Table 3). This translated into a substantial

428   magnitude of 22 Mg C ha^{-1} being derived specifically from tree biomass (Table 3). Moreover, as

429   stated above (Method section 2.3), the rest of the soil C was attributed to remaining prairie-C.

430   The existing soil C beneath trees in the Norfolk shelterbelt allotted to remaining prairie-C was

431   only 45.5 $\pm$ 0.3 % of the whole soil C typically found in the adjacent annual crop field (Table 3).

432   This indicated that 54.5% of the soil C (equivalent to 13.0 - 5.9 = 7.1 $\pm$ 0.4 Mg C ha^{-1}, Table 3)

433   that existed under the long-term annual cropland (i.e., assumed to be at steady state) prior to tree

434   planting has been lost from the topsoil over the 70 years of afforestation at Norfolk. This net

435   decline in remaining prairie-C is attributable to CO_2 respiratory losses from enhanced biological

436   activity beneath the trees that gradually accessed, mobilized, cycled and partly mineralized this

437   legacy prairie-C pool. These results indicated that the turnover of remaining prairie-C in

438   afforested soils can be even faster than in open cropland fields.

439   Because the crop field south from the shelterbelt in Norfolk was dedicated to perennial

440   cropping of alfalfa forage – a C3 species, we undertook a mass balance to distinguish and

441   allocate the C sources as alfalfa-C vs. remaining prairie-C, with a similar approach as in the

442   afforested areas (i.e., Eq. [4] and [5]). Within the 0 to 15 cm depth increment, soil C storage

443   under alfalfa (i.e., 13.1 ± 1.0 Mg C ha^{-1}) was the same as in the annual cropland on the north side

444   of the shelterbelt (Fig. 4A; Table 3). However, the soil δ^{13}C shifted to -21.6 ± 0.4 ‰, which

445   resulted in a 41.2 ± 0.4% replacement of the whole soil C storage being derived specifically from

446   recent contributions of alfalfa-C in this perennial forage field.

447   Similar to Norfolk, the shelterbelt at Mead also showed a major contribution of

448   afforestation to whole soil C storage between tree rows in the 0 to 15 cm depth increment (i.e.,

449   17 Mg C ha^{-1}, Table 3), which corresponded to 37% of the whole soil C. It is noted that although

450   the magnitude of original prairie-C lost after 35-yr of afforestation at Mead (i.e., 36.2 - 29.3 =

451   6.9 ± 0.7 Mg C ha^{-1}, Table 3) was comparable to Norfolk, the proportion of this original prairie-

452   C lost from afforested soils in Mead was much smaller with only 19% (i.e., 100 – 80.9; Table 3).

453   This can indicate that the net changes in whole soil C storage under afforestation (i.e.,

454   simultaneously encompassing the noted prairie-C losses and the asymmetrically-larger tree-C

455   gains) did not follow a fixed proportionality to the initial prairie-C.

456   At the 19-year-old forest plantation at Huron, trees also increased soil C storage while

457   decreasing the δ^{13}C signature, in particular in the 0 to 15 cm depth increment (Fig. 4B, Fig. 4D).

458   When contrasting the afforested soils collected at least 17 m away from the plantation boundary

459   vs. topsoils taken within the adjacent cropland from the sampling locations that were farthest

460   removed from the trees, δ^{13}C changed from -20.7 ± 0.2 ‰ to -17.3± 0.1 ‰ ‰, respectively.

461   Although trees were much younger in Huron than in Mead and Norfolk, a considerable

462   magnitude of tree-C was found (i.e., 12 Mg C ha^{-1}, Table 3). This indicated that direct tree

463   contributions to soil C storage can take place rather quickly, within a few decades following

464   afforestation. However, the changes in the remaining prairie-C pool beneath the trees at Huron

465   apparently differed from what was found in both Mead and Norfolk. While soils beneath trees at

466   both Mead and Norfolk showed declines in remaining prairie-C relative to whole soil C stored in

467   open cropland fields at the time of sample collection, the afforested soil at Huron showed no

468   change in the magnitude of remaining prairie-C in the 0 to 15 cm depth increment. In fact, it is

469   striking how similar the whole C in the cropped soil (25.6 ± 0.7 Mg C ha^{-1}) was to the C

470   allocated to the remaining prairie-C pool beneath the trees (25.3 ± 0.7 Mg C ha^{-1}, Table 3). What

471   is more, the soils sampled from the adjacent native grassland within the Huron site also returned

472   a very consistent magnitude of soil C storage, with 25.4 ± 1.9 Mg C ha^{-1} (n= 3, data not shown).

473   Provided the uncertainty of field sampling, this evidence strongly indicated that all or nearly all

474   the original prairie-C was retained and still present in the Huron soils under both annual cropping

475   and afforestation (Table 3). Likewise, when examining the 0 to 30 cm soil depth increment at

476   Huron (i.e., aggregating the two sampled soil layers shown in Fig. 4B), we further corroborated

477   this similarity in soil C storage between native grassland and annual cropland (45.4 ± 1.0 vs. 44.2

478   ± 1.1 Mg C ha^{-1}, respectively, data not shown)

479   Significant regressions revealed the consistent dependency of C source allocations on

480   time since tree planting (Fig. 5). Over time following afforestation, tree-C source increased

481   linearly from an assumed null contribution at planting to become 79% of the whole soil C after

482   70 years of tree planting in Norfolk (R^2= 0.95, Fig. 5A). We also evaluated changes over time for

483   the remaining prairie-C in afforested soils relative to the corresponding adjacent croplands within

484   each study site. Linearity of these prairie-C proportions as a function of time was also observed

485   when encompassing the 3 study sites (R^2= 0.999, Fig. 5B). As described above, the more

486   recently-afforested soils at the Huron site kept the entire prairie-C, while the oldest afforested

487   soils at the Norfolk site retained less than half of the whole soil C present in the adjacent annual

488   croplands (45.5%).

489   We focused on estimating the turnover rates of soil C mass density derived directly from

490   tree-C sources while encompassing the range of conditions in the 3 studies. As most of the

491   beneficial effects of tree planting across the sites in the United States were detected in the 0 to 15

492   cm soil depth increment, further examination of accretion rates of soil C storage focused on this

493   specific topsoil layer. Upon assembling the magnitudes of tree-C contributions over time since

494   afforestation, unified first-order kinetics modelling converged and emerged robustly (R^2= 0.997,

495   Fig. 6). The k rate constant of 0.0552 years^{-1} corresponds to a half-life of 12.6 years, which

496   indicates that more than half of the accrual tree-C occurred within less than two decades (when

497   accounting for a lag phase of 5 years following tree planting) (Fig. 6). This further substantiated

498   the rapid contributions of afforestation to increase soil C storage quickly until reaching a new

499   dynamic equilibrium. This generalized relationship enabled projecting tree-C accruals in

500   afforested soils within the assessed time range of 70 years. We further implemented this robust

501   k-progression to simultaneously depict the gains in tree-C while also representing the declines in

502   prairie-C in afforested soils for each study site separately (Fig. 7). This approach accounts for the

503   C that is being lost from net mineralization of pre-existing C in the remaining prairie SOM (Fig.

504   7). It was noticeable that the afforested soils at Mead showed faster turnover rate of the

505   remaining prairie-C than the other two sites by approximately two-fold. The k rate constant of

506   net mineralization of prairie-C beneath trees at Mead was 0.145 years^{-1} (Fig. 7B), which

507   corresponded to an MRT of about 7 years. This implies that the average time for prairie-C to be

508   lost from Mead afforested soils was well within one decade, whereas prairie-C in afforested soils

509   in Huron and Norfolk showed longer residence times by about double.

510

511   **4 Discussion**

4.1 Carbon contributions from trees to SOM sequestration

Planting trees in croplands creates substantial sinks of atmospheric C in the soil profile (Sauer et al., 2007; Khaleel et al., 2020; Zhang et al., 2020). Current knowledge of this important benefit of afforestation has been deepened and reinforced earlier literature (Post and Kwon, 2000; Hernandez-Ramirez et al., 2011; Chendev et al., 2015b). It is noticeable that having long-term annual croplands as the land use system prior to establishing trees particularly enlarges the soil C sink and replenishment caused by afforestation (Guo and Gifford, 2002; Laganiere et al., 2010; Sauer et al., 2012). Overall results indicate that across tree species and local edaphic-climatic conditions at the studied sites, the massive tree-C contributions through decaying roots and litter (Li et al., 2012; Amadi et al., 2016) can saturate the soil with C substrates in surplus to the capacity of microbial decomposition (Li et al., 2018; Deng et al., 2014), which collectively incline the C balance towards net C accrual (Hernandez-Ramirez et al., 2011). Our quantification of these tree biomass-C contributions to newly-accrued soil C further expand this growing body of knowledge. The proportion of new soil C originated from tree biomass were shown to increase significantly with time (Fig. 5A).

By contrast to afforestation, long-term annual cropping implies recurrent soil mixing, disruption of any preexistent vertical stratification, microclimate fluctuation, and exposure of SOM to decomposition, which collectively shift the C balance and predispose towards depletion of soil C (Post and Kwon, 2000; Hernandez-Ramirez et al., 2009; Curtin et al., 2014). In addition to this disturbance and exposure of SOM to decomposition, low C inputs and high C removals via harvest are also characteristic of conventional annual cropping systems. Moreover, results indicated that C from crop residues was largely lost back to the atmosphere every year with no significant net contribution to the soil organic C pools in the long term (Fig. 2, Fig. 3). A

535   reduction of C inputs in croplands once the native vegetation (roots and aboveground biomass)

536   have been removed (Hu et al. 2013) is also typically followed by alterations in soil physical

537   properties such as decreases in porosity and gas exchange which can become detrimental to plant

538   primary productivity and soil biology (Kiani et al., 2017).

539   Our study explicitly examined and quantified for first time in literature the losses of

540   remaining prairie-C directly beneath trees across afforested soils (Fig. 3, Fig. 5B, Fig. 7). This

541   analysis showed that under afforestation, soil C remaining from original native grasslands

542   continues to be lost from the profile, likely via microbial mineralization (Fig. 3, Fig. 7). It is

543   noted that the accretion of recently-added tree-C is much faster than these observed losses of

544   remaining prairie-C beneath trees as the recently-added plant-C is considered relatively more

545   labile than prairie-C. The noted decline in remaining prairie-C beneath young afforestation

546   agrees well with a decomposition of SOM in the early stage of tree growth as previously

547   deliberated by Paul et al. (2002), Garten (2002), and Xiong et al. (2020). At the Norfolk site,

548   tree-C contributions effectively replenished and greatly surpassed the gradual losses of

549   remaining prairie-C in the soil (Fig. 7C). In the case of the Huron site, afforestation conserved

550   the initial prairie-C while also contributing directly to additional tree-C accrued in an overall

551   increasing SOM pool (Fig. 7A)

552   It is noted that although the 3 US sites (Norfolk, Huron and Mead) shared a common

553   trajectory of tree-C accretion with time (Fig. 5, Fig. 6), their k turnover rates of remaining

554   prairie-C differed (Fig. 7). These apparent divergences are potentially attributable to differences

555   in temperature and moisture regimes across the region. In further details, in the case of accretion

556   of tree-C in afforested soils, this response to land use change seems governed mostly by the

557   change into tree vegetation and the duration of afforestation; therefore, it became feasible for us

558 to establish a unified, robust k model across a range of afforestation ages (Fig. 6). Conversely,

559 loss rates of remaining prairie-C in afforested soils appeared to be mostly contextual and even

560 site specific, likely as a function of local climatic conditions (Chendev et al. 2014, 2015b).

561 Relative to both Norfolk and Huron (Fig. 7), warmer-wetter conditions in Mead could have led

562 to the faster C turnover rate and mineralization of the remaining prairie-C in these afforested

563 soils (Fig. 7B). Overall, these results exemplify how analyzing the compartments of soil C

564 turnover – evaluating separately tree-C contributions and remaining prairie-C, instead of

565 studying only the whole soil C – can provide further insights into SOM dynamics following land

566 use conversions. Future research could address the potential existence of underlying thresholds

567 of heat and moisture availabilities that are conducive to retain and converse pre-existing prairie-

568 C in afforested soils while simultaneously enabling soil C accretion directly from new tree-C

569 contributions. Likewise, SOM fractionation approaches offer excellent avenues for further

570 unravelling the stabilization mechanisms of C in the soil.

571   Based on the kinetics-modelled reconstruction of soil C storage over time in the Russian

572 land use chronosequences (i.e., encompassing a range of different ages since conversion to

573 annual cropping), over the 55 years that elapsed since tree planting until soil sample collection,

574 the remaining soil C from the original native grassland was shown to be lost continually (Fig. 3).

575 Our turnover estimations using kinetics modelling suggested that only 1.7% of the initial

576 grassland-C was lost over these 55 years following shelterbelt afforestation (Fig. 3). Based on

577 these results from the Russian chronosequences, the relatively small grassland-C loss is in part

578 because soil C had been depleted over nearly two centuries of annual cropping prior to tree

579 planting. Nevertheless, mycorrhizae activity in afforested soils can preferentially access and

580 utilize remaining grassland-C beneath trees (Mellor et al., 2013). Hence, this biological effect

581 could contribute to gradual decreases in remaining grassland-C in afforested soils.

582 Chendev et al. (2014, 2015b) further addressed differences in soil C accrual across

583 afforested sites in Russia and the United States, also attributing them primarily to differences in

584 moisture regimes. Within each geographic region as well as in the collective of both countries,

585 they explained that cooler-moister conditions led to increases in overall soil C accrual beneath

586 trees. This postulate is clearly in line with earlier results by Garten (2002). Potential increases in

587 plant primary productivity with increasing moisture as well as reductions in microbial

588 mineralization of the overall SOM with colder conditions can shift and drive the C balance in the

589 soil towards net C accrual.

590 Of the 3 paired sites in Russia (Table 2), the Yamskaya site is probably the most

591 representative and closely related to the 3 long-term chronosequences evaluated in this study.

592 This is because Yamskaya and the 3 chronosequences are all geographically located within the

593 Belgorod oblast, and hence, they share a more similar regional climate. It was striking that the

594 afforested soil at Yamskaya site had a soil C accretion even greater than the native grassland

595 reference, which strongly indicated the high capacity of shelterbelts to sequester C even beyond

596 the capacity of the corresponding native ecosystem (Table 2). After noticing this finding, it can

597 also be anticipated that although the drier Kamennaya site showed at the present the slowest soil

598 C accretion following afforestation of SOM-depleted croplands (i.e., 5% of C restoration; Table

599 2), it is possible that in the long term, this drier environment can gradually sequestered even

600 more soil C than the moist sites located in Belgorod oblast (e.g., Yamskaya). This is suggested as

601 the Kamennaya site exhibited the highest soil C storage when comparing across all the native

602 grasslands compiled in our study (i.e., 152.5 Mg C ha^{-1}; Table 2 and Fig. 2).

4.2 Turnover rates of soil carbon as a function of land use changes

603

604   This study clearly confirms that the long-term dynamics of soil C is consistently

605   nonlinear, either during decline or accumulation of soil C as a function of land use choices. As

606   deliberated earlier by Post and Kwon (2000) and Garten (2002), erroneously assuming linearity

607   in depicting these trajectories of soil C would lead to underestimating the rates of soil C changes

608   during the first decades following a land use conversion as well as overestimating the turnover

609   rates of soil C after multiple decades once the ecosystem has actually reached stability and a

610   balance between their C inputs and outputs. This latter notion essentially applies when long-term

611   cropland or afforested fields have become mature (Hernandez-Ramirez et al., 2011)

612   Earlier chronosequence and stable isotope analyses by Arrouays et al. (1995) in

613   Southwest France further support that land use effects on soil C changes take place rather

614   quickly. They reported that a new equilibrium in soil C storage was reached within only few

615   decades of a land use change from forest to annual croplands, and with about half of the C loss

616   occurring rapidly within few years (< 10) of beginning cultivation (Arrouays et al., 1995).

617   Similarly, as in our study, only a few decades seems to be required to reach equilibrium when

618   switching from cropland to trees (Richter et al., 1999; Paul et al., 2002; Guo and Gifford, 2002).

619   Likewise, comprehensive results by Dhillon and Van Rees (2017) depicting soil C accretion

620   caused by afforestation in the Canadian Prairies can be interpreted as net C losses taking place

621   over the first several years after tree planting, and subsequently, an ensuing fast accrual of soil C

622   until tree ages of about 35 years when new equilibria or C sequestration ceilings under

623   afforestation can be reached. In our study, MRT of soil C beneath trees were in general

624   determined to be about two decades (Fig. 6). Furthermore, the evaluation of our two scenarios of

625   asymptotic equilibria of C accrual in afforested soils (i.e., trajectories A and B in the normalized

626 Russian chronosequences; Fig. 3) can provide the boundaries of faster vs. slower accretion rates

627 of soil C with the corresponding MRTs of one vs. three decades.

628 In the case of long-term annual croplands in Russia, the soil C MRT of 100 years found

629 in our study (i.e., associated with a k rate constant of 0.010 years^{-1}, Fig. 3) is comparable to

630 findings by Huggins et al. (1998) who registered MRTs of 91 and 143 years in annual cropping

631 systems in Minnesota, but overall longer than a report by Collins et al. (1999) who found a wide

632 range of MRTs between 18 to 96 years for sites with 8 to 33 year-old continuous maize cropping

633 across the Central United States, respectively.

634 It is noted that the exponential first-order trajectory of soil C turnover in the Russian

635 chronosequences (Fig. 3) was captured better with Eq. [1] than the simplistic $C_{(t)} = C_o$ x e^{-kt}

636 previously used by Hernandez-Ramirez et al. (2011). While Eq. [1] provided an R^2 of 90% (Fig.

637 3), $C_{(t)} = C_o$ x e^{-kt} returned an R^2 of 68% (data not shown). With two fitting parameters (i.e., C_o,

638 C_e), first-order kinetic modelling with Eq. [1] represented reasonably well the assumptions of

639 steady state and new equilibrium at the end of the evaluated time series (Fig. 3, Fig. 6).

640 Further kinetics modelling efforts of soil C increases in afforested systems can take the

641 form of two functional C pools where inputs and outputs to labile and recalcitrant SOM can be

642 predicted (Arrouays et al., 1995; Garten, 2002; Hernandez-Ramirez et al., 2009; Xiong et al.,

643 2020). Preferentially accruing C into recalcitrant vs. labile SOM pools in afforested soils can be

644 interpreted as tree-C contributions towards long- vs. short-term stability of soil C storage,

645 respectively, with crucial ramifications for mitigation of future climate change (Laganiere et al.,

646 2010; Hernandez-Ramirez et al., 2011; Deng et al., 2014). Future investigations can also focus

647 on the protection and stabilization mechanisms of SOM as created by soil aggregate formation

648 beneath trees (Kiani et al., 2017; Quesada et al., 2020). Once soils subjected to long-term annual

649   cropping are converted to permanent vegetation, fungal hyphae can become an important means

650   that mediates C accretion by enhancing soil aggregation (Jastrow et al., 1996; Kiani et al., 2017).

651   Jastrow et al. (1996) indicated that fungal hyphae could improve macroaggregation and hence

652   indirectly enhance C accrual. Using phospholipid fatty acid biomarkers, Kiani et al. (2017)

653   identified a linkage between presence of fungal biomass and increases in hierarchical fractal

654   aggregation specifically in forest soils, while this association was absent in the adjacent

655   annually-cropped soils in their study. Furthermore, Quesada et al. (2020) recently discussed the

656   mechanisms for soil C accretion in tropical forests. In line with earlier findings by Wang et al.

657   (2016), Quesada et al. (2020) stated that SOM physical protection provided by the formation of

658   soil aggregates slows decomposition of SOM within aggregates, and hence, it becomes a second

659   layer of stabilization after realizing the primary SOM stabilizing effects caused by mineral

660   surfaces of fine soil particles such as silt and clay. Further studies can focus on the effects of

661   inherent mineralogy and texture as well as clay lessivage processes on C dynamics and storage in

662   afforested soils (Chendev et al., 2020; Quesada et al., 2020).

663

664   **5 Summary**

665   Nonlinear turnover rates of soil C revealed an MRT of a century in long-term croplands

666   as soil C slowly undergoes depletion and losses to the atmosphere. Likewise, when croplands

667   were afforested, nonlinear accretion rates of soil C indicated a MRT of approximately two

668   decades following afforestation. Soil C showed to be rapidly accrued as trees remove CO_2 from

669   the atmosphere and contribute C substrates for SOM accumulation and stabilization. While our

670   study confirmed these substantial C accruals in the soils under the trees, the overall gain at the

671   actual landscape scale will depend in part on the proportion of farmland dedicated to

672   afforestation, with afforested areas typically accounting for up to 5% of the farmlands (Amadi et

673   al., 2016).

674   Our focus on soil organic C behavior in soils under shelterbelts is only part of a broader

675   range of studies evaluating the overarching impacts of agroforestry on soil quality and crop yield

676   across the landscape. Beyond C sequestration, the benefits of shelterbelts can also be manifested

677   in improving the local climate as well as increasing crop yields. Results collectively

678   substantiated that in addition to multiple benefits by trees such as providing air quality,

679   microclimate regulation and erosion control (Sauer et al., 2007; Hernandez-Ramirez et al., 2012),

680   C sequestration in afforested lands is a suitable means to proactively address and effectively

681   mitigate ongoing climate change within a person's lifetime.

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Data availability

Data are available on request.

Author contribution

GHR: Conceptualization; Methodology; Formal analysis; Funding acquisition; Resources;

Visualization; Writing the original draft; Review and editing new versions; Corresponding

author role.

TJS and YGC: Conceptualization; Data curation; Formal analysis; Funding acquisition;

Investigation; Methodology; Supervision; Project administration; Resources; Visualization;

Review and editing new versions.

ANG: Investigation; Methodology; Funding acquisition; Review and editing new versions.

Competing interests.

The authors declare that they have no conflict of interest.

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Table 1. Topsoil pH and textures at the 9 study sites. Numerals correspond to sites as shown from west to east in Fig. 1.

Site	Soil pH	Soil texture
1. Huron†	7.0	sandy loam
2. Norfolk†	6.8	loamy sand
3. Mead†	6.1	silty clay loam
4. Streletskaya Steppe‡	7.0	loam
5. Ivnyanskiy‡	7.5	silt loam
6. Prokhorovskiy‡	7.2	loam
7. Gubkinskiy‡	7.4	clay loam
8. Yamskaya Steppe‡	7.2	loam
9. Kamennaya Steppe‡	7.6	clay loam

† correspond to the cropland location within this site.
‡ correspond to the native grassland location within this site.

Table 2. Soil organic carbon storages and differences within the 0 to 30 cm depth increment under three land uses (i.e., native grasslands, annual croplands and afforestation) in Russia. The nine values of soil C storage across the nine site-land uses were previously presented and discussed in Chendev et al. (2015b) and are repeated here for informing first-order kinetic modelling and estimations of C accretion rates when converting from annual croplands to afforestation as shown in Fig. 3. At time of soil sample collection, the ages of the annual croplands in Streletskaya and Yamskaya were at least 140 years, and at least 145 years of age in Kamennaya. In all 3 sites, the shelterbelts had been planted 55 years prior to soil sample collection.

Land use or descriptor	Streletskaya Steppe site, Kursk	Yamskaya Steppe site, Belgorod	Kamennaya Steppe site, Voronezh	3-sites mean	Standard error
	Soil C mass density (Mg C ha^{-1})				
Native grassland (G)	126.2	138.0	152.5	138.9	7.61
Annual cropland (C)	109.3	127.2	123.6	120.0	5.47
Shelterbelt (Trees)	126.4	142.1	125.0	131.2	5.48
Net decrease G-to-C	-16.9	-10.8	-28.9	-18.9	5.32
Net increase C-to-Trees	17.1	14.9	1.40	11.1	4.91
G-to-C / C-to-Trees†	1.01	1.38	0.05	0.81	0.40

†Ratio representing the replenishing of depleted soil C by tree planting. These ratios were calculated as the absolute values of net increase from cropland to shelterbelt (trees) divided by net decrease from grassland to cropland.

Table 3. Soil organic carbon storage within the 0 to 15 cm depth increment contrasting annual croplands and afforestation in United States. Sources of soil C storage beneath the trees were allocated as tree-C versus remaining prairie-C using Eq. [4] and [5] and associated assumptions. Error bars are standard errors of the means.

Site	Tree age	Whole soil C mass density		Tree-C contribution to soil C		Remaining prairie-C		Remaining prairie-C / whole C in cropland†
	(years)	Annual cropland (Mg C ha^{-1})	Beneath trees (Mg C ha^{-1})	Mass density (Mg C ha^{-1})	Proportion (%)	Mass density (Mg C ha^{-1})	Proportion (%)	Proportion (%)
Huron, S. Dakota	19	25.6 ± 0.7	37.5 ± 1.2	12.2 ± 0.8	32.6 ± 1.5	25.3 ± 0.7	67.4 ± 1.5	98.8 ± 2.8
Mead, Nebraska	35	36.2 ± 0.4	46.7 ± 1.5	17.4 ± 1.2	37.2 ± 1.8	29.3 ± 1.9	62.8 ± 1.8	80.9 ± 3.6
Norfolk, Nebraska	70	13.0 ± 1.3	28.0 ± 1.4	22.1 ± 1.1	78.9 ± 1.0	5.90 ± 0.42	21.1 ± 1.0	45.5 ± 0.3

† This ratio represents the proportion of remaining prairie-C relative to whole soil C in annual cropland. The magnitudes of both 'remaining prairie-C' and 'whole soil C in annual cropland' are shown in other columns of this same table. It is noted that balance (e.g. in Mead, 100 - 81 = 19) represents to the proportion of prairie-C lost in afforested soils since tree planting.
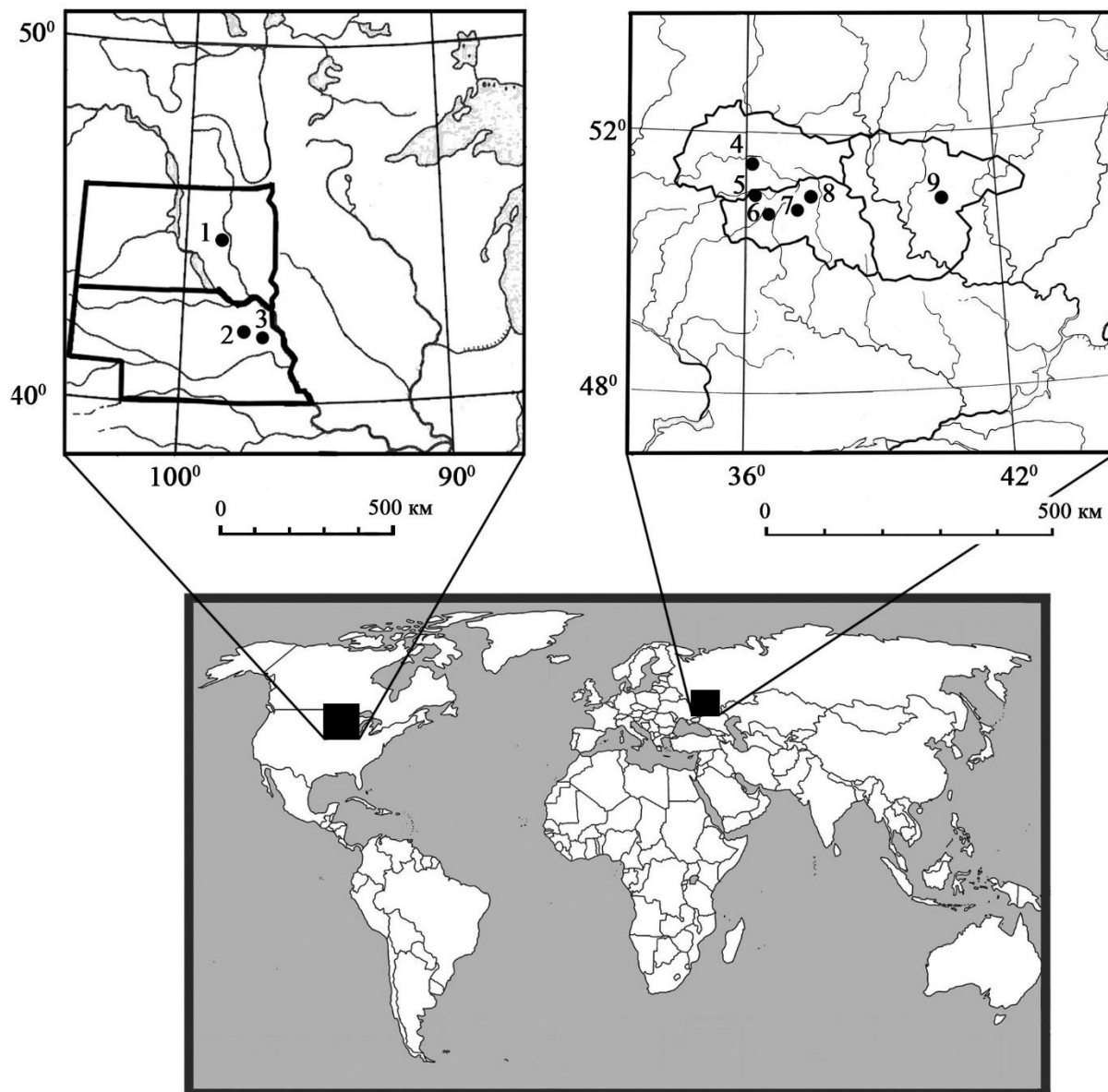
Fig. 1. Geographical location of the nine study sites within the United States (1. Huron, 2. Norfolk, 3. Mead) and Russia (4. Streletskaya Steppe, 5. Ivnyanskiy, 6. Prokhorovskiy, 7. Gubkinskiy, 8. Yamskaya Steppe, 9. Kamennaya Steppe). Within Russia, 5, 6, and 7 are sites with chronosequences of land use conversion, while 4, 8 and 9 are paired sites (native grasslands, annual croplands versus shelterbelts). The 3 sites in the United States are paired sites (afforestation vs. adjacent annual croplands).
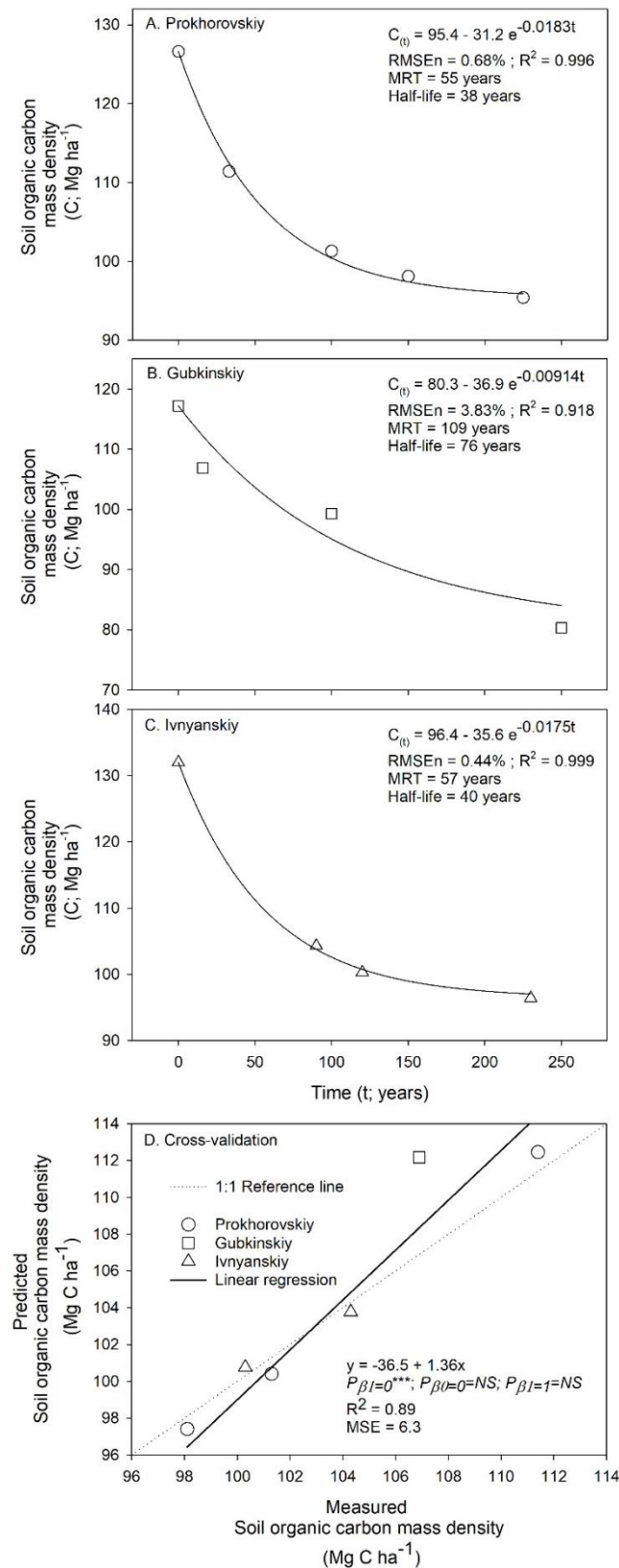
Fig. 2. Land use chronosequences of soil organic carbon storage within the 0 to 30 cm depth increment after converting native grassland to annual croplands in Belgorod oblast, Russia. These showed soil C declines over time. (A) Prokhorovskiy, (B) Gubkinskiy and (C) Ivnyanskiy districts. In Panels A, B and C, first-order kinetic models are described by the solid curvilinear fittings and equations in the form $C_{(t)} = C_e + (C_o - C_e) e^{-kt}$ where C_e is C at new dynamic equilibrium (inputs = outputs), C_o is initial C at time of land use conversion (time zero), k is the first-order kinetic rate constant equivalent to turnover rate. As reciprocal of k, MRT stands for mean residence time, while half-life equates to ln(2)/k. (D) Cross-validation of first-order predicted C versus measured C encompassing the 3 chronosequences within the age range from 10 to 200 years. The subscripts of the *P* values denote the null hypotheses for testing the regression coefficient (β_1) and intercept (β_0), where *** is a p-value of <0.001 and *NS* is non-significant. First-order kinetic modelling was supported by this performance evaluation.
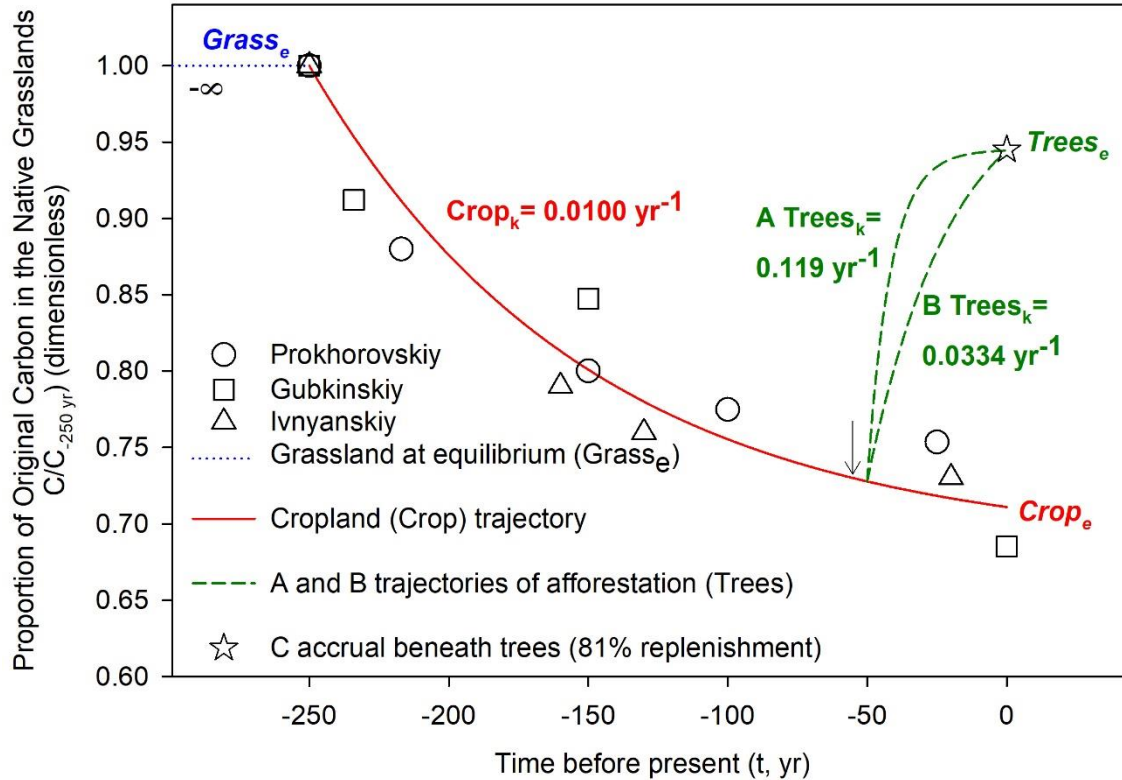
Fig. 3. Reconstruction of soil organic carbon storage within the 0 to 30 cm depth increment following land use conversions from native grassland to annual cropland and subsequently into afforestation with shelterbelts in Russia. This assemblage assumed that native grasslands were at dynamic equilibrium and steady state prior to conversion to annual croplands. Likewise, nonlinear k estimates of turnover rates in croplands and accretion rates under afforestation also required the assumptions of reaching new dynamic equilibria and steady state at zero year (i.e., time of soil sample collection). The cropland trajectory of soil C over time was derived from chronosequence data presented in Fig. 2A, Fig. 2B and Fig. 2C. The soil C accrual beneath trees at time zero (open star) was estimated from measured data presented in Table 2 (i.e., of the soil C that had been depleted by cropping, afforestation replenished 81%, based on 3-sites mean). Note that although trees were planted on year -55 (vertical arrow ↓), tree-C contributions to soil C accrual were accounted for starting from year -50 based on a literature review by Paul et al. (2002) that suggested a lag phase of 5 years. The trajectory 'B Trees' (dashed red line with k= 0.0334 yrs⁻¹) assumed that the soil C storage had asymptotically reached 95% of a theoretical equilibrium (i.e., 'Trees$_e$' / 0.95). First-order kinetic modelling was used to derive these three nonlinear trajectories of soil C (Eq. [1]). With the aim of integrating information from the 3 available chronosequences (Fig. 2), all soil C storage data were normalized (i.e., zero to one) and presented here as fractions of C storage at time of conversion from native grassland to annual cropland (shown as -250 years before time of soil sample collections).
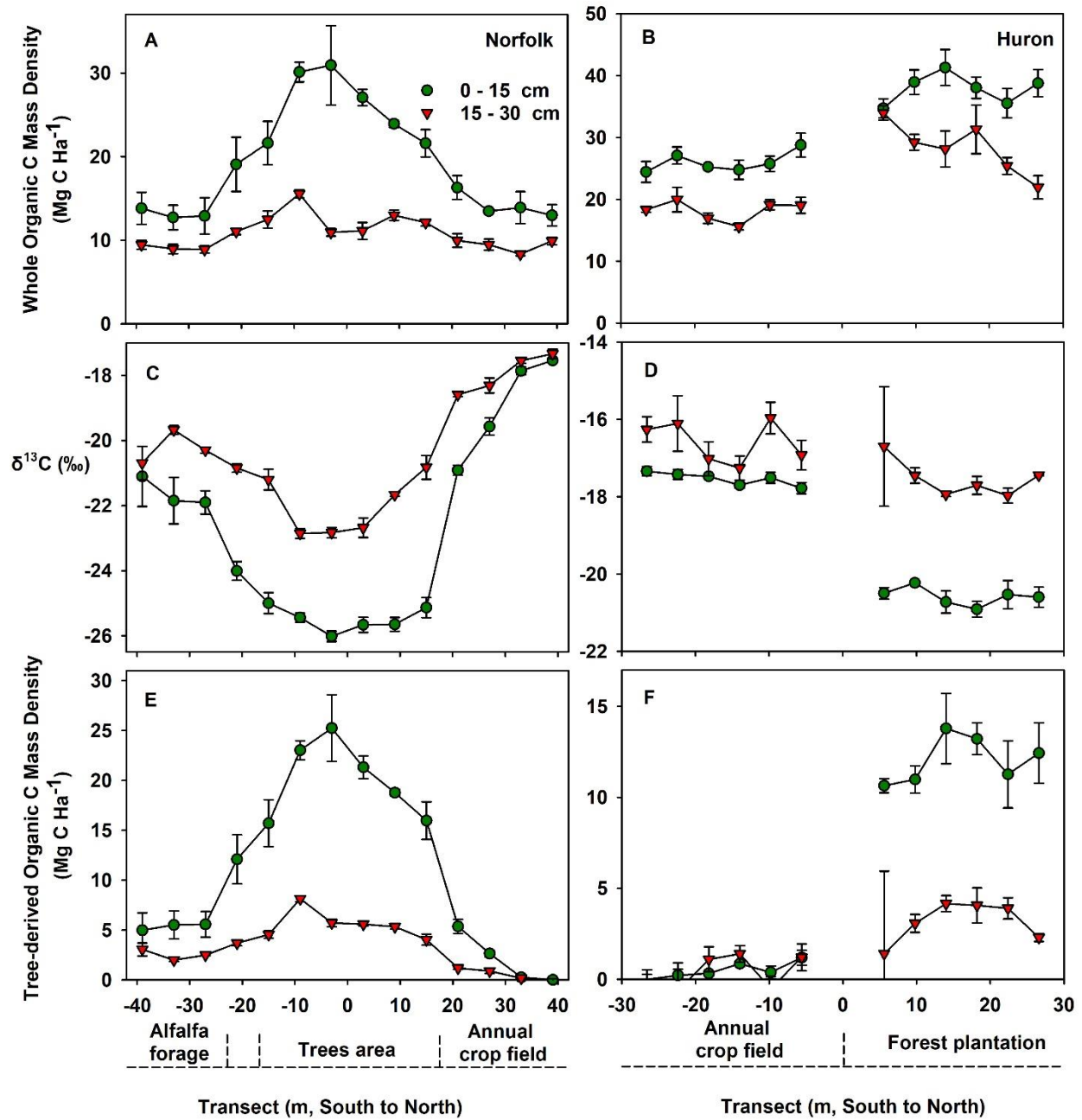
Fig. 4. (A, B) Soil organic carbon mass storage, (C, D) stable isotope ratios of organic C (δ^{13}C), and (E, F) organic C mass derived from C3 plants across transects at Norfolk shelterbelt (left panels) and Huron forest plantation (right panels) for the 0 to 15 and 15 to 30 cm soil depth increments. Adjacent cropped fields were also included. Within the afforested areas in panels E and F, the reported organic C masses are primarily attributed to direct tree contributions. Contributions of tree-C to soil C storage were clearly discernable within the 0 to 15 cm depth increment. Note the difference vertical y-scales across panels. Error bars are standard errors of the means, with sample sizes of 4 for Huron and 3 for Norfolk.
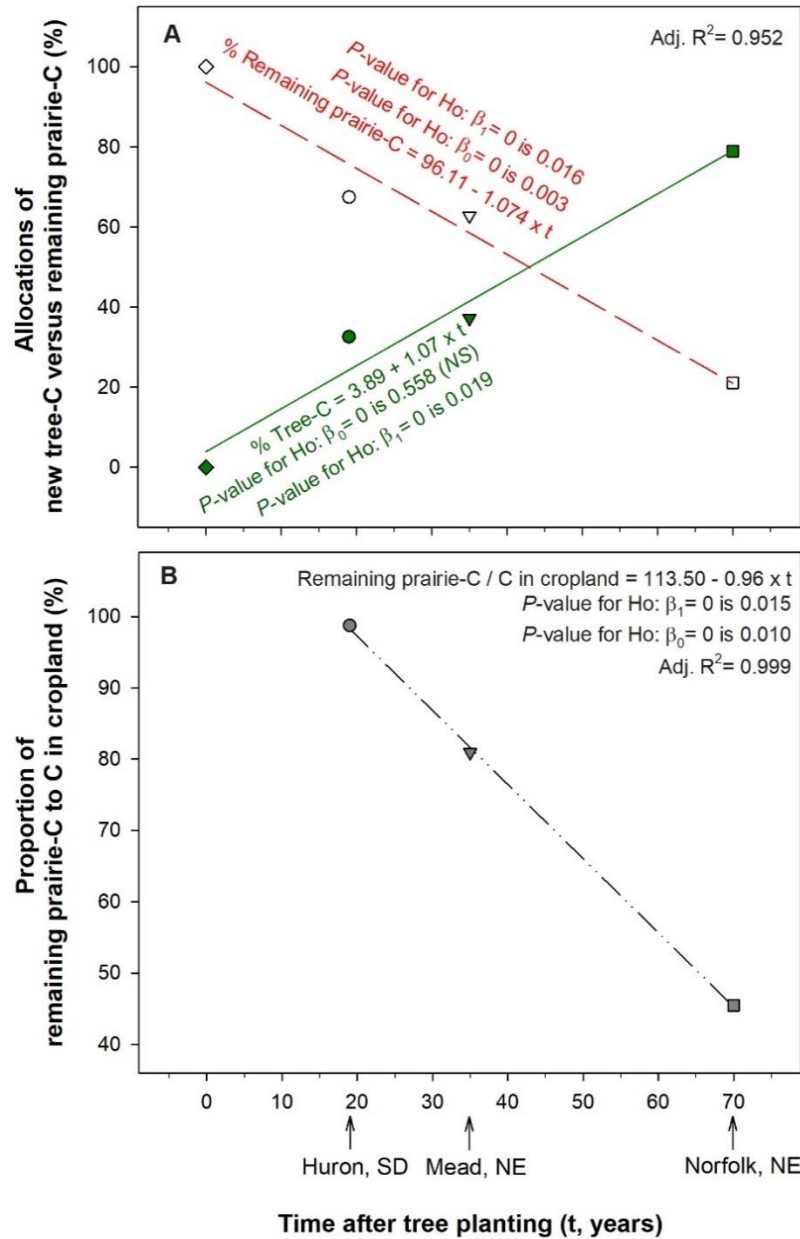
Fig. 5. Proportions of soil organic carbon within 0 to 15 cm depth increment (A) from tree-C versus remaining prairie-C relative to whole soil C stored directly beneath trees, and (B) remaining prairie-C in afforested soils relative to whole soil C in the adjacent croplands. It was inferred that all soil C different from new tree-C was preexisting soil C attributable to remaining prairie-C. (A) Mead data was recalculated from Hernandez-Ramirez et al. (2011) as compiled in Table 3. The tree-C source increased to become 79% of the whole soil C over 70 years after tree planting. (B) The proportion of prairie-derived C beneath trees declined over time to become less than half (45%) of the whole soil C in the adjacent annual croplands, which were assumed to be at steady state.
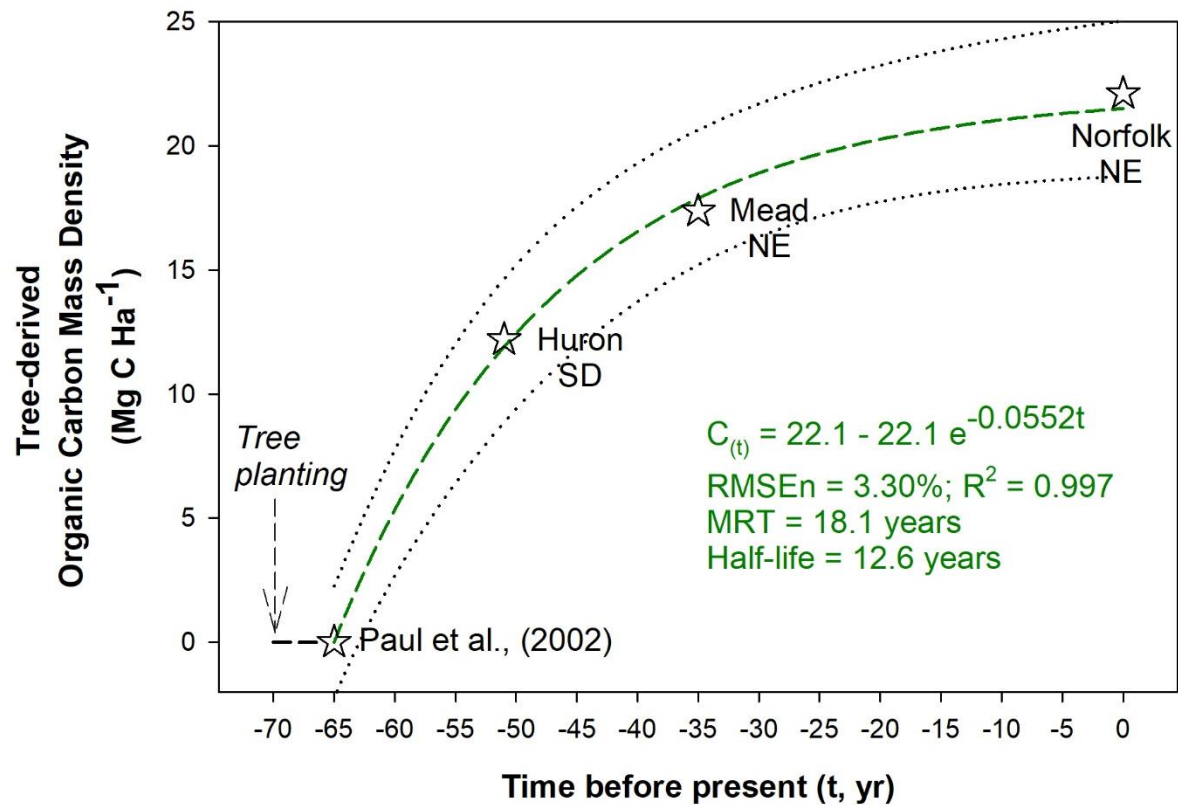
Fig. 6. Reconstruction of soil organic carbon storage within the 0 to 15 cm depth increment following land use conversion from annual cropland to afforestation United States. The arrow indicates the time of tree planting (-70 years). Based on a literature review by Paul et al. (2002), we included a lag phase of 5 years following tree planting. Huron and Norfolk data were derived from results presented in Fig. 4. Mead data was recalculated from Hernandez-Ramirez et al. (2011) as compiled in Table 3. This assemblage supports that afforested soils were approaching steady state at nearly 70 years after tree planting, as required for first-order kinetic modelling. The first-order kinetic model (Eq. [1]) is depicted by the solid curvilinear fitting, where k is the first-order kinetic rate constant equivalent to accretion rate under afforestation, and MRT stands for mean residence time. Normalized root mean square error (RMSEn) and coefficient of determination (R^2) for the k model are also provided. The 95% prediction bands of this k model are provided as dotted lines. This nonlinear trajectory describes and highlights the contribution of trees to soil C accrual.
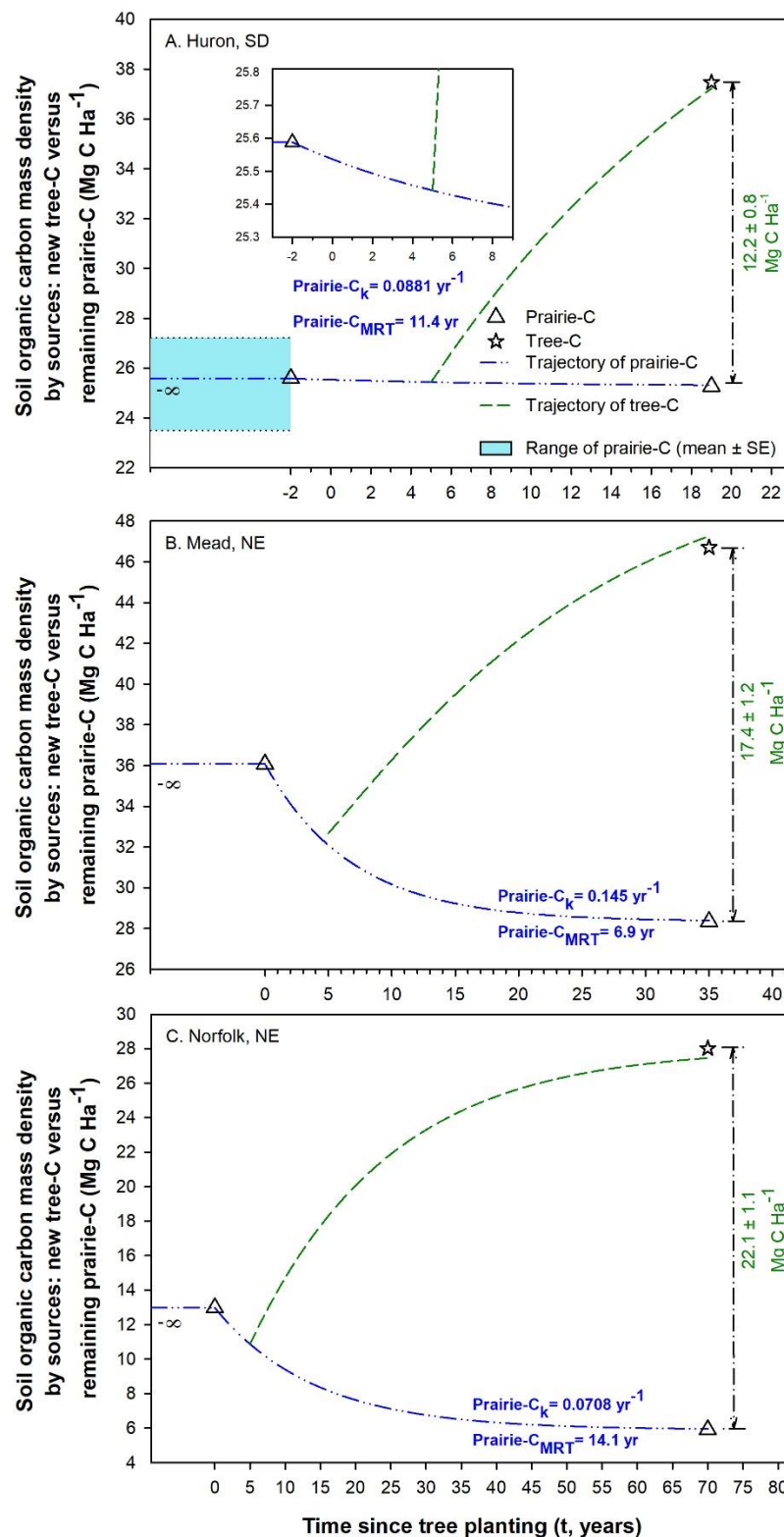
Fig. 7. Net carbon accretion in afforested soils caused by simultaneous C gains from substantial tree-C contributions and smaller C losses from mineralization of remaining prairie-C. It was inferred that all soil C different from new tree-C was preexisting soil C attributable to remaining prairie-C. Time of tree planting was set at the year zero. Based on a literature review by Paul et al. (2002), we included a lag phase of 5 years following tree planting. The accretion trajectories of tree-C presented within each Panel were projected using the unified first-order model developed in Fig. 6. The magnitude of tree-C gains as well as the turnover rates (k) and mean residence times (MRT) of prairie-C trajectories are provided within each panel. The Huron site (Panel A) had an adjacent native grassland available, which here is provided as a reference and plotted prior to the time of conversion into annual cropland. Note the different x and y scales across panels.