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Master thesis

Carbon Storage in Orchards

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List of Abbreviations

f  field
p  pasture
s  streuobst

B1  Breitenfurt 1
K1  Klosterneuburg 1
K2  Klosterneuburg 2
M1  Mauerbach 1
M2  Mauerbach 2
R1  Roppersberg 1

g  grammme
kg  kilogramme
Gt C  Gigatonne Carbon
Mg C  Megagrammes Carbon

cm  centimetre
cm³  cubic centimetre
m  metre
m²  square metre
ha  hectare

ppb  parts per billion
W  watts
RF  Radiative Forcing
Introduction

In the light of climate change, the carbon storage in terrestrial ecosystems is of great interest. Terrestrial ecosystems have presumably taken up a large part of anthropogenic carbon dioxide emissions in the past. Human activities partly control the amount of emissions released into the atmosphere on the one hand and on the other hand, they can, for example through land management, influence the quantity of emissions that is taken up. This uptake occurs both in the vegetation and the soil and is influenced by a variety of factors (IPCC 2007a).

Land use plays a role in both terrestrial carbon sinks, and is analysed in an increasing number of studies dealing with different land use types. The exact quantification of the carbon stored in vegetation and soil is essential for valid results, which might be used for political decisions concerning the mitigation of and adaptation to climate change.

Agroforestry systems have been recognized as storing a larger amount of carbon than most other land use systems. An agroforestry system which has so far not been studied is the streuobst system. Streuobst orchards are an old European land use system and were widespread in France, Germany, Switzerland, Austria and Poland. Due to their low economic profitability, the streuobst area has been in a continuous decline since 1930. From an ecological point of view, streuobst orchards should be preserved because they provide many environmental benefits, especially in terms of biodiversity (HERZOG & OETMANN 2001).

A high carbon storage in streuobst systems would give a further reason for their conservation. Besides, the determination of the carbon storage in streuobst contributes to the inventory of carbon sources and sinks which is required from Annex I countries by the Kyoto Protocol (UNITED NATIONS 1998).

Aboveground carbon in agroforestry and other tree-based systems is commonly determined by using allometric equations which relate the biomass to easily measurable variables. Allometric equations are species, site and management specific. Although a wide range of equations exists, none could be found for the quantification of the biomass of fruit trees in streuobst orchards.

The development of such an equation was therefore the first purpose of the current study. The second purpose was to compare the carbon stock of streuobst orchards with adjacent land uses, pasture and field, to evaluate the carbon storage potential of streuobst as a land use system. In addition to soil sampling, the formerly derived allometric equation was used to quantify the carbon storage in streuobst systems. The main research question was therefore if land use, either streuobst or pasture and field, significantly influences the carbon storage. The hypothesis was that the total carbon stock as well as the soil organic carbon stock is higher in the land use streuobst than in the adjacent pasture followed by field.
The introduction gives a broad background concerning the subjects climate change, carbon storage and land use. Starting with climate change and the global carbon cycle, the issue of terrestrial carbon storage is embedded into the global context and its relevance for the earth’s climate system. By illustrating the carbon dynamics and potential responses to climate change, the seriousness of carbon storage is highlighted.

The two components of terrestrial ecosystems, soil and vegetation, are described in regard of the most important factors that influence their carbon storage. After giving a short description of the carbon measurement methods, agroforestry systems in general and the European systems streuobst orchards are characterised. At last, an overview of political measures is presented. The chapter Materials & Methods is divided into the parts Tree Carbon Equation and Streuobst Carbon Storage, as are the other chapters. The part Streuobst Carbon Storage comprises the subparts Tree Carbon, Soil Analysis and Soil & Tree Carbon. A detailed description of the practical and mathematical/ statistical processes can be found in the Materials & Methods chapter, followed by the presentation of the results. The chapter Discussion picks up issues mentioned in the Introduction and compares the results to the ones from other studies to explain the findings and to answer the research questions. The part Streuobst Carbon Storage focuses on the difference between the streuobst and the pasture carbon storage. Although a field was also included in the sample, the difference in carbon stock between cropland and other land uses has been amply discussed in other studies. The thesis ends with the Conclusion where main results are summed up and the significance of agroforestry for climate change is briefly described.
**An Introduction to Climate Change**

Climate change is one of the most prominent subjects in today’s scientific discussion. Even though the first scientific paper about human-induced climate change was already published in 1896 by Arrhenius (RAMANATHAN & VOGELMANN 1997), it was not before 1988 that an intergovernmental organisation was set up to assess information about climate change on a scientific base. The Intergovernmental Panel on Climate Change (IPCC) assesses information from the fields of science, technology and socio-economics related to climate change, evaluates the consequences of climate change for the environment and humans and formulates response strategies (IPCC 2007b). Its definition of climate change is “a change in the state of climate [...] that persists for an extended period, typically decades or longer” (IPCC 2007b). The definition comprehends both natural causes and human-induced changes. Natural causes include variations in the Earth’s orbit, solar radiation variability and volcanic aerosols. The main impact of human activities is the change of the atmosphere, predominantly through the emission of greenhouse gases. The three greenhouse gases of primary concern are carbon dioxide (CO$_2$), methane (CH$_4$) and nitrous oxide (N$_2$O). By affecting the absorption, scattering and emission of radiation, they alter the energy balance of the Earth-atmosphere system. This change can be either positive or negative and is expressed in the measurement unit Radiative Forcing (RF). It is an index of the importance a factor has as climate change mechanism and is given in watts m$^{-2}$. The net radiative forcing of the effect of human activities since 1750 is +1.6 (+0.6 to +2.4) W m$^{-2}$. The greenhouse gases CO$_2$, CH$_4$ and N$_2$O contributed +2.3 (± 0.2) W m$^{-2}$ to this. In comparison, the effect of changes in solar irradiance is far less important with a RF of +0.12 (+0.06 to +0.3) W m$^{-2}$ (IPCC 2007b).

An example for a negative RF is the cooling effect produced by aerosols such as sulphate, black carbon or dust with a RF of −0.7 (-0.9 to -0.1) W m$^{-2}$ (IPCC 2007c).

Directly observable consequences of climate change are higher air and ocean temperatures, declining glaciers and ice caps, rising sea levels and an increase of the frequency of extreme weather events. Independent from the scenario assumed, these effects are predicted to last and probably to intensify at least until the end of the 21$^{st}$ century. The warming and sea level rise are assumed to continue for more than a millennium due to the persistence of CO$_2$ in the atmosphere (IPCC 2007b).

**Methane, Nitrous Oxide & Carbon Dioxide**

70 % of CH$_4$-emissions originate from biogenic sources, e. g. wetlands, livestock, rice agriculture and landfills. Non-biogenic sources include mining and burning of fossil fuel, waste treatment and geological sources (IPCC 2007a). In 2005, the global concentration of CH$_4$ was 1,774 ppb, which is an increase of about 250 % compared to the pre-industrial concentration in 1750. It exceeds the natural range for the last 650,000 years of 320-790 ppb by far (IPCC 2007c).
The increase of the atmospheric concentration of N\textsubscript{2}O from 270 ppb in 1750 to 319 ppb in 2005 is smaller. But its atmospheric lifetime\textsuperscript{1} of 131 (± 10) years is much longer than the lifetime of CH\textsubscript{4} of 9.1 (± 0.9) years (PRATHER et al. 2012). The largest source of CH\textsubscript{4}-emissions is agricultural soil management with 68 %, followed by industry and chemical production with 8 % (EPA 2012). CH\textsubscript{4} has a radiative forcing of +0.48 (± 0.05) W m\textsuperscript{-2} which is larger than the one for N\textsubscript{2}O with +0.16 (± 0.02) W m\textsuperscript{-2}, but which is still exceeded by the RF of CO\textsubscript{2} with +1.66 (± 0.26) W m\textsuperscript{-2} (IPCC 2007d).

CO\textsubscript{2} is thereby defined as the most important greenhouse gas. Its concentration has increased from a pre-industrial value of about 280 to 379 ppm in 2005 (IPCC 2007c), thereby reaching the highest concentration of the last 650,000 years (IPCC 2007a). Fossil fuel combustion is the greatest source of CO\textsubscript{2}, accounting for 75 % of the atmospheric CO\textsubscript{2} increase since 1750 or 7.2 (± 0.3) Gt C y\textsuperscript{-1}. The remainder originates from land use\textsuperscript{2} change, primarily deforestation with 1.6 (± 1.1) (IPCC 2007a). There are different estimates in literature about the atmospheric lifetime of CO\textsubscript{2} ranging from 31-92 years (MOORE & BRASWELL 1994) over 5-200 years (IPCC 2001) to centuries-millennia (MOORE 2009) and millennia (ARCHER & BROVKN 2008).

The Global Carbon Cycle

Even though fossil fuel emissions amount to 7.2 (± 0.3) Gt C y\textsuperscript{-1}, the atmospheric increase is only 4.1 (± 0.1) Gt C y\textsuperscript{-1}. This is due to the natural sinks of carbon which take up about 3.3 Gt C y\textsuperscript{-1}, namely the terrestrial biosphere and the oceans (Picture 1). The net land-to-atmosphere flux based on data from 2005 is -0.9 (± 0.6) Gt C y\textsuperscript{-1}, the net ocean-to-atmosphere flux is -2.2 (± 0.5) Gt C y\textsuperscript{-1}. Therefore, only 45 % of the combined anthropogenic CO\textsubscript{2}-emissions have remained in the atmosphere. Oceans have taken up about 30 %, the terrestrial ecosystems the remainder.

On land, plants take up CO\textsubscript{2} from the atmosphere to convert it to plant biomass by photosynthesis (= Gross Primary Production, GPP). This process captures 120 Gt C y\textsuperscript{-1} using the values from Picture 1 from the 1990s. 119.6 Gt C y\textsuperscript{-1} is returned to the atmosphere through plant, animal and soil respiration. In vegetation, soil and detritus, an amount of 2300 Gt C is stored. From this, the cumulative losses from land use change are 140 Gt C which is a flux of 1.6 Gt C y\textsuperscript{-1}.

In the oceans, carbon is also taken up through photosynthesis by phytoplankton (50 Gt C y\textsuperscript{-1}) and partly respired (39 Gt C y\textsuperscript{-1}). The remaining 11 Gt C y\textsuperscript{-1} sink to the intermediate and deep ocean as dead organisms and particles, from where a small

\textsuperscript{1} The atmospheric lifetime is the mean time a molecule resides in the atmosphere before it is removed by chemical reactions or by deposition (MCGRAW-HILL 1982); defined as T = M / S where T: Turnover time, M: Mass of a reservoir, S: total rate of removal (IPCC 2007e).

\textsuperscript{2} In natural sciences, land use is defined in terms of syndromes of human activities, e. g. agriculture and building construction, which alter land surface processes, encompassing biogeochemistry, hydrology and biodiversity (ELLIS 2012).
fraction reaches the ocean surface sediment (0.2 Gt C y\(^{-1}\)). \(\text{CO}_2\) additionally reacts with water, forming bicarbonate and carbonate ions. To calculate the terrestrial biosphere sink of 101 Gt C, the emissions from fossil fuels and land use change were added and the sinks of the ocean and the atmosphere subtracted. The flux “land sink” of 2.6 Gt C y\(^{-1}\) represents this uptake by terrestrial ecosystems which could so far not be explained by scientists. The net land sink or net land-to-atmosphere flux shown in Picture 1 from the 1990s of 1.0 Gt C y\(^{-1}\) (Land sink 2.6 Gt C y\(^{-1}\) minus 1.6 Gt C y\(^{-1}\) Land use change) was corrected for 2005 to 0.9 Gt C y\(^{-1}\) as mentioned above (IPCC 2007a).

**Picture 1:** The global carbon cycle for the 1990s. Pre-industrial fluxes and pools are shown in black, human perturbations are shown in red. GPP: Gross Primary Production.

Numerous feedbacks exist between the fluxes and between climate, the carbon cycle and human activities (GRUBER 2004, SABINE et al. 2004). SCHIMEL (1995a) for example points out that the carbon storage in terrestrial ecosystems is affected by climate because both ecosystem carbon gain, i. e. photosynthesis, and carbon loss, i. e. respiration, are influenced by temperature, moisture and radiation. Because of the uncertainty about the future development of the climate, the complexity and the sensitivity of the global carbon cycle and the unpredictability arising hereof, precise quantitative assessments including these feedbacks do not exist yet (GRUBER 2004).
Responses to Climate Change
The global carbon cycle indicates two possibilities to reduce CO\textsubscript{2} in the atmosphere: Sources of carbon have to be reduced or sinks have to be enlarged. This is described by the IPCC as mitigation response. For each sector, different technologies and practices exist to decrease greenhouse gas emissions or to enlarge sinks. Examples for the energy sector are the use of renewable energies, improved efficiency in supply and distribution and fuel switching from coal to gas. For the transport sector, hybrid and more fuel efficient vehicles as well as shifts from road transport to rail and public transport systems are key mitigation technologies. For the sectors of interest, agriculture and forestry, a number of technologies is listed. Examples for agriculture are restoration of cultivated peaty soils and degraded lands, improved nitrogen fertiliser application techniques and improved crop and grazing land management for increased soil carbon storage. For forestry, technologies include forest management, afforestation and reforestation.
Beyond the recommendation of technologies, the IPCC also proposes policies, measures and instruments. Examples for the energy sector are a reduction of fossil fuel subsidies and renewable energy obligations, for the transport sector, CO\textsubscript{2} standards for road transport and taxes on vehicle purchase are named. For agriculture, financial incentives and regulations for improved land management, for maintaining the soil carbon content and for the efficient use of fertilisers and irrigation are recommended. Financial incentives to increase the forest area, to reduce deforestation and to maintain and manage forests as well as regulations concerning land use and their enforcement are policies regarding the forestry sector (IPCC 2007e).
Referring to the IPCC, the increase of the carbon storage in soils by agriculture is therefore a way to mitigate climate change, recommended by the IPCC. Increasing the global forest area is another way.

Terrestrial Carbon Storage
The amount of carbon stored in terrestrial ecosystems depends significantly on the type of land use (e. g. GRUBER 2004, IPCC 2000, SCHIMEL 1995a). To estimate the quantity of carbon associated with a certain type of land use, the carbon stored aboveground, i. e. in the vegetation (also including roots), and carbon stored belowground, i. e. in the soil, has to be taken into account.
The share of carbon stored in the soil is much larger than the share stored in the vegetation: According to the IPCC (2000), the global soil carbon stock down to a depth of 1 m amounts to 2,011 Gt C. Compared to this, carbon stored in the vegetation is only 466 Gt C. The U.S. DEPARTMENT OF ENERGY (2008) found similar values of 2300 Gt C for the soil pool and of 550 Gt C for the plant biomass.
The storage of carbon in the pools varies with different factors. In the following, a selection of the most influencing factors is presented. The list does not intend to be exhaustive.
Vegetation Carbon

**Latitude** is one of the factors that control vegetation as well as soil carbon storage. Even though exact estimates of the vegetation carbon stock in literature differ, tropical forests are commonly declared as being the largest carbon pool (e.g. Dixion et al. 1994, Sabine et al. 2004, Schimel 1995b) with a proportion of 59-63% of global forest carbon (Dixon et al. 1994, Sabine et al. 2004) and 52% of global vegetation carbon (Sabine et al. 2004). Together with boreal forests, they account for 75-80% of all aboveground carbon (Sabine et al. 2004, Dixon et al. 1994). Dixon et al. (1994) give a value of 212 Gt C for the aboveground and the belowground living and dead forest vegetation mass of low latitude forests. High latitude forests, including forests in Russia, Canada and Alaska are supposed to contain 88 Gt C which concurs with Goodale et al.’s (2002) estimate of 88.4 Gt C.

The process by which plants gain energy is photosynthesis. The base material necessary for building biomass is taken up in the form of nutrients and water from the soil. All factors that exert an influence on these two processes therefore affect plant biomass.

Photosynthesis depends, as the name already says, on **light**. It is the dominant carbon metabolic process in plants, and carbon gains were found to increase linearly with solar radiation (Gough et al. 2008). Changes in light intensity are especially important for light-deficient plants, for example for plants from the arctic (Shafer et al. 1992). However, the short-term increase in productivity with increased light is assumed not to be sustained in the long run due to nutrient limitation, especially nitrogen, in arctic ecosystems (Shafer et al. 1992).

**Temperature** affects all biological processes. As it increases, the metabolic rate\(^3\) also increases exponentially at first until an optimum temperature is reached, and then falls rapidly (Rastetter et al. 1991). Two main processes, photosynthesis and respiration, are sped up with increasing temperature below the optimum temperature. Thus it is the ratio of photosynthetic to respiring (non-photosynthetic) tissue that determines the net carbon response of a plant (Rastetter et al. 1991) respectively of an ecosystem (Raich et al. 2002). This is the reason why forests experience a net carbon loss in winter when the canopy is leafless and a net carbon gain in summer when carbon gain through photosynthesis outweighs carbon loss through respiration (Gough et al. 2008).

The extent to which an increased temperature can cause plant growth through increased photosynthesis depends on plant species, plant condition and other

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\(^3\) The *metabolic rate* describes the rate at which chemical transformations of nutrients (metabolism) occur in an organism. It is determined by the size of the organism and its species (Demetrius & Tuszyński 2010).
environmental factors such as water and nutrient supply (Peltonen-Sainio et al. 2010, Rastetter et al. 1991, Shaver et al. 1992).

The amount as well as the seasonal distribution of precipitation are major determinants of the plant primary production (Friedlingstein et al. 2006), especially so in semiarid and arid regions (Ojima et al. 1993). Water is the most limiting factor in the productivity of natural ecosystems (Taiz & Zeiger 2007). In general, the aboveground annual primary production is lower when the precipitation amount declines (Fang et al. 2005, Ládány et al. 2011, Ojima et al. 1993, Woodwell et al. 1983).

The two substrates for the process of photosynthesis are water and atmospheric CO$_2$. It is therefore obvious that an elevated CO$_2$-concentration raises net photosynthesis. The one reason is the increased substrate supply, the other is the decreased photorespiration$^4$ (Poorter 1993). At the same time, plant transpiration is reduced (Strain 1985) because the stomatal conductance$^5$ of water is lower (Poorter 1993). These two processes lead to an increase of net primary production and therefore of carbon storage with elevated CO$_2$-concentration (Rastetter et al. 1991, Schimel 1995b). But similar to the effect of increased light intensity, this is only the initial, short-term ecosystem response and vegetation acclimates soon due to nitrogen limitation (Rastetter et al. 1991).

Nutrients are required in numerous biological functions and processes. Certain elements are essential, meaning that they are intrinsic components in the plant’s structure or metabolism. An essential element deficiency causes abnormalities in growth, development or reproduction. Nitrogen belongs to this group of elements and is the mineral element required in the greatest amounts. It is a main constituent of almost all plant cell components such as free amino acids, proteins and also chloroplasts, the sites of photosynthesis. Therefore, nitrogen deficiency always causes a decrease in photosynthesis (Taiz & Zeiger 2007).

Most natural and agricultural ecosystems are nitrogen-limited in the sense that the addition of nitrogen will increase plant growth and thereby carbon storage (Schimel 1995b, Woodwell et al. 1983). Fertilisation is possible through anthropogenic activities, either direct fertiliser application or indirect enhancement of atmospheric deposition, e. g. of N$_2$O, or through biological processes, nitrogen mineralisation being the most important (Schimel 1995b).

As pointed out above, many factors are connected. This becomes evident when the response of plant growth to climate change is observed: Even though the sole increase in atmospheric CO$_2$ caused an increased primary production (Poorter 1993), the combined effects of CO$_2$, precipitation and temperature with climate

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$^4$ Photorespiration is the diametrically opposite process to photosynthesis in which atmospheric O$_2$ is taken up by the plant and CO$_2$ is released, thereby causing a loss of CO$_2$ (Taiz & Zeiger 2007).

$^5$ The stomatal conductance is a measure for the flow of water and CO$_2$ through the stomata in and out of a leaf (Taiz & Zeiger 2007).
change were simulated to cause a general decrease in net primary production (FRIEDELINGSTEIN et al. 2006).
Furthermore, plants can only reach optimal growth when all growth factors are fulfilled (SCHUBERT 2006). An example for this is that an increase in light or atmospheric CO$_2$ only results in enhanced plant growth when the subsequent provision of nitrogen is secured (SHAVER et al. 1992).

**Soil Carbon**

The soil carbon pool is composed of inorganic and organic carbon (CONANT et al. 2011). The inorganic carbon makes up only a small fraction and mainly consists of carbonates, e. g. calcium carbonate (CaCO$_3$) or sodium carbonate (Na$_2$CO$_3$). It is very little involved in carbon exchanges between the global carbon pools (GRINAND et al. 2012). Litter and all substances therefrom derived form the soil organic carbon pool. Litter is dead organic matter originating from plants, animals and microorganisms. For this reason, soil organic carbon is closely related to the primary production of plants. The conversion of litter through bio-chemical processes is termed humification and the amorphous, macromolecular substances hereby produced are called humic substances. Soil organic carbon therefore encompasses a wide range of very different organic compounds (STAHR et al. 2008).

Several factors determine the amount of organic carbon stored in a soil. One of them is, as mentioned before, latitude. The soil carbon stock is much larger in higher latitudes (IPCC 2000, OECHEL & VOURLITIS 1994). Main determinants are the effects of high soil moisture and low temperature on the decomposition of organic matter (CELIK 2005, NADELHOFER et al. 1991). By way of example, the IPCC (2001) gives a value of 344 Mg C ha$^{-1}$ for the soil of boreal forests while the soil of tropical forests contains 123 Mg C ha$^{-1}$ down to a depth of 1 m. MILLER et al. (1983) furthermore point out that northern ecosystems contain about 25 % of the global soil carbon even though they represent only 14 % of the global land area. Additionally, large amounts of soil organic biomass are accumulated in the humus layer and shallow moors which can quickly be decomposed by microorganisms if temperature rises (BLUME et al. 2019). The predominance of permafrost along with the high amount of soil carbon and the expectation that climatic changes will be the greatest in this region (SHAVER et al. 1992) makes high-latitude ecosystems extremely vulnerable to climate change (OECHEL & VOURLITIS 1994).

The climate is, especially in the face of climate change, another important factor (JOBBAGY & JACKSON 2000).
**Temperature** can have a positive and a negative effect: It can increase both the primary production and the heterotrophic respiration of soil organisms, two counteracting processes in terms of soil carbon storage (SHAVER et al. 1992).

In wet tundras, the largest carbon store of arctic ecosystems, plants’ nitrogen demand is almost entirely provided by recycled soil organic matter and plant litter
Thus, an enhanced primary production will lead to an increased nitrogen supply. As the primary production in most ecosystems is nitrogen-limited (Schimel 1995b), the increased nitrogen supply will in turn stimulate net primary production (Raich et al 1992). Since temperature along with moisture controls the rates of microbial activity, organic matter decomposition is enhanced when temperature rises. On the one hand, this leads to a larger amount of plant-available nitrogen through increased nitrogen mineralisation. On the other hand, the uptake of nitrogen by microorganisms leads to nitrogen immobilisation. The relationship of mineralisation to immobilisation is controlled by the C:N ratio of organic matter, with a high ratio leading to greater immobilisation (Schimel et al. 1990). As the net effect, mineralisation is assumed to outweigh immobilisation, leading to a fertilisation effect (Schimel et al. 1994, Schimel 1995a) which increases plant biomass and thereby soil carbon storage.

The increased microbial activity not only leads to an enhanced rate of organic matter decomposition but also to increased heterotrophic respiration. Together with root respiration, this is termed total soil respiration. Because the two processes are difficult to separate in field observations of soil respiration, they are usually considered together (Peng et al. 2009, Raich & Schlesinger 1992). Soil respiration accounts for a large part of total terrestrial respiration: While plant, animal and soil respiration is assumed to be 119.6 Gt C y\(^{-1}\) (see Picture 1, IPCC 2007a), global soil respiration is estimated to average 80.4 (79.3 to 81.8) Gt C y\(^{-1}\) for the same time period (Raich et al. 2002). Even though studies modeling the response of soil respiration to rises in temperature come to different conclusions concerning the extent of the response, all of them agree in the fact that rising temperatures increase soil respiration (Friedlingstein et al. 2006, Raich et al. 2002). Raich et al. (2002) for example expect global soil respiration to increase by 3.3 Gt C y\(^{-1}\) per °C increase in the short term. Long-term models that include cumulative C losses through time yield carbon losses of 22 to 33.3 Gt C y\(^{-1}\) per °C (Schimel et al. 1994, Jenkinson et al. 1991). This carbon loss might be reduced by production increases caused by the enhanced mineralisation of soil organic nutrients as described above. Schimel et al. (1994) estimated this effect to reduce carbon losses from 22 to 11.1 Gt C y\(^{-1}\) per °C. The extent of the soil respiration response to increased temperatures also differs among ecosystems (Peng et al. 2009, Shaver et al. 1992).

Besides temperature, moisture or precipitation is another factor profoundly influencing soil respiration (Schimel et al. 1990). This is why soil respiration is largest in biomes with high temperature and moisture availability year-round, e. g. in tropical moist forests (Raich & Schlesinger 1992). Another study found this positive correlation of precipitation with soil respiration only within specific biomes, i. e. on a regional scale, and within a year, and not on an annual-global scale (Raich et al. 2002). In contrast to temperature, soil respiration is not linearly correlated with precipitation. A linear relationship only exists up to a certain threshold, followed by a decrease in the increase rate. Wei et al. (2010) quantified this threshold to be 813 mm in naturally-regenerated forest soils. Below the breakpoint, soil respiration increased by 75 g C m\(^{-2}\) y\(^{-1}\) per 100 mm increase in mean annual precipitation; above
the breakpoint, the rate was only a third with 20.3 g C m\(^{-2}\) y\(^{-1}\). Thresholds and increase rates differed with forest type. This non-linearity is explained by the fact that at a certain level of cloudiness, plant activity is decreased and thereby reduces autotrophic respiration. 

The counteracting process to carbon loss caused by increased precipitation is the enhanced production of litter through increased primary production (OJIMA et al. 1993, POST & KWON 2000, WEI et al. 2010).

The effect of precipitation depends on the vegetation, with soils under grasslands being more responsive by a factor of 2.6 than soils under shrublands/ woodlands (JACKSON et al. 2002). In most cases, increased precipitation leads to a net soil carbon storage (SCHIMEL 1995a). GUO & GIFFORD (2002) found precipitation and climate to be the best predictors for soil organic carbon in the top 20 cm worldwide.

The increase of plant primary production is also the main reason why soil carbon storage might be increased with elevated \textit{atmospheric CO}_2-concentration (DE GRAAF et al. 2006). A reason why soil carbon storage might decrease is that the rate of soil carbon mineralisation outbalances the increase in carbon input (RAICH & SCHLESINGER 1992). Very divergent responses of ecosystems to an elevated atmospheric CO\(_2\)-concentration have been observed and reasons for variable results are not quite clear (DE GRAAF et al. 2006). However, a meta-analysis by DE GRAAF et al. (2006) showed a significant soil carbon increase of 1.2 % y\(^{-1}\) under elevated CO\(_2\). But this was only the case when nitrogen availability was high and the increase was only significant for soils under herbaceous species.

The average increase of belowground biomass was 28 %. This is composed of increased root biomass, subsequent increase in rhizodeposition\(^6\) and an increase of 7.7 % in microbial carbon. The increased microbial activity of 17.7 % and the consequent losses in soil carbon through mineralisation reduced the average increase of 28 % to 1.2 % y\(^{-1}\). Additionally, RASTETTER et al. (1991) found a high C:N ratio in plant litter with elevated CO\(_2\)-concentrations which has a higher nitrogen-immobilization potential. This suggests that elevated CO\(_2\) may lead to a decrease in nitrogen availability, confining plant growth and thereby soil carbon storage (DE GRAAF et al. 2006).

The \textit{pH} is only a minor factor influencing soil carbon storage (LEIFELD et al. 2006), even though the activity of the two largest groups of soil microorganisms, the bacteria and the fungi, depends strongly on pH. While the pH-optimum for bacteria is between 5-7, fungi prefer acid soils (STAHR et al. 2008). The low impact on soil carbon mineralisation despite the pH-optima can be explained by the fact that these two major decomposer groups are, at least partially, functionally redundant (ROUSK et al. 2009).

\(^6\) \textit{Rhizodeposition} is defined as the „loss of organic material from the roots“ and encompasses water-soluble exudates, e. g. organic acids and secretions, e. g. carbohydrates and enzymes (WHIPPS 1990). These substances are readily available for microorganisms.
All effects described above are confined by soil characteristics. The share of clay particles in a soil for example has been named to be another determinant of soil organic matter, the clay shielding soil organic matter from decomposition (RASTETTER et al. 1991). This causes fine-textured soils to have higher carbon contents than coarse-textured soils (POST & KWON 2000, SCHIMEL et al. 1990). The proportion of organic horizons containing large carbon and nitrogen stocks (RASTETTER et al. 1991), as well as total soil depth, are further criteria (JOBBÁGY & JACKSON 2000).

The amount of total soil carbon storage arises from the balance of soil carbon gains and losses. The amount and type of the primary production is the main determinant of the gains, while the losses are determined by erosion, leaching and the oxidation of soil organic matter, the decomposition (HAN et al. 2010, Post & Kwon 2000). A variety of interactions exist between the single factors and between the climate system, the described factors and soil carbon storage. Ecosystem models are used to simulate the response of soil carbon to a combination of climate factors. Using these models, the soil of global grassland ecosystems was simulated to lose carbon on a range from 0 to 14 % for the first 20 cm soil depth with climate change, causing a flux to the atmosphere of 0.06 to 0.08 Gt C y\(^{-1}\) (OJIMA et al. 1993). Similarly, investigations of arctic ecosystems showed that they are currently a source of carbon and will most probably remain one, at least in the near future (OECHEL & VOUURLITIS 1994). The net carbon loss is supposed to be caused indirectly by an enhanced drainage, soil aeration and a decrease in the water table (OECHEL et al. 1993). Most models agree in an enhanced soil carbon loss with climate change in the long run, even though the extent of loss differs (SCHIMEL 1995a).

**Net Carbon Storage**

To calculate the net carbon storage of terrestrial ecosystems, both changes in soil carbon and vegetation carbon have to be taken into account. Results for net carbon storage differ between studies, depending on whether calculations were made globally or only for the northern hemisphere. For studying the coupling between climate change and the *global* carbon cycle, FRIEDLINGSTEIN et al. (2006) suggested to subtract the amount of carbon produced by heterotrophic respiration of the carbon from the net primary production. In their simulations, this resulted in a positive feedback on climate change and therefore an increase of the airborne fraction of CO\(_2\)-emissions. Other ecosystem models came to similar conclusions: Even though some of them assumed the terrestrial biosphere to be a net carbon sink at first, it was reversed in the long run to a net carbon source (SCHIMEL 1995a, WOODWELL et al. 1983). The three reasons for this are (i) the limitation of ecosystems’ capacity to additionally take up carbon by nutrients and other biophysical factors, (ii) the fact that once the threshold of maximum photosynthesis is reached, photosynthesis will not increase anymore whereas heterotrophic respiration will continue to rise with increasing temperatures, and (iii) the possible degradation of ecosystems through climate change (IPCC 2000).
When only the northern hemisphere was analysed, results differed: GOUGH et al. (2008), investigating northern forests, assumed the net primary production to average 6.5 Gt C y\(^{-1}\) while heterotrophic respiration was 5 Gt C y\(^{-1}\), thereby resulting in a net carbon storage rate of 1.5 Gt C y\(^{-1}\). This coincides with DIXON et al.’s (1994) findings of the mid- to high-latitude forests being a net sink of 0.7 (±0.2) Gt C y\(^{-1}\) and SCHIMEL’s (1995a) estimation of 0.5 (±0.5) Gt C y\(^{-1}\) for northern hemisphere forests.

**Measurement Methods Carbon**

For the quantification of the carbon stored in vegetation and soil, scientifically sound measurement methods are required. The method for the belowground carbon consists in taking soil samples and analysing them for their carbon content. For the aboveground carbon, grasses and herbs are harvested, dried and weighed and the carbon content determined by chemical analysis. Carbon mass is then estimated by multiplying the measured biomass by its carbon content (SHARROW & ISMAEL 2004).

The common tool for the determination of the biomass of trees are allometric equations. Allometric equations relate easily measurable variables, e.g. tree diameter or height, with the aboveground biomass of the entire tree (TER-MIKAELIAN & KORZUKHIN 1997, ZAPATA-CUARTAS et al. 2012). They are generated from a small sample of trees by taking measurements, determining the total tree biomass and then calculating a regression of the measured values on the biomass (KUYAH et al. 2012a, STERBA & NACHTMANN 2009). This equation can in the following be used to estimate the biomass at forest or landscape scale in a non-destructive way (KUYAH et al. 2012a). As 50% of the dry weight of wooden plant parts is commonly assumed to be carbon, the multiplication of the biomass with 0.5 gives the carbon amount (IPCC 1996).

The exact estimation of the aboveground biomass is necessary in many respects. First, the biomass determination allows the quantification of the carbon storage and can indicate the carbon sequestration potential as well as the CO\(_2\)-emissions to the atmosphere through deforestation (CHAVE et al. 2004, KUYAH et al. 2012a, ZIANIS et al. 2005). The Kyoto Protocol and the United Nations Framework Convention on Climate Change (UNFCC) require countries to monitor the forest carbon stock, emissions and removals (ZIANIS et al. 2005). This includes biomass changes through forest management, forest harvesting, plantation establishment, abandonment of lands that regrow to forests and forest conversion to other land uses (BROWN 2002). Second, in order to establish an optimal, sustainable forestry or agroforestry management, the prediction of standing biomass via some simple measurements is crucial (CHAVE et al. 2004, DROPPELMANN & BERLINER 2000, ZIANIS et al. 2005). Third, the establishment of coppicing or energy crop forests due to the increasing demand for bioenergy requires the estimation of their current and potential biomass to determine optimal harvest time (SUCHOMEL et al. 2012 in press). Fourth and last, the practical application for various studies is given which might lead to further insights.
Introduction - Land Use Change

into topics related to carbon storage and climate change (DROPPELMANN & BERLINER 2000).
To accurately predict tree biomass, allometric equations have to be specifically developed according to species, site, environment and management. Most equations are set up for a certain tree species, few for relatively homogeneous ecosystems, which obviously derives from the fact that tree species differ in growth rate and tree shape (CHAVE et al. 2004, KUYAH et al. 2012a). Site is another prominent factor as climate and soil properties exert a large influence on tree growth (BROWN 2002, LEVIA Jr. 2008, ZAPATA-CUARTAS et al. 2012). Besides biomes, local conditions also need to be taken into account. This includes the landscape, as e. g. trees in agricultural landscapes show a growth pattern and stature different from that of trees in forests (KUYAH et al. 2012a, SAJDAK & VELAZQUEZ-MARTI 2012). Also, the species mix and the stand properties (closed – not closed – isolated trees) have an influence (KUYAH et al. 2012a). Finally, the management should not be neglected, such as pruning, coppicing and the cutting of crowns (KUYAH et al. 2012a, SAJDAK & VELAZQUEZ-MARTI 2012, SUCHOMEL et al. 2012 in press). DROPPELMANN & BERLINER (2000) for example found that pruning changes the internal carbohydrate partitioning, with pruned trees spending a larger share of photosynthates on leaf regrowth than on biomass accumulation in the trunk.
It can therefore be concluded that the development of local, species- and site-specific allometric equations is vital for the precise determination of aboveground tree biomass.

Land Use Change

The estimations of the ecosystem studies described above where made for a certain type of land use. But results might differ when land use types are allowed to migrate (SCHIMEL 1995).
As visible in Picture 1, land use change contributes considerably to the loss of carbon from terrestrial ecosystems to the atmosphere with an annual flux of 1.6 Gt C and is responsible for all human-caused CO₂-emissions not generated through fossil fuel combustion (IPCC 2007a). In contrast to burning of fossil fuel, land use change does not create a new carbon source but changes the natural carbon exchange rate between the terrestrial biosphere and the atmosphere (IPCC 2000).
This takes place either through direct physical impacts such as clearing of forest or through indirect consequences of human activities such as climate change (FOSTER et al. 2003). Direct physical impacts, meaning a change in land management or land use, lead in most cases to a change in land cover as well (IPCC 2007f). Land cover describes the physical and biological cover of land, being either vegetation or man-made features (IPCC 2000). Due to the tight relationship between vegetation and soil as described above, land use change also affects the soil by altering land cover. These effects on soil properties are supposed to be even stronger and more abiding
for the function and dynamics of successive ecosystems than the effects on land
cover (JACKSON et al. 2000, RICHTER et al. 2000).

In the past, the effect of land use change on terrestrial ecosystems has been one of
net carbon loss to the atmosphere, mainly through the conversion of forests to
croplands and pasturelands (JACKSON et al. 2000). This must not necessarily remain
so as the net carbon effect depends on the category of land use change.

GUO & GIFFORD (2002) reviewed the effects of land use changes on soil carbon
stocks. The overall average across all land use change categories was calculated as
being a decrease in soil carbon stock by 9 %. According to category, they found a
significant decline of soil carbon for the changes from pasture to plantation (-10 %),
from native forest to plantation (-13 %) and from forest and pasture to crop (-42 and
-59 %, respectively). Significant soil carbon increases were listed for the changes
from forest and crop to pasture (+8 and +19 %, respectively) and from crop to
plantation and to secondary forest (+18 and +53 %, respectively). Hence, the land
use pasture seems to be the best regarding soil carbon storage, followed by forest.

BROWN & LUGO (2000) found the same result and suggested that grasses may be
more effective at storing soil carbon than trees in some environments. Another
explanation could be that grassland soils have a substantially higher humus content
than all other well-aerated soils (STEVenson 1982).

However, as 80 % of all aboveground carbon is stored in the world’s forests (DIXON et
al. 1994), the land use forest is the preferred one in terms of total carbon (e. g. DIXON

Changes in land management that affect carbon storage without altering land cover
are, among others, improvements of agricultural practices. A prominent example is
the introduction of conservation tillage in the USA that increased soil organic matter
by 0.05 Gt C y\(^{-1}\) since 1975 (IPCC 2007a).

Besides the land uses mentioned above, another one exists which used to be less
often included in analyses of land use effects on carbon sequestration: agroforestry.

Probably due to its low economic importance and the somewhat vague definition, it
has rarely showed up in global land use change studies. This has been changing
recently with the realisation that agroforestry systems are capable of storing
significant amounts of carbon (NAIR 2011a).

**Agroforestry**

The term agroforestry encompasses all land use systems that combine woody
components such as trees or shrubs with herbaceous plants such as crops or
pastures and/ or livestock. This combination evokes ecological and economic
interactions between the components and is possible via a spatial arrangement, a
rotation or both (YOUNG 1989).

The definition includes a large variety of systems, and even among similar
agroforestry practices, high levels of spatial heterogeneity exist. This is due to the
fact that agroforestry systems are established and managed according to local socio-
cultural norms and traditions and are adapted to the local environment (NAIR 2011a). The broad definition might be a reason for the difficulties to precisely assess the extent of the area under agroforestry. The IPCC (2000) estimates the global area as roughly 400 million hectares in 2000, which is about 8% of the global agricultural area (FAO 2012).
Agroforestry has been prevalently practiced for many centuries all over the world (NAIR 2011b), also as a way to increase agricultural sustainability and to slow the negative effects of agriculture, e.g. soil erosion (ALBRECHT & KANDJI 2003). Agroforestry systems are still a common land use system in the tropics, and certain types exist also in the northern hemisphere. In North America, these are for example windbreaks, shelterbelts, riparian buffers and forest farming systems (MONTAGNINI & NAIR 2004). The prevalent reason for the establishment of agroforestry systems in the industrialised nations is the environmental protection, either directly through the prevention of soil erosion or indirectly through carbon storage (NAIR 2011b). By comparison, the improvement of soil quality and the provision of food and nutritional security are the motivation of small-scale family farms in the tropics (NAIR 2011b).
In southern latitudes, agroforestry systems are extremely complex and diverse (ALBRECHT & KANDJI 2003, THEVATHASAN & GORDON 2004). A common system is alley cropping: annual or perennial arable crops are cultivated between tree hedgerows (ALBRECHT & KANDJI 2003). Trees are either periodically pruned to decrease light and water competition and to use their mulch as fertiliser for the crops or left to grow fully for the production of firewood and timber. The combination of cacao (*Theobroma cacao* L.) with the shade tree Spanish Elm (*Cordia alliodora* Oken) is a perennial system often used in South America and Africa (MONTAGNINI & NAIR 2004). Among the trees’ various functions, the diversification of farm products to increase the household income and the exploitation of the benefits of tree-crop interactions are most important (ALBRECHT & KANDJI 2003).

The productivity of agroforestry systems depends on a number of factors such as age, structure and particularly management (ALBRECHT & KANDJI 2003). To establish a successful tree-based intercropping system, the positive interactions between the components need to be exploited and negative interactions need to be minimized (THEVATHASAN & GORDON 2004).

**Benefits & draw-backs of agroforestry systems**

Agroforestry systems provide a broad range of environmental benefits compared to conventional land use systems (NAIR 2011b, THEVATHASAN & GORDON 2004).

**Carbon storage**

The benefit currently attracting most attention is the carbon storage. Even though the carbon accumulation of agroforestry is not as high as that of natural forests from the same region, agroforestry systems still exceed most other land uses (DIXON et al. 1994, HOWLETT et al. 2011, MONTAGNINI & NAIR 2004). This is directly due to the large volume of aboveground biomass consisting of multiple plant species and to the deep
root system of the trees (Montagnini & Nair 2004, Nair 2011b). The large biomass quantity stems from fast growth, high system productivity and the fact that only small biomass proportions are exported from the system through harvest (Montagnini & Nair 2004). The high productivity is made possible through the ability for greater capture and better utilisation of growth resources (Nair 2011b). The amount of stored carbon depends on the plant species used, management practices and the agroforestry system, which is in turn determined by environmental and socio-economic factors (Albrecht & Kandji 2003, Tumwebaze 2011).

Probably due to the wider distribution in the tropics, most studies concerning the carbon storage in agroforestry treat tropical systems. Often, the carbon storage of an agroforestry system is compared to an adjacent treeless system as in Andrade et al. 2008, Soto-Pinto et al. 2010 or Tumwebaze et al. 2011, whose results point out a larger carbon storage potential in the agroforestry plot. Studies from the temperate region, namely Canada (Sharrow & Ismael 2004, Thevatasan & Gordon 2004), Spain (Howlett et al. 2011) and Europe (Palma et al. 2007, Eichhorn et al. 2005) show the same, although the relevance of these findings is lower because of the smaller land area under agroforestry and the lower potential for establishing agroforestry plots (Dixon 1995, Reisner et al. 2007).

Indirectly, agroforestry can decrease pressure on primary forests, the largest sink of terrestrial carbon (Montagnini & Nair 2004).

The current carbon gain of agroforestry systems ranges from 0.29 to 15.21 Mg C ha\(^{-1}\) y\(^{-1}\) aboveground and from 30 to 300 Mg C ha\(^{-1}\) y\(^{-1}\) including the soil down to a depth of 1 m (Nair 2011b). The greatest potential for carbon accretion via biomass production lies within tropical latitudes (Dixon 1994).

**Microclimate**

The microclimate of a site is strongly affected by the tree canopy of the woody component of an agroforestry system. It decreases evaporation and provides shade, thereby reducing temperature variations and improving the water balance (Ong et al. 2002, Tumwebaze et al. 2012). The litter layer also contributes to an improved water balance (Lott et al. 2003). Furthermore, the provision of shelter from wind and the protection of the soil from the impacts of heavy rain help to reduce soil erosion (Schroeder 1993, Tumwebaze et al. 2012). Erosion is also reduced through leaf litter which acts as mulch and beyond that reduces evaporation and surface runoff. All these processes lead to an increase in soil organic matter, thereby making agroforestry systems more sustainable and resistant to climate change than single cropping systems (Schroeder 1993).

**Biological effects**

A number of biological effects distinguish agroforestry from monocropping systems. Activity and biomass of the soil fauna are increased under agroforestry (Rao et al. 1998). Especially earthworms play a role in this respect as they are responsible for the comminution and mixing of soil organic matter with the soil, among others (Rao et al. 1998, Thevatasan & Gordon 2004). Arthropod, parasitoid and detritivore number and diversity were also found to be higher as well as the ratio of parasitoids
to herbivores, presumably due to the provision of habitat. In addition, a greater number of bird species occurs (MANNING et al. 2006, THEVATHASAN & GORDON 2004).

**Biodiversity**

Silvoarable agroforestry systems were found to have a higher landscape biodiversity (PALMA et al. 2007, REISNER et al. 2007) and a higher species biodiversity (NAIR 2011b, REISNER et al. 2007). The mechanisms here are (i) the reduced exploitation of protected areas through the intensification of agroforestry systems, (ii) an increased biodiversity in working landscapes through the expansion of agroforestry systems into traditional farmlands and (iii) an increased tree species diversity in farming systems. This is vitally important in landscapes that are occupied to a large part by farmlands and in the light of climate change (NAIR 2011b).

**Nutrient cycling**

Agroforestry can have effects on the nitrogen cycle. Through the addition of plant litter, the need for nitrogen fertilisation decreases, thereby reducing $\text{N}_2\text{O}$-emissions and potential nitrogen leaching (PALMA et al. 2007, THEVATHASAN & GORDON 2004). As tree roots reach far deeper than crop roots, they are capable to absorb nutrients from layers not accessible to agronomic crops (EICHHORN et al. 2006, RAO et al. 1998). By recycling these nutrients via leaf litter and root turnover, trees can improve the nutrient-use efficiency of the whole system (NAIR 2011b, RAO et al. 1998).

**Reduction of soil erosion**

As 35% of the global land area is at risk to degradation and desertification, the reduction of soil erosion is of far-reaching importance (DIXON et al. 1994). The trees in agroforestry systems improve land cover and provide organic matter inputs to the soil. Through the return of litter and pruning biomass and the decay of roots, the soil’s physical and chemical properties are improved (ALBRECHT & KANDJI 2003). These processes not only prevent and reduce soil erosion, they are also ways to rehabilitate degraded land (NAIR 2011b).

In addition to the environmental benefits, agroforestry systems have other advantages to monocropping which are of great importance for poorer households (NAIR 2011b). These are mainly (i) the production of animal fodder and nutritious foods such as meat and fruits, (ii) the production of fuelwood from thinning and pruning, either for own needs or as additional source of farm income, and (iii) the production of valuable goods, e.g. timber or its raw materials such as branches to make baskets (KÜRSTEN 2000, MONTAGNINI & NAIR 2004, NAIR 2011b).

**Draw-backs**

Among the various interactions between the components of an agroforestry system, there are also factors that negatively influence its productivity and the environment. This is, to a large part, competition for light, water and nutrients.
Competition for light causes yield reductions in that it decreases photosynthesis through effects of shading (Shaver et al. 1992). Water competition is a common problem in semi-arid regions. Even though agroforestry can also improve the water balance, competition for water can seriously decrease the production when tree and/or crop water demand exceeds the supply (Lott et al. 2003). Nutrient competition is related to water competition as both water and nutrients are taken up by the root (Rao et al. 1998).

The extent of competition can be regulated through the selection of tree and crop species and through management methods such as pruning. The species selection determines root and canopy architecture, thereby influencing light and nutrient competition. When species are chosen in such a way that the water demand of the trees peaks at another time of year as that of the crops, water competition can be reduced to a great extent. Tree pruning is a management strategy to reduce light competition, while root pruning decreases nutrient competition (Rao et al. 1998, Thevathasan & Gordon 2004).

Agroforestry systems can also be sources of greenhouse gases, especially ruminant-based agrosilvopastoral systems in tropical regions. Tillage, burning, chemical fertilisation and manuring are practices that can lead to emissions of CO₂, CH₄ and N₂O; the unsustainable management of silvopastoral systems can entail soil compaction and erosion (Montagnini & Nair 2004).

In spite of this variety of environmental benefits, there is still a vast area available worldwide for agroforestry establishment. In North America, this area is considered to be as large as 140 million ha. Constraints to large-scale implementation can be classified into three groups: (i) social conditions, including population pressure, issues of land tenure and lack of infrastructure; (ii) economic obstacles such as inadequate markets for agroforestry products, capital requirements and the lack of financial incentives; and (iii) ecological considerations, encompassing limited knowledge of the impacts and the sustainability of these systems (Dixon et al. 1994).

**Agroforestry in Europe**

In Europe, 90 million ha have been identified as being potentially suitable for (silvoarable) agroforestry systems (Reisner et al. 2007), capable of storing 0.77-3 Mg C ha⁻¹ y⁻¹ over a 60 year period (Palma et al. 2007). Throughout historical times, agroforestry systems have formed key elements of the European landscape (Eichhorn et al. 2006). Silvoarable agroforestry systems, the combination of widely spaced trees with annual or perennial crops, were widespread. Yield diversification and the production of a short-term return on land were perceived as the principal advantages. The three main purposes of the tree component in agroforestry systems were the production of fruits, fodder and wood, wood being used as either fuel, litter or timber. Additional services were seen in the provision of
shade and shelter for workers and livestock and of fertiliser, either in the form of litter or in the form of manure resulting from woodland grazing (EICHHORN et al. 2006). Throughout the last century, most agroforestry systems have been abandoned. Even though systems and their production purposes differ between European countries, the reasons for the decline of agroforestry are the same:

(i) Trees in arable landscapes impede the mechanisation of agriculture which leads to higher efforts and costs.
(ii) The focus on the maximisation of yield leads to single cropping systems to increase productivity.
(iii) High labour costs and the concomitant substitution of manpower through machinery limit the commercial viability of systems requiring high labour inputs, e. g. standard fruit tree orchards.
(iv) Land consolidation programmes remove boundary trees and hedgerows.
(v) Political incentives promote single cropping to increase agricultural production.
(vi) Stricter quality norms regarding fruits and vegetables standardise the production in intensively managed systems (EICHHORN et al. 2006).

In northern Europe, light is the limiting factor in most agroforestry systems (EICHHORN et al. 2006). The best known systems in this zone are hedgerows and windbreaks (HERZOG 1998). In Mediterranean Europe, water availability imposes greater constraints, so that trees are more dispersed due to the greater root systems. Also, a wider range of economically valuable trees and crop species exists, leading to a greater diversity of systems (EICHHORN et al. 2006). Common systems are olive (Olea europaea L.) groves with cereals, vegetables, fodder legumes or grasses sown in-between, and fruit tree systems, predominantly almond (Prunus dulcis D.A.Webb) and fig trees (Ficus carica L.), intercropped with cereals, fodder legumes and grasses (EICHHORN et al. 2006).

A comparable style of silvoarable orchards in temperate Europe is the streuobst system (EICHHORN et al. 2006).

**Streuobst Systems**

In contrast to other agroforestry practices or agroecosystems, streuobst has attracted interest from various groups throughout its history until today. As most other agroforestry systems in Europe, it has decreased significantly in area and economic and productive importance in the last 50 years (HERZOG & OETTMANN 2001), but is still widespread in Southern Germany and certain regions in Austria, e. g. the “Mostviertel” (EICHHORN et al. 2006). LUCKE et al. (1992) defined streuobst as “tall trees of different types and varieties of fruit, belonging to different age groups which are dispersed on cropland, meadows and pastures in a rather irregular pattern. This also includes single trees on roads, streets and banks, small groves, tree alleys and extensive plantings with a rather regular but spacious planting distance.” ZEHNDER & WELLER (2011) give a broader
definition, which is also used for this study: “The combination of sparsely
distributed fruit trees having large crowns with permanent grassland”.
The most common fruit types are apple (*Malus domestica* Borkh.), pear (*Pyrus
communis* L.), plum (*Prunus domestica* L.) and mazzard cherry (*Prunus avium* L.)
(HERZOG 1998). Robust varieties well-adapted to local conditions did and still do play
a large role (HERZOG & OETMANN 2001). Tree shape is usually the standard fruit tree
with a log length of 1.6 m or more, but half-standard trees with a log length of
1.0-1.2 m occur as well (HERZOG 1998). Trees are usually scattered in the agricultural
landscape (HERZOG & OETMANN 2001), with tree densities ranging from
20-100 trees ha⁻¹ (HERZOG 1998).

**History**

While the domestication of fruit trees started as early as 3500 to 2000 BC, the formal
development of the system streuobst began in the 17th century with the increasing
importance of market production (HERZOG 1998, HERZOG & OETMANN 2001). Governmental orders, which had so far mostly concerned the royal orchards, started
to involve the population in fruit tree planting and tending (ZEHNDR & WELLE 2011). Monks supported the process by providing extension service. As a result, the
streuobst area increased until 1950. At that time, most streuobst systems were
converted to fruit plantations to facilitate farming and to increase production. This
process was supported by national governments and the European Economic
Community which subsidised the clearing of standard fruit trees in streuobst systems
(HERZOG 1998). In Baden-Wurttemberg, Germany, this halved the number of
streuobst fruit trees from 25 million in 1950 to about 11 million in 1990 (HERZOG &
OETMANN 2001). Germany is the country estimated to have the largest streuobst area
with 225,000-500,000 ha, followed by Poland and France with 198,770 ha and
186,282 ha, respectively; Austria lies with 8,564 ha in the lower range (HERZOG 1998,
data from 1993-1998). In the still existent streuobst orchards, the undergrowth
beneath the fruit trees is used either as pasture or as meadow (EICHHORN et al.
2006).

**Functions**

Whereas the main purpose of streuobst orchards has always been the production of
fruits, they have nowadays been recognized as fulfilling a broad range of functions
(EICHHORN et al. 2006, HERZOG 1998, HERZOG & OETMANN 2001, ZEHNDR & WELLE 2011). These can be grouped into three categories: economic importance, socio-
cultural properties and ecological functions.

(i) The economic importance of streuobst orchards arises mainly from the fruit
production. The total production of streuobst fruits in Germany is higher than that of
intensively managed orchards. Only half of the fruits are sold, mainly for juice
production, the rest is used for household consumption (HERZOG 1998). Due to the
low labour productivity combined with the relatively high European wages, streuobst
has an insufficient operational efficiency, compared to monocropping systems (HERZOG 1998, HERZOG & OETMANN 2001). However, under specific conditions, streuobst can have comparative advantages over other land use types: For a medium-sized family farm with cattle and crop production in the hilly parts of temperate Europe, e. g. in southern Germany, Switzerland or northern France, streuobst might be a profitable option (HERZOG 1998). The production of timber is another economic option, but as there is currently only a very small market for fruit tree timber, this is negligible (HERZOG 1998, ZEHNDER & WELLER 2011). In contrast, streuobst does play a role in relation to tourism. Certain regions in Europe use streuobst orchards in combination with other elements such as local gastronomy, cultural programme and infrastructure to attract tourists. Examples are the “Albtrauf” in Germany, the “Mostviertel” in Austria and “Mostindien” in Switzerland (ZEHNDER & WELLER 2011).

The second category, socio-cultural properties, includes functions like landscape aesthetics and recreation. Streuobst orchards characterise the landscape and give a feeling of spatial depth and diversity (LUCKE et al. 1992). This is why streuobst regions count to the most diverse European cultural landscapes (ZEHNDER & WELLER 2011). Streuobst orchards can also act as “buffer strips” around villages, integrating and connecting settlements to the open agricultural land (LUCKE et al. 1992). The natural scenery is assumed to affect the well-being of humans, so that landscapes characterised by streuobst can enhance the recreational value of a region, thereby promoting tourism (HERZOG 1998, ZEHNDER & WELLER 2011).

Through its long history and its formerly great importance for the food supply, streuobst has also cultural importance and can create regional identity as e. g. in the “Mostviertel” (HERZOG 1998, ZEHNDER & WELLER 2011).

The ecological functions can again be divided into three groups: protection of resources, biodiversity and gene pool. As an agroforestry system, the ecological benefits described above also apply to streuobst (EICHHORN et al. 2006, HERZOG 1998, HERZOG & OETMANN 2001, LUCKE et al. 1992, ZEHNDER & WELLER 2011). Again the background that 5 million hectares of the arable land in Europe is at high or very high risk of soil erosion (REISNER et al. 2007) and that the land use streuobst occurs predominantly in the hilly parts of temperate Europe which are especially prone to soil erosion, streuobst plays a vital role in soil protection (HERZOG 1998).

As already described, streuobst leads to a greater landscape diversity. Higher species diversity, a property of agroforestry systems, is not only limited to animal species, but also applies to plant species (HERZOG 1998, ZEHNDER & WELLER 2011). Tree varieties used in streuobst differ from the ones used in intensively managed fruit orchards. They have to meet different requirements: While fruit production is the most important aspect in the choice of trees for fruit orchards, the usage and storage properties, the seasonal spreading of the harvest and the resistance against fungal and virus diseases and against weather extremes is of greater importance for trees in streuobst systems (HERZOG 1998, LUCKE et al. 1992). Streuobst led thereby to a tremendous number of local varieties adapted to specific site conditions, providing a large gene pool which is inter alia valuable for breeding new varieties (HERZOG & OETMANN 1997, ZEHNDER & WELLER 2011).
In conclusion, streuobst systems provide a broad range of environmental and socio-cultural benefits, especially in the light of climate change. Due to their low economic profitability, these systems have been largely abandoned.

**Political Influence**

As the public and the government become more aware of the environment and the impacts of human activities thereon, primarily regarding CO$_2$-emissions and their effects on climate change, viable options to reduce the human impact gain in importance. Regulations concerning land use change exist on a European as well as on a national basis. Only recently has Commission Regulation No 1122/2009, which obliged member states to maintain the ratio of land under permanent pasture to the total agricultural area (EC 2009), led to the prohibition of the ploughing up of permanent grassland in several federal states of Germany (LANDESREGIERUNG NRW 2011, LANDESREGIERUNG BW 2011).

On the European scale, the promotion of the extensification of crop farming, including the conversion of arable land into extensive grassland, has been named as one of the aims in Regulation No 2078/92 (EEC 1992). In Germany, this regulation has led to the nation-wide distribution of agri-environment measures, reaching 29 % of the agricultural area (BUNDESREGIERUNG 2000). EC Regulation No 1698/2005 on the support for rural development by the European Agricultural Fund for Rural Development (EAFRD 2005) also included the extensification of forest resources as eligible for support. This encompasses the first afforestation of agricultural and non-agricultural land and the restoration of forestry potential. Furthermore, the establishment of agroforestry systems on agricultural land is listed as a measure that is supported by the EU, and agroforestry systems are described as having “a high ecological and social value by combining extensive agriculture and forestry systems” (EAFRD 2005).

Through the Clean Development Mechanism under the Kyoto Protocol, industrialized countries can contribute to compliance with their emission limitation by investing in mitigation projects in developing countries (UNITED NATIONS 1998). These projects can be agroforestry activities, so that farmers in developing countries would be compensated for the carbon they sequester (NAIR et al. 2009).

These prohibitions and subsidies from the EU and European governments can be seen as efforts to internalise the externalities of land use changes. However, they seem to be not yet high enough for streuobst, as this agroforestry system is currently not a profitable land use. Further research into the environmental benefits of streuobst to allow for the economic quantification of these benefits might lead to higher subsidies and might thereby increase the cost-effectiveness of streuobst.
One such research area is the determination of the carbon storage of streuobst systems and thereby their contribution to the mitigation of climate change. This has been done by choosing six streuobst orchards in the vicinity of Vienna and comparing their carbon storage to adjacent land use systems. For the quantification of the aboveground carbon, an allometric equation for fruit trees was required. Since even an extensive literature review did not result in finding a fruit tree equation and since the application of a local, species- and site-specific allometric equations is crucial for the precise determination of aboveground tree biomass, it was developed in the current study. 18 apple trees were used for a destructive sample inventory, the common procedure to derive an allometric equation. This includes measuring, felling and weighing the trees, drying subsamples of tree parts and calculating a regression equation. The equation could then be used to calculate the aboveground carbon of the streuobst orchards. Furthermore, soil samples were taken from orchards and adjacent pastures respectively a field and analysed for their organic carbon content. The statistical analysis tested the hypothesis that soil as well as total carbon storage was influenced by land use. Additionally, the effect of pH, bulk density and sampling location on the plot on the soil organic carbon content was analysed.
Materials & Methods

Carbon Equation

In order to derive the carbon content of fruit trees, the dry weight of a number of fruit trees is required. For reasons of practicality and comparability with other studies, only aboveground biomass was considered.

The simplest method of biomass determination, the felling and weighing of the trees, was not practical in this work because the trees in question were situated on streuobst orchards privately owned. Since it could be assumed that the owners would not agree with felling the trees and because the efforts of felling all trees would be disproportionate, only a non-destructive method was possible. In environmental studies and forest management, allometric models are commonly used to quantify aboveground biomass of trees and forest ecosystems (Zapata-Cuartas et al. 2012) on the basis of easily quantifiable metrics, e.g. tree height or breast height diameter (Levia Jr. 2008). However, only models for tree species of silvicultural importance are reported in literature, so that the development of an allometric equation for fruit trees was necessary. Only apple trees were used for this equation to allow for the transferability of the equation to other cases. Nevertheless, the carbon content of other tree species, e.g. mazzard cherry, was also calculated with the equation because growth habits were assumed to be similar.

A destructive sample inventory was carried out with 18 trees from different locations, applying the common technique for destructive sample inventories as described in Hochbichler et al. (2006) or in the agroforestry guidelines of Dietz & Kuyah (2011). Hereby, several size characteristics of these trees were taken, afterwards trees were felled, separated in different parts according to diameter, weighed and dried. In the following, a three-step-procedure was set up to determine total carbon. The first step was to determine the dry weight of a fruit tree by multiplying the observed fresh weight with the dry matter content. As second step, the size data taken before felling was used to fit a power function for tree dry weight as dependent variable with a linear regression. It was assumed that 50 % of the dry weight of wooden plant parts is carbon. Therefore, the factor 0.5 was used to convert dry matter into carbon in the third step. This factor is widely used as general default value for the conversion of biomass dry matter into total carbon (IPCC 1996).

Procedure

18 trees from three different locations were chosen to ensure a large variation of tree size and form. 2 trees were single trees surrounded by fields close to Nordheim, Hesse, Germany, 5 trees grew on a streuobst orchard close to the Rhine river in the municipality of Biblis, Hesse, Germany, and 11 trees were taken from a fruit orchard
close to Öhringen, Baden-Württemberg, Germany. The streuobst orchard in Biblis
had not been tended for the last few years, and tree ages, sizes and conditions
varied widely. While the trees in Nordheim had not been pruned for at least 20 years,
the fruit orchard in Öhringen had been under constant management. The trees here
were planted in 1970, individually replaced when necessary and have been managed
organically since 1990. Due to regular planting distances and pruning, the trees
developed evenly and were similar in shape and size in each age group.
From all trees, diameter at breast height diameter (dbh), tree height, crown height
and crown radius were measured. For the dbh, the diameter of the stem at a height
of 1.3 m was taken with a measuring tape. Crown radius, tree and crown height were
assessed with a Vertex IV and a Transponder T3 (Haglöf, Sweden). The crown
radius was calculated as the mean from four measurements in each geographic
direction.
The tree was cut at ground line and separated into three components. These were
stem, branches with a diameter larger 5 cm (in the following: branches > 5) and
branches with a diameter smaller 5 cm (in the following: branches < 5). All three parts
were weighed separately. The fresh weight of the trunk and the branches > 5 was
obtained using a decimal scale. The branches < 5 from the trees in Nordheim and
Biblis were shredded and the chaff was put in a large bin which was weighed on the
decimal scale as well. For the trees in Öhringen, the branches < 5 were directly
placed in the bin.
Wood samples were taken from each component. Stem discs with a thickness of
about 2-4 cm were used as samples for the stem. Similarly, several branch discs with
a thickness of about 1-2 cm were produced for the branches > 5, while samples from
the branches < 5 were small twigs. These were taken from different parts of the tree.
The wood samples were stored in plastic bags and hermetically sealed to prevent
humidity to escape. Fresh weight of samples was determined, and samples were
oven-dried in a drying cabinet (Ehret Trockenschrank TKL 3200 So.) at 105 °C until
constant weight was reached. The drying process was observed by repeatedly
weighing the samples.

**Calculations**
The arithmetic mean, the standard deviation and the coefficient of variation were
calculated for the tree dimensions. To compare the drying behaviour of the different
tree parts, the weight loss over time was calculated numerically as well as in percent
of original weight.
The dry matter content was determined by dividing the dry weight of the sample by
its fresh weight for each of the three parts for each tree individually (HOCHBICHLER et
al. 2006, HULTNÄS 2011). By adding up the dry weight of the three components, the
total dry biomass of each tree is calculated:
\[ dw_{\text{tree}} = f_{\text{trunk}} \times \frac{dw_{\text{sample trunk}}}{fw_{\text{sample trunk}}} + f_{\text{branches}>5} \times \frac{dw_{\text{sample branches}>5}}{fw_{\text{sample branches}>5}} \]

\[ + f_{\text{branches}<5} \times \frac{dw_{\text{sample branches}<5}}{fw_{\text{sample branches}<5}} \]

where

- \( dw \): dry weight [kg]
- \( fw \): fresh weight [kg]

For the prediction of dry weight of a fruit tree depending on the surveyed variables, the following allometric equation similar to the one proposed in HOCHBICHLER et al. (2006) was used:

\[ dw_{\text{tree}} = \exp(a_0) \times h^{a_1} \times dbh^{a_2} \times ch^{a_3} \times cr^{a_4} \]

where

- \( h \): tree height [m]
- \( dbh \): diameter at breast height [cm]
- \( ch \): crown height [m]
- \( cr \): crown radius [m]

The measured values for dbh, tree height, crown height and crown radius with the corresponding dry biomass were log-transformed to fit the model with linear regression.

**Statistics**

Data was first tested for normal distribution and homoscedasticity. The Pearson product-moment correlation was calculated to detect linear relationships between variables. A value close to 1 indicates a strong positive correlation between two variables, a value close to -1 a strong negative correlation. If either one is the case, one of the two variables can be omitted to avoid overfitting of the model. A value around 0 shows that the variables are independent and are therefore both necessary for the model. Furthermore, the variables were tested for significance with F-tests.

For all statistical analyses in this study, the software SAS (SAS version 9.1; SAS Institute, Cary, NC, USA) was used unless otherwise indicated.

Normal distribution was tested with the Shapiro-Wilk statistic from PROC UNIVARIATE. A residual plot was created with PROC GPLOT to assess homoscedasticity visually. PROC CORR was applied to compute Pearson correlation coefficients. To fit the equation parameters with a linear regression and to test variables for significance, PROC GLM was used with a significance level of 0.05.

As data had to be log-transformed, estimates of dry weight-values were skewed. This bias was estimated through:
Materials & Methods - Carbon Equation

\[
    \text{bias} = \frac{\sum_{i=1}^{n}(\hat{y}_i - y_i)}{n}
\]

where 
\( n \): number of trees
\( \hat{y} \): dry weight estimated by regression function [kg]
\( y_i \): surveyed dry weight [kg]

To correct the bias, the correction factor \( \lambda \) was introduced as suggested by Condés & Sterba (2005). This was calculated as follows:

\[
    \lambda = \frac{\sum_{i=1}^{n} y_i}{\sum_{i=1}^{n} \hat{y}_i}
\]

Since the intention of developing the model was not the description of the tree data used for the regression but the practical application to other trees, an extrapolation of the model was carried out. Extreme values not within the range of the data used were inserted into the model to test if it yielded reasonable values for dry weight. This model was then used to calculate the biomass of the trees on the sampling plots. As a measure of fit, the standard error of the estimates was calculated, both for the model with and without \( \lambda \):

\[
    sd(\hat{y}) = \sqrt{\frac{\sum_{i=1}^{n}(\hat{y}_i - y_i)^2}{n - k}}
\]

where 
\( k \): number of variables

Most studies concerning allometric equations give the coefficient of determination, the \( R^2 \), as error estimate. To allow for comparison, it was therefore also calculated according to Dufner et al. (2002) as in the following:

\[
    R^2 = \frac{(SS_{CTotal} - SS_{Error})/SS_{CTotal}}{SS_{CTotal}}
\]

with \( SS_{CTotal} \) being the sum of squares corrected total which is

\[
    SS_{CTotal} = \sum_{i=1}^{n}(y_i - \bar{y})^2
\]

and \( SS_{Error} \) being the sum of squares error, calculated as

\[
    SS_{Error} = \sum_{i=1}^{n}(y_i - \hat{y}_i)^2
\]
Streuobst Carbon Storage

Study Area

Samples and measurements were taken on 6 randomly selected streuobst orchards and 6 adjacent pastures as reference areas. All of them are located in the same region, the Eastern Wienerwald, Lower Austria, to minimize the impact of location factors such as climate. Latitude ranges from 48°8’ to 48°16’ N, longitude from 16°7’ to 16°6’ E. According to the Köppen-Geiger Climate Classification, Lower Austria is classified as Cfb climate group, where C stands for “warm temperate”, f for “fully humid” and b for “warm summer” (KOTTEK et al. 2006). The average temperature is 9.7 °C, the average annual precipitation 607 mm (CLIMATE DATA 2012). The orchards are situated between 400 and 500 m above sea level.

While the area of the streuobst orchard was mostly fixed by the distribution of the fruit trees, the pasture area was measured off as such as to be about the same size as the orchard where possible. Additionally, one field neighbouring an orchard was also taken into the sample.

For the determination of the soil type and certain additional information such as soil depth, the Austrian soil information system BORIS (Bodeninformationssystem) from the environment agency Austria was used (AMT DER NÖ LANDESREGIERUNG & BA F. BODENWIRTSCHAFT 1994). Several soil profile analyses as close to the sampling sites as possible were chosen and therefrom, inferences were drawn about the soil types of the sampling sites. As the soil type from all surrounding sites is the pseudogley or a variation thereof and as this fact was also confirmed by two owners of streuobst orchards, it can safely be assumed that the soil type of the sampled streuobst orchards and pastures is also the pseudogley. The parent material is Flysch which consists mainly of sandstones and sandy marls (AMT DER NÖ LANDESREGIERUNG & BA F. BODENWIRTSCHAFT 1994, BERGER et al. 2010, GEOLOGISCHE BUNDESANSTALT WIEN 1999).

All orchards are privately owned and are managed extensively.

1st study site: Breitenfurt (B1)

The study site B1 is located close to Breitenfurt bei Wien (48°08’07” N, 16°07’19” E; 467 m a.s.l.), the slope faces north-west. The soil is a pseudogley or a pseudogleyificated loose sediment brown earth free from carbonates. The deep soil (> 70 cm) exhibits no concretions or spotting (AMT DER NÖ LANDESREGIERUNG & BA F. BODENWIRTSCHAFT 1994). The streuobst orchard belonging to the restaurant Schoeny has been an orchard since 1910. In 1990, most of the trees were replaced and have since been regularly pruned. Therefore, most trees are of a relatively small size except for the few old trees which were left from the first planting. Old cultivars like Bohnapfel, Berner Rosenapfel, Gelber Bellefleur, Schweizer Wasserbirne, Speckbirne, Sommer-birne, Winterbirne and Landbirne were planted. The owner uses the fruit yield of 300-400 kg per tree for juice.
The 56 trees are arranged in 5 rows on 2153.5 m². The grass on the orchard is cut twice a year, no fertilizer is and has been applied. According to the proprietor, the soil is clayey.

For the reference plot, the same area was measured off from a pasture close-by which is of the same soil type, but faces the other direction, south-east. This is also cut twice and fertilized once a year with cattle slurry.

2nd and 3rd study site: Klosterneuburg (K1 and K2)

20 km to the northwest of Vienna in the Wien-Umgebung District, two streuobst orchards are situated on a north facing slope (48°16’56” N, 16°13’35” E; 491 m a.s.l.). The parent material is a carbonate-free quartz sandstone and is part of the Flysch-zone. The soil is a deep (> 70 cm), periodically wet pseudogley with intense root penetration but only minor earthworm activity (AMT DER NÖ LANDESREGIERUNG & BA F. BODENWIRTSCHAFT 1994). The pastures and orchards are cut twice and fertilized once a year with manure. Trees are infrequently pruned. The pear cultivars Mostbirne, Spätbirne and Speckbirne occur; as apple trees, only the Boskop could be identified. Fruits are used for juice.
The streuobst orchard of the plot K1 has a size of 1261 m² with 26 trees of similar appearance, the corresponding pasture a size of 975 m². The other orchard of K2 is 1582 m² large and holds 15 trees. In contrast to the first streuobst orchard, trees here vary widely in shape and size with some trees being very old and large. As pasture area, 1367 m² were measured off.

4th and 5th study site: Mauerbach (M1 and M2)

The 4th and 5th study sites are located on a large clearing in the Schönwald (48°16’55” N, 16°10’29” E; 401 m a.s.l. and 48°16’49” N, 16°10’20” E; 394 m a.s.l.). The slopes face west. They belong to the village Mauerbach in the Tulln District. Soil on all streuobst orchards, pastures and the pasture is a clayey, periodically wet pseudogley. The quartz sandstone is free from carbonates (AMT DER NÖ LANDESREGIERUNG & BA F. BODENWIRTSCHAFT 1994). On streuobst orchard M1, the trees are grown with only small distances in between and are arranged in rows. The apple trees were planted in 1993 so that except for a few trees being replanted later, size does not differ much. However, some variation is existent due to the different cultivars. The orchard is mulched twice a year and once a year, liquid manure is applied, mainly to ensure a sufficient water supply for the trees. These are pruned every 2 to 3 years; the fruits, when fallen down, are used to make juice and liqueur. The total number of trees is 81, the size of the streuobst orchard is 8939 m².
As there is a field right next to the orchard, this was also included in the soil sampling. This has been managed conventionally for the last 100 years and has mostly been planted with cereals. At the point of soil sampling, summer barley (*Hordeum vulgare* L.) had reached growth stage 10 according to BBCH-classification (Lancashire et al. 1991). Residues of last year’s crop, maize (*Zea mays* L.), could still be found on the field. Its area is 8711 m² and had been fertilized with diammonium phosphate and lime a few days before soil sampling.

The reference pasture has only a size of 2125 m². It is used as pasture for cows and cut once a year. As a fertilizer, manure and 100 kg lime nitrogen per ha are applied. On the 5th streuobst orchard, M2, the trees are planted in lines, with large variation in age, size and shape. The orchard contains extensive areas without trees, so that only a part of the whole orchard was measured off and dedicated as the sampling site. This had a size of 4600 m² with 21 apple and pear trees on it. Age ranges from 25 to more than 80 years. Identified cultivars include Pittenfelder Sämling, Kronprinz Rudolf, Apfel aus Croncels, Morgenduft, Danzinger Mostapfel and Berner Rosen for apple; for pear, Speckbirne and Pastorenbirne could be identified (personal records provided by ARCHE NOAH). The orchard is used as pasture for about 60 cows during the summer and fall months, with an aftermath being applied once a year. It is also fertilized with 100 kg lime nitrogen per ha and year. Fallen fruits are used to make must.

The corresponding pasture, with 4176 m² of almost the same size, is managed in a similar manner as the streuobst orchard: Being used as pasture during summer and fall, it is cut and fertilized once a year with 100 kg lime nitrogen per ha. Additionally, manure is applied once a year.
6th study site: Roppersberg (R1)

The 6th study site is located in Roppersberg in the Mödling District (48° 09'42" N, 16° 09'12" E; 394 m a.s.l.) in the Eastern Wienerwald, the slope facing south. The soil is a pseudogley on quartz sandstone which belongs to the Laaber Decke in the Flysch zone. It is a deep soil (> 70 cm) free from carbonates and has a small to moderate coarse fraction (AMT DER NÖ LANDESREGIERUNG & BA F. BODENWIRTSCHAFT 1994). According to the owner, the groundwater table at the site is very low at approximately 4.30 m. This has caused drought problems for the fruit trees during the last 20 years. The pasture has been in the ownership of Mr. Krischke for 30 years and has a size of 4 ha. It is used as pasture for sheep and goats, with two aftermaths per year. The trees are not tended except for the lopping of dead branches. Several fruit species could be found: apple, pear, mazzard cherry and plum. As fruit cultivars, Schmidberger Reinette, Cox Orange, Boskop and Bohnapfel exist. Apples and pears are used as dessert fruits and juice. For the soil samples, an area of 3081 m² for the streuobst orchard and of 2552 m² for the pasture was measured off, the streuobst orchard containing 34 scattered trees.

In the following, only the abbreviations of the place names will be used.

Tree Carbon

Procedure

10 trees were measured on each orchard except for one orchard with only 26 trees of similar appearance where only 6 trees were randomly picked. The mean of the 10 respectively 6 trees was entered into the formerly derived equation to calculate dry biomass. This value was multiplied by 0.5 to derive the carbon content per tree. To calculate the total tree carbon, the mean carbon content per tree was multiplied by the total number of trees on the orchard. The total tree carbon was finally divided by the area, resulting in the kilogramme carbon per square metre.
Statistics

The arithmetic mean, the standard deviation and the coefficient of variation of the carbon content per tree were calculated for each place. Additionally, correlations between the mean carbon content and the variables total number of trees on a plot, standard deviation of carbon per tree, total carbon per plot, plot area and carbon per square meter were determined. A box-plot of the tree carbon was created to visualize the variance and the mean carbon content of each place.

The statistical programme R (GNU General Public License, Boston, MA, USA) was used to create the box-plots, while the other statistical measures were calculated with SAS.

Soil Analysis

pH and bulk density

Procedure

On each plot, 30 soil samples were taken for the analysis of the carbon content and an additional four soil samples were used for the determination of pH and bulk density. Samples were taken in April and May 2012. For the bulk density, a sampling ring with a volume of 91.6088 cm³ was pushed into the soil on two randomly selected spots cleared from grass and litter to gain a sample from the topsoil. On the same spots, a hole was dug to a depth of about 50 cm and the sampling ring pushed into the soil for a sample from the subsoil. These samples were stored in plastic bags and their fresh weight was determined. They were sieved with a 2 mm sieve, stones, roots and dead organic matter were removed and a sample of 20 g was taken. After being oven-dried for 24 hours at 60 °C, the quotient of the two weights was formed and used to convert the total weight of the sample into its dry weight. Divided by the volume of the sampling ring, this calculation yielded the bulk density. The arithmetic mean of the two values for the topsoil was used for further calculations concerning the soil layer 0-10 cm, the mean value from the subsoil was used for the lowest layer of 50-60 cm. For the soil layer from 20-30 cm, the arithmetic mean of all four values was calculated.

From each sieved sample, two subsamples were taken to measure the pH with H₂O and CaCl₂. Where the mean of the four values for one site and one soil depth exceeded 6.5 with the extraction of CaCl₂, all soil carbon samples from this site were tested for calcium carbonate with the Scheibler apparatus. The principle of the Scheibler apparatus is to measure the volume of the carbon dioxide that is produced when calcium carbonate reacts with 10 percent hydrochloric acid solution. The acid is added in excess to the soil sample in an Erlenmeyer flask and reacts with the calcium carbonate to carbon dioxide. Thereby, an equivalent
amount of air is displaced. The air exerts pressure on a liquid column so that the amount of produced carbon dioxide can be read off the measuring burette containing the liquid. Since a gas volume depends on pressure and temperature, these two parameters must be measured and factored in (Loeppert & Suarez 1996). With the following formula, the percentage of calcium carbonate in the soil sample can be calculated:

\[
\% \text{CaCO}_3 = \frac{p \times V}{(273 + t) \times m \times R}
\]

where

- \(p\): air pressure [mbar]
- \(V\): volume CO\(_2\) [ml]
- \(t\): room temperature [\(^\circ\)C]
- \(m\): weight soil sample [g]
- \(R\): gas constant = 8.3144621 J mol\(^{-1}\) K\(^{-1}\)

### Statistics

It was tested if pH and bulk density differed significantly between land uses and between places.

Values for pH (measured with H\(_2\)O and with CaCl\(_2\)) and bulk density were first tested for normal distribution. As they were non-normally distributed, different transformations such as the log- or the arcsin-transformation were applied, but without reaching a normal distribution. Secondly, homogeneity of variance was tested for all three variables, transformed and non-transformed, both for groups according to land use and according to place. This was done with the Levene-test. In addition, the Levene-test modified by Brown and Forsythe was used because it can be applied to a larger variety of probability distributions and is also more selective (Dufner et al. 2002).

Because homogeneity of variances could not be assumed, a statistical method which is independent from a specific distribution assumption had to be applied. Therefore, a nonparametric test, the Kruskal-Wallis test, was used. The Kruskal-Wallis test is the nonparametric equivalent to the one-way analysis of variance and is based on the ranks of the sample values instead of the sample values themselves (Dufner et al. 2002). If only two groups had to be analysed, the Wilcoxon-test was applied. Since these two tests only detect differences between groups, group means were additionally calculated and a test for correlation between pH and bulk density was run.

The test for normality was performed with the Shapiro-Wilk statistic from PROC UNIVARIATE. PROC GLM with the options HOVTEST = LEVENE and HOVTEST = BF was used to test for homogeneity of variance. Group differences were examined with PROC NPAR1WAY, with the option WILCOXON producing the Kruskal-Wallis and the Wilcoxon test. Differences were considered significant when the p-value was
smaller than 0.05. To calculate group means, PROC UNIVARIATE was applied. PROC CORR was used to calculate the correlation between pH and bulk density.

**Soil carbon**

**Procedure**

5 spots distributed widely over the whole plot were chosen in the tree rows; at a distance of about 2-4 m to each of these spots, depending on the distance between the rows, 5 more samples were drawn being located between the tree rows. To reduce edge effects, a distance of at least 5 m was kept from the boundaries of the plots. A soil corer was driven down to a depth of 60-70 cm. The soil core was separated into three soil layers from 0-10 cm, 20-30 cm and 50-60 cm depth. On the field and the pastures, 10 evenly distributed spots were chosen and the same procedure applied. Soil samples were stored in hermetically sealed plastic bags and, after being transferred to plastic cups, air-dried at 35 °C for about 24 hours. They were sieved with a 2 mm sieve, the remaining fine roots were removed with forceps and the samples oven-dried at 60 °C for at least 24 hours.

For the determination of the carbon content, a Leco TruSpec CN Carbon/ Nitrogen Determinator (Leco Corporation St. Joseph, MI, USA) was used. This required subsamples between 50 and 250 mg being weighed in and placed in tin foil cups. The carbon analysis consists of three phases:

1. **Purging phase:** The sample in the tin capsule is entered into the machine and purged of any gases. Besides, the ballast volume and gas lines are purged as well.
2. **Combustion phase:** After being dropped into a furnace heated to 950 °C, the sample is combusted with pure oxygen. The gases hereby produced pass through the afterburner where they are further oxidised and particulates are removed, before moisture is removed by a furnace filter and a thermoelectric cooler. A collection vessel, the ballast, collects the combustion gases.
3. **Analyzing phase:** Again oxygen is added to the ballast and gases are purged through an CO\(_2\) infrared detector which measures the carbon as CO\(_2\).

*(LECO CORPORATION, operating instructions)*

**Calculations**

The measurement unit of the soil carbon from the C/ N Determinator is percent carbon in the sample. To calculate the amount of carbon per square metre soil, this value was first multiplied by 10 to obtain the unit gramme carbon per kilogramme soil. Since the stone and gravel content of the soil had not been determined, a default value of 5 % was assumed (GERZABEK 2005). Therefore, 5 % of the gramme carbon per kilogramme soil were subtracted. This value was then multiplied by the corresponding bulk density, so that measurement unit was kilogramme carbon per cubic metre. As sampling depth was 10 cm for every soil sample, the multiplication by
0.1 resulted in the kilogram carbon per square metre and was used for all further calculations.

**Statistics**

Soil carbon values were first tested for normal distribution and homoscedasticity and then log-transformed to increase normality of distribution. Since a normal distribution and homoscedasticity could not be assumed for all subsets of data, the Kruskal-Wallis test was used when necessary and also in addition to check the results obtained by other tests.

Two statistical approaches were applied. For the first approach, a paired site design was assumed: Places were considered to be independent replicates, land use was taken as the treatment factor. Thereby, the statistical design for the analysis of variance consisted of six blocks, the places, and two treatments, land use streuobst and land use pasture. Soil organic carbon was seen as a function of place, land use, pH and bulk density. For reasons of simplicity and because the measurement method did not change the relation of pH-values, only pH measured with H$_2$O was included in the model. The land use field was also omitted.

These effects were tested for significance with F-tests. Least squares means were calculated for the main effects.

As the land use field existed only at one place, a Kruskal-Wallis test was used to include the data from the field at place M1. The arithmetic mean, the median, the standard deviation and the coefficient of variation were calculated for each land use and depth layer. These calculations were performed additionally without place R1.

Since the main purpose was to detect differences between land uses and not to detect differences with soil depth, each depth layer was analysed separately.

To examine the effect of pH and bulk density on the carbon content, regressions of these parameters on soil organic carbon were calculated. The data set was the same as the one for the analysis of variance. The F-test was used as decision criterion. Where subsets of data were non-normally distributed, a log-transformation was done prior to regression analysis.

Of interest was furthermore the question if the location where the soil sample was taken, either in the tree row or between tree rows, had an effect on the carbon content. Analysis of variance was carried out, assuming a model where log-transformed carbon values were determined by the place and the interaction of place and location. The effect place×row was therefore taken up into the model. Row is a binary coded variable that takes the values “r” for the sample being taken in the tree row and “b” being taken between tree rows. The data set for the soil depth 20-30 cm was normally distributed and had equal variances, the data set for soil depth 0-10 cm had to be log-transformed. For the soil layer 50-60 cm and additionally for the other soil layers, Wilcoxon tests were carried out.
As the second statistical approach, each place was analysed separately. Land use was again taken as treatment factor, and the same calculations as described above were performed for each place.

To test for normality and homoscedasticity, residual plots were created with PROC UNIVARIATE and PROC GPLOT for visual assessment. Where results were ambiguous, the Shapiro-Wilk test from PROC UNIVARIATE and the Levene-test from PROC GLM with the options HOVTEST = LEVENE and HOVTEST = BF were used. PROC MIXED was applied to perform F-tests for significance of effects and to calculate least square means for the main effects. PROC TTEST was used for t-tests, PROC NPAR1WAY with the option WILCOXON was used to produce the Kruskal-Wallis and Wilcoxon test. Mean and median values and standard deviations were calculated with PROC GLM, regressions with PROC REG. Microsoft Office Excel 2007 (Microsoft Corporation Redmond, WA, USA) was used to calculate coefficients of variation. The level of significance was set at 0.05.

**Soil & Tree Carbon**

For a full carbon comparison of land uses, the carbon provided by the trees on the streuobst orchard was added to the soil carbon down to a depth of 60 cm and contrasted with the land use pasture. Biomass from grasses and other plants was considered to be equal for both land uses and was therefore omitted for the calculations. Tree roots were also disregarded due to reasons of practicality. The comparisons were performed over all places as well as separately for each place.

**Calculations & statistics**

Soil samples were only taken from depth layers 0-10, 20-30 and 50-60 cm. Therefore, soil carbon from depths in between had to be estimated. For each place and land use, mean values for each depth layer were calculated first. For the soil depth 10-20 cm, the arithmetic mean of the mean carbon content of layers 0-10 and 20-30 cm was then calculated. For the layer 30-40 cm, two thirds of the carbon content of layer 20-30 cm were added to one third of the carbon content of layer 50-60 cm. For layer 40-50 cm, one third of the carbon content of layer 20-30 cm was added to two thirds of the carbon content of layer 50-60 cm. Thereby, the carbon content was determined for a depth of 60 cm. Mean and median values for land uses were calculated.

The share of the tree carbon on total carbon was calculated for all places and separately for each place as well as arithmetic means of soil carbon down to 60 cm plus tree carbon for land uses. Data was tested for normal distribution with the Shapiro-Wilk test and for homoscedasticity with the Levene-test and the Levene-test modified by Brown and Forsythe. Since these tests were positive, an analysis of variance was conducted.
and the parameter land use tested for significance with the F-test and the t-test. Because group comparisons in most statistical analyses before were carried out with the Kruskal-Wallis and the Wilcoxon test, these two were also done to ensure comparability between analyses and to increase consistency. Analysis of variance, Kruskal-Wallis and Wilcoxon tests were performed for only soil carbon down to a depth of 60 cm and for soil carbon plus tree carbon. Again arithmetic mean and standard deviation was calculated. In addition, it was tested if results differed when place R1 was left out.

Tests for normality and homoscedasticity were done using the Shapiro-Wilk statistic from PROC UNIVARIATE respectively PROC GLM with the options HOVTEST = LEVENE and HOVTEST = BF. PROC MIXED was applied to perform F-tests and t-tests, PROC NPAR1WAY with the option WILCOXON to carry out Kruskal-Wallis and Wilcoxon tests. For arithmetic mean and standard deviation, PROC GLM was used, level of significance was 0.05.
Results

Tree Carbon Equation

Data Description

The 18 trees measured range from 5.4 m to a maximum of 9.0 m in tree height with a median of 7.2 m (Table 1). Variation in diameter at breast height was greater with the smallest value being 23.2 cm and the largest being 56.3 cm, expressed by the coefficient of variation which is 0.13 for tree height and 0.26 for dbh. It was only exceeded by the one for crown height with 0.38.

Table 1: Size characteristics of trees from destructive sample inventory.

<table>
<thead>
<tr>
<th>Statistical measure</th>
<th>Tree no.</th>
<th>Tree height [m]</th>
<th>Dbh [cm]</th>
<th>Crown height [m]</th>
<th>Crown radius [m]</th>
<th>Dry weight [kg]</th>
</tr>
</thead>
<tbody>
<tr>
<td>Tree no.</td>
<td>1</td>
<td>7.0</td>
<td>56.3</td>
<td>1.0</td>
<td>3.74</td>
<td>592.08</td>
</tr>
<tr>
<td></td>
<td>2</td>
<td>6.9</td>
<td>36.0</td>
<td>1.3</td>
<td>3.93</td>
<td>431.23</td>
</tr>
<tr>
<td></td>
<td>3</td>
<td>7.7</td>
<td>41.8</td>
<td>1.8</td>
<td>3.07</td>
<td>279.01</td>
</tr>
<tr>
<td></td>
<td>4</td>
<td>6.2</td>
<td>44.9</td>
<td>1.7</td>
<td>4.56</td>
<td>453.12</td>
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<tr>
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<td>5</td>
<td>6.5</td>
<td>46.3</td>
<td>1.6</td>
<td>3.09</td>
<td>378.58</td>
</tr>
<tr>
<td></td>
<td>6</td>
<td>8.1</td>
<td>30.1</td>
<td>1.8</td>
<td>2.19</td>
<td>150.60</td>
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<td>7</td>
<td>5.4</td>
<td>42.7</td>
<td>1.7</td>
<td>3.46</td>
<td>277.94</td>
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<tr>
<td></td>
<td>8</td>
<td>7.2</td>
<td>29.3</td>
<td>1.1</td>
<td>3.07</td>
<td>305.82</td>
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<tr>
<td></td>
<td>9</td>
<td>6.0</td>
<td>23.2</td>
<td>1.0</td>
<td>3.56</td>
<td>204.81</td>
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<td>10</td>
<td>7.5</td>
<td>23.6</td>
<td>1.9</td>
<td>3.27</td>
<td>178.61</td>
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<td>11</td>
<td>7.2</td>
<td>29.2</td>
<td>2.0</td>
<td>3.01</td>
<td>227.66</td>
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<td></td>
<td>12</td>
<td>7.7</td>
<td>29.9</td>
<td>2.5</td>
<td>2.72</td>
<td>305.07</td>
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<td></td>
<td>13</td>
<td>8.5</td>
<td>37.2</td>
<td>2.6</td>
<td>3.26</td>
<td>388.38</td>
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<tr>
<td></td>
<td>14</td>
<td>8.4</td>
<td>32.8</td>
<td>2.6</td>
<td>3.08</td>
<td>349.21</td>
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<td>7.2</td>
<td>33.2</td>
<td>3.2</td>
<td>2.92</td>
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<td>16</td>
<td>9.0</td>
<td>34.9</td>
<td>2.7</td>
<td>3.11</td>
<td>340.83</td>
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<td>17</td>
<td>8.3</td>
<td>29.0</td>
<td>1.0</td>
<td>3.51</td>
<td>373.52</td>
</tr>
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<td></td>
<td>18</td>
<td>7.0</td>
<td>23.8</td>
<td>1.0</td>
<td>2.71</td>
<td>215.53</td>
</tr>
</tbody>
</table>

| Arithmetic mean     | 7.32     | 34.68           | 1.81    | 3.24            | 321.1           |
| Standard deviation  | 0.94     | 8.89            | 0.68    | 0.52            | 108.92          |
| Coefficient of variation | 0.13   | 0.26            | 0.38    | 0.16            | 0.34            |

During the drying process, the two tree parts branches < 5 and branches > 5 showed a similar drying behaviour: Independent from their fresh weight, the constant weight was reached after 24 hours (Figure 1). Samples always comprised several pieces from different branches and twigs, so that sample fresh weight from branches < 5 was between 0.1 and 0.3 kg and from branches > 5 between 0.2 and 0.85 kg.
44 % of the original fresh weight was lost in the first 24 hours, followed by only minor losses of fewer than 1 % in the following 44 hours (Figure 1). Total mean weight loss added up to around 45 %. From this it follows that dry matter content is 55 and 57 % for branches < 5 and branches > 5, respectively. The lowest content was 50 %, the largest 63 % for both parts. Wood moisture content, calculated as the difference of fresh and dry weight divided by the dry weight, was between 0.61 and 1.01 with a mean value of 0.82, corresponding to 82 %.

Meanwhile, the stem parts took a longer time to reach constant weight. After 24 hours, they had only lost about 36 % of their fresh weight. During the next 22 hours, they lost an additional 15 % (Figure 2). This slower drying process resulted in a less pronounced curvature compared to the one from the branches. The arithmetic mean of the total weight loss was 47 %, so that dry matter content can be concluded to be 53 %. The maximum value here was 63 % as well, the lowest was 44 %.

Wood moisture content was between 0.59 and 1.26 with a mean of 0.91 or 91 %.
Calculations & Statistics

The Shapiro-Wilk test showed a normal distribution of the log-transformed values. The computation of Pearson correlation coefficients resulted in a slight negative correlation of -0.23 between the variables tree height and dbh (Table 2). Tree height and crown radius were also negatively correlated with -0.43 while a positive correlation of 0.45 could be found between tree and crown height. Dbh and crown height were not correlated in contrast to dbh and crown radius (0.44). The coefficient for crown height and crown radius showed a moderate negative correlation of -0.31. Noticeable was also the very low correlation of -0.02 respectively -0.07 between the pair height and dry weight and the pair crown height and dry weight, while dbh and crown radius showed a high positive correlation with dry weight.

Table 2: Pearson correlation coefficients for the variables height, dbh, crown height, crown radius and dry weight.

<table>
<thead>
<tr>
<th>Variable</th>
<th>Height</th>
<th>Dbh</th>
<th>Crown height</th>
<th>Crown radius</th>
<th>Dry weight</th>
</tr>
</thead>
<tbody>
<tr>
<td>Height</td>
<td>1.0000</td>
<td></td>
<td></td>
<td></td>
<td>-0.0184</td>
</tr>
<tr>
<td>Dbh</td>
<td></td>
<td>1.0000</td>
<td></td>
<td>0.4378</td>
<td>0.7717</td>
</tr>
<tr>
<td>Crown height</td>
<td></td>
<td></td>
<td>1.0000</td>
<td>-0.3103</td>
<td>-0.0690</td>
</tr>
<tr>
<td>Crown radius</td>
<td></td>
<td></td>
<td></td>
<td>1.0000</td>
<td>0.6286</td>
</tr>
<tr>
<td>Dry weight</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>1.0000</td>
</tr>
</tbody>
</table>

The F-test from the regression analysis confirmed the insignificance of the crown height for the estimation of the dry weight. The p-value of 0.6117 for the F-statistic exceeded the significance level of 0.05 and crown height was therefore dropped for
further regression analysis. In contrast to the low correlation coefficient, tree height proved to be a significant term.

For the extrapolation, the model was extrapolated in both directions: extreme values were inserted into the equation that were lower and higher than the ones from the destructive sample inventory. As the model yielded reasonable values for dry weight, it can be assumed to be valid for the application to tree forms and sizes different from the ones used.

The estimates for the parameters are shown in Table 3 along with the standard errors, t-values and p-values.

| Parameter      | Estimate       | Standard error | t-value | Pr > |t| |
|----------------|----------------|----------------|---------|-------|---|
| ln (a₀)        | -0.32463360    | 1.10302031     | -0.29   | 0.7728  |
| a₁             | 0.89994067     | 0.37148053     | 2.42    | 0.0296  |
| a₂             | 0.81727675     | 0.19573794     | 4.18    | 0.0009  |
| a₄             | 1.18911520     | 0.32853326     | 3.62    | 0.0028  |

a₀ was included in the model although it was not significant (p 0.7728) because the hypothesis that a₀ is equal to 1 is negligible due to the fact that data was log-transformed to facilitate linear regression for an exponential model. Additionally, the correction factor λ was introduced in any case.

The bias caused by the logarithmic transformation of the data, the average deviation, was -3.464.

To correct the bias and thereby minimize the average deviation, the correction factor λ was calculated. The division of the sum of the surveyed dry weights by the sum of their estimates resulted in a λ of 1.0109.

The final equation that can be used to calculate the dry biomass of an apple tree from its tree height, dbh and crown radius is therefore:

\[
dw_{tree} = 1.0109 \times \exp(-0.3246) \times h^{0.8999} \times dbh^{0.8172} \times cr^{1.1891}
\]

\[
= 0.9965 \times h^{0.8999} \times dbh^{0.8172} \times cr^{1.1891}
\]

where

- \( dw \): dry weight [kg]
- \( h \): tree height [m]
- \( dbh \): diameter at breast height [cm]
- \( cr \): crown radius [m]

The calculation of the measure of fit yielded a se(\( \hat{y} \)) of 49.8979 for the model without lambda and of 49.7966 for the model with lambda, the R-Square was 0.8148 without lambda and 0.8156 with lambda.
Streuobst Carbon Storage

Tree Carbon

Data description

The number and the dimensions of the trees varied widely between the different places. While only 15 trees grow on plot K2, more than 80 are located on plot M1 (Table 4, Figure 3). The largest trees could be found on plot K2 with a mean tree height of 9.91 m, a mean dbh of 43.75 cm and a mean crown radius of 3.93. Compared with K2, the trees on plot M1 were the smallest with a tree height of 5.78 m, a dbh of 20.25 cm and a crown radius of 2.72 m (Figure 3).

Calculations & statistics

When applying the formerly derived equation for fruit tree biomass, these dimensions resulted in a mean dry weight of 745.74 kg for K2 and 150.70 kg for M1. The mean carbon content of the trees on these two plots is therefore 372.87 and 75.35 kg C (Table 4). Taking into account the total number of trees on the plots and the plot area, a value between 0.68 and 3.28 kg C m$^{-2}$ was found. The tree carbon m$^{-2}$ averaged over all places was 2.17 kg C m$^{-2}$ or 21.7 Mg C ha$^{-1}$.

Table 4: Number of measured trees (n), total number of trees on plot (N), plot area, tree density, calculated mean tree carbon, standard deviation of tree carbon, cv, total tree carbon, and tree carbon per m$^2$ for every place.

<table>
<thead>
<tr>
<th></th>
<th>B1</th>
<th>K1</th>
<th>K2</th>
<th>M1</th>
<th>M2</th>
<th>R1</th>
</tr>
</thead>
<tbody>
<tr>
<td>n</td>
<td>10</td>
<td>6</td>
<td>10</td>
<td>10</td>
<td>10</td>
<td>10</td>
</tr>
<tr>
<td>N</td>
<td>56</td>
<td>16</td>
<td>15</td>
<td>81</td>
<td>21</td>
<td>34</td>
</tr>
<tr>
<td>Plot area [m$^2$]</td>
<td>2153.50</td>
<td>1261.00</td>
<td>1582.00</td>
<td>8938.50</td>
<td>4600.00</td>
<td>3081.00</td>
</tr>
<tr>
<td>Tree density [N ha$^{-2}$]</td>
<td>260.04</td>
<td>126.88</td>
<td>94.82</td>
<td>90.62</td>
<td>45.65</td>
<td>110.35</td>
</tr>
<tr>
<td>Arithmetic mean tree carbon [kg]</td>
<td>98.64</td>
<td>258.70</td>
<td>372.87</td>
<td>75.35</td>
<td>252.04</td>
<td>163.27</td>
</tr>
<tr>
<td>Standard deviation tree carbon [kg]</td>
<td>108.29</td>
<td>124.13</td>
<td>277.95</td>
<td>43.10</td>
<td>263.70</td>
<td>57.76</td>
</tr>
<tr>
<td>Coefficient of variation</td>
<td>1.10</td>
<td>0.48</td>
<td>0.75</td>
<td>0.57</td>
<td>1.05</td>
<td>0.35</td>
</tr>
<tr>
<td>Total tree carbon [kg]</td>
<td>5523.76</td>
<td>4139.21</td>
<td>5593.03</td>
<td>6103.34</td>
<td>5292.79</td>
<td>5551.09</td>
</tr>
<tr>
<td>Tree carbon m$^2$ [kg C m$^2$]</td>
<td>2.57</td>
<td>3.28</td>
<td>3.54</td>
<td>0.68</td>
<td>1.15</td>
<td>1.80</td>
</tr>
</tbody>
</table>
A strong negative correlation of -0.89 could be observed for mean tree carbon and number of trees (N). The standard deviation of tree carbon and tree carbon m$^{-2}$ showed a positive correlation with mean tree carbon of 0.83 and 0.63, respectively. Tree density and tree carbon m$^{-2}$ were negatively correlated with a coefficient of -0.82.

**Soil Analysis**

**pH and bulk density**

pH-values range from 6.00 to 7.49 for the measurement with H$_2$O and from 5.10 to 6.86 for the CaCl$_2$-solution. A high value for the measurement with H$_2$O was usually reflected in a high value for the measurement with CaCl$_2$ and the other way around. An exception were the pH-values from the field: The pH of the field was with 6.02 higher than streuobst and pasture with 5.86 and 5.80 respectively when measured with CaCl$_2$. In contrast, the field and the streuobst had the same pH of 6.65 and the pasture a slightly lower with 6.56 when H$_2$O was used.

Statistical tests showed that pH and bulk density were non-normally distributed, both the non-transformed and the transformed values. According to the Levene-test and
the Levene-test modified by Brown and Forsythe, the null hypothesis of homoscedasticity had to be rejected ($p < 0.05$) too. Consequently, a nonparametric test had to be applied. The Kruskal-Wallis test showed differences between land uses for all soil depths taken together and separately at soil layers 20-30 and 50-60 cm for both pH-measurement methods. Mean values from all soil depths together of the $\text{H}_2\text{O}$-measurement method were 6.74 for streuobst, 6.57 for pasture and 6.71 for field. The $\text{CaCl}_2$-method resulted in mean values of 6.06 for streuobst, 5.87 for pasture and 6.21 for field. When soil layers were analysed separately, the pH of the land use pasture was always lower than that of the land use streuobst (Figure 4).

**Figure 4:** Arithmetic mean and standard deviation of pH measured with $\text{H}_2\text{O}$ at all three soil depths for land uses streuobst (s), pasture (p) and field (f).

Besides, the pH was found to increase with soil depth. The difference between pH at depth layer 0-10 cm and pH at depth layer 50-60 cm was the largest for the land use streuobst with 0.22, whereas it was only 0.09 for pasture. The difference between streuobst and pasture increased with increasing soil depths: While the difference was 0.11 at soil layer 0-10 cm with the pH for streuobst being 6.64 and the pH for pasture being 6.53, it rose to 0.24 at soil layer 50-60 cm with a pH of 6.86 for streuobst and 6.62 for pasture.

When the land use field was left out of the statistical analysis, the Wilcoxon test showed significant differences between land uses over all depths and for each soil layer with the $\text{H}_2\text{O}$-method. Results for the $\text{CaCl}_2$-method were different: Only for all layers together and at layer 50-60 cm could differences be detected between only streuobst and pasture.

The minimum value for bulk density was 0.80 g cm$^{-3}$ and the maximum value was 1.54 g cm$^{-3}$. At the layers 0-10 and 20-30 cm, mean bulk density was lowest for the
land use streuobst and highest for the land use field. At layer 50-60 cm, the mean streuobst bulk density was slightly above that of the field (Figure 5). Standard deviation was always lower for streuobst than for pasture.

Figure 5: Arithmetic mean and standard deviation of bulk density at all three soil depths for land uses streuobst (s), pasture (p) and field (f).

Bulk density differed only significantly between all three land uses when all depths were taken together (p 0.0085) and at the soil layer 0-10 cm (p 0.0003), but not at layers 20-30 (p 0.2153) and 50-60 cm (p 0.0522). For all depths, calculations yielded a mean bulk density of 1.13 g cm\(^{-3}\) for streuobst, of 1.14 g cm\(^{-3}\) for pasture and of 1.23 g cm\(^{-3}\) for field.

If the land use field was omitted and only the groups streuobst and pasture were compared, bulk density was not significantly different between pasture and streuobst at depth layers 0-10 (p 0.1141), 20-30 cm (p 1.0000) and over all layers (p 0.7611). The p-value for the comparison at soil layer 50-60 cm showed a significant difference with 0.0085, compared to 0.0522 for all three land uses. Just as pH-values, bulk density-values tended to rise with increasing soil depth.

When comparing the places without taking into account the different land uses, all three variables pH measured with H\(_2\)O, pH measured with CaCl\(_2\) and bulk density showed significant differences over all soil layers and separately at each layer (p < 0.0001). When the land use field was omitted, a difference existed with a p-value of 0.0111.

Over all soil layers, place R1 had the lowest mean pH of 6.21 for H\(_2\)O and of 5.41 for CaCl\(_2\), while B1 reached the highest values of 7.09 for H\(_2\)O and 6.37 for CaCl\(_2\) (Figure 6). For bulk density, B1 had the lowest value of 0.92 g cm\(^{-3}\), M1 the highest of 1.31 g cm\(^{-3}\).
A test for correlation between pH and bulk density showed no significance for any soil depth.

Soil samples from the streuobst orchard at place B1 reached a mean pH of 6.63 with the CaCl₂-measurement method which exceeds the limit of 6.5, so that all soil carbon samples were tested for calcium carbonate with the Scheibler apparatus. However, none of the samples showed a noteworthy reaction, so that calcium carbonate content could be assumed to be negligible.

Soil carbon

First statistical approach

Data description
Soil organic carbon content was in the range of 0.249 to 7.420 %. When transformed into kg carbon per m², values were between 0.2782 and 6.3089 kg C m⁻². When the arithmetic mean of the three analysed soil depths 0-10, 20-30 and 50-60 cm was calculated for each land use, streuobst and pasture had a very similar carbon content of 2.1611 kg C m⁻² and of 2.1683 kg C m⁻², respectively, and field had 1.3001 kg C m⁻².
Separate analysis of soil layers showed a strong decrease of soil organic carbon with soil depth (Figure 7): At layer 0-10 cm, streuobst had a mean value of 3.2214 kg C m$^{-2}$, pasture of 3.2425 kg C m$^{-2}$ and field of 1.8864 kg C m$^{-2}$. At layer 50-60 cm, corresponding values were 1.0877, 1.0325 and 0.5707 kg C m$^{-2}$.

![Figure 7: Arithmetic mean and standard deviation of soil organic carbon at all three soil depths for land uses streuobst (s), pasture (p) and field (f).](image)

Median values differed slightly with 3.2650, 2.7900 and 1.5645 % at layer 0-10 cm, 1.8450, 1.6500 and 1.5645 % at layer 20-30 cm and 0.8605, 0.7855 and 0.4795 % at layer 50-60 cm for streuobst, pasture and field, respectively. If the unit kg carbon m$^{-2}$ was used, median values were 3.0949, 2.9826 and 1.8268 kg C m$^{-2}$ at layer 0-10 cm, 1.9315, 1.8793 and 1.3893 kg C m$^{-2}$ at layer 20-30 cm and 1.0240, 0.8371 and 0.5597 kg C m$^{-2}$ at layer 50-60 cm for streuobst, pasture and field, respectively.

The decrease of the mean soil carbon content with depth was less than the decrease of the standard deviation, so that the coefficient of variation increased with depth. It was much higher for the land uses streuobst and pasture than for field over all depths: By way of example, in the top layer, the coefficient of variation was 0.33 respectively 0.38 for the land uses streuobst and pasture but only 0.16 for field.

Because the carbon content of the pasture on place R1 seemed to be unreasonably high, arithmetic means, standard deviations and medians were additionally calculated without values from this place. This resulted in the fact that streuobst was then the land use with the highest soil organic carbon values at each depth layer (Figure 8). They were 3.2400, 2.2151 and 1.2412 kg C m$^{-2}$ at soil depths 0-10, 20-30 and 50-60 cm, respectively, compared to pasture with 3.0083, 2.0610 and 1.0352 kg C m$^{-2}$. The standard deviation of the pasture decreased by up to 14 % at depth layer 50-60 cm. Also the coefficient of variation declined from a mean coefficient of 0.47 to 0.45.
Results - Streuobst Carbon Storage

Figure 8: Arithmetic mean and standard deviation of soil organic carbon at all three soil depths for land uses streuobst (s), pasture (p) and field (f) without place R1.

Median values were then 3.0911, 2.5642 and 1.8087 kg C m\(^{-2}\) at soil layer 0-10 cm, 2.0836, 1.8218 and 1.3893 kg C m\(^{-2}\) at 20-30 cm and 1.2100, 0.9292 and 0.5597 kg C m\(^{-2}\) at 50-60 cm for streuobst, pasture and field, respectively.

Statistics

Effect of land use on soil organic carbon storage
To test if differences between land uses were significantly different, a model containing the variables place, land use (streuobst and pasture), bulk density and pH measured with H\(_2\)O was assumed and soil layers were analysed separately. Visual assessment showed a normal distribution and homoscedasticity of residuals at the depth layers 0-10 and 20-30 cm. Values from depth layer 50-60 cm had to be log-transformed.

At depth layer 0-10 cm, the variables place, land use and pH were significant with p-values for the F-test of < 0.0001, 0.0053 and 0.0008, respectively. The p-value for bulk density was with 0.7143 above the confidence limit of 0.05. Calculation of least squares estimates yielded an estimate of 3.6296 for land use streuobst and of 2.8343 for land use pasture. When the effect bulk density was omitted, the F-values increased slightly for the other effects and estimates changed to 3.6114 for streuobst and 2.8526 for pasture.

Land use was also a significant effect at depth layer 20-30 cm (p 0.0013). Least squares estimates here were 2.6121 for streuobst and 1.7917 for pasture. Besides place and pH, bulk density was significant too (p < 0.0001, 0.0000 and 0.0395, respectively).
Results were different for depth layer 50-60 cm. Only the effect place was significant (p < 0.0001), the other factors land use, pH and bulk density had p-values of 0.2663, 0.8679 and 0.3016, respectively. The differences in carbon content between land uses were still not significant when the effects pH and bulk density were omitted (p 0.2316).

As the land use field was not included in the model described above, another procedure comparing group means was applied. The Shapiro-Wilk test showed a non-normal distribution of the residues for this data set. Also the group variances were not equal as pointed out by the Levene test and the modified Levene test. Thus, Kruskal-Wallis tests were carried out. According to the tests performed over all places, there was no significant difference between the soil carbon content on the land uses streuobst and pasture for any of the three soil layers analysed. Only when the land use field was taken into account, the Kruskal-Wallis test showed differences. At layer 0-10 cm, the p-value was 0.0002 when comparing all three land uses; when the land use field was omitted, the p-value was only 0.8092. For the other two layers, the situation was similar: The carbon content of the field was about half of the carbon of the other two land uses and was thereby responsible for the significant difference between land uses. p-values for the Kruskal-Wallis test at soil layer 20-30 cm were 0.0212 for all three land uses and 0.7171 for the comparison streuobst – pasture. For the depth from 50-60 cm the p-value was 0.0492 for the three land uses and 0.5742 for streuobst and pasture.

**Effect of pH and bulk density on soil organic carbon storage**

For the quantification of the effect that pH and bulk density have on the carbon content, regression analysis was carried out. At depth layer 0-10 cm, the regression function for pH measured with $H_2O$ on soil organic carbon was

\[ y = -2.1451 + 0.8332 \times pH \]

where \( y \): carbon \([\text{kg C m}^{-2}]\)

The R-Square was rather low with 0.053. The function for the depth layer 20-30 cm was similar, the R-Square was 0.1023.

According to the analysis of variance, bulk density was only significant at depth layer 20-30 cm. The regression function was here

\[ y = 5.4738 - 2.8785 \times bd \]

The R-Square was with 0.3096 larger than the two coefficients of determination for pH.
Effect of sampling location on soil organic carbon storage
To test if the location of the soil sample on the plot had an effect on the carbon content, analysis of variance was performed with the two effects place and place×row. Data from depth layer 0-10 cm was non-normally distributed and was log-transformed. The p-value from the F-test was 0.0365, so that the effect that the location of the soil sample has, either drawn in the tree row or between tree rows, is significant. At depth layer 20-30 cm, this effect was not significant (p 0.0680). Data from depth layer 50-60 cm was not normally distributed, so a Wilcoxon test was carried out. It showed no significant difference between the carbon content of the soil samples taken in the tree row and the samples taken between tree rows (p 0.5298). To facilitate comparison between depth layers and because p-values from the F-test were only just above the confidence limit, the Wilcoxon test was additionally run for the other two soil layers. No significant difference could be shown here with the p-value at layer 0-10 cm being 0.0863 and that at layer 20-30 cm being 0.2340. However, a general tendency towards more carbon in the tree rows is observable (Figure 9). The mean carbon content in the tree row over all depths was 2.2875 kg C m\(^{-2}\) compared with 2.0584 kg C m\(^{-2}\) between the tree rows. The carbon content was between 7-16 % lower between the rows than in the rows when soil layers were analysed separately.

Second statistical approach

Data description
The separate analysis of the carbon content for each place showed large differences between places, as was indicated by the analysis of variance (Figure 10). Mean soil organic carbon values range from 2.0310 to 5.0164 kg C m\(^{-2}\) at depth layer 0-10 cm,
from 1.3905 to 3.4435 kg C m\(^{-2}\) at depth layer 20-30 cm and from 0.5332 to 1.6730 kg C m\(^{-2}\) at depth layer 50-60 cm. Places that have a high soil carbon content at one layer also have high contents at the other layers and vice versa. Places B1, K2, M1 and M2 have higher soil carbon contents at the streuobst orchard at all soil layers. Place K1 has larger values for pasture at soil depth layers 0-10 and 50-60 cm, while R1 is the only place with larger values for pasture at all soil depths. Standard deviation is also rather high, especially at soil depth 50-60 cm.

Figure 10: Arithmetic mean and standard deviation of soil organic carbon at depth layers 0-10 cm, 20-30 cm and 50-60 cm for land uses streuobst (s), pasture (p) and field (f), separately for places.
Statistics

Effect of land use on soil organic carbon storage

The Shapiro-Wilk test for normality done separately for each place and soil layer showed a non-normal distribution in three cases, each time concerning the layer 50-60 cm. Both the Levene-test and the Levene-test modified by Brown and Forsythe indicated normality of variance for these places and layers, but heteroscedasticity for the layer 0-10 cm on place M1 (p 0.0001 and 0.0004 for Levene-test and for Levene-test modified by Brown and Forsythe, respectively). So the Kruskal-Wallis and the Wilcoxon test were used again to ascertain the results obtained by t-tests for log-transformed data.

The analysis of soil carbon separately for each place showed in most cases no significant difference between the soil carbon means from the groups pasture and streuobst (p > 0.05). Only at place R1 for depth layers 0-10 and 20-30 cm, at place M1 for layer 20-30 cm and at place K2 for layers 0-10 and 20-30 cm were p-values for the t-test smaller than 0.05. These results were confirmed by the Wilcoxon tests except for the layer 0-10 cm at place K2 (p 0.0588). Instead, Wilcoxon tests showed a significance difference between the land uses streuobst and pasture at layer 50-60 cm (p 0.0494).

The calculation of the arithmetic means of these places and soil layers identified the land use pasture as having higher carbon contents at places R1 and M1: At R1 at the soil layer 0-10 cm, the pasture had a mean carbon content of 4.4137 kg C m⁻² while the streuobst orchard had only a mean value of 3.1282 kg C m⁻². The difference was even larger at the layer 20-30 cm where the mean value for the pasture was 3.0734 kg C m⁻² and 1.9694 kg C m⁻² for the streuobst orchard. At place M1, the mean pasture carbon content was 1.7676 kg C m⁻², the mean streuobst carbon content was 1.3905 kg C m⁻² and the mean field carbon content was 1.4432 kg C m⁻². The difference between land uses was significant at this place between all three land uses (p 0.0331) as well as between pasture and streuobst (p 0.0191). The situation was different at place K2. Here, the mean carbon content was higher in the streuobst orchard in all soil depths: At layer 0-10 cm, streuobst reached a mean value of 3.1002 kg C m⁻² while that of pasture was 2.3544 kg C m⁻²; in the layer 20-30 cm, the streuobst orchard had 2.1687 kg C m⁻² and the pasture 1.5531 kg C m⁻²; in layer 50-60 cm, the carbon content was 1.6589 kg C m⁻² in the streuobst and 1.1904 kg C m⁻² in the pasture.

Effect of sampling location on soil organic carbon storage

Differences between samples taken in the tree row and between tree rows were tested for significance separately for each place. Normal distribution and homoscedasticity could not be assumed for this dataset as the Shapiro-Wilk test respectively the Levene-test showed. Therefore, the Wilcoxon test was used to detect differences between the groups of samples being taken in the tree row and the groups of samples being taken between tree rows. No differences could be identified for any place and soil layer except for place M1 where differences existed at all soil layers and over all layers (p 0.0159 at layers 0-10 and 20-30 cm, p 0.0317 at layer
50-60 cm and p 0.0182 over all layers). Calculation of mean values showed that soil organic carbon content was higher in the tree row than between tree rows in all cases.

**Soil organic carbon storage with soil depth**
As was already determined with the first statistical approach, the soil carbon content decreases with depth (Figure 11). This is the case for all places even though quantitative differences of the decrease exist. The streuobst orchard at place R1 has the greatest decrease of soil organic carbon with depth. With a carbon content of 3.1285 kg C m$^{-2}$ in the top layer and of 0.5332 kg C m$^{-2}$ in the bottom layer, the decrease amounts to 83.0 %. The carbon decrease is the lowest at the streuobst orchard at place K2 with 46.5 %. Furthermore it is visible that the difference in carbon content between the land uses is more pronounced in the upper soil layers and decreases with soil depth. Also the standard deviation diminishes with increasing soil depth.
Figure 11: Decrease of soil organic carbon with soil depth for land uses streuobst (s), pasture (p) and field (f), separately for places. Bars represent standard errors of the mean.
Soil & Tree Carbon

To calculate the total carbon of a plot, the soil carbon down to a depth of 60 cm was converted into carbon per square meter and the above ground carbon, namely the carbon provided by the trees, was added. This resulted in soil organic carbon values between 7.58 kg C m\(^{-2}\) or 75.79 Mg C ha\(^{-1}\) and 19.48 kg C m\(^{-2}\) or 194.79 Mg C ha\(^{-1}\) (Table 5). A comparison of the mean soil carbon values for each land use showed that the land use field had the lowest value of 7.58 kg C m\(^{-2}\) and pasture the highest with 13.09 kg C m\(^{-2}\) while the mean soil carbon value for streuobst was slightly lower with 12.51 kg C m\(^{-2}\). When taking into account the median values, streuobst had a marginally higher median of 11.79 kg C m\(^{-2}\) than pasture with 11.32 kg C m\(^{-2}\).

The smallest value for the carbon provided by the trees was 0.6828 kg C m\(^{-2}\) at place M1, the largest was 3.5354 kg C m\(^{-2}\) at place K2. The share of the trees on the total carbon for the land use streuobst was lowest at place M1 with 7.43 % and amounted to 20.89 % at place K2. The share of tree carbon on places M2 and B1 was on the lower range with 10.2 and 11.64 %, respectively, while the trees on R1 had a share of 14.43 %. The percentage of tree carbon on total carbon on K1 was rather high with 20.28 %. All in all, tree carbon amounted to a mean share of 14.15 % on total carbon.

Table 5: Carbon from field soil, pasture soil, streuobst, trees and total carbon streuobst separately for places. Arithmetic means and sums of field, pasture, streuobst soil carbon, tree carbon and streuobst.

<table>
<thead>
<tr>
<th>Statistical measure</th>
<th>Place</th>
<th>Soil carbon field [kg C m(^{-2})]</th>
<th>Soil carbon pasture [kg C m(^{-2})]</th>
<th>Soil carbon streuobst [kg m(^{2})]</th>
<th>Carbon trees [kg C m(^{-2})]</th>
<th>Total carbon streuobst [kg C m(^{-2})]</th>
</tr>
</thead>
<tbody>
<tr>
<td>Arithmetic mean</td>
<td></td>
<td>7.5793</td>
<td>13.0888</td>
<td>12.5140</td>
<td>2.1697</td>
<td>14.6837</td>
</tr>
<tr>
<td>Sum</td>
<td></td>
<td>7.5793</td>
<td>78.5329</td>
<td>75.0841</td>
<td>13.0180</td>
<td>88.1021</td>
</tr>
<tr>
<td>B1</td>
<td></td>
<td>18.3600</td>
<td>19.4792</td>
<td>2.5650</td>
<td>22.0442</td>
<td></td>
</tr>
<tr>
<td>K1</td>
<td></td>
<td>12.8512</td>
<td>12.9020</td>
<td>3.2825</td>
<td>16.1845</td>
<td></td>
</tr>
<tr>
<td>M1</td>
<td></td>
<td>7.5793</td>
<td>8.7268</td>
<td>8.5042</td>
<td>0.6828</td>
<td>9.1870</td>
</tr>
<tr>
<td>M2</td>
<td></td>
<td>8.9310</td>
<td>10.1262</td>
<td>1.1506</td>
<td>11.2768</td>
<td></td>
</tr>
<tr>
<td>R1</td>
<td></td>
<td>19.8687</td>
<td>10.6826</td>
<td>1.8017</td>
<td>12.4844</td>
<td></td>
</tr>
</tbody>
</table>

When the above ground carbon from the trees is added to the soil carbon, all places had a higher carbon content on the streuobst orchards except for place R1 (Figure 12). Maximum values here were 22.04 kg C m\(^{-2}\) or 220.4 Mg C ha\(^{-1}\) at place B1 and 16.93 kg C m\(^{-2}\) or 169.3 Mg C ha\(^{-1}\) at place K2. The arithmetic mean of the streuobst carbon increased to 14.68 kg C m\(^{-2}\) or 146.7 Mg C ha\(^{-1}\), therewith being higher than the one for pasture of 12.51 kg C m\(^{-2}\) or 125.1 Mg C ha\(^{-1}\). The standard deviation for both mean values was 4.65 for streuobst and 4.92 for pasture. The addition of the tree carbon did not change the ranking order of the places according to soil carbon: place B1 still had the highest amount of carbon, followed by...
places K2 and K1. Third ranks R1, while M2 and M1 had the lowest amounts of carbon.

![Figure 12: Soil organic carbon for a soil depth of 60 cm for land uses streuobst (s), pasture (p) and field (f) and tree carbon (trees), separately for all places.]

Statistical analysis showed that data was normally distributed and had equal variances, so that analysis of variance could be used. When all places were included and only the soil carbon down to a depth of 60 cm was taken into account, neither the F-test nor the t-test showed a significant effect of land use on total carbon (p 0.8264, p 0.6083). The Kruskal-Wallis and the Wilcoxon test also showed no group differences (p 0.8728, p 0.9372). The same situation was the case when carbon from both sources, soil and trees, was taken into the analysis: p-values were 0.5887 for the F-test and 0.9048 for the t-test. p-values from the Kruskal-Wallis and the Wilcoxon test decreased, but still were above the significance level of 0.05 with 0.4103 and 0.4848.

Tests were also run without place R1, but even though p-values declined again to a value as low as 0.2306 for the Kruskal-Wallis, they were still above the significance level (p 0.2759 for F-test, p 0.6795 for t-test, p 0.3095 for Wilcoxon).

However, a clear trend was visible when R1 was omitted as can be seen in Figure 13. The mean carbon value for streuobst was then 15.12 kg C m\(^{-2}\) or 151.2 Mg C ha\(^{-1}\) with a standard deviation of 5.06, while that of pasture was 11.73 kg C m\(^{-2}\) or 117.3 Mg C ha\(^{-1}\) with a standard deviation of 4.06.
When all places were taken together, soil carbon from the land use streuobst amounted to 75.08 kg C m\(^{-2}\), slightly less than soil carbon from the pasture with 78.53 kg C m\(^{-2}\) (Figure 14a). When the carbon provided by the trees on the streuobst orchard was added, streuobst was the land use having more carbon with 88.1 kg C m\(^{-2}\).

As was mentioned before, the soil carbon for the pasture at place R1 seems to be extraordinary high. At the other places, the soil carbon content of the pasture was similar to the one of streuobst which was not the case at place R1. If R1 is therefore assumed to be an outlier and left out of the analysis, soil carbon content on the streuobst orchard was higher than on the pasture. It was then 75.62 kg C m\(^{-2}\) while that of pasture was 58.66 kg C m\(^{-2}\) (Figure 14b).
Discussion

Tree Carbon Equation

Sample Characterisation

The trees used for the derivation of the allometric equation should provide a representative sample of the respective tree species under given climatic and management conditions. Apple trees have a growth height of 10-12 m (FISCHER et al. 2007), and trees on streuobst orchards are typically standard trees with log lengths of 1.6-1.8 m or more (HERZOG 1998). Therefore the maximum height of 9.0 m in the destructive sample inventory seems appropriate for the general application to apple trees grown on streuobst orchards in the temperate region of Europe. Maximum tree height of the trees described in the second part of this study, in the following called streuobst sample trees, was larger with 13.6 m, but the extrapolation of the model ensured that it could also be used for trees not within the sampled range. Compared to other destructive sample inventories, the maximum height covered in relation to the common growth height is rather large: SAJDAK & VELAZQUEZ-MARTI (2012), investigating the biomass of the Pagoda tree (Sophora japonica (L.)) which grows to a height of about 20 m (CSU 2010) only used values from trees of up to 12.4 m. Similarly, the maximum height of the spruces (Picea abies (L.) H. Karst) in HOCHBICHLER et al.’s (2006) study was only 31.2 m although spruces commonly reach heights of 50 m (SULLIVAN 1994, TU DRESDEN 2000). The range of tree height was rather small, resulting in a low coefficient of variation of 0.13.

Even more important than the range of tree heights for the precision of biomass regression equations is the dbh range, because the trees with a large diameter usually account for a comparatively great proportion of the aboveground biomass (BROWN 2001, CHAVE et al. 2004, HOUGHTON et al. 2001). The arithmetic mean of the dbh of the destructively sampled trees was with 34.68 cm larger than the mean dbh of the streuobst trees of 32.9 cm. Also, the sample inventory included 9 trees with a larger dbh than 32.9 cm, the mean dbh of the streuobst sample trees. 5 out of these were above 40 cm, and 1 even exceeded 50 cm. Although the largest tree in the streuobst sample had a dbh of 67.5 cm, the range covered in the destructive sample inventory seems sufficient. This impression was supported by the relatively high coefficient of variation of 0.26 and a comparison with literature values, which showed that the range of 33.1 cm was similar to the dbh ranges used by other authors (HOCHBICHLER et al. 2006, LEVIA JR. 2008, SUCHOMEL et al. 2012 in press). The coefficient of variation was largest for crown height with 0.38, indicating the broad range covered, which also included half-standard trees with a crown height below 1.8 m.
Whether the dry weights from the destructively sampled trees are representative is
difficult to assess because the biomass of a tree respectively of a plant is influenced
by many factors (see “Introduction - Vegetation Carbon”) and varies widely even
between trees grown under the same environmental conditions. The mean dry weight
of 321.1 kg per tree is close to the 343 kg which STERBA & NACHTMANN (2009) found
for Austrian oaks (Quercus L.) with a mean dbh of 24 cm and a mean tree height of
16 m. Since apple trees and oaks have similar growth habits, this could point to
representative dry weights.

The number of trees used for the development of a regression equation varies
greatly. ZIANIS et al. (2005) in their review of equations for tree species in Europe
found numbers from 3 to 1503. The most usual tree number was 11-20 which
coincides with TRITTON & HORNBECK’s (1982) review concerning major tree species of
the northeastern United States. SUCHOMEL et al. (2012 in press) used 12 trees per
species and site for their destructive sample inventory of coppiced oaks in Germany,
STERBA & NACHTMANN (2009) used 15-24 for six tree species in Austria. It can
therefore safely be assumed that the 18 trees used for the sample inventory in this
study are enough.

Drying Behaviour

The differences in the drying behaviour of the wood compartments can mainly be
ascribed to the different initial moisture contents (BERBEROVIC & MILOTA 2011,
MÖTTÖNEN 2006). With 0.91, the average stem moisture content was larger than that
of the branches with 0.82, a matter of fact commonly observed in different tree
species (DIBDIAKOVA 2011, GIBSON et al. 1986). Besides, the stem discs were thicker
than branch samples which might also have protracted the drying process. The
moisture contents were in the same range as birch wood (Betula pendula Roth)
(MÖTTÖNEN 2006), higher than the ones for citrus wood (SAJDAK & VELAZQUEZ-MARTI
2012) and lower than those for western hemlock (Tsuga heterophylla (Raf.) Sarg.)
(2012a) found the same drying time of 24 hours at 105 °C for branch wood samples.
These similarities suggest that the procedure was carried out correctly and that
results are representative.

Variables

The calculation of Pearson correlation coefficients showed that dry weight was most
strongly correlated with dbh. This is in accordance with all other studies treating the
subject of regression equations for tree biomass (e. g. BROWN 2001, HÄGER 2012 in
KORZUKHIN 1997, TRITTON & HORNBECK 1982, ZIANIS et al. 2005). If not used as the
single one, the dbh was at least the best predictor of biomass in these studies. In
most of them, the correlation coefficient was between 0.77 and 0.98 (KUYAH et al.
2012a). Crown radius, which had a correlation coefficient of 0.63 with dry weight, is commonly not used for the development of allometric equations as it is labour-intensive to measure and not necessary because using solely dbh and tree height already results in high R-Square values. In some cases however, the variable crown area substantially improved the fit of the regression equation (Dietz & Kuyah 2011a, Parvaresh et al. 2012) or was used to calculate the crown diameter (Sajdak & Velazquez-Martí 2012). The correlation of 0.45 between tree and crown height arises from the fact that most of the tall trees sampled were standard trees so that crown height was then usually above 1.8 m.

Regression Equation

Most regression equations contain only the variable dbh, while tree height is added as second variable in about one third of the equations (Zianis et al. 2005). Beyond that, wood density is sometimes included (Chave et al. 2004, Nair 2011). The prevalent equation form is

\[ dw_{\text{tree}} = A \times dbh^B \]

where
- \( dw \): dry weight [kg]
- \( A, B \): estimated parameters
- \( dbh \): diameter at breast height [cm]

(Ter-Mikaelian & Korzukhin 1997, Zianis et al. 2005). This is the form used for the current study, extended by the variables height and crown radius. A reason to include several descriptive variables for the equation of streuobst fruit tree biomass is that pruning causes changes in tree biomass without substantially influencing certain variables such as dbh. The biomass determination should therefore rely on several factors which describe the total tree extent better than just one variable.

Several authors have pointed out the need to include a correction factor, as data is usually first log-transformed and after regression converted back to arithmetic units which introduces a systematic bias (Chave et al. 2004, Levia Jr. 2008, Ter-Mikaelian & Korzukhin 1997). The calculated correction factor 1.0109 is similar to the ones used by Levia Jr. (2008) for eastern white pine which are in the range of 1.026 and 1.071.

The R-Square reported in literature for the estimation of biomass by empirical models varies from 0.012 to 0.99 (Zianis et al. 2005). Low values occurred for example for equations estimating the branch biomass of Norway spruce (Picea abies (L.) H. Karst.) (Zianis et al. 2005), high values for the tree biomass of black birch (Betula lenta L.) (Ter-Mikaelian & Korzukhin 1997). The R-Square for the estimation of tree biomass is usually in the upper range, between 0.8 and 0.99 (Brown 2001, Sterba 2006, Suchomel et al. 2012 in press, Ter-Mikaelian & Korzukhin 1997).
The R-Square depends therefore on the biomass component modeled and the tree species (TER-MIKAEKIAN & KORZUKHIN 1997) as well as on site characteristics, climatic conditions and management practices such as fertilizing and pruning (KEITH et al. 2000, KUYAH et al. 2012a, PINKARD et al. 2004). Furthermore, the silvicultural system has an influence through its determination of the competition between plant species (SUCHOMEL et al. 2012 in press). Different management practices, especially pruning, might have had the largest impact on the comparatively low R-Square of 0.82 in this study because trees with similar size characteristics can have differing weights when pruned. Also, the different growth habits of the cultivars and the differing tree densities might explain some of the variability of the streuobst tree biomass. It could be observed that the R-Square increased slightly from 0.8148 to 0.8156 with inclusion of the correction factor.

**Procedure of Destructive Sample Inventory**

The procedure of the destructive sample inventory was in principal the same as described in HÖCHBICHLER et al. (2006), KUYAH et al. (2012a) and DIETZ & KUYAH (2011). However, measurements were taken before the trees were felled and not afterwards as in HÖCHBICHLER et al. (2006). Only few trees with small dbh or tree height were included so that the range of sizes spanned by the trees was also not completely covered due to practical reasons. But this also arose from the facts that it was assumed that streuobst trees are usually standard trees which implies a log height of at least 1.8 m and that they are relatively old because they are seldom replaced, so that small trees rarely occur.

All three procedure descriptions include the determination of wood density. Although wood density has been reported to be a supporting parameter, it is not vital for the development of a suitable regression equation (CHAVE et al. 2004) and therefore left out in a number of studies (LEVIA JR. 2008, SAJDÁK & VELAZQUEZ-MARTI 2012, STERBA & NACHTMANN 2009). Furthermore, trees have to be damaged by drilling for its determination, which was not possible in the current study because streuobst trees were privately owned.

DIETZ & KUYAH (2011) also include the quantification of the belowground biomass of a tree, the root system. This is only seldom assessed because the process is very cost, effort and time intensive as the full portion of the root system needs to be unearthed, measured off and weighed. Studies including the assessment reported the belowground biomass of forests to average 21 % (range 13 to 26%) of aboveground living biomass (HOUGHTON et al. 2001). A similar percentage of 26 % (range 20 to 30 %) is reported by CAIRNS et al. (1997) who also pointed out that 84 % of the variation can be explained by aboveground biomass density, age and latitudinal category. KUYAH et al. (2012b) added the tree species, silvicultural practices, site differences and vegetation type to the list of factors affecting the root:shoot ratio. They found the root:shoot ratio to predict belowground biomass in agricultural landscapes in Western Kenya to be 0.28. The relatively high ratio is attributed to management practices, especially pruning and coppicing and might therefore be a
good estimate for streuobst trees (see also “Effect of land use on soil organic carbon storage”).

Process Evaluation

The method of the destructive sample inventory to estimate tree biomass is, though extremely time and labour intensive, relatively accurate and seems to be the most appropriate way for tree biomass determination in a forestry system (NAIR 2011a) such as streuobst orchards. As has been pointed out by several authors, a representative sample, especially including trees with large dbhs, is essential for a precise equation (DIETZ & KUYAH 2011). This requirement has been fulfilled in the current study by using 18 trees with a diameter of up to 56.3 cm and a height of up to 9.0 m. The drying time for the stem discs exceeded by large the drying time of 24 hours for the branch samples. KUYAH et al. (2012a) for example extended the drying time of 24 hours only by another 12 hours for wood samples, which seems rather short compared to the additional 65 hours that were required for stem wood samples from this study. No further weight changes were indeed reported in KUYAH et al.’s (2012a) study, but a sufficient drying time until constant weight is reached should in any case be ensured.

The derivation of part-specific values for the dry matter content increased the accuracy of the conversion from fresh to dry weight.

Although the correction factor to correct the bias introduced by log-transforming values was included, the variable wood density was not. Opinions about its importance differ, but it seems to have less influence on the precision of the equation than other factors. In view of the utilisation of the equation for the eventual estimation of the carbon quantity of streuobst trees, it might have been more important to determine the carbon content in biomass. Although a general default value provided by the IPCC was used, some studies assume different contents depending on ecosystem, tree species and tree size. DIETZ & KUYAH (2011) gave a value of 48 % for different tree species in Kenya, LAMLOM & SAVIDGE (2003) reported the carbon content in wood from hardwood species to range from 46-50 % and from softwood species to be between 47 and 55 % in North America. Thus, the 50 % used for fruit trees probably overestimates the carbon contained in streuobst trees by about 2 %.

Streuobst Carbon Storage

Tree Carbon

The largest trees and therefore the largest carbon amount per tree occurred on the streuobst orchard with the lowest tree density and vice versa. This is not surprising as the level of competition among trees is lower at low tree densities (BALANDIER &
DUPRAZ 1998, DROPPELMANN & BERLINER 2000), which also explains the strong negative correlation between mean tree carbon and total number of trees. The tree carbon per area is with an average of 21.7 Mg C ha\(^{-1}\) much lower than in a forest, which ranges from 32 Mg C ha\(^{-1}\) in Europe (DIXON et al. 1994) to 271 Mg C ha\(^{-1}\) in Brazil’s Amazonian forests (HOUGHTON et al. 2001). The amount of carbon stored in the aboveground biomass of an agroforestry system depends largely on the system applied and the latitude. DIXON (1995) reported values of 12 Mg C ha\(^{-1}\) for agrisilvicultural systems in the humid tropical region, e. g. China, and up to 228 Mg C ha\(^{-1}\) for systems in humid tropical low ecoregions such as India, including belowground tree biomass.

A similar tree density as in the streuobst system occurs in systems where scattered trees are planted in corrals or in silvicultural systems together with annual crops. With a tree density of 35 trees ha\(^{-1}\), similar to that of plot K2, KÜRSTEN & BURSCHEL (1993) reported a carbon storage of 25 Mg C ha\(^{-1}\) which is comparable to the 35.4 Mg C ha\(^{-1}\) found in this study. With a higher density of 113 trees ha\(^{-1}\) as on plot B1, PALMA et al. (2007) found a carbon storage between 32 and 54 Mg C ha\(^{-1}\), exceeding the 25.7 Mg C ha\(^{-1}\) from plot B1. This can be explained by differences in the management since the trees from PALMA et al.’s study were not pruned and solely grown for timber production.

The positive correlation between the standard deviation of tree carbon and the mean tree carbon is visualised in the box-plot: Place K2 has the highest mean tree carbon and at the same time the greatest deviation. This points to the fact that the larger the tree and therefore the mean carbon content, the larger is also the variability. It seems therefore advisable to include several descriptive variables in the allometric equation for a precise quantification of the carbon amount stored in large trees, especially against the background of the relatively great share of carbon stored in the larger trees.

The negative correlation between tree density and tree carbon m\(^{-2}\) along with the positive correlation between mean tree carbon and tree carbon m\(^{-2}\) suggests that the size of a tree is more important for the quantity of carbon storage on a plot than the number of trees and that in terms of carbon storage, the planting of widely spaced trees, promoting the individual tree growth, is preferable to small planting distances. Besides the diverse ecological functions that old trees fulfill such as providing habitats for animals and insects, the high carbon storage would be another argument for keeping old trees even though productivity and/ or fruit quality have decreased.
Soil Analysis

pH and bulk density

pH

The pH-values classify the soils as being moderate to slight acid (BLUME et al. 2010). They are high for the soil type pseudogley, which usually has a more acid pH of about 5 (AMT FÜR UMWELTSCHUTZ & ENERGIE 2012, UNI BONN 2011). As no calcium carbonate could be found with the Scheibler apparatus, other reasons for the high pH-values must exist.

One might be that pastures have in general a high amount of soil organic matter (GUO & GIFFORD 2002, MERINO et al. 2004, UMWELTUNIBUNDESAMT 2004). Soil organic matter functions as ion-exchange compound and is thereby one of the most important buffer systems in the temperate regions (JONES et al. 2005, SCHROEDER 1992). By electrostatically binding H⁺-ions in exchange for other positively charged ions such as K⁺, ion-exchange systems remove H⁺-ions from the soil solution and thereby buffer against the decrease of the pH (STAHR et al. 2008). The higher the base saturation of the exchange systems, the higher the buffer capacity (BLUME et al. 2010). A high base saturation also indicates that the soil is relatively young and little weathered (BLUME et al. 2010). BERGER et al. (2010) indeed found the Flysch bedrock from the study region as being base- and nutrient-rich.

Every buffer system has a main activity range; the one of the ion-exchange system is in the acidic range. Still, all buffer systems operate over the whole pH range, and the systems that are not dissipated such as the exchange system are active over a broader pH range. The pH of around 6.5 (H₂O) points to the activity of the silicate buffer system. Here, metallic cations are detached from the silicate lattice and exchanged for H⁺-ions (STAHR et al. 2008).

Since study sites are located in the Flysch zone which consists mainly of sandstones and clayey marls (BERGER et al. 2010), a large percentage of parent material is composed of silicates (BLUME et al. 2010), so that the silicate buffer system can be assumed to be the predominant system.

The high pH of the field is probably due to the application of lime a few days before sampling.

The higher pH of the streuobst orchard compared to the pasture might be caused by the enhanced bioturbation by the soil fauna which can impede the dislocation and/ or eluviation of nutrients (BLUME et al. 2010). Moreover, the crop load that is carted off might be greater on the pastures than on the orchards due to harvest impediments through the trees. Nutrients are thus removed, which are mostly cations in humid climates (STAHR et al. 2008). Another cause could be that the trees, reaching deeper soil layers than grasses, pump up nutrients and thereby increase the pH.

With the highest value being 7.28 (H₂O), the pH is still within the range of 5-7.5 which is generally seen as the most favourable for plant production because the nutrient
availability is highest and at the same time, negative effects such as aluminium-toxicity are lowest (SCHROEDER 1992).

The increase of the pH with increasing soil depth is commonly observed in most soils that are not limed because the production of H\(^+\)-ions and the loss of cations with alkaline effects is greater in the topsoil than in the subsoil and because leaching progresses from the topsoil to the subsoil. It is assumed that the steeper the gradient, the more alkaline is the original parent material (SCHROEDER 1992). But since the pairs of streuobst orchard and pasture both had the same parent material, this cannot explain the steeper gradient in the streuobst orchards. Therefore, the above described possible explanations for a higher pH at the orchards should have a greater impact in the topsoil than in the subsoil. This is indeed the case for the activity of soil organisms (BLUME et al. 2010). The nutrient extraction via grass harvest has also a greater impact on the upper soil layer since grass plants root very shallow, so that they extract nutrients primarily from the top layer (JACKSON et al. 1996).

The difference in the pH-values between the two measurement methods H\(_2\)O and CaCl\(_2\) is explained by the fact that Ca\(^{2+}\)-ions replace H\(^+\)-ions on the exchange system, which leads to an increase of H\(^+\)-ions in the solution and subsequently to a lower pH of the CaCl\(_2\)-method (SCHROEDER 1992). The CaCl\(_2\)-method therefore measures the exchange acidity while the H\(_2\)-method measures the active acidity (BLUME et al. 2010, HLFS 2012). The values of the exchange acidity are usually 0.3-1.0 units lower than the active acidity (SCHROEDER 1992), which was confirmed in this study. A larger difference might point out that many nutrient ions have already been replaced by H\(^+\)-ions, which would in the following lead to a decrease in nutrient availability and a reduced buffering capacity. The mean difference between the measurement methods was 0.71 for the land use pasture and 0.66 for the land use streuobst, suggesting a slightly lower nutrient availability of the pasture.

**Bulk density**

The range of the bulk density values is in accordance with the values commonly observed for grassland soils in Austria (GERZABEK et al. 2005). Since the top layer is the soil depth influenced the most by land management, the difference in bulk density between the land uses is most pronounced in this soil layer (EMADI et al. 2008, VANDENBYGAART et al. 1999). Land under cultivation has, at least in the long term, been observed to have a higher bulk density than other land uses such as forest or grassland, mainly due to compaction from heavy machinery, the breaking up of pores and disturbance of soil life (CELIK 2005, VANDENBYGAART et al. 1999). The fact that the streuobst orchards had lower bulk density values than the pastures in the two soil layers 0-10 and 20-30 cm might be due to a better microclimate provided by the trees which dampen temperature extremes and reduce winds, so that microbial activity is higher (see “Effect of land use on soil organic
carbon storage”). This might facilitate aeration and water entry by forming biopores and decrease the bulk density (DAVIS 1994, VANDENBYGAART et al. 1999) (see “Effect of pH and bulk density on soil organic carbon storage”).

Differences in pH and bulk density between places

The differences in pH and bulk density between the places are likely due to a variety of reasons. Place R1, which had the lowest mean pH, also had very low values for bulk density, the pH of the pasture increased with soil depth and the soil organic carbon values of the pasture were exceptionally high. These abnormalities cannot be explained and place R1 was therefore left out in some of the statistical analyses. Place B1 in contrast showed the highest pH values, especially in the streuobst orchard. A reason here might be the soil type which tends towards a loose sediment brown earth. Brown earths can have high pH values if they are base-rich (KOPPE 2012). The comparatively low difference between the active and the exchange acidity of 0.56 for the orchard points to a high base saturation. The low bulk density might be caused by the long history of the plots being a streuobst orchard respectively a pasture and the extensive management without heavy machinery and with a low number of passes. In addition, orchard and pasture are not grazed as are some plots on other places, so that no compaction through grazing can have occurred.

A striking feature of plot K2 is the reversed pH-gradient of the pasture for which no explanation could be found. In a forest, the root system reaches in deeper soil layers and as roots release H+ ions, this could lead to an acidification in these layers (SCHROEDER 1992). But even though the pasture is located right next to a forest, this effect will not extend over the whole width of the pasture.

On plot M1, the bulk density gradient of the pasture is also reversed, decreasing with increasing soil depth. As this plot is used as a pasture for cows, the grazing might have caused the compaction of the topsoil (CELIK 2005). The rather low pH of the orchard and the pasture can be explained by the application of liquid manure which commonly leads to a soil acidification as ammonium is converted into nitrate (STAHR et al. 2008).

Soil carbon

First statistical approach

The soil organic carbon values of 0.249-7.420 % concur well with the results of GERZABEK et al. (2005) who investigated the soil organic carbon storage in Austria’s agricultural soils. For intensive grasslands in Upper Austria, they found a median value of 3.52 % at the soil depth 0-10 cm, which was slightly higher than the median value for streuobst orchards of 3.27 % and higher than the median for pasture of 2.79 % from this study. The median at soil depth 20-40 cm in GERZABEK et al.’s (2005) study of 0.99 % was lower than the values at depth 20-30 cm for streuobst of
1.85 % and for pasture of 1.65 %, probably because sampling depth was 20-40 cm instead of 20-30 cm. LEIFELD & KÖGEL-KNABNER (2005), analysing soils in south east Germany, found values of 1.39 % for arable land and of 2.29 % for grassland at soil depth 0-30 cm, which is somewhat lower than the 1.43 and 2.80 % from this study. The higher deviation of the streuobst and pasture carbon values compared to the field values was also observed by GERZABEK et al. (2005). The reason might be the cultivation of fields, which leads to a more even distribution of nutrients and organic substrates (BLUME et al. 2010). In the current study, the number of soil samples of the field was moreover much smaller than the one of the pasture and streuobst. JOBBÁGY & JACKSON (2000) noted a coefficient of variation of 0.64 for the soil organic carbon of the first soil metre globally which is, presumably due to the greater soil depth and the higher amount of soil samples included and the larger area sampled, larger than coefficients from this study where the highest values were 0.54 for streuobst and 0.53 for pasture (without place R1) at depth layer 50-60 cm.

The analysis of variance identified the variables place, land use and pH as having a significant effect on the soil carbon content at depth layer 0-10 cm; at depth layer 20-30 cm, bulk density was also significant. The additional calculation of the Kruskal-Wallis test showed that the land use field differed significantly from streuobst and pasture over all depths, although no difference could be found between streuobst and pasture. These results are explained in the following section.

Effect of land use on soil organic carbon storage
It has been observed in a wide range of studies that land use has a significant effect on the soil carbon content. Litter biomass, the quantity as well as the quality, is one of the major explanations for differences in the soil organic carbon content between land uses. This encompasses both root and leaf litter. The large share of litter that is removed from cropland through harvest is the main reason why cropland has a much lower soil carbon content than other land uses (GUO & GIFFORD 2002, HAN et al. 2010, TUMWEBAZE et al. 2012), as could also be observed in this study.

In northern temperate grasslands, fine roots account for 92 % of the total litter production (STEINAKER & WILSON 2005), so that aboveground litter biomass is rather small. According to the hypothesis of a higher carbon storage on streuobst orchards than on pastures, trees planted onto grasslands will enhance this aboveground production and might thereby influence soil organic carbon (NAIR 2011b, TUMWEBAZE et al. 2012). This impact depends on tree density and shape. With increasing tree density, light competition between grasses and trees increases and thus grass growth decreases (ROZADOS-LORENZO et al. 2007). The tree density at which pasture production starts to decrease depends on tree shape which is mainly determined by species and age. ROZADOS-LORENZO et al. (2007) found this threshold to be 952 trees ha⁻¹ for the species Douglas fir (Pseudotsuga menziesii (Mirb.) Franco) and Maritime Pine (Pinus pinaster Aiton) and 427 trees ha⁻¹ for Monterey Pine (Pinus radiata Aiton), all aged four years. SHARROW & ISMAIL (2004)
examined pasture production under 14 year old Douglas fir and found no difference between open pasture and silvopasture at a tree density of 570 trees ha\(^{-1}\). Tree densities in streuobst systems are around 20-100 trees ha\(^{-1}\) (HERZOG 1998). Even though the densities from this study are already on the upper range with a maximum of 260 trees ha\(^{-1}\), they are still well below the densities where light competition occurs, so that the trees on the streuobst orchard can be assumed to add aboveground biomass through leave and branch litter without decreasing grass biomass to a large extent. As this seems not to be true for the current study, a possible explanation could be that tree density was too low to substantially increase aboveground litter production. All streuobst orchards were also, though extensively, well managed; no branches or twigs could be found on the ground. Fruits are harvested or gathered up, so that the tree litter addition might be limited to leaves. The insignificance of the increased aboveground litter production supports STEINAKER & WILSON’s (2005) findings that fine roots account for 80 % of total litter production in forests and for 92 % in grasslands, so that the litter quantity provided by the trees is too small to have an impact.

Tree age is not only important in terms of light competition but also in terms of quantity of carbon inputs (CORRE et al. 1999, GUO & GIFFORD 2002, JACKSON et al. 2002) and the microclimatic regime (LAGANIÈRE et al. 2010). This is why afforested areas experience a soil carbon loss in the first few years, which is then followed by a gradual return to the level of the control plots and finally leads to a net carbon gain (LAGANIÈRE et al. 2010, PAUL et al. 2002). The combination of tree age and tree density might explain the low streuobst carbon content in spite of the trees: Although some streuobst orchards contain a certain number of old trees, these are also the ones that have only a low tree density. Place K2 for example had the oldest trees, but the lowest tree density, in contrast to place M1 where the highest tree density could be found, but where trees were also the youngest with an age of 19 years, so that this orchards might not have reached the phase of net carbon gain yet.

Besides the quantity of litter, its quality is also of importance. Studies differ in the definition of aboveground “litter quality” in the respect of carbon storage. POST & MANN (1990) term litter with high nitrogen and low lignin contents, i.e. a low C:N ratio, as best for the accumulation of carbon and nitrogen in and on soils. This hypothesis is supported by GUO et al. (2007) and YAMASHITA et al. (2006), who pointed out that pine respectively Norway spruce (Picea abies L.) litter is of low bioavailability and has therefore a low quality compared to grass or crop litter. HOWLETT et al. (2011) named the low C:N ratio and low lignin content as the reasons for the higher accumulation of soil organic carbon under a birch pasture as compared to a pine pasture.

In contrast, GUO & GIFFORD (2002) give the relatively low decomposability of forest litter as explanation for the higher amounts of soil organic matter in forest surface soils in comparison to grassland soils, where the higher decay rates due to the low C:N ratio are supposed to enhance the carbon output. MELILLO et al. (1982) found the
ratio of initial lignin concentration to initial nitrogen concentration to be positively correlated with the loss of dry mass for hardwood leaf litter as well. It can be assumed that streuobst trees add litter with a low C:N ratio since branches are usually removed from the orchards, so that leaves and, if they are not harvested or picked up, fruits make up the largest part of aboveground litter (Mungai & Motavalli 2006). The opinion first described, that a low ratio is more promotive for carbon storage, supports the hypothesis that streuobst carbon storage is larger than pasture carbon storage. As this is not the case, the second opinion might explain results from this study. The low C:N ratio might not only promote the rapid decomposition of tree litter, but also attract a larger amount of decomposing microorganisms and soil animals so that grass litter decomposition is also enhanced.

Besides aboveground litter, **belowground litter** plays a role, especially in temperate grasslands which have a very high root:shoot ratio of 3.7 (Jackson et al. 1996). Although root length and root mass is greater in forests (Nair 2011b, Steinaker & Wilson 2005) by a factor of 3 (Jackson et al. 1996), grasslands have a greater root production and turnover (Guo et al. 2007, Steinaker & Wilson 2005). Guo et al. (2007) estimated the root production of red grass (Themeda triandra Forsk.) in Australia to be 3.6 Mg C ha\(^{-1}\) yr\(^{-1}\), compared to 2.7 Mg C ha\(^{-1}\) yr\(^{-1}\) for Monterey Pine. Episodic cutting or grazing is suggested to increase root production as it is followed by a rapid death of grass roots and subsequent root regrowth (Guo & Gifford 2002). The great root production and the amount of dead roots thereby incurred might function as continuous delivery of belowground litter and thus explain the comparatively high carbon content detected in this study and in general on the land use pasture (Guo & Gifford 2002).

The **root system** of temperate grasslands is very shallow, with about 83 % of root biomass in the upper 30 cm of the soil (Jackson et al. 1996). Yakimenko (1998 in Guo & Gifford 2002) suggested that the dense root net of grasses might reduce water and gas exchange and thus decrease the rate of decomposition in meadow ecosystems. The shallow root system also allows trees from the temperate zone which have a deeper rooting system (Jackson et al. 1996) to grow on the same plot without substantially decreasing soil space for grass roots. The niche differentiation by rooting depths might lead to an increased belowground litter input and subsequent storage as soil organic carbon (Häger 2012 in press). Moreover, species complementarity can facilitate processes such as nutrient cycling (Häger 2012 in press), thus increasing nutrient availability and consequently plant growth for all system components (Davis 1994, Dixon et al. 1994).

The deeper and more extensive root system is also the major reason why in agroforestry systems, woody species are capable of a better spatial capture and utilisation of growth resources than annual species such as many grasses (Nair 2011b, Tumwebaze et al. 2012). This leads to a higher biomass production per unit of area and subsequently to a higher soil organic carbon content (Haile et al. 2008), supporting the main hypothesis.
Most fruit trees form both taproots and lateral roots (ZEHNDER & WELLER 2011). The apple tree for example builds more shallow roots with 60 % of roots occurring in a soil depth of 5-60 cm (FISCHER et al. 2007) while pear is a deep-rooting plant (ZEHNDER & WELLER 2011). In view of the soil type being a pseudogley, the rooting system of the fruit trees might be more shallow than it would be under better soil conditions. Thus in this case root competition might exist between trees and grasses which does not exploit benefits through species complementarity. Therefore, total root production is not sufficient to increase soil organic carbon storage.

The amount of tree root biomass depends, amongst others, on tree species. Broadleaves are commonly observed to produce more root biomass, especially fine-root biomass, than conifers (GUO et al. 2007, GUO & GIFFORD 2002). TUMWEBAZE et al. (2012) suggest that the effect of tree species on soil carbon is due to the species’ varying abilities to capture soil nutrients, to the amount and quality of litter fall and to root distribution. PAUL et al. (2002) and LAGANIÈRE et al. (2010) add that the rate of carbon transfer from litter to soil, the humification rate, might also account for the variation.

Another reason for the difference in soil carbon storage of the land uses streuobst and pasture might be the effect the fruit trees have on the microclimate. Firstly, the trees might decrease water consumption of the grass through shading effects on soil evaporation and rainfall redistribution (ONG et al. 2002, ONG & LEAKEY 1999, TUMWEBAZE et al. 2012). Since the fruit trees also consume water, the net water balance effect depends on the tree density (ONG & LEAKEY 1999). ONG & LEAKEY (1999) found this effect to be negative in alley cropping with fast growing trees and positive in parkland agroforestry with widely spaced trees. As trees in streuobst systems are usually scattered, the effect here might be a positive one. Secondly, the tree canopy also reduces temperature extremes and impacts of heavy rain and winds and might thereby increase grass production and microbial activity (LAGANIÈRE et al. 2010, TUMWEBAZE et al. 2012).

The provision of shelter from heavy rains and winds also decreases soil erosion (ONG et al. 2002). However, the effect might be negligible in this study because pastures, through their year-round vegetation cover and dense root net, under given climatic conditions, do not have a high potential for soil erosion (NRPH 2007). As mentioned above, agroforestry systems can enhance nutrient cycling by species complementarity. Again, this depends on tree density and evidence exists that the effect is a positive one for systems similar to streuobst orchards (ONG & LEAKEY 1999, TUMWEBAZE et al. 2012).

On the one hand, these reasons support the hypothesis that streuobst systems have a larger soil organic carbon stock than pastures because primary production is increased and organic matter decomposition is enhanced, leading to a greater soil carbon storage. On the other hand, they might also lead to an increase in microbial activity in such a way that the decomposition of organic matter causes a larger carbon loss through the respiration of soil organisms than is gained through carbon fixation in the soil. Further factors promoting a high activity of soil organisms in
streuobst systems are the lower bulk density and the higher pH, a larger amount of roots which creates a greater habitat for microorganisms in the rhizosphere and an additional nutrient input through tree leaves.

Besides the reduced litter input, another reason for the lower soil organic carbon content of the field compared to pasture and streuobst orchard are soil disturbances. The yearly cultivation for crops accelerates organic matter decomposition and oxidises carbon (PAUL et al. 2002, RICHTER et al. 2000). Soil organisms which promote the formation of stable aggregates are disturbed (see “Effect of pH and bulk density on soil organic carbon storage”) and soil structure is destroyed, so that carbon stabilisation mechanisms are reduced (DIXON et al. 1994, LAGANIÈRE et al. 2010). Moreover, the periodically lacking vegetation cover increases climate impacts on the soil such as variation in temperature (BROWN & LUGO 1990).

Summing up, there are three possible explanations for the slightly higher soil carbon storage in pastures than in streuobst orchards.

The first explanation is that the sampling size of six streuobst-pasture pairs was too small. Additionally, one of the six sites, R1, had peculiar values not only for soil carbon so that the soil organic carbon values from R1 might have distorted the results. The fact that median values clearly showed a higher soil carbon storage in streuobst than in pasture when place R1 was omitted bear out the bias of R1. The strong reduction in the median of the pasture carbon storage also points to the fact that the pasture’s arithmetic mean was largely influenced by few very high values.

The second explanation is that the described processes supporting the main hypothesis are not strong enough to cause the expected results or that other, unknown processes affect soil carbon storage to a larger extent in general or only at the study sites.

Thirdly, the factors such as enhanced litter production, better microclimate and lower bulk density provide a very promotive habitat for soil organisms so that decomposition of organic matter is very fast and leads to carbon losses either through respiration of soil organisms to the atmosphere or through translocation of organic compounds to deeper soil layers.

For a full comparison of land uses, the inclusion of dead aboveground biomass and grass biomass could have been considered. Houghton et al. (2001) reported that dead aboveground biomass averaged 9 % of aboveground live biomass in a number of studies concerning Brazilian forests. Besides the obvious difference in ecosystem and climatic region, most of the surveyed streuobst orchards are, though extensively, well managed, and sampling time was in spring, so that almost no twigs, branches or leaves were visible on the ground of the orchards. Therefore dead aboveground biomass might be much smaller, but it should still be considered to be included in the determination of aboveground biomass. Grass biomass was assumed to be equal on streuobst orchards and pastures, and visual assessment confirmed this hypothesis. SHARROW & ISMAEL (2004) also pointed out that carbon storage on pastures is almost
entirely belowground due to the extensive root system of grass plants. However, an exact quantification might reveal differences in biomass between land uses.

**Effect of pH and bulk density on soil organic carbon storage**

Soil organic carbon was also significantly affected by the pH. Although other studies came to the same conclusion, the relationship between pH-values and soil carbon content was usually a weak one (Bell & Worrall 2009) or not even significant (Laganière et al. 2010). However, the pH is supposed to control microbial activity, thereby determining on the one hand organic matter mineralisation (Bell & Worrall 2009), on the other hand the bioturbation and eventually the formation of stable aggregates (Laganière et al. 2010). This positive relationship between soil organic carbon content and pH is amplified by the fact that a low pH restrains tree growth which in turn reduces carbon inputs to the soil (Paustian et al. 1997). This is especially true for fruit trees which prefer a higher pH; mazzard and sour cherry trees (Prunus cerasus L.) for example need a pH between 6 and 7 while plum trees prefer values around 7 (Fischer et al. 2007, Zehnder & Weller 2011). Although Paustian et al. (1997) point out that low pH-values might also reduce decomposition rates of soil organic matter and thus lead to soil carbon accumulation, Rousk et al. (2009) found this not to be true because the major decomposer groups, bacteria and fungi, are functionally redundant.

Bulk density was only a significant factor at depth layer 20-30 cm and was negatively correlated with soil organic carbon. Tumwebaze et al. (2012) and Emadi et al. (2008) also observed the decreasing bulk density with increasing organic carbon. This can be ascribed to the promotion of aggregate formation by organic matter in an indirect way by increasing microbial activity. The organic compounds produced as microbial metabolites and decomposition products have the ability to adhere inorganic particles. Small soil animals, particularly annelids (Annelida Lamarck) which feed on organic matter excrete more resistant aggregates due to the combination of organic and inorganic compounds and their mechanical mixing in the intestinal tract. As a third group, fungal hyphae, actinomycetes, bacteria and hair roots have an aggregating effect. This process of ecological engineering leads to the formation of the crumb structure which characterises the topsoil of grasslands and the humus type mull. The high pH, the large amount of nutrients in the soil and the easily decomposable litter are further characteristics of mull and are all present at the study sites (Blume et al. 2010). The relationship between bulk density and soil organic carbon might therefore be the other way around than suggested by results: It is not a low bulk density that increases organic carbon, but a high carbon content that decreases bulk density.

**Effect of place on soil organic carbon storage**

The factor place was significant over all soil layers. As already described, pH and bulk density can significantly influence soil organic carbon in different ways. Since previous analyses showed differences between pH and bulk density between places, their variation could be an explanation for the effect of place on soil carbon.
Another factor potentially important that has not been investigated in this study is soil texture. CELIK (2005) for example attributes the control of the formation and of turnover rates of soil organic matter pools to soil texture. JOBBAGY & JACKSON (2000) found that soil organic carbon in the first soil meter was significantly correlated with climate and soil texture globally. Although PAUL et al. (2002) confirmed in their study that soil texture influences carbon dynamics, the relation was not significant and differed with soil depth: While increasing clay content decreased soil organic carbon at depth layer 0-10 cm, it increased soil carbon at layer 10-30 cm. Similarly, MÜLLER & HÖPER (2004) detected a decrease in specific microbial turnover and an increase of soil microbial biomass with increasing clay content, but at the same time deduced that no or only weak relationships between clay content and soil humus carbon can be expected in practical agriculture and forestry. According to LAGANIÈRE et al. (2010), clay soils have the theoretical potential for greater soil organic carbon accumulation than coarse-textured soils because fine particles contribute to the formation of stable organo-mineral complexes by associating with organic compounds. This potential however has rarely been shown in studies (BROWN & LUGO 1990, KRISHNAN et al. 2007, LAGANIÈRE et al. 2010, MÜLLER & HÖPER 2004). LAGANIÈRE et al. (2010) hypothesise that the soil clay content might interact and be confounded with other factors influencing carbon storage, e.g. primary productivity and initial organic carbon content. Soil texture seems therefore not to be a sufficient explanation for the differences between places. Management differences could also play a role, but as all pastures and orchards receive an annual application of manure except for place R1 (which is grazed), this does not apply here. Moreover, GUO & GIFFORD (2002) reported that pasture fertilisation had no effect on soil organic carbon. Slope aspect might influence soil organic carbon as well. HAN et al. (2010) and LIAN et al. (2006) found a tendency to higher organic carbon contents in soils on northern slopes, which was reflected in this study: Places K1 and K2 which have higher soil carbon contents are situated on slopes facing north while places M2 and M1 have lower carbon contents, their slopes facing west.

**Effect of sampling location on soil organic carbon storage**
Statistical tests have shown that the difference between sampling locations at soil depth 0-10 cm is only just, respectively non-significant depending on the test. Similar results were reported by HAILE et al. (2008) who investigated soil carbon in a silvopasture based on slash pine (*Pinus elliottii* Engelm.) and bahiagrass (*Paspalum notatum* Flüggé) in Florida. Although they not only found differences in the top layer but also in deeper layers going down to 100 cm, soil organic carbon content in-between tree rows was higher in 8 cases while it was higher in 6 cases in the center of an alley, so that no clear trend of sampling location was visible. TUMWEBAZE et al. (2012) detected no change in soil organic carbon with distance from various tree species in a linear simultaneous agroforestry system in Uganda at all, just as
Guo et al. (2007) did not find differences between soil cores taken in the tree row and between rows on a pine plantation in Australia. The tendency to more soil carbon in the row might be due to a larger amount of litterfall closer to the tree. Protection from wind, temperature variations and heavy rain as well as shade effects might also be greater closer to the tree. Finally, the streuobst orchards that are grazed (places M2 and R1) could have higher soil carbon contents around the trees because these areas are preferred as resting places by grazing animals so that the amount of animal excrements is larger at these spots.

Second statistical approach

Effect of land use on soil organic carbon storage

With the second statistical approach, only few significant differences between land uses could be detected. At place R1, soil organic carbon values at layers 0-10 and 20-30 cm were larger for the land use pasture than for streuobst. These results should be regarded with reserve, because also other measurements at place R1 differed from the rest. Especially values from the pasture seemed to be skewed, as is demonstrated by the decrease of pH and bulk density with depth in contrast to the commonly observed increase. Contrary to the other plots, the pasture was on a relatively steep slope, so this might have affected the results. The soil also seemed to be different because it was very dry and stony in places. The otherness was also reflected in the mean and median values of soil organic carbon. Omitting values from R1 reduced the standard deviation and the coefficient of variation. Especially median values reacted, with a decrease of about 0.4 kg C m\(^{-2}\) for pasture at depth layer 0-10 cm and 0.06 kg C m\(^{-2}\) at depth layer 20-30 cm which amounts to about 15 % respectively 3 %.

The difference at place M1 might be due to different fertilisation regimes: While the streuobst orchard only receives one annual application of liquid manure, the pasture is fertilised with manure and lime nitrogen annually and is used as pasture for cows, so that the pasture receives a greater amount of fertiliser in total.

At place K2, streuobst was the land use having a significantly higher soil organic carbon content than the pasture. The management and the slope angle are the same at the orchard and the pasture. A conspicuous feature at K2 was the decreasing pH with increasing soil depth of the pasture which might have negatively influenced the soil organic carbon content. The pH at depth 0-10 cm was also higher than that of the streuobst which is atypical compared with the other places. The pH was found to be positively correlated with soil carbon, so either the carbon content should have been higher at the upper soil layer or the pH should have been lower to explain the low carbon content in spite of the high pH. A comparison of pH-values shows that the pasture at M2 has about the same pH at depth layer 50-60 cm as K2. Although soil carbon values are lower at M2 at this soil depth, they are about the same for layers 0-10 and 20-30 cm. The pH-values at the upper soil layers are much lower at M2 than at K2 and show the usual increase with soil depth. It might therefore be possible that the pH at the upper soil layers at place K2 is subject to measurement errors. If
that is the case, the low pH at the upper soil layers could explain the significant differences between the soil organic carbon content between the streuobst orchard and the pasture. Another explanation that cannot be completely be excluded is that soil samples were interchanged.

**Effect of sampling location on soil organic carbon storage**
The only significant differences were detected at place M1. This might be caused by the tree arrangement. Clear tree rows exist at this plot while at the others, trees are more widely scattered or tree rows are incomplete because shattered trees were not replaced. Due to the young tree age, crown radiiuses were the smallest, so that the predominant share of litter will accrue close to the stem and might not reach the center of the tree row.

**Soil organic carbon storage with soil depth**
The decrease of soil organic carbon with soil sampling depth is common knowledge in the field of soil carbon research (Corre et al. 1999, Gerzabek et al. 2005, Guo et al. 2007, Haile et al. 2008, Jobbágy & Jackson 2000). The main reason is the reduction of organic inputs (Brown & Lugo 1990) and a concomitant decrease in the rate of biological and physical activity (Tumwebaze et al. 2012). The deeper distribution of soil organic carbon compared to biomass is explained by a reduced soil organic carbon turnover in greater depths, leaching and vertical mixing by soil organisms (Jobbágy & Jackson 2000).

For the mentioned reasons, the topsoil is more reactive to land use changes than the subsoil, and the differences of soil organic carbon between land uses are more pronounced in the upper than in the lower soil layers (Gerzabek et al. 2005, Guo & Gifford 2002).

As mentioned before, soil organic carbon is sensitive to environmental and climatic factors and changes such as pH or temperature. In contrast to the former view that the persistence of organic matter is due to intrinsic properties of organic matter, recent research has revealed that it is primarily an ecosystem property (Schmidt et al. 2011). A number of stabilisation processes leads to the formation of three organic matter carbon pools with different turnover times. The first pool, the active pool, has a characteristic turnover time of less than 10 years (Von Lützow et al. 2008). It is composed of plant residues, microbial and faunal biomass and products and rhizodeposits (Von Lützow et al. 2006). The selective preservation of recalcitrant compounds is the responsible mechanism leading to a relative accumulation of recalcitrant compounds during the initial decomposition stages. During later phases, spatial inaccessibility to decomposer groups and organo-mineral interactions are dominating processes (Von Lützow et al. 2006). These mechanisms form the intermediate pool with a turnover time of 10-100 years (Von Lützow et al. 2008). The third, passive pool is created through occlusion of organic matter by clay microstructures, the development of hydrophobic properties, the production of charcoal and complexation (Von Lützow et al. 2006) and has a turnover time of
more than 100 years (VON LÜTZOW et al. 2008). The stabilisation of organic matter in the soil therefore depends on a number of processes which are influenced by soil characteristics and environmental conditions. The fraction of biomass that is eventually stored in the soil as soil organic carbon is thus site-dependent (NAIR 2011a) and cannot be predicted solely from the data that was collected in the current study. This should be kept in mind when regarding and assessing the results. While carbon sequestration describes the process of removing carbon from the atmosphere and storing it in long-lived pools (LAL 2008), the subject of this thesis was the momentary carbon storage and should be regarded as this.

**Soil & Tree Carbon**

The soil organic carbon values for a soil depth of 60 cm are well within the range reported by other authors. HOWLETT et al. (2011), investigating soil organic carbon storage to a depth of 50 cm under a birch pasture in Spain, reported a value of 128 Mg C ha\(^{-1}\), SHARROW & ISMAIL (2004) found a value of 102.5 Mg C ha\(^{-1}\) for the first 45 cm of a pasture in Oregon, USA and DIXON et al. (1994) estimated the carbon content of Europe’s forest soils down to a depth of 1 m to be 90 Mg C ha\(^{-1}\). The same sampling depth as in this study (60 cm) was used by ANDRADE et al. (2008) who reported a mean value of 110.3 Mg C ha\(^{-1}\) for a silvopasture in Costa Rica. Values from similar experimental designs differ with soil sampling depth, with larger sampling depths yielding higher soil carbon values of around 130 Mg C ha\(^{-1}\) (SAHA et al. 2009) and smaller sampling depths yielding lower values of around 65 Mg C ha\(^{-1}\) (HÄGER 2012 in press, SOTO-PINTO et al. 2010). These similarities suggest that sampling and carbon analysis in the current study were carried out properly and that results are valid.

The amount of tree carbon per area is, as already discussed, lower than that in forests, but corresponds well with findings from similar agroforestry systems. One of these systems is coffee production under shade trees, where tree carbon storage is estimated to be 18 (± 12) Mg C ha\(^{-1}\) in Costa Rica (HÄGER 2012 in press) which is slightly lower than the average of 22 Mg C ha\(^{-1}\) from this study. SHARROW & ISMAIL (2004) give an even lower value of 12 Mg C ha\(^{-1}\) for the carbon stored in Douglas firs of a silvopastoral system in Florida, while KÜRSTEN & BURSCHEL (1993) estimated the tree carbon accumulation in tropical agroforestry systems to average 22 Mg C ha\(^{-1}\). Since tropical latitudes are known to have a much larger biomass production, the mean value from the current study can be assessed as being rather high for a temperate region (DIXON et al. 1994, SABINE et al. 2004, SCHIMEL 1995b).

The share of tree carbon on total carbon covers with 7 to 21 % a relatively small range. Percentages in literature range from 3 % for a silvopasture in Costa Rica (ANDRADE et al. 2008) over 11 % for a North American silvopasture (SHARROW & ISMAIL 2004) and 21 % for a coffee agroforestry system in Costa Rica (HÄGER 2012 in press) up to 52 % for a pasture with scattered trees in Mexico (SOTO-PINTO et al. 2010). The scattered tree pasture in Mexico is described as densely-forested and its
share is therefore close to values for forests, which are between 36 (DIXON et al. 1994) and 59 % (GOODALE et al. 2002). The tree carbon percentage seems to depend on tree age and agroforestry system, with older trees and systems with higher tree densities reaching a greater share of tree carbon on total carbon. Although tree density is rather low in the sampled streuobst orchards, trees are comparatively old; thus this combination might explain the medium tree carbon percentage. Of importance is furthermore the latitude of the ecosystem: While biomass production is greatest in tropical regions, soil carbon storage is much greater in high latitudes (DIXON et al. 1994, IPCC 2000).

The total carbon is slightly higher than values from similar systems reported in literature. Silvopasture is assumed to be a comparable system, and values here are e.g. 116-121 Mg C ha\(^{-1}\) in Costa Rica (ANDRADE et al. 2008) or 107 Mg C ha\(^{-1}\) in Florida (SHARROW & ISMAIL 2004). SOTO-PINTO et al. (2010) gave a carbon content of 143 Mg C ha\(^{-1}\) for a pasture with scattered trees in Mexico, which is still below the arithmetic mean of the streuobst orchard of 146.7 Mg C ha\(^{-1}\). The mean might be higher because the fruit trees were much older than for example in ANDRADE et al.’s (2008) study which were only 4 years old or in SHARROW & ISMAIL's (2004) which were aged 11.

For the pasture mean of 125 Mg C ha\(^{-1}\), comparable values are 103 Mg C ha\(^{-1}\) for a North American pasture (SHARROW & ISMAIL 2004) and 79 Mg C ha\(^{-1}\) for a pasture without trees in Mexico (SOTO-PINTO et al. 2010). Again reported values are lower which is probably due to a different fertilisation regime. While all pastures except one from the current study are fertilised once a year, the last fertiliser application at the North American pasture was 21 years before measurements were taken (SHARROW & ISMAIL 2004); no information was available for the pasture in Mexico, though one may conclude from the description that management is rather extensive and that fertiliser application is therefore rather low respectively non-existent (SOTO-PINTO et al. 2010).

The fact that the addition of tree carbon to the soil carbon for each place did not change the ranking order of places according to soil carbon suggests that the amounts of soil and tree carbon are connected. This is in accordance with findings from HAILE et al. (2010) who used stable C isotope signatures to trace carbon plant sources in soil organic carbon under silvopasture and an adjacent open pasture. The silvopasture soils contained more C3-derived soil organic carbon and accumulated higher C3-derived soil organic carbon in the silt and clay-sized (< 53 µm) fraction. This indicates that most of the soil organic carbon was derived from tree components (C3 plants) and not from grasses (C4 plants), including the soil carbon contained in the fraction < 53 µm which is commonly assumed to be more stable than carbon in other fractions (NAIR et al. 2009).

The quantification of the tree root system might have amplified this relationship. Furthermore, its inclusion in the statistical analysis might turn streuobst orchard in the land use storing significantly more carbon than pastures.
As mentioned above, the root:shoot ratio can be used to predict belowground biomass of fruit trees. The ratio of 0.28 established by KUYAH et al. (2012b) seems to be the most suitable. Even though their study area was in Kenya, the ratio was developed for an agricultural landscape with scattered trees, including different species and covering a wide dbh range from 2.5 cm to more than 60 cm. The destructive sample inventory was based on a relatively high tree number and a comparatively large soil radius and depth, so that results can be assumed to be reliable. Other important points were that the ratio is a conservative way of estimating biomass and that trees were managed through pruning, which is supposed to greatly influence root:shoot ratios (KUYAH et al. 2012b). Using the ratio of 0.28, belowground biomass carbon ranges from 0.19 kg C m\(^{-2}\) at plot M1 to 0.99 kg C m\(^{-2}\) at plot K1 with a mean of 0.61 kg C m\(^{-2}\) for all plots. Including these calculations in statistical analyses should be considered in future studies. Simultaneously, belowground biomass of grass roots might then to be taken into account.

Carbon storage trends observed in comparable studies support the main hypothesis: Most of them have shown a slightly higher, if not insignificantly higher, soil organic carbon concentration on grasslands with trees compared to adjacent treeless pastures (HAILE et al. 2008). This increase with the inclusion of trees in grassland systems was as high as 43 % for certain grass and tree species combinations as observed by ANDRADE et al. (2008), although it was in this case not significant. When tree carbon was included, the difference in the total carbon stock turned out to be significant in the majority of studies. In the study conducted by SHARROW & ISMAIL (2004), the difference between the silvopasture and the treeless pasture amounted to 5.8 Mg C ha\(^{-1}\) or an increment difference of 0.5 Mg C ha\(^{-1}\) y\(^{-1}\). However, they also found that the treeless pasture had a numerically lower soil organic carbon content than the silvopasture. JACKSON et al. (2002) and GUO & GIFFORD (2002) differentiated the change in soil carbon storage according to precipitation. JACKSON et al. (2002) concluded that woody plant invasion of grasslands in Texas and New Mexico increased the soil organic carbon content at drier sites (mean annual precipitation 200-350 mm) and decreased it at wetter sites (650-1,050 mm). GUO & GIFFORD (2002) suggested that the conversion of forest to pasture increased soil organic carbon exclusively in areas with 2000-3000 mm precipitation, while trees planted onto pasture land reduced soil carbon stocks. However, this is only true for the conversion of pasture to plantation and not for the conversion pasture to secondary forest which is ascribed to the soil disturbances associated with establishing a plantation. Additionally, only planting conifer trees onto pasture or forest land reduced soil carbon stocks, planting broadleaves had little effect (GUO & GIFFORD 2002, LAGANIÈRE et al. 2010). Since precipitation in the surveyed area is only 600 mm, since trees have been planted with lower densities than those used for plantations and soil disturbances can therefore be assumed to be smaller and since all trees are broadleaves, negative effects of the tree component in streuobst systems on soil organic carbon content can be excluded in this respect.
GUO & GIFFORD (2002) further point out that the conversion of pasture to plantation only causes carbon losses if the litter layer is not included in calculations. This issue has been raised by several authors and PAUL et al. (2002) estimated the covariable ‘inclusion of the organic layer’ as the variable explaining the greatest proportion of variability in a data set dealing with changes in soil carbon following afforestation. As described by GUO & GIFFORD (2002), the inclusion of the organic layer reversed the decrease in soil carbon with afforestation of pastures to an insignificant change or a net increase in PAUL et al.’s (2002) study. LAGANIÈRE et al. (2010) calculated the contribution of the organic layer to total soil organic carbon stock as high as 17%. Although decomposition of litter might involve less formation of soil organic matter, some of it will eventually be incorporated in deeper soil horizons and thereby become more stable (CERLI et al. 2006, LAGANIÈRE et al. 2010).
Conclusion

Several processes have been identified that likely influence the carbon storage in land use systems. Although most of them suggest a higher soil organic carbon storage in streuobst orchards, this could not be confirmed by the results from this study. The main hypothesis that soil organic carbon storage is larger in streuobst than in pasture has therefore been disproved. Compared to tropical ecosystems, carbon storage in soil is high whereas the storage in biomass is low in temperate ecosystems. Therefore, the soil organic carbon storage in pastures might already be at its upper limit and can only be marginally increased, so that the main focus in temperate regions should be on increasing the aboveground carbon storage e.g. through the planting of trees. While the inclusion of tree carbon turned streuobst into the land use storing more carbon than pasture, the difference in total carbon storage between land uses was not statistically significant and thereby refutes also the second part of the hypothesis. As main reasons, the low tree density and the young tree age respectively the combination of both were identified. It can therefore be concluded that the management of streuobst orchards which also includes the replacement of shattered trees is crucial for the carbon storage potential of streuobst systems. Furthermore, the tree size was found to be more important for the carbon storage per unit area than the tree density. Thus, especially old trees should be well tended.

The other purpose of the current study, the development of an allometric equation applicable to streuobst trees in Austria was well fulfilled and proved to yield reasonable results. As discussed, it could have been improved by including certain parameters, most importantly belowground biomass which can make up a substantial share of total tree biomass. The derived equation can therefore be assumed to be a very conservative estimate of tree carbon.

Although streuobst systems could not be characterised as storing more total carbon than pastures, their storage was still high compared to other land use systems, even compared to the ones from other ecoregions. This might be a further reason to promote the establishment and the conservation of streuobst orchards in Europe. Moreover, results from the current study are yet another confirmation of the supposition that agroforestry systems are the preferred choice of land use system in terms of carbon storage.

The contribution of agroforestry to carbon storage at a global scale has been estimated to range from 12 to 228 Mg C ha^{-1} depending on ecoregion, with a median of 95 Mg C ha^{-1} (DIXON 1995). NAIR et al. (2009) estimate the area currently under agroforestry as being 1,023 million ha worldwide, which yields a current carbon storage of 97.185 Gt C in agroforestry systems. The land area technically suitable for
the establishment of agroforestry systems is estimated to be 585-1215 million ha in Africa, Asia and North and South America. This means the potential carbon storage is 55.575-115.425 Gt C (DIXON 1995), which is about 40-80 % of the current carbon loss through land use change (IPCC 2007a). Although the estimates of carbon storage in agroforestry systems and of the area currently under agroforestry as well as the area potentially available vary largely with system, ecoregion and definition of agroforestry area, agroforestry has been recognized as a carbon sequestration strategy and thereby as playing a considerable role in climate change mitigation (ALBRECHT & KANDJI 2003, HOWLETT et al. 2011, MONTAGNINI & NAIR 2004, NAIR 2011a, NAIR et al. 2009).
Declaration

I, Christiane Peßler,

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hereby declare on my honor that the attached Master thesis has been independently prepared, solely with the support of the listed literature references and that no information has been presented that has not been officially acknowledged. The thesis was not written for another course as part of an examination.

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Thesis topic   Carbon Storage in Orchards
Semester   4/ 5

I declare, here within, that I have transferred the final digital text document (in the format doc, docx, odt, pdf, or rtf) to my mentoring supervisor and that the content and wording is entirely my own work. I am aware that the digital version of my document can and/ or will be checked for plagiarism with the help of an analyses software program.
Appendix

Picture 8: Four examples for streuobst trees from destructive sample inventory.

Source: Own illustration.
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