agronomy

MDPI

Article

Comparative Study between Silvopastoral and Agroforest Systems on Soil Quality in a Disturbed Native Forest of South-Central Chile

Juan Ortiz [1,†], Francis Dube [1], Pablo Neira [2], Rafael R. Hernández Valera [3], Pedro M. de Souza Campos [4], Marcelo Panichini [4], Andrés Pérez-San Martín [5], Neal B. Stolpe [6], Erick Zagal [6] and Gustavo Curaqueo [7,*]

1 Department of Silviculture, Faculty of Forest Sciences, University of Concepción, Victoria 631, Concepción 4030000, Chile
2 Institut de Recerca de la Biodiversitat (IRBio), Universitat de Barcelona, 08028 Barcelona, Spain
3 Colegio de Postgraduados, Campus Montecillo, Mexico-Texcoco, Highway (km 36.5), Texcoco 56230, Mexico
4 National Agricultural Research Institute (INIA), Carillanca, km 10, Cajón Road, Vilcún 4880000, Chile
5 Doctoral Program in Agricultural Sciences, Catholic University of Temuco, Rudecindo Ortega 02950, Temuco 4813302, Chile
6 Soils and Natural Resources Department, Faculty of Agronomy, University of Concepción, Vicente Méndez 595, Casilla 537, Chillan 3812120, Chile
7 Department of Agricultural and Aquaculture Sciences & Food Production Research Nucleus, Catholic University of Temuco, Rudecindo Ortega 02950, Temuco 4813302, Chile
* Correspondence: gcuraqueo@uct.cl
† This paper is a part of the Ph.D. Thesis of Juan Ortiz.

Abstract: Agroforestry systems (AFSs) have gained recognition as a land use strategy to address food security and climate change. They involve intentionally cultivating trees alongside crops and/or animals. AFSs cover approximately 5% of the global forest area and promote sustainable soil conservation, including soil organic carbon (C) sequestration (C_{SEQ}). In some areas of Chile, AFSs are used to preserve the ecological value of native forests. This study evaluates the effects of two AFSs, namely, an agroforest for fodder production ($A_{GRO}F_{RST}$) and Silvopastoral (SPS), within a degraded native forest (*Nothofagus obliqua* sp.). The evaluation focuses on their impact on C_{SEQ} capacity and soil quality (SQ), using soil quality indexes (SQIs) derived from 30 soil quality indicators (SI_{NDs}) related to physical, chemical, and microbiological properties at two depths (0–5 and 5–20 cm). The results for the total depth analyzed (0–20 cm) indicate an average C_{SEQ} of 6.88 and 4.83 Mg C yr^{-1} and a global SQI of 37.8% and 31.0% for $A_{GRO}F_{RST}$ and SPS, respectively. Among the thirteen SI_{NDs} that demonstrated significant differences ($p < 0.05$), five were associated with the considered depths (P^+, Ca^{2+}, S, ECEC, and Al_{SAT}), three differed between $A_{GRO}F_{RST}$ and SPS (BD, NH_4^+, NO_3^-), while SOC, K^+, and Mg^{2+} varied across all conditions (e.g., combinations of systems and depths), and β-$_{GLU}$ and N_{MIN} differed in a single condition. However, almost all 30 SI_{NDs} analyzed showed higher values at the 0–5 cm depth, indicating the positive effects of soil organic matter (SOM)/SOC additions. Significant interactions (Pearson's correlation) revealed that SOC correlated with most SI_{NDs} (e.g., N, NH_4^+, P^+, K^+, Ca^{2+}, Mg^{2+}, S, ECEC, N_{MIN}). These findings suggest that both $A_{GRO}F_{RST}$ and SPS systems have similar capabilities in restoring the ecological value of native *Nothofagus* forests while providing conditions for productive and complementary use. This sustainable option offers opportunities for cattle production alongside ecological restoration efforts and provides a possible strategy to generate public policies related to the ecosystem services of agroforestry systems.

Keywords: agroforestry; C sequestration; *Nothofagus* sp.; food security; climate change mitigation

Citation: Ortiz, J.; Dube, F.; Neira, P.; Hernández Valera, R.R.; de Souza Campos, P.M.; Panichini, M.; Pérez-San Martín, A.; Stolpe, N.B.; Zagal, E.; Curaqueo, G. Comparative Study between Silvopastoral and Agroforest Systems on Soil Quality in a Disturbed Native Forest of South-Central Chile. *Agronomy* **2023**, *13*, 2683. https://doi.org/10.3390/agronomy13112683

Academic Editors: Gianni Bellocchi and Jiafa Luo

Received: 19 January 2023
Revised: 12 October 2023
Accepted: 15 October 2023
Published: 25 October 2023

1. Introduction

Nowadays, about 51.4–52.7 Pg CO_{2eq} y^{-1} are globally emitted by anthropogenic activities [1–3], where 7.1–8 CO_{2eq} y^{-1} are generated via agriculture, including 3.2–5.7 Pg released to land use changes [1,4]. Livestock symbolizes probably the most relevant

scenarios mentioned above, occupying 33% of the total land area [5] and generating 14.5% of global CO_{2eq} emissions [6]. Moreover, uncontrolled livestock (overgrazing), resulted in an overall degradation of about 73% of pastures [7], also involved in 69% of deforestation processes (48,000 km^2 y^{-1}) [8], consequently being primarily responsible for 5% of historical SOC losses (including 2.4 Pg CO_2 y^{-1}) [9]. Therefore, unplanned grazing within a forest leads to direct C losses via plant biomass removal, limiting the formation of SOC by reducing C inputs [10]. Consequently, there is an urgent need for the implementation of sustainable land management promoting C_{SEQ}.

The last is of ecological relevance since forests represent principal ecosystems as C and SOC reservoirs since central SOC stabilization mechanisms are present: (i) litterfall and understory plants, promoting ground cover (e.g., forest-floor or O pedogenic horizons), associated with practical rainfall effects (e.g., regulating soil temperature, reducing decomposition rates—SOC losses), (ii) rich lignin-based inputs such as woody debris, which increase SOC recalcitrance, (iii) a reduction in the accessibility of microorganisms to sources of labile SOC via aggregation processes mediated by precipitation–sorption-complexation processes and soil animal and root activities and byproducts, and (iv) interactions among soil inorganic and organic substances (e.g., microbial activity), resulting in the alteration of decomposition rates and/or re-synthesis of new organics [11,12]. Likewise, such increase in soil organic matter reservoirs (SOC), or "SOC fertilization", is of ecological relevance since it modifies different soil properties, ultimately controlling the availability of nutrients (e.g., N, S. P, Fe, Cu, and partially, Zn) and water [11,13,14].

Agroforestry systems (AFSs), regarded as global recommended land uses, can offset up to 2% of the aforementioned global emissions [15] since they can store nearly 300 Mg SOC ha^{-1} 0–1 m depth [16], including all the benefits described above. About 14 ecological mechanisms for C_{SEQ} have been identified in AFSs, including tree–herbaceous combinations (e.g., agroforest); meanwhile, tree–herbaceous–animal associations (e.g., SPS) reached 25 [17].

AFSs cover 1023–1600 Mha worldwide, including 700, 450, 300, 100, and 50 Mha for alley cropping (including agroforest for fodder production [$A_{GRO}F_{RST}$]), Silvopastoral systems (SPSs), protective AFSs (e.g., windbreaks, riparian buffers), multi-strata, and disperse trees in agro-systems, respectively [18]. SPSs have a remarkable C_{SEQ} capacity, reaching 1.8–6.1 Mg ha^{-1} y^{-1} [19–21], which is about 25% more than $A_{GRO}F_{RST}$ [16].

Multiple benefits of SOC pool improvement would allow sustainability in the long-term for animal and plant productivity, improved water and air quality, and support human health improvement, collectively known as soil quality (SQ) [22,23]. Soil properties sensitive to spatio-temporal variations due to land use changes or management are potentially valuable tools to estimate SQ indexes (SQIs) and are designed as SQ indicators (SI_{NDs}) [24]. An SQI is regarded as an interrelated set of parameters that provide numerical data expressing the different aspects of soil productivity (e.g., fertility status), ecological functions, and time-lapses in which soil has changed [25]. Consequently, SQIs should include the determination of physical, chemical, and biological SI_{ND} [24,26,27]. In addition, an SI_{ND} should be easily measured and replicated to be eventually integrated into databases to: (i) define, implement, and monitor local strategies of conservation and (ii) identify regional ecosystem patterns (e.g., acidification and/or flooding) [28,29].

In Chile, mainly due to anthropogenic activities, 37.8% of the national territory is under moderate to severe land degradation [30,31], where the Andean foothills are the most susceptible areas due to the high erosive potential of the soils, which endanger their crucial role as genetic reservoirs and the regulation of the water cycle.

In this respect, at least 44% (8.1 Mha) of the native forest in southern Chile has been partially replaced due to the implementation of different land uses, including (i) 0.5 Mha of urban areas, (ii) 2.8 Mha of croplands, (iii) the expansion of about 3 Mha of grasslands, and (iv) 2.1 Mha to forest plantations, where the *Nothofagus* genus the most strongly affected (70% of its original area) [32]. However, considering the extensively reported benefits of AFSs, very few but relevant experiences of using AFSs in temperate native forests have been reported in Chile. For instance, it has been observed that these systems promote a range of

30–50% of solar irradiance, which enhances the plant diversity of the understory, increases prairie productivity, and reduces wind speed by up to 200% in Ñirre and Nothofagus forests (*Nothofagus antartica*) [33–35], promotes greater SOC Q_{SEQ} accumulation rates than observed in *Nothofagus obliqua* forests [36], and doubles the SOC Q_{SEQ} capacity compared with commercial forestry plantations [37].

Therefore, this study aimed to evaluate the comparative effect of two principal and novel AFSs in Chile (SPS and $A_{GRO}F_{RST}$) 5 years after their establishment in a degraded *Nothofagus* sp. native forest on C_{SEQ} and SQ with different physical, chemical, and microbiological—or early response—SI_{ND}, subsequently grouped and weighted according to their current status and functional relevance into an SQI, which expresses soil health associating soil properties and its capacity to provide ecosystem services and/or to evaluate determined management propose.

2. Materials and Methods

2.1. Site Description and Experimental Design

Ranchillo Alto is located within a *Nothofagus* sp. forest at Ñuble Region, which is under severe degradation processes resulting from: (i) over-logging (e.g., uncontrolled woodlots), (ii) overgrazing, (iii) browsing, limiting forest regeneration, and (iv) allotment and over-utilization of areas destined to agriculture. To address this situation, two different AFSs were established in 2016, an SPS and $A_{GRO}F_{RST}$, each divided into 3 plots of 1.33 ha [34,38,39] (Figure 1).

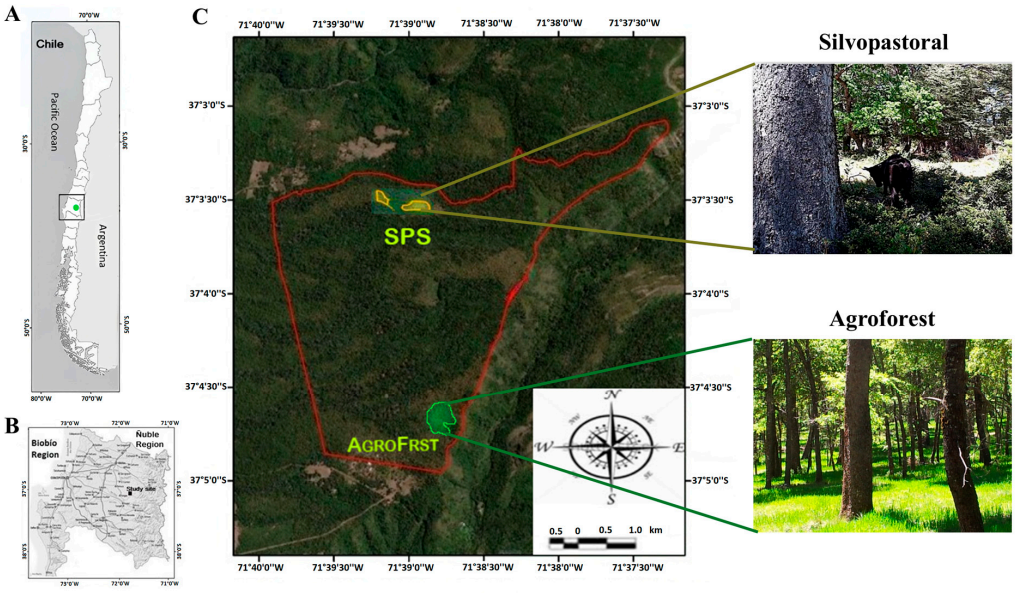

Figure 1. Approximation maps of the study site at (**A**) national and (**B**) regional levels (Source: Maps from Nations Online Project 2016, http:///www.nationsonline.org/oneworld/map/chile-political-map.htm, accessed on 18 January 2023). (**C**) Satellite image of Ranchillo Alto (demarcated in red), a 635 ha property consigned to the Universidad de Concepción in the Ñuble Region (120 km east of Concepcion City), and the study sites $A_{GRO}F_{RST}$ (37°04′43″ S, 71°38′41″ W; 1260 m.a.s.l.) and Silvopastoral (37°03′36″ S, 71°39′14″ W; 1290 m.a.s.l.), both 6 ha in size (each subdivided into 2 ha plots). (After Dube et al. [38] and Leal et al. [40]). Photographs of the two systems above are the (i) SPS, composed of oak (*Nothofagus obliqua*) and coihue (*Nothofagus dombeyi*) at a tree density of 173.3 stems ha^{-1} (133.3:40, respectively), corresponding to 65–75% of solar irradiance, whereas the understory includes introduced associations of Vicia (*Fabaceae purpurea*), clover (*Trifolium incarnatum*, *T. subterraneum*, and *T. vesiculosum*), *Lolium multiflorum westerwoldicum*, *Phalaris acuatica*, *Lolium perenne*, *Festuca arundinacea*, *Dactylis glomerata*, natural re-sprouting Radal (*Lomatia hirsuta*), and sprouts of Quila (*Chusquea quila*). Below is the (ii) $A_{GRO}F_{RST}$, composed of oak (*Nothofagus obliqua*) at a tree density of 446.6 stems ha^{-1} (35–45% of solar irradiance) and oats (*Avena sativa*)-vetch (*Vicia atropurpurea*) as an herbaceous component (Photos credit: F. Dube, 2019).

The $A_{GRO}F_{RST}$ is used as a complementary food source for cattle (Red Angus) of the SPS during winter, yielding up to 24 t ha^{-1} yr^{-1} (dry basis) (2017–2018), which is stored in bundles to be used chiefly during winter as a staple food. Regarding the grazing season, up to 12 cows remained foraging for 11–15 days within each plot of the SPS. The soils were classified as medial, amorphic, mesic Typic Haploxerands corresponding to the "Santa Barbara" Series according to the USDA [41] and Stolpe [42]. Andisols are of crucial ecological and economic importance in Chile (particularly at 35–49° S), not only supporting about 60% of the national cropland area, according to Besoain and Sepulveda [43], but also covering 60–70% of Andean foothills [42,44].

2.2. Soil Sampling and Characterization

In January 2019, 3 bulk soil samples (2 kg) from 3 plots corresponding to the two AFSs (SPS and $A_{GRO}F_{RST}$) were obtained within two different depths (0–5 cm and 5–20 cm) and three replications (n = 36). The soil samples were taken from a low slope site under light clearings and far from tree trunks (and with no apparent influence of trampling or presence of feces, in the case of SPS) according to Ortiz et al., 2020 [36].

After that, the samples were air-dried and mixed and then ground and passed a 2 mm sieve for later analysis, except for biological trials, where in such cases, the samples were stored unaltered under cold conditions. Soil physical determinations were performed as follows: (a) Bulk density was determined according to Stone (1991), where samples were taken with a cylindrical soil core (211 cm^3) and dried at 105 °C until reaching a constant weight. (b) Soil particle density (PD) was estimated using the pycnometer method according to Blake and Hartge [45]. (c) Net pore space (P_{OR}) was calculated from the BD and PD values as follows:

$$P_{OR} = [PD - BD/PD] \times 100 \quad (1)$$

(d) Penetration resistance ($P_{EN}R$) was evaluated using a penetrometer model Soil Compaction Tester Dickey–John (Auburn, IL, USA). A total of 60 measurements in each plot were taken across longitudinal transects to ensure representativeness. (e) Water-stable aggregates (WSA) (%) were determined according to the method proposed by Kemper and Rosenau [46]. Each sample was placed in a 0.250 mm sieve and then immersed for 3 min (35 rep. min^{-1}) in an aluminum chamber containing distilled water. Dispersed soil was dried at 105 °C, while the remaining soil was placed into another aluminum chamber containing sodium hexametaphosphate (2 g L^{-1}) for 15 min (35 rep min^{-1}), and the dispersed soil was dried at 105 °C. After both procedures, samples were weighed to determine each proportion compared to the original sample. (f) A hydraulic conductivity (unsaturated) (K) trial ($I_{NF}Vk$) was conducted using an infiltrometer model Mini Disk Infiltrometer S (Pullman, WA, USA). The determination of K (cm day^{-1}) was estimated according to Zhang [47] using a sequence of cumulative infiltration measurements. (g) Water holding capacity (WHC) was tested based on the method proposed by Zagal et al. [48]. Each sample was placed into a plastic cone (1:2 soil water ratio), which was previously sealed at the bottom with adhesive tape, for 12 h. After that, the tape was carefully perforated, allowing the soil solution to drain, which was collected in a plastic bottle and eventually measured.

Chemical characterization ($pH_{(water)}$, NH_4^+, NO_3^-, P, K^+, Ca^{2+}, Mg^{2+}, effective cation-exchange capacity (ECEC), Na^+, S, $Al_{EXCH,}$ and % Al_{SAT}) was conducted at the Agricultural Research Institute of Chile (INIA-Quilamapu, Chillán, Chile) according to the methods proposed by Sadzawka et al. [49]. Total N and SOC content determinations were performed at the Soil and Natural Resources Laboratory (Faculty of Agronomy, University of Concepcion), according to Wright and Bailey [50].

Microbiological parameters were assessed as follows: (a) Soil microbial respiration (S_{RESP}) was determined using the substrate-induced method according to Anderson and Domsch [51], where 10 g of dry soil was incubated for 24 h and then placed into a gas-tight container where liquid glucose amendments were added [52,53] to bring the slightly dried soil to 60% of water-filled pore space [54]. A subsequent CO_2 measurement was performed using a CO_2 analyzer (LI-COR LI-820, Lincoln, NE, USA). The minimal glucose concentration (able to

produce the maximal respiratory rates) was added to each sample (5 and 10 μMole g^{-1} at 0–5 and 5–20 cm depths, respectively). Microbial Biomass C (M_{SOC}) was determined using the fumigation-extraction method according to Jenkinson et al., 2004 [52], where 100 g of soil was divided into 4 equal parts and 50 g was fumigated with chloroform ($CHCl_3$) free of ethanol in a vacuum desiccator at room temperature, while the rest were incubated. After 24 h, the chloroform was removed, and all samples were extracted with K_2SO_4 (100 mL, 0.5 M) added, stirring for 45 min at 140 rpm. Then, the supernatant was filtered, and 1 mL of reagent N ninhydrin was added (0.8 g ninhydrin + 0.12 g hydridantine hydrate + 30 mL of organic solvent dimethyl sulfoxide + 10 mL lithium acetate buffer) and then placed in a water bath for 20 min. Later, ethanol + distilled water (15 mL, 1:1 ratio) was added for further analysis in an ultraviolet/visible light spectrophotometer (UV/Vis) at 568 ηm both fumigated samples and controls. Finally, the difference between fumigated and no fumigated samples was multiplied by a factor (Equation (2)) to obtain M_{SOC}.

$$x = 31(\text{N ninhydrin}) = \text{dry weight}(31) = x\ \mu g\ \text{Msoc}\cdot(\text{dry soil g})^{-1}. \quad (2)$$

where x means the total microbial biomass C expressed as μg C g^{-1} dry soil and 31 refers to a factor used to calculate the biomass size based on the amount of N ninhydrin.

(b) A potential N mineralization (N_{MIN}) trial and nitrification (N_{NIT}) estimation were conducted according to Linn and Doran [54], where three subsamples were incubated at 22 °C for 10 days at 60% moisture, including another three subsamples consisting of 5 g dry soil used as controls. After that, each sample was put into 150 mL plastic flasks with 25 mL of K_2SO_4 (0.5 M) and shaken for 1 h at 180 rpm. The resulting extract was decanted, filtered, and analyzed using colorimetry using a UV-visible spectrophotometer (AA3, BRAN + LUEBBE, Norderstedt, Germany). According to Alef [55], Nessler and sulfosalicylic reagents were utilized to determine the N mineralization as NH_4^+ and NO_3^-. Finally, N_{MIN} and N_{NIT} were calculated using the following equations:

$$N_{MIN} = [(N-NH_4 + N-NO_{3^-})_f - (N-NH_4 + N-NO_{3^-})_i]/Td \quad (3)$$

$$N-NO_{3^-} = [(N-NO_{3^-})_f - (N-NO_{3^-})_i]/Td \quad (4)$$

where f and i subscripts refer to concentrations measured before and after incubation, respectively, and Td indicates the incubation time in days. Both parameters were expressed as μg N g^{-1} dry soil d^{-1}.

Three enzymatic activities were determined as follows: (c) The β-glucosidase activity was determined using the method proposed by Tabatabai [56], where after an incubation period of soil samples in a buffer solution (pH 6), there was a subsequent determination of p-nitrophenol released using the colorimetric at 400 nm. (d) Urease activity was determined using incubation of soil samples into urea solution (0.7 M) for 2 h at 37 °C, according to the method proposed by Kandeler et al. [57]. (e) A phosphatase trial was conducted according to Tabatabai and Bremner [58], Tabatabai [56], and Bello et al. [59], where, to 1 g of soil, 4 mL of buffer solution (pH4) was added to adjust the pH to 6.5, and then 0.3 mL of toluene and 1 mL of 0.12 M disodium *p*-nitrophenyl phosphate tetrahydrate were incorporated and mixed and eventually incubated at 37 °C for 1 h. Then, 1 mL of 0.5 M calcium chloride and 4 mL of 0.5 M sodium hydroxide were added to each sample. The resulting mixed suspension was filtered and measured in a spectrophotometer at 420 nm. (f) Fluorescein diacetate hydrolysis (FDA) was assayed using the method of Alef [60] and Green et al. [61], where 50 mL of 60 mM Na-phosphate solution (buffered at pH 7.5) was added to 1.0 g of soil and then incubated at 37 °C for 3 h. Then, 20 mL of acetone was added to stop the reaction until a 50% v/v ratio was reached. Shortly after that, the suspension of each sample was centrifuged at 4000 rpm for 10–15 min, and using the clear supernatant, FDA hydrolysis was determined using spectrophotometry at 490 nm. Both enzyme activities

and FDA absorbance were measured using a UV-visible spectrophotometer (AA3, BRAN + LUEBBE, Norderstedt, Germany).

2.3. SQI Estimation

Each SI_{ND} was selected based on projected goals for both SPS and $A_{GRO}F_{RST}$ (soil reclamation, gradual improvement of soil fertility, and C_{SEQ}). After the analytical characterization of any single SI_{ND}, a numerical value (normalized on a scale of 0–100) or sub-index was assigned depending on its relevance to the overall SQ, from optimal to critical ranges according to the methodology proposed by Amacher et al. [62] (Section A). Before SQ estimation, this was divided into chemical (SQ_{CHE}), physical (SQ_{PHY}), and biological (SQ_{BIOL}) in order to elucidate the contribution of a particular group of pedogenetic processes. Therefore, the SQI_{PHY} was estimated as follows:

$$SQI_{PHY} = \Sigma sub - index[I_{NF}V + \%WSA + WHC + P_{EN}R + BD + PD + P_{OR}] \quad (5)$$

where $SQI_{PHYSICAL}$ is the physical soil quality index, $I_{NF}V$ is infiltration, %WSA is the water-stable aggregates %, WHC is the water holding capacity, $P_{EN}R$ is the penetration resistance, BD is the bulk density, PD is the particle density, and P_{OR} is the total porosity.

SQ_{CHE} was calculated as follows:

$$SQI_{CHEM} = \Sigma sub - index(pH + \%SOC + C:N + N + NH_{4+} + NO_{3-} + CEC + P + K + Ca^{2+} + Mg^{2+} + Na + S + Al_{EXCH} + Al_{SAT}) \quad (6)$$

where SQI_{CHEM}: chemical soil quality index; pH: soil reaction; %SOC: percentage of SOC; C/N: C:N ratio; N: total N (%); NH_4^+: available ammonium; NO_3: available nitrate; ECEC: effective cation exchange capacity, P^+: available phosphorus; K^+: potassium content; Ca^{2+}: calcium content; Mg^{2+}: magnesium content; Na: sodium content; Al_{EXCH}: exchangeable aluminum; and Al_{SAT}: aluminum saturation (%).

The SQ_{BIOL} was calculated as follows:

$$SQI_{MBIOL} = \Sigma sub - index[M_{SOC} + M_N + MR_{ESP} + N_{MIN} + \beta_{-GLU} + U_{RS} + P_{HOSP} + FDA] \quad (7)$$

where SQI_{MBIOL} is the microbiological soil quality index; M_{SOC}: microbial biomass C; M_N: microbial N; MR_{ESP}: microbial respiration; N_{MIN}: N mineralization; β-$_{GLU}$: β-glucosidase activity; U_{RS}: urease activity; P_{HOSP}: phosphatase activity; and FDA: fluorescein diacetate hydrolysis.

The overall SQI for each system (SPS and $A_{GRO}F_{RST}$) was calculated as follows:

$$SQI_{TOTAL} = \Sigma[SQI_{CHEM} + SQI_{PHYSICAL} + SQM_{BIOL}]/3 \quad (8)$$

Finally, the percentage of % SI_{ND} at critical levels (%SQ) was determined as:

$$\%\ SQ = [\text{number of } SI_{ND} \text{ at critical level/number of } SI_{ND} \text{ estimated}] \times 100 \quad (9)$$

2.4. Statistical Analysis

The dataset resulting from the transformation of analytical results for every SI_{ND} into sub-indexes (Sections A–C) and the estimation of the partial SQ indexes (SQI_{CHEM} + $SQI_{PHYSICAL}$ + SQ_{BIOL}) and SQI_{TOTAL} were performed using Microsoft Excel 365 v. 2309. Since not all the variables analyzed in the dataset complied with the normality of residuals and homogeneity of variance, a one-way analysis, ANOVA, and Kruskal–Wallis test were performed as appropriate, followed by Tukey's or Dunn's HSD test, respectively. A Pearson's correlation analysis was conducted between the different SI_{NDs}. The data analysis and graphical representation were conducted using R (statistical software V4.1.0, the R Core Team 2021).

3. Results and Discussion

3.1. Soil Characterization

The results of the analytical procedures are shown below. The different SI_{NDs} were grouped according to the type of functions they perform and their effect and interpretation respecting CSEQ and SQ.

3.1.1. Physical Properties as SI_{NDs}

There were no significant differences ($p > 0.05$) found in any of the physical SI_{NDs} between systems and depths (Table 1), except for BD (0.6 and 0.7 g cm^{-3} for SPS and $A_{GRO}F_{RST}$, respectively). Those differences may be related to the greater previous intensive-mechanized management, "consisting on chisel plowing, debris dragging, both at an early stage and subsequent seasonal operations of harvesting, bundle production and their transportation", leading to lower tree ha^{-1} stocking [63]. The same trend is also mirrored in the low WSA means: <50% in all cases (48.8 and 49.0% for $A_{GRO}F_{RST}$ and SPS for the total depth 0–20 cm). The SI_{NDs} BD, PD, and P_{OR} resulted in optimal values within all conditions (Table 1, Appendix A—Table A1), ranging from 0.6 to 0.7 g cm^{-3}; 2.05 to 2.12 g cm^{-3}; and 70.7, 66.6% for SPS and $A_{GRO}F_{RST}$ (0–20 cm), respectively.

Table 1. Characterization results of physical SI_{NDs}.

	System			
SI_{ND}	**SPS 0–5**	**SPS 5–20**	**$A_{GRO}F_{RST}$ 0–5**	**$A_{GRO}F_{RST}$ 5–20**
[A] $I_{NF}Vk$ *	$19.41 \pm 1.2 \times 10^{-5}$	$19.41 \pm 1.2 \times 10^{-5}$	$13.08 \pm 1.3 \times 10^{-5}$	$13.08 \pm 1.3 \times 10^{-5}$
WHC **	70.83 ± 1.67 a	65.97 ± 1.86 a	74.44 ± 0.78 a	65.97 ± 2.75 a
BD ***	0.57 ± 0.01 a	0.61 ± 0.01 a	0.65 ± 0.02 b	0.73 ± 0.02 b
PD ***	1.91 ± 0.01 a	2.09 ± 0.01 a	2.0 ± 0.01 a	2.16 ± 0.02 a
[B] P_{OR} (%) **	70.21 ± 0.79 a	70.83 ± 0.87 a	67.09 ± 0.71 a	66.21 ± 0.66 a
WSA **	49.20 ± 0.05 a	48.65 ± 0.02 a	49.74 ± 0.43 a	48.82 ± 0.20 a
PEN_{RES} ****	100–200 ± 0.00 a	100–200 ± 0.00 a	100–200 ± 0.00 a	100–200 ± 0.00 a

[A] Relative to the total depth, [B] Estimated using the Formula (1), *: (cm d^{-1}); **: (%); ***: (g cm^{-3}); ****: (PSI). Identical lowercase letters mean that according to Tukey's mean comparison analysis ($p < 0.05$), there are no significant differences.

Mainly, BD was in the typical range for low-nonallophanic/C rich (SOC ≥ 6%) volcanic soils (≤0.9 g cm^{-3}), which is consistent with the PD estimations, where all the values that were observed were similar to the reported mean for PD of condensed SOC (1.5 g cm^{-3}), and consequently, with a wide range of P_{OR} [61]. Similar findings were reported by Nanzyo [64] and Ortiz et al. [36], who estimated ranges from 1.9 to 2.1 (0–15 cm) and 1.9 to 2.0 (0–20 cm) for PD and 73.9; 68.3 to 74.2% for P_{OR}, respectively, in volcanic soil from a native forest and a native *Nothofagus* sp. forest under SPS management, both in South Central Chile.

The SI_{NDs} related to soil hydraulic capacities showed similar WHC values (0–20 cm) of 67.19 and 68.09 $A_{GRO}F_{RST}$ > SPS, while there was a remarkable difference for $I_{NF}Vk$ (19.41 and 13.8 for the SPS and $A_{GRO}F_{RST}$, respectively) and PEN_{RES} 100–200 psi (except for $A_{GRO}F_{RST}$ 0–5), suggesting a probable structural degradation/net disaggregation due to past anthropogenic disturbances. However, the former did not affect water entrance into soil and storage and movement within the soil matrix because of the distinctive properties of Andisols and the successive re-aggregation processes expected to occur due to the constant C inputs in both systems. For instance, Panichini determined similar means for WHC (70.69%) in conterminous Andisols to our study site (same series) for wheat production that was managed during four years under stubble incorporation (10 t ha^{-1} yr^{-1}) [65].

3.1.2. Chemical Properties as SI_{NDs}

From the 15 chemical SI_{NDs} analyzed, pH, C:N, N, Na, and Al_{EXCH} showed no statistical differences ($p > 0.05$) at any condition (Table 2).

Table 2. Characterization results of chemical SI_{NDs}.

	System			
SI_{ND}	**SPS 0–5**	**SPS 5–20**	**$A_{GRO}F_{RST}$ 0–5**	**$A_{GRO}F_{RST}$ 5–20**
•pH	6.03 ± 0.07 a	6.06 ± 0.06 a	5.62 ± 0.17 a	5.51 ± 0.04 a
SOC *	13.94 ± 0.02 a	10.65 ± 0.16 b	14.63 ± 0.03 a	11.95 ± 0.15 c
C:N	12.73 ± 0.25 a	13.48 ± 1.00 a	10.47 ± 0.15 a	10.90 ± 0.11 a
N *	1.10 ± 0.02 a	0.80 ± 0.05 b	1.40 ± 0.02 c	1.10 ± 0.02 a
•P^{+} **	3.11 ± 0.05 a	1.89 ± 0.13 a	4.88 ± 1.12 a	1.83 ± 0.17 a
NH_4^{+} **	12.13 ± 0.53 a	9.30 ± 0.37 b	11.62 ± 0.64 ab	9.59 ± 0.08 ab
•N-NO_3^{-} **	3.32 ± 0.34 ab	2.46 ± 0.41 b	28.70 ± 4.15 a	22.69± 3.12 ab
K **	47.37 ± 1.57 ab	30.72 ± 4.58 c	51.33 ± 1.69 a	37.58 ± 0.11 bc
•Ca^{2++} **	5.48 ± 2.29 a	0.97 ± 0.22 a	6.66 ± 0.59 a	1.50 ± 0.32 a
Mg^{2+} **	0.20 ± 0.01 a	0.08 ± 0.03 b	0.36 ± 0.03 c	0.15 ± 0.07 ab
S **	7.71 ± 0.44 ab	2.10 ± 0.10 c	7.80 ± 0.49 a	2.87 ± 0.40 c
•ECEC ***	9.22 ± 0.38 a	1.77 ± 0.52 b	7.68 ± 1.69 ab	3.19 ± 0.30 ab
Na *	1.61×10^{-6} ± 0.00 a	1.61×10^{-6} ± 0.00 a	1.38×10^{-6} ± 0.00 a	1.38×10^{-6} ± 0.00 a
•Al_{EXCH} ***	0.09 ± 0.01 ab	0.18 ± 0.01 a	0.14 ± 0.03 b	0.29 ± 0.15 ab
•Al_{SAT} *	0.86 ± 0.05 b	4.68 ± 1.57 ab	1.57 ± 0.21 ab	6.45 ± 0.45 a

*: %; **: mg kg^{-1}; ***: cmol (+) kg^{-1}), SI_{ND}: soil indicator. Identical lowercase letters mean that according to Tukey's mean comparison analysis ($p < 0.05$; n = 18), there are no significant differences. •X: the points to the left of the names of some indicators refer to the fact that they were analyzed using the Kruskal–Wallis method because they did not meet the conditions of an ANOVA test.

In the case of pH, the low differences that were observed (5.54, 6.05) for the $A_{GRO}F_{RST}$ and SPS, respectively (0–20 cm) (weighted), could be explained by the narrow divergence in precipitation regimes (north > south) and basic cations scavenging by plants [64,66].

Regarding total N (0.88, 1.18%) and C:N ratios (13.30, 10.80) (0–20 cm) for SPS and $A_{GRO}F_{RST}$, respectively, the comparative greater values that were observed in $A_{GRO}F_{RST}$ (also by depth) expressed a potentially faster SOM cycling than SPS, which probably reflects both the differences in N fertilization practices and labile C inputs. A similar pattern occurred with P, which varied both between depths and systems ($p > 0.05$) and was accentuated within $A_{GRO}F_{RST}$ (2.20 and 2.59 ppm at 0–20 cm for the SPS and $A_{GRO}F_{RST}$, respectively). Regarding available N forms, the SI_{ND} NH_4^{+} had average weighted values (0–20 cm) of 10.01 and 10.10 ppm for the SPS and $A_{GRO}F_{RST}$, respectively, and showed significant differences ($p \leq 0.05$) only in the SPS (by depth). Such differences were probably associated with urine inputs from the animal component and a lower intake by the herbaceous plant mosaic, which was composed of both annual and perennial species, contrary to the $A_{GRO}F_{RST}$, although the $A_{GRO}F_{RST}$ means were intermediate between those of the SPS. In the case of NO_3^{-}, significant differences ($p \leq 0.05$) were found between systems having SPS:$A_{GRO}F_{RST}$ ratios of 1:8.6 and 1:9.2 for 0–5 cm and 5–20 cm depths, respectively. Since NO_3^{-} is highly mobile, the remarkable differences could be attributed to high fertilization practices and a net NH_4^{+} consumption by herbaceous components (intensive oats and vetch production) within the $A_{GRO}F_{RST}$. Contrary to our results, Ortiz et al. [36] found C:R ratios of 26.3/25.37 (0–5/5–20 cm) in a 5-year-old SPS established over a degraded forest with *Nothofagus obliqua* (134 stems ha^{-1}).

These differences could be explained by litter quantity/quality variations and root exudates related to tree-specific and vegetation/understory types, environmental factors such as temperature and precipitation, topography (e.g., latitude, elevation), and soil texture [67]. The uniformly low Na^{+} values (0.07, 0.06 $cmol^{(+)}$ kg^{-1}) are related to the relatively high precipitation level, which can leach out basic cations, as previously stated [66]. A similar condition may explain the moderate Al_{EXCH} concentrations (0–20 cm) of 0.26 and 0.16 $cmol^{(+)}$ kg^{-1} for the $A_{GRO}F_{RST}$ and SPS, respectively, despite doubling the concentration at the 5–20 cm depth (regardless of the similar pH values). The same pattern was also observed for the basic cations Ca^{+2}, Mg^{+2}, K^{+}, and ECEC, showing significant differences ($p > 0.05$) between the two depths of both systems, probably due to remarkably higher

SOC content at 0–5 cm of the two systems. The SI_{ND} S varied only at the 0–5 cm depths in both systems ($p \leq 0.05$), and presented ratios of 3.7:1, 2.7:1 (0–5:5–20 cm) for the SPS and $A_{GRO}F_{RST}$, respectively, demonstrating: (i) the very leachable condition of the bioavailable form of this nutrient (SO_4^{2-}) and (ii) the importance of SOM as being responsible for the retention of this anion via organic mineralization and immobilization, apart from inorganic adsorption–desorption mechanisms [68]. Concerning Al_{SAT}, the means of 3.73 and 5.23 (%) at 0–20 cm depth were observed for the SPS and $A_{GRO}F_{RST}$, respectively, and significant differences were observed varying between depths and systems ($p > 0.05$). Since this SI_{ND} expresses the Al_{EXCH}:ECEC ratio, it can provide insights into periodic variations in acidification/potential Al toxicity risks and nutrient depletion/enlargement processes that may ultimately affect crop yield, biomass production, and C_{SEQ} potential [69].

3.1.3. Microbiological Properties as SI_{NDs}

No significant differences ($p > 0.05$) were found in the microbiological SI_{NDs} M_N, MR_{ESP}, U_{RS}, or P_{HOSP}, and FDA showed no statistical differences ($p > 0.05$) at both depths and systems (Table 3). In the case of M_{SOC} and β-$_{GLU}$, differences ($p > 0.05$) were found at SPS 0–5 concerning the rest of the conditions. The same occurred in the case of N_{MIN}, although the significant difference ($p > 0.05$) was between $A_{GRO}F_{RST}$ 0–5 cm and the rest of the conditions.

Table 3. Characterization results of microbiological SI_{NDs}.

	System			
SI_{ND}	**SPS 0–5**	**SPS 5–20**	**$A_{GRO}F_{RST}$ 0–5**	**$A_{GRO}F_{RST}$ 5–20**
•M_{SOC}	1889.50 ± 373.68 b	315.23 ± 292.00 ab	463.43 ± 109.34 ab	69.6 ± 23.76 a
M_N	196.67 ± 72.74 a	46.77 ± 19.16 a	68.77 ± 6.88 a	10.30 ± 1.06 a
MR_{ESP}	0.18 ± 0.01 a	0.13 ± 0.01 a	0.14 ± 0.03 a	0.13 ± 0.01 a
•β-$_{GLU}$	2.43 ± 0.23 b	1.15 ± 0.32 ab	1.34 ± 0.60 ab	0.94 ± 0.26 a
U_{RS}	1247.55 ± 1.33 a	994.69 ± 2.20 a	1063.50 ± 2.68 a	650.07 ± 0.12 a
P_{HOSP}	713.62 ± 0.09 a	699.71 ± 0.07 a	759.52 ± 0.02 a	740.05 ± 0.04 a
FDA	55.85 ± 0.51 a	56.74 ± 1.15 a	54.64 ± 4.71 a	39.93 ± 3.48 a
N_{MIN}	19.36 ± 1.93 ab	1.78 ± 2.13 a	18.21 ± 4.86 ab	8.41 ± 1.49 ab

Measurement units: M_{SOC}: μg C g dw^{-1}; M_N: μg N g dw^{-1}; MR_{ESP} mg: CO_2 g dw^{-1}; β-$_{GLU}$: μg PNF g dw^{-1} h^{-1}; U_{RS}: μg N-NH_4 g dw^{-1} h^{-1}; P_{HOSP}: μg PNF g dw^{-1} h^{-1}); FDA: μg F g dw^{-1}; N_{MIN}: μg N g dw^{-1} d^{-1}. Where dsw: grams dry weight; PNF: p-nitrophenol; F: fluorescence. Identical lowercase letters mean that according to Tukey's mean comparison analysis ($p < 0.05$), there are no significant differences. •X: the points to the left of the names of some indicators refer to the fact that they were analyzed by the Kruskal–Wallis method because they did not meet the conditions of an ANOVA test.

However, a noticeable vertical variability was observed in all the SI_{NDs} (Table 3). The N_{MIN}, MR_{ESP}, and M_{SOC} means were remarkably higher than those estimated by Alfaro et al. [70] in: (i) the same SPS that was analyzed in our study (estimates made in 2015), averaging 1.01/0.20 μg N g dw^{-1} d^{-1}; 0.067/0.040 mg CO_2 g dw^{-1} and 1450.95/881.76 μg C g dw^{-1} and (ii) a conterminous SPS to our study site $A_{GRO}F_{RST}$, both under *Nothofagus obliqua* as a tree component, with means of 1.67/0.48 μg N g dw^{-1} d^{-1}; 0.081/0.048 mg CO_2 g dw^{-1} and 1774.15/1042.98 μg C g dw^{-1} for 0–5 cm/5–20 cm depths in both systems (i,ii), respectively.

The former suggests a temporal enhancement in microbial activity (in the case of the same SPS). Concerning depth differences, they may be due to greater amounts of fresh substrates/relative proportion of labile organic materials (e.g., M_{SOC}), whereas, between the two systems, they might be caused by substrate quality differences including variations in substrate inputs, chemical/physical leaf composition, since deciduous leaf litter has relatively lower lignin content and consequently lower C:N ratios [70,71]. This is consistent with Decker and Boerner [72], who determined lignin N ratios of 21.7 and 27.1 for *N. obliqua* and *N. dombeyi*, respectively. The latter could also explain the significantly higher N_{MIN} values ($p \leq 0.05$) at the 5–20 depth in the $A_{GRO}F_{RST}$ compared with their counterpart in the SPS and was probably due to the historical evolution of litter quality inputs, mineralization–

immobilization processes (e.g., differences on M_N), fertilization practices, and species preferences for available N forms, where the NO_3^- ratio was 1:3.39 at the 5–20 cm depth, favoring the uptake of this chemical species by the herbaceous component, then promoting the N_{MIN}.

Regarding enzymatic activities, the more significant β-$_{GLU}$ activity in the SPS could be explained by the continuous closed canopy sectors given by the evergreen trees (compared with the total deciduous condition in the $A_{GRO}F_{RST}$), that limited solar irradiance, which contributed to preserving the moisture and temperature conditions in the leaf-litter for subsequent fungal proliferation (also correlated with FDA) [72], and consequently, the occurrence of ligninolytic enzymes (e.g., manganese peroxidase), which may influence positively other SI_{NDs} such as M_{SOC} and MR_{ESP} as in the case of our results (Table 3). Despite the U_{RS} activity being well-correlated with the presence of labile N forms and a combination of high temperature–low moisture conditions, the higher biomass activity found in the SPS (e.g., M_{SOC} and MR_{ESP}), may explain the higher U_{RS} values because it was able to release this enzyme [73]. The P_{HOSP} exhibited greater activity in the $A_{GRO}F_{RST}$, probably related to a combination of the more acidic conditions and greater water retention capacity in this system (e.g., SI_{NDs} pH, WHC, and I_{NF}Vk).

In the case of FDA, the factors controlling the increases in activity of this SI_{ND} were directly proportional to pH, temperature, and ECEC [60]. In the case of U_{RS}, the higher C and N biomass in the SPS (M_{SOC} M_N) may stimulate the FDA activity [72], which may explain the differences that were observed mainly at the 5–20 depth between systems. In Andisols (0–15 cm), from a relict-native forest consisting of the plant associations: *Aextoxicon punctatum*, *Nothofagus obliqua*,*Eucryphia cordifolia*, *Laurelia sempervirens* and *Persea lingue* in Temuco, Chile, Reyes et al. [74] observed similar trends to our findings in M_{SOC} (1245 µg C g dw^{-1}), M_N (99.5 µg N g dw^{-1}), and FDA (51.5 µg F g dw^{-1}), although there were remarkable differences for β-$_{GLU}$ and P_{HOSP} activities, with means of 7.3 and 68.2 µmol g dw^{-1} h^{-1} PNF, respectively, virtually tripling and exceeding by a 10 order of magnitude our measurements. This may be related to the maturity stage of the forest, compared with our study sites.

3.2. Interactions among Soil Quality Indicators (SI_{NDs})

A total of 73 interactions with high correlations were detected (with inverse) as follows: 57, 14, and two for chemical, microbiological, and physical SI_{NDs}, respectively (Figure 2). The SOC was the most associated SI_{ND} and was correlated with total N (R = 0.70), available N (NH_4^+ R = 0.76), and other relevant plant nutrients such as P (R = 0.71), K (R = 0.90), Ca^{2+} (R = 0.91), and Mg^{2+} (R = 0.90), S (R = 0.84), including the ECEC (R = 0.89), which also influences N_{MIN} (R = 0.77). This demonstrates the pivotal role of SOM in mediating nutrient storage and supply, contrary to Al, where increases in soil solution may lead to nutrient deficiencies (e.g., ECEC–Al, R = −0.83; NH_4^+–Al_{SAT}, R = −0.83). Most of the microbial SI_{NDs} were strongly correlated, for instance, (i) MR_{ESP}–M_{SOC} (R = 1.0), MR_{ESP}–β-$_{GLU}$ (R = 0.8), and M_{SOC}–β-$_{GLU}$ (R = 0.8), which are responsible for C cycling and (ii) indicators related to the N cycle, including N–N_{MIN} (R = 0.74), M_N–U_{RS} (R = 0.72), and MR_{ESP}–M_N (R = 1.0).

3.3. Carbon Sequestration

Based on previous on-site studies [38,70], it was possible to determine temporal changes in SOC concentrations and stocks (0–20 cm) and their respective accumulation rates (Table 4). Both SOC density and accumulation were greater in the $A_{GRO}F_{RST}$ than the SPS, reflected in SOC stock variations: 21.93 Mg C ha^{-1} for the SPS (5 yr period), while 27.52 Mg C ha^{-1} for the $A_{GRO}F_{RST}$ (4 yr period). The latter could be related to the record-periodicity of fertilization and more labile plant residues in the $A_{GRO}F_{RST}$, promoting faster cycling of plant detritus into the soil matrix (ultimately meaning larger inputs of soil organic matter).

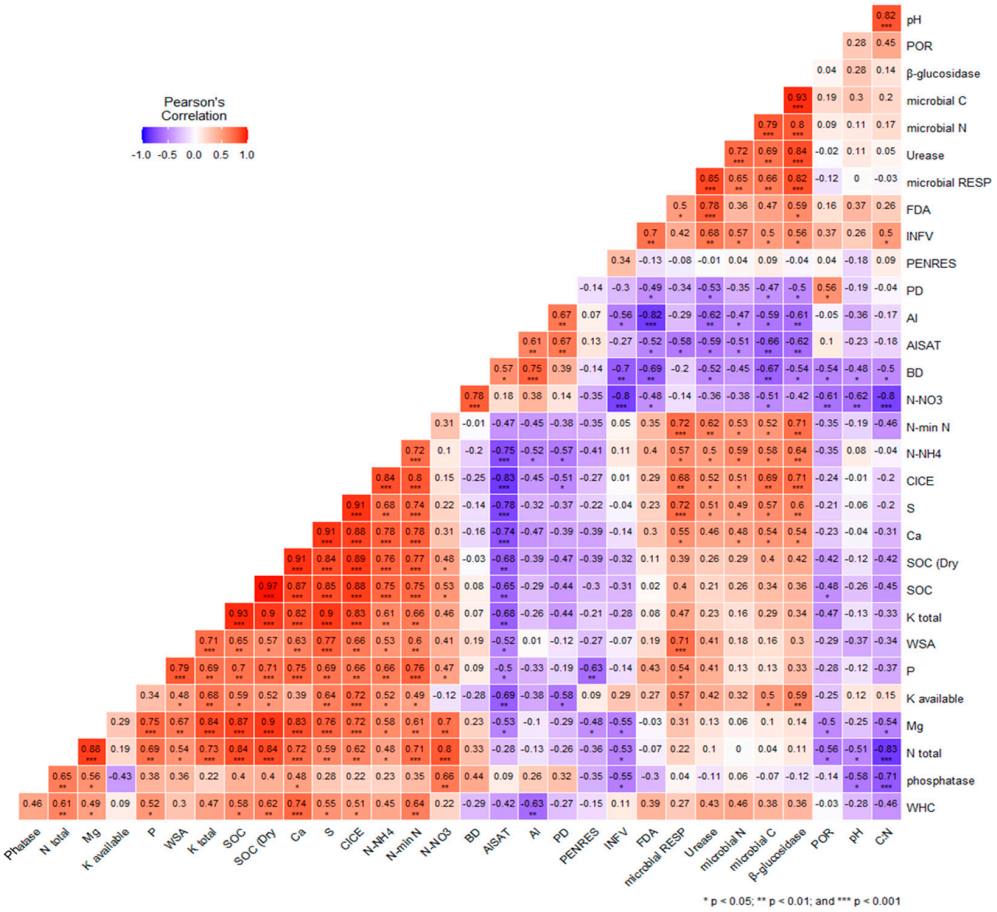

Figure 2. Heat map representing Pearson's correlation coefficients for the different SI_{NDs} analyzed. The marks *, **, and *** indicate *p*-values below 0.05, 0.01, and 0.001, respectively. Reddish tones correspond to positive correlations and blue tones correspond to negative; color scales refer to different levels of correlation.

Table 4. Temporal variation in SOC density, stocks, and C_{SEQ}.

	System	
	SPS 0–20	**$A_{GRO}F_{RST}$ 0–20**
Previous SOC (%)	8.5 *	9.2 **
SOC_{2019}	11.5	12.6
Previous SOC stock (Mg ha^{-1})	61.09 *	79.92 **
SOC stock 2019 (Mg ha^{-1})	83.02	107.44
Theoretical annual C_{SEQ}	5.48	5.50

All data reported correspond to weighted values for the 0–5 and 5–20 cm depths. * For the year 2015, source: Alfaro et al. [69] and ** year 2014 source: Dube et al. [38].

In nearby areas to our study site that were devoted to conservation agriculture (Andisols), Muñoz et al. [75] reported favorable variations in SOC (33.1 to 35.5 Mg ha^{-1}) after 16 years of no-tillage, while Panichini estimated a SOC stock mean of 106.83 Mg ha^{-1} in systems with stubble incorporation for wheat production [65]. Regarding C_{SEQ}, our estimates were in concordance with Ortiz et al. [36], who determined favorable variations in C_{SEQ} of +4.83, +7.5 and 1.6 Mg C ha^{-1} yr^{-1} during the period 2015–2018 in three different SPS within a native *Nothofagus obliqua* degraded forest over Andisols in south-central Chile, having three tree densities of 60, 134 and 258 stems ha^{-1} (corresponding to 85–95%, 65–75%, and 45–55% of solar irradiance), respectively.

3.4. Determination of SQI

After the estimation of all the sub-indexes (Figure 3), different trends were observed by type of SI_{ND}: (i) physically, all the indicators in all conditions (between systems and depths) were inadequate to optimal conditions for plant growth (except for I_{NF}Vk and WSA), representing the highest SQI type (Table 5). However, the critical status for the aggregation is the limit with a higher category, while water movement is at a similar level from acceptable near to optimal, based on the fact that this occurs in both cases, evidencing the historic degradation processes, although gradual changes will probably be observed due to the ongoing SOM inputs. Chemical SQIs presented the highest variation among the systems and depths, having accentuated critical levels that mainly were related to nutrient availability in all conditions (e.g., NH_4^+, P, K, Ca^{2+}); however, the SPS presented differences in the percent of −19.7 and −15.2 for 0–5 and 5–20 cm, respectively, probably due to a combination of natural (e.g., base cation leaching) and anthropogenic (e.g., acid reaction fertilizers use) processes. A lightly Al^{3+} toxic threat was observed, where, despite exchangeable Al^{3+} (Al_{EXCH}) being in moderate concentrations in all conditions except for the SPS at the 0–5 cm depth (optimal), Al saturation (Al_{SAT}) was critical and at high-level risk for the SPS and $A_{GRO}F_{RST}$ at 5–20 cm, respectively, which also corresponded to the condition with high nutrient depletion, alluding to the previous statement. Accordingly, Casanova et al. [76] mention that acidification processes reduce ECEC values, which were lower than 2 cmol kg^{-1} and associated not only to limited bio-availability of certain nutrients but Al^{3+} availability since the exchange complex releases cations to buffer H^+ ion production via leaching.

Table 5. Soil quality indexes (%) by condition.

	Systems			
	SPS 0–5	**SPS 5–20**	**$A_{GRO}F_{RST}$ 0–5**	**$A_{GRO}F_{RST}$ 5–20**
(5) Physical SQI	64.26	64.26	64.26	64.26
(6) Chemical SQI	9.09	1.52	28.79	16.67
(7) Microbiological SQI	41.86	25.58	41.86	23.26
(8) Global SQI	38.41	30.46	44.98	34.74
(9) % SQI	37.9	44.8	31.0	37.9

The number inside each parenthesis corresponds to the equation from which the values were calculated.

Microbiological activity had equivalent SQI at 0–5 depth of both systems, whereby the SI_{NDs} M_N M_{SOC}, MR_{ESP}, and β-GLU had critical status, while the U_{RS} P_{HOSP} was optimum, and the N_{MIN} in FDA was acceptable. Similar trends were observed at the 5–20 depth except for FDA in the $A_{GRO}F_{RST}$, which was in a limited range, while N_{MIN} was critical in both systems.

The critical category common to all conditions could be related to the substrate quality of fresh input organic material. Meanwhile, the optimal category may depict P and N mining, generating a net immobilization process linked to their chemical SI_{ND} concentrations counterparts.

Specifically, the limited FDA activity that was previously discussed indicates a less diverse microbial community that is able to produce proteases, lipases, and esterases (responsible for hydrolyzing the fluorescein diacetate or FDA) [77]. At the same time, critical N_{MIN} at the 5–20 depth may be caused by a combination of lower N concentrations (SI_{ND} N%) and focal compaction generating lower aeration limiting O^+ availability, thus slowing nitrification processes (transformation of NH_4^+ into NO_2^-/NO_3^-) and/or due to higher lignin (SI_{ND} C:N).

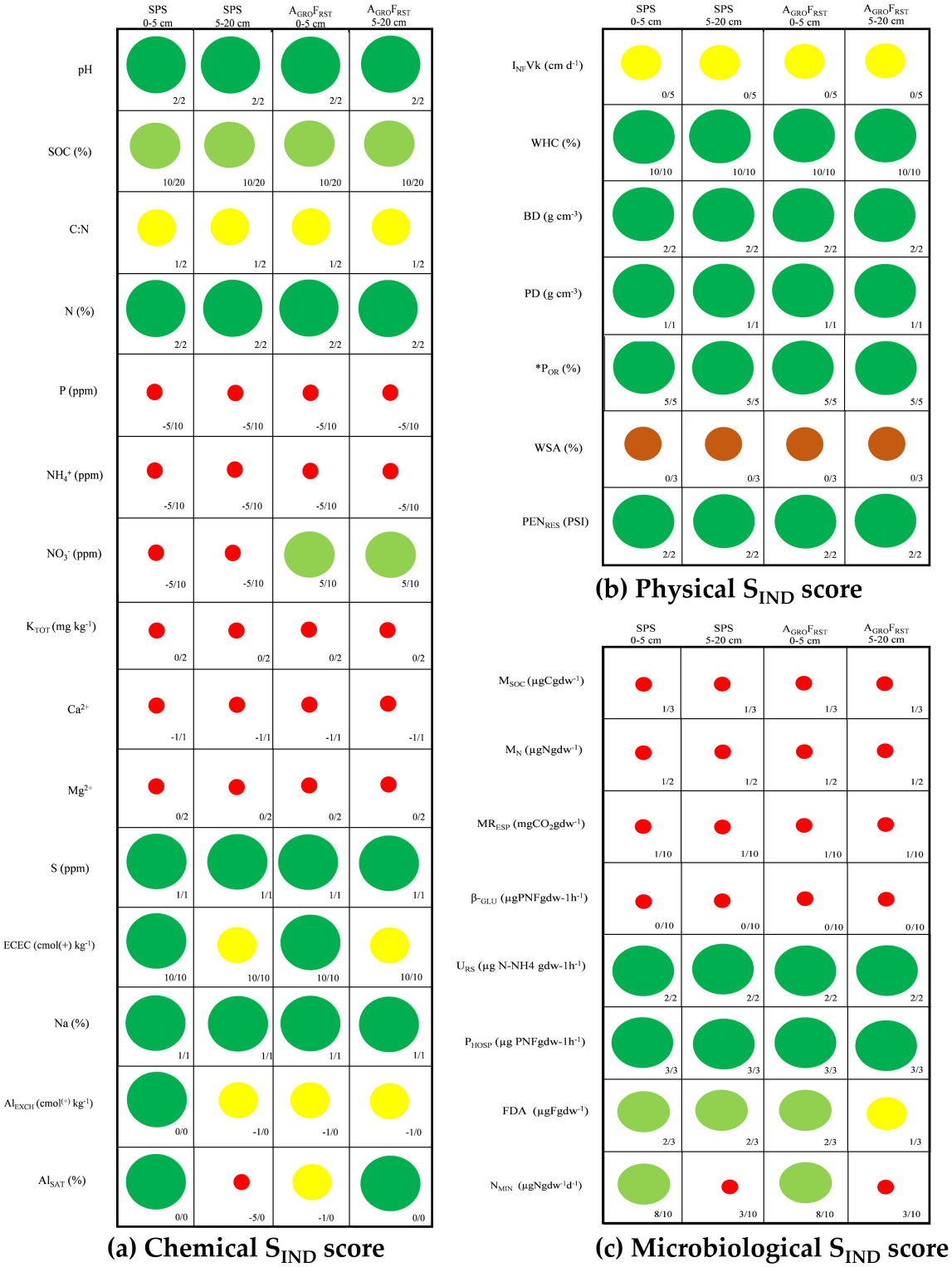

Figure 3. Bi-indicative illustration of SI_{ND} sub-indexes using dots for the estimation of SQI. The combination of each color and diameter range represents the spectrum defined in the Appendix A. Larger diameters and greenish hues correspond to acceptable–optimal ranges (distinguishable by minor to main intensity, respectively), and yellowish tones and medium diameters point out intermediated to limiting soil quality values. Finally, orangey-reddish colors and smaller diameters correspond to undesirable critical levels. The values adjacent to each circle (x/y) symbolize the quality sub-index score (x) and the maximum possible sub-index value (y) (Appendix A). (**a**–**c**) refer to chemical, physical, and microbiological SI_{ND} scores. * P_{OR} = Net pore space.

The estimated overall SQI for the system (0–20 cm) was 31% for the SPS and 37.8 for the $A_{GRO}F_{RST}$, while SQI% reached 41.4% and 31.5%, respectively; this was probably due to a combination of greater leaf-litter production and fertilization practices. However, if fewer SI_{NDs} were in a critical state, a pattern showing higher SQI in the upper depths of both systems was observed, which may be evidence of the positive effects of AFS management–SOC generation.

Broadly, although the general SQI could be explained mainly by native SQ (e.g., distinctive properties of Andisols) and possibly to a lesser extent by the effects of anthropogenic activities, the differences in SOM and most of the SI_{NDs} that were evaluated between the upper and lower depths (both systems) finally resulted in a positive change over the SQI and showed a sensitiveness to AFS management.

Moreover, to establish comparative advantages between AFS and (i) natural regeneration–secondary ecological succession after degradation processes, merely in terms of soil properties, in adjacent plots but outside of our study sites (with the same historical record of pure extractive practices), we estimated the following values: WHC: 40.4/38.6%; WSA 42.63/41.53%; $P_{EN}R$ 200 mm: 300 PSI; and SOC: 11.26/7.63% (0–5/5–20 cm), and (ii) plots with 50-year-old secondary forest SOC: 10.2/9.7%, WSA 52.5/50.8%; WHC: 39.3/34.9% (0–5/5–20 cm), and $P_{EN}R$ 100 mm: 200 PSI (both cases noticeably lower than those observed for the SPS and $A_{GRO}F_{RST}$.)

In addition, a dense undergrowth was observed, dominated almost entirely by *Chusquea quila*, which is considered a species of indirect medical importance (e.g., habitat of the longtailed lobster *Oligoryzomys longicaudatus*, vector of the hantavirus *Bunyaviridae*), also having opportunistic habits, persistence, and low potential nutritional value in addition to the high presence of the locally denominated "blond scorpion" *Brachystostenus negrei*, of specific medical importance.

Although it is beyond the scope of the present work to prove its occurrence and distribution in that site due to natural regeneration, it is possible to state that no specimens were found in any other study sites and that it could be a future line of research [78]. Therefore, it is possible to conclude that under the described scenarios, AFS can improve soil conditions in shorter periods and prevent the massive development of undesirable species.

4. Conclusions

Following the assessment of the proposed 30 SI_{NDs}, it was determined that 79.3% did not exhibit significant variation under any conditions. Specifically, 41% fell within the acceptable–optimal interval, while 6.9% were ranked as intermediate-limiting, and 31.0% were classified as restrictive-critical levels. This analysis revealed an overall SQI (0–20 cm) of 34.5 for the SPS and 41.4% for the $A_{GRO}F_{RST}$. The physical SQI remained within the optimal range, except for two SI_{NDs} (WSA and $I_{NF}Vk$), which were in ranges near the desirable level, ensuring adequate water storage, movement, and root exploration. In contrast, the chemical SQI exhibited high variability and more critical aspects. SI_{NDs} related to cationic nutrients (K^+, Mg^{2+}) and P^+ were found to be at critical levels, reflecting the inherent properties of Andisols and the risk of Al^{3+} toxicity. Deficiencies in assimilable N were attributable to net N immobilization and limited microbial activity, indicating critical levels for the SI_{NDs} M_N, M_{SOC}, and M_{RESP}. These conditions influenced microbial SQI and were influenced by the combined effects of fertilization patterns, substrate quality, and labile C provided by the herbaceous component (e.g., rhizodeposition). The $A_{GRO}F_{RST}$ showed some comparative advantages, but it also displayed a dependency on external inputs. In contrast, positive changes in nutrient status and microbial dynamics were expected to be enhanced in the SPS in the medium-term due to manure contribution. Furthermore, the addition of 5.48 and 5.50 Mg C ha yr^{-1} to the SPS and $A_{GRO}F_{RST}$, respectively, significantly improved virtually all the SI_{NDs} ($p \leq 0.05$) or simply increased their magnitude in the upper depth studied (0–5 cm). This highlights the pivotal role of soil organic matter (SOM) in influencing pedogenic processes and functions. These results align with the growing scientific evidence supporting the recognized capabilities of

agroforestry systems (AFSs) in addressing global challenges such as climate change and food security. Additionally, the complementarity of these systems in forested areas for native forest reclamation–conservation and multi-purpose land management for smallholder agriculture-focused food production was evident. Future research should include the periodic seasonal quantification and characterization of leaf-litter stock input, microbial communities (including critical groups related to metabolic activity), and SOC fractionation to assess the involvement of C stabilization processes and the specific contribution of each C pool to the estimated C_{SEQ}. Additionally, a general C balance should be determined to establish comparative ecological parameters.

Author Contributions: Conceptualization, F.D. and J.O.; methodology, M.P., J.O., F.D., E.Z. and G.C.; software, J.O., P.N., R.R.H.V., A.P.-S.M. and P.M.d.S.C.; validation, E.Z., F.D., M.P., G.C. and N.B.S.; formal analysis, J.O., F.D. and P.N.; investigation, J.O., P.N., A.P.-S.M., P.M.d.S.C. and R.R.H.V.; resources, F.D., M.P., G.C. and E.Z.; data curation, J.O., M.P., P.N., A.P.-S.M., P.M.d.S.C. and R.R.H.V.; writing—original draft preparation, J.O.; writing—review and editing, J.O., P.N., A.P.-S.M., P.M.d.S.C., R.R.H.V. and N.B.S.; visualization, J.O., F.D., E.Z., G.C., M.P. and N.B.S.; supervision, J.O., F.D., M.P., G.C., N.B.S. and E.Z.; project administration, F.D.; funding acquisition, F.D., M.P., G.C., J.O. and E.Z. All authors have read and agreed to the published version of the manuscript.

Funding: This research was funded by: Concurso VRDI Multidiscliplinaria (UdeC), no. 219.142.040-M and CONAF (Fondo de Investigación del Bosque Nativo), no. 001-2014.

Data Availability Statement: Not applicable.

Acknowledgments: We would like to express our sincere thanks to the staff of the Ranchillo Alto Research Center of the Universidad de Concepcion. We thank the Agroecology laboratory of INIA (Quilamapu) and, in particular, Cecilia Céspedes for the support provided for this research, as well as the Soil and Environment Research Laboratory of the Faculty of Agronomy of the University of Concepcion and, in particular, Katherine Rebolledo for her valuable collaboration. This research is part of J.O.'s doctoral thesis, and we thank the University of Concepcion for approving its publication.

Conflicts of Interest: The authors declare no conflict of interest.

Abbreviations

Abbreviation	Description	Abbreviation	Description
AFS	agroforestry system	Mg^{2+}	magnesium
SPS	silvopastoral systems	Na^{+}	sodium
$A_{GRO}F_{RST}$	agroforest	S	sulphur
RA	Ranchillo Alto	Al_{EXCH}	exchangeable Al
Pg	petagrams	$\%Al_{SAT}$	% of Al saturation
C	carbon	ECEC	effective cation exchange capacity
CO_{2eq}	carbon equivalent	pH	soil reactivity
SOM	soil organic matter	PD	particle density
SOC	soil organic carbon	BD	bulk density
C_{SEQ}	carbon sequestration	P_{OR}	total porosity (%)
SQ	soil quality	WSA	% of water stable aggregates
SQ_{CHE}	chemical soil quality	$I_{NF}V$	infiltration velocity
SQ_{PHY}	physical soil quality	WHC	water holding capacity
SQ_{BIOL}	biological soil qualit	$P_{EN}R$	penetration resistance
SQI	soil quality index	M_{BIOMSS}	microbial biomass
SI_{ND}	soil quality indicator	M_{SOC}	microbial biomass C
N	nitrogen	M_{N}	microbial N
C/N	carbon–nitrogen ratio	MR_{ESP}	microbial respiration
NH_4^{+}	ammonium	N_{MIN}	N mineralization
NO_3^{-}	nitrate	β-GLU	β-glucosidase activity
P	phosphorous	U_{RS}	urease activity
K^{+}	potassium	P_{HOSP}	phosphatase activity
Ca^{2+}	calcium	FDA	fluorescein diacetate hydrolysis

Appendix A. Subindices for Each Soil Quality Indicator (SI_{ND}) Considered for the Soil Quality Index (SQI) Estimation

Table A1. Soil quality levels and their associated sub-index values for physical SI_{NDs}.

PHYSICAL SI_{ND}	Level	Interpretation	Subindex	Source
PD (g cm^{-3})	<2	Desirable	1	[64]
	>2	Without effect	0	
BD (g cm^{-3})	<1.10	Optimum	2	[77]
	1.10−1.47	Desirable	1	
	>1.47	Low	0	
P_{OR} (%)	<5	Critical	−5	[79]
	5–10	Restrictive	0	
	10–25	Acceptable	1	
	25–40	Desirable	2	
	>40	Optimum	5	
WHC (%)	>60	Optimum	10	[80]
	51–60	Acceptable	5	
	41–50	Low	0	
	<40	Critical	−10	
$I_{NF}Vk$ (cm day^{-1})	<8.64	Undesirable	−5	[81]
	8.64–20	Acceptable	0	
	20–43.2	Optimum	5	
$P_{EN}R$ (psi)	>300	Undesirable	0	[82,83]
	200–300	Acceptable	1	
	100–200	Optimum	2	
WSA (%)	<50	Undesirable	0	[82]
	50–70	Medium	1	
	70–90	High	2	
	>90	Optimum	3	

Appendix B

Table A2. Soil quality levels and their associated sub-index values for chemical SI_{NDs}.

CHEMICAL SI_{ND}	Level	Interpretation	Sub-Index	Source
pH	<3.0	Super critical	−1	[84]
	3.01–4.0	Critical	0	
	4.01–5.5	Limiting	1	
	5.51–6.8	Desirable	2	
	6.81–7.2	Optimum	2	
	7.21–7.5	Acceptable	1	
	7.51–8.5	Limiting	1	
	>8.5	Critical	0	
SOC (%)	>15	Excellent	20	[80]
	5–15	High	10	
	3–5	Moderate	1	
	<2	Low	−10	
N (%)	>0.5	Desirable	2	[84]
	0.1–0.5	Adequate	1	
	<0.1	Insufficient	0	
NO_3^- (mg kg^{-1})	<10	Critical	−5	[80]
	10–20.1	Insufficient	0	
	20.1–40	Adequate	5	
	>40	Desirable	10	

Table A2. *Cont.*

CHEMICAL SI_{ND}	Level	Interpretation	Sub-Index	Source
NH_4^+ (mg kg^{-1})	<25 25–50 51–75 >75	Critical Insufficient Adequate Desirable	−5 0 5 10	[80]
C:N ratio	1–10 10–20 >20	Adequate Moderate Insufficient	2 1 0	[84]
P (mg kg^{-1})	>15 5–15 <5	Adequate Moderate Insufficient	10 1 −5	[62]
K (mg kg^{-1})	>500 100–500 <100	Adequate Moderate Insufficient	2 1 0	[84]
S (mg kg^{-1})	>100 1–100 <1	Insufficient Adequate Insufficient	0 1 0	[84]
Ca (mg kg^{-1})	>1000 101–1000 10–100 <10	Desirable Adequate Insufficient Critical	2 1 0 −1	[84]
Mg (mg kg^{-1})	>500 50–500 <50	Adequate Moderate Insufficient	2 1 0	[84]
ECEC (cmol kg^{-1})	>6.27 1.65–6.27 <1.65	Adequate Moderate Insufficient	10 5 0	[75,85]
Exchangeable % Na	<15 ≤15	Critical Acceptable	−10 1	[84]
Al_{EXCH} (cmol kg^{-1})	<0.1 0.11–0.51 0.51–0.81 >0.81	Adequate Moderate Undesirable Critical	0 −1 −2 −3	[85]
Sat Al (%)	1.1–3.1 3.2–6.1 6.2–12 >12	Adequate Moderate High Critical	0 −1 −2 −5	[85]

Appendix C

Table A3. Soil quality levels and their associated sub-index values for biological SI_{NDs}.

BIOLOGICAL SI_{ND}	Level	Interpretation	Subindex	Source
M_N (µg N g dw^{-1})	>4067 <4067	Desirable Undesiderable	2 1	[86]
M_{SOC} (µg C g d w^{-1})	>28,608 2814–28,608 <2814	Adequate Moderate Low	3 2 1	[87]

Table A3. *Cont.*

BIOLOGICAL SI_{ND}	Level	Interpretation	Subindex	Source
MR_{ESP} (mg CO_2 g dw^{-1})	<0.3	Critical	1	[88]
	0.3–0.5	Restrictive	3	
	0.5–0.65	Limited	5	
	0.65–0.85	Desirable	8	
	>0.85	Optimum	10	
N mineralization N-min N (μg N kg dw^{-1})	<9	Critical	1	[89]
	9–13	Restrictive	3	
	13–17	Limited	5	
	17–21	Desirable	8	
	>21	Optimum	10	
β-$_{GLU}$ (μg PNF g dw^{-1} h^{-1})	<14,304	Undesirable	0	[88]
	14,304–28,608	Acceptable	5	
	>28,608	Optimum	10	
U_{RS} (μg N-NH_4 g dw^{-1} h^{-1})	<28	Undesirable	0	[89,90]
	28–560	Acceptable	1	
	>560	Optimum	2	
P_{HOSP} (μg PNF g dw^{-1} h^{-1})	<60	Undesirable	0	[91]
	60–170	Medium	1	
	>170	Optimum	3	
FDA (μg F g dw^{-1})	>66	Optimum	3	[60]
	50–66	Adecuate	2	
	33-50	Average	1	
	<33	Limited	0	

References

1. Poore, J.; Nemecek, T. Reducing food's environmental impacts through producers and consumers. *Science* **2018**, *360*, 987. [CrossRef] [PubMed]
2. Mbow, H.O.P.; Reisinger, A.; Canadell, J.; O'Brien, P. *Food Security in Climate Change and Land in: Special Report on Climate Change and Land*; Cambridge University Press: Cambridge, UK; New York, NY, USA, 2019; p. 32. Available online: https://www.ipcc.ch/site/assets/uploads/2018/07/sr2_background_report_final.pdf (accessed on 18 January 2023).
3. Rosenzweig, C.; Mbow, C.; Barioni, L.G.; Benton, T.G.; Herrero, M.; Krishnapillai, M.; Liwenga, E.T.; Pradhan, P.; Rivera-Ferre, M.G.; Sapkota, T.; et al. Climate change responses benefit from a global food system approach. *Nat. Food* **2020**, *1*, 94–97. [CrossRef] [PubMed]
4. Crippa, M.; Solazzo, E.; Guizzardi, D.; Monforti-Ferrario, F.; Tubiello, F.N.; Leip, A. Food systems are responsible for a third of global anthropogenic GHG emissions. *Nat. Food* **2021**, *2*, 198–209. [CrossRef] [PubMed]
5. Giraldo, C.; Escobar, F.; Chará, J.D.; Calle, Z. The adoption of silvopastoral systems promotes the recovery of ecological processes regulated by dung beetles in the Colombian Andes: Ecological processes regulated by dung beetles. *Insect Conserv. Divers.* **2011**, *4*, 115–122. [CrossRef]
6. Gerber, P.J.; Hristov, A.N.; Henderson, B.; Makkar, H.; Oh, J.; Lee, C.; Meinen, R.; Montes, F.; Ott, T.; Firkins, J.; et al. Technical options for the mitigation of direct methane and nitrous oxide emissions from livestock: A review. *Animal* **2013**, *7*, 220–234. [CrossRef]
7. FAO. *Livestock's Long Shadow: Environmental Issues and Options*; Steinfeld, H., Ed.; Food and Agriculture Organization of the United Nations: Rome, Italy, 2006; p. 390.
8. O'Mara, F.P. The role of grasslands in food security and climate change. *Ann. Bot.* **2012**, *110*, 1263–1270. [CrossRef]
9. Sarabia, L.; Solorio, F.J.; Ramírez, L.; Ayala, A.; Aguilar, C.; Ku, J.; Almeida, C.; Cassador, R.; Alves, B.J.; Boddey, R.M. Improving the Nitrogen Cycling in Livestock Systems through Silvopastoral Systems. In *Nutrient Dynamics for Sustainable Crop Production*; Meena, R.S., Ed.; Springer Singapore: Singapore, 2020; pp. 189–213. [CrossRef]
10. Li, B.V.; Jiang, B. Responses of forest structure, functions, and biodiversity to livestock disturbances: A global meta-analysis. *Glob. Chang. Biol.* **2021**, *27*, 4745–4757. [CrossRef]

11. Sollins, P.; Homann, P.; Caldwell, B.A. Stabilization and destabilization of soil organic matter: Mechanisms and controls. *Geoderma* **1996**, *74*, 65–105. [CrossRef]
12. Amerongen-Madison, J.; Bulmer, C.; Trofymow, T.; Prescott, C.; Wallace, B.; Philpott, T.; Dymond, C.; Fredeen, A. Soil Carbon in Forest Ecosystems: Pools and Processes. Forest Carbon Initiative. 2021. Available online: https://a100.gov.bc.ca/pub/eirs/finishDownloadDocument.do;jsessionid=8F62A9315428CC8F653587E211134F7F?subdocumentId=21641 (accessed on 18 January 2023).
13. Blanco-Canqui, H.; Lal, R. *Principles of Soil Conservation and Management*; Springer: Berlin/Heidelberg, Germany, 2008; p. 617.
14. Juhos, K.; Madarász, B.; Kotroczó, Z.; Béni, Á.; Makádi, M.; Fekete, I. Carbon sequestration of forest soils is reflected by changes in physicochemical soil indicators—A comprehensive discussion of a long-term experiment on a detritus manipulation. *Geoderma* **2021**, *385*, 114918. [CrossRef]
15. Lorenz, K.; Lal, R. Soil organic carbon sequestration in agroforestry systems. A review. *Agron. Sustain. Dev.* **2014**, *34*, 443–454. [CrossRef]
16. Nair PK, R. Carbon sequestration studies in agroforestry systems: A reality-check. *Agrofor. Syst.* **2012**, *86*, 243–253. [CrossRef]
17. Ortiz, J.; Neira, P.; Panichini, M.; Curaqueo, G.; Stolpe, N.B.; Zagal, E.; Dube, F.; Gupta, S.R. Silvopastoral Systems on Degraded Lands for Soil Carbon Sequestration and Climate Change Mitigation. In *Agroforestry for Sustainable Intensification of Agriculture in Asia and Africa*; Dagar, J.C., Gupta, S.R., Sileshi, G.W., Eds.; Sustainability Sciences in Asia and Africa; Springer Nature Singapore: Singapore, 2023; pp. 207–242. Available online: https://link.springer.com/10.1007/978-981-19-4602-8_7 (accessed on 18 January 2023).
18. Somarriba, E.; Beer, J.; Alegre-Orihuela, J.; Andrade, H.J.; Cerda, R.; DeClerck, F.; Detlefsen, G.; Escalante, M.; Giraldo, L.A.; Ibrahim, M.; et al. Mainstreaming Agroforestry in Latin America. In *Agroforestry—The Future of Global Land Use*; Nair, P.K.R., Garrity, D., Eds.; Springer: Dordrecht, The Netherlands, 2012; Volume 9, pp. 429–453. [CrossRef]
19. Alonso, J. Silvopastoral systems and their contribution to the environment. *Cuba. J. Agric. Sci.* **2011**, *45*, 9.
20. Udawatta, R.P.; Jose, S. Carbon Sequestration Potential of Agroforestry Practices in Temperate North America. In *Carbon Sequestration Potential of Agroforestry Systems*; Kumar, B.M., Nair, P.K.R., Eds.; Springer: Dordrecht, The Netherlands, 2011; Volume 8, pp. 17–42. [CrossRef]
21. Feliciano, D.; Ledo, A.; Hillier, J.; Nayak, D.R. Which agroforestry options give the greatest soil and above ground carbon benefits in different world regions? *Agric. Ecosyst. Environ.* **2018**, *254*, 117–129. [CrossRef]
22. Karlen, D.L.; Mausbach, M.J.; Doran, J.W.; Cline, R.G.; Harris, R.F.; Schuman, G.E. Soil Quality: A Concept, Definition, and Framework for Evaluation (A Guest Editorial). *Soil Sci. Soc. Am. J.* **1997**, *61*, 4–10. [CrossRef]
23. Arshad, M.A.; Martin, S. Identifying critical limits for soil quality indicators in agro-ecosystems. *Agric. Ecosyst. Environ.* **2002**, *88*, 153–160. [CrossRef]
24. Doran, J.W.; Jones, A.J. (Eds.) *Methods for Assessing Soil Quality*; Soil Science Society of America: Madison, WI, USA, 1997. [CrossRef]
25. Muñoz-Rojas, M. Soil quality indicators: Critical tools in ecosystem restoration. *Curr. Opin. Environ. Sci. Health* **2018**, *5*, 47–52. [CrossRef]
26. Tale, K.S.; Ingole, S. A review on role of physico-chemical properties in soil quality. *Chem. Sci. Rev. Lett.* **2015**, *4*, 57–66.
27. Li, J.; Nie, M.; Powell, J.R.; Bissett, A.; Pendall, E. Soil physico-chemical properties are critical for predicting carbon storage and nutrient availability across Australia. *Environ. Res. Lett.* **2020**, *15*, 094088. [CrossRef]
28. Karlen, D.L.; Andrews, S.S.; Doran, J.W. Soil quality: Current concepts and applications. In *Advances in Agronomy*; Elsevier: Amsterdam, The Netherlands, 2001; Volume 74, pp. 1–40. [CrossRef]
29. Drobnik, T.; Greiner, L.; Keller, A.; Grêt-Regamey, A. Soil quality indicators–From soil functions to ecosystem services. *Ecol. Indic.* **2018**, *94*, 151–169. [CrossRef]
30. Flores, J.P.; Martínez, E.; Espinosa, M.; Ahumada, I.; Avendaño, P.; Henríquez, G.; Torres, P. Determinación de la erosión actual y potencial de los suelos de Chile: Región de La Araucanía, 2010. Síntesis de Resultados (Pub. CIREN N° 149). Available online: https://bibliotecadigital.ciren.cl/handle/20.500.13082/2133 (accessed on 15 January 2023).
31. Casanova, M.; Salazar, O.; Seguel, O.; Luzio, W. Human-Induced Soil Degradation in Chile. In *The Soils of Chile*; Casanova, M., Salazar, O., Seguel, O., Luzio, W., Eds.; Springer: Dordrecht, The Netherlands, 2013; pp. 121–158. [CrossRef]
32. Lara, A.; Solari, M.E.; Prieto MD, R.; Peña, M.P. Reconstrucción de la cobertura de la vegetación y uso del suelo hacia 1550 y sus cambios a 2007 en la ecorregión de los bosques valdivianos lluviosos de Chile (35°—43°30′ S). *Bosque (Valdivia)* **2012**, *33*, 3–4. [CrossRef]
33. Schmidt, A.; Alonso, V.; Schmidt, H. *Manejo Silvopastoril de los Bosques de Ñirre en la XII Región de Magallanes*; Fondo de Investigación del Bosque Nativo: Santiago, Chile, 2013; p. 63. Available online: https://investigacion.conaf.cl/archivos/repositorio_documento/2018/10/Informe-final-029-2013_final_observaciones.pdf (accessed on 18 January 2023).
34. Neira, P.; Henríquez-Castillo, C.; Ortiz, J.; Stolpe, N.; Dube, F. Do different densities of tree cover affect pasture biomass and soil microbial communities? *Agrofor. Syst.* **2021**, *95*, 1465–1478. [CrossRef]
35. Sotomayor, A.; Schmidt, H.; Salinas, J.; Schmidt, A.; Sánchez-Jardón, L.; Alonso, M.; Moya, I.; Teuber, O. Silvopastoral Systems in the Aysén and Magallanes Regions of the Chilean Patagonia. In *Silvopastoral Systems in Southern South America*; Peri, P.L., Dube, F., Varella, A., Eds.; Springer International Publishing: Berlin/Heidelberg, Germany, 2016; Volume 11, pp. 213–230. [CrossRef]

36. Ortiz, J.; Dube, F.; Neira, P.; Panichini, M.; Stolpe, N.B.; Zagal, E.; Martínez-Hernández, P.A. Soil Quality Changes within a (*Nothofagus obliqua*) Forest Under Silvopastoral Management in the Andes Mountain Range, South Central Chile. *Sustainability* **2020**, *12*, 6815. [CrossRef]
37. Dube, F.; Thevathasan, N.V.; Zagal, E.; Gordon, A.M.; Stolpe, N.B.; Espinosa, M. Carbon Sequestration Potential of Silvopastoral and Other Land Use Systems in the Chilean Patagonia. In *Carbon Sequestration Potential of Agroforestry Systems*; Kumar, B.M., Nair, P.K.R., Eds.; Springer: Dordrecht, The Netherlands, 2011; Volume 8, pp. 101–127. [CrossRef]
38. Dube, F.; Sotomayor, A.; Loewe, V.; Müller-Using, B.; Stolpe, N.; Zagal, E.; Doussoulin, M. Silvopastoral Systems in Temperate Zones of Chile. In *Silvopastoral Systems in Southern South America*; Peri, P.L., Dube, F., Varella, A., Eds.; Springer International Publishing: Berlin/Heidelberg, Germany, 2016; Volume 11, pp. 183–211. [CrossRef]
39. Dube, F.; Stolpe, N.B.; Zagal, E.; Figueroa, C.R.; Concha, C.; Neira, P.; Carrasco, C.; Schwenke, J.M.; Schwenke, V.; Müller-Using, B. Novel agroforestry systems in temperate Chile. In *Temperate Agroforestry Systems*, 2nd ed.; Gordon, A.M., Newman, S.M., Coleman, B.R.W., Eds.; CABI: Wallingford, UK, 2018; pp. 237–251. [CrossRef]
40. Leal, F.; Aburto, F.; Aguilera, N.; Echeverría, C.; Gatica-Saavedra, P. Forest degradation modifies litter production, quality, and decomposition dynamics in Southern temperate forests. *Front. Soil Sci.* **2023**, *3*, 1111694. [CrossRef]
41. United States Department of Agriculture (USDA). *Soil Survey Staff: Keys to Soil Taxonomy*; USDA: Washington, DC, USA, 2014.
42. Stolpe, N. *Descripciones de los Principales Suelos de la VIII Región de Chile*; Department of Soils and Natural Resources, Faculty of Agronomy, Universidad de Concepción: Chillán, Chile, 2006.
43. Besoain, E.; Sepúlveda, G. Minerales secundarios. In *Suelos Volcánicos de Chile*; Tosso, J., Ed.; INIA: Santiago, Chile, 1985; pp. 153–214.
44. CIREN (Centro de Información de Recursos Naturales). Descripciones de suelos, materiales y símbolos. In *Estudio Agrológico VIII Región*; CIREN: Santiago, Chile, 1999; Volume 1, p. 121.
45. Blake, G.R.; Hartge, K.H. Particle Density 1. In *Methods of Soil Analysis: Part 1—Physical and Mineralogical Methods*, 2nd ed.; Klute, A., Ed.; American Society of Agronomy, Soil Science Society of America Inc.: Madison, WI, USA, 1986; pp. 377–382. [CrossRef]
46. Kemper, W.D.; Rosenau, R.C. Aggregate Stability and Size Distribution. In *SSSA Book Series*; Klute, A., Ed.; Soil Science Society of America, American Society of Agronomy: Madison, WI, USA, 2018; pp. 425–442. [CrossRef]
47. Zhang, R. Determination of Soil Sorptivity and Hydraulic Conductivity from the Disk Infiltrometer. *Soil Sci. Soc. Am. J.* **1997**, *61*, 1024–1030. [CrossRef]
48. Zagal, E.; Longeri, L.; Vidal, I.; Hoffman, G.; González, R. Influencia de la adición de nitrógeno y fosforo sobre la descomposición de paja de trigo en un suelo derivado de cenizas volcánicas. *Agric. Técnica* **2003**, *63*, 403–415. [CrossRef]
49. Sadzawka, A.; Carrasco, M.A.; Grez, R.; Mora, M.L.; Flores, H.; Neaman, A. *Métodos de Análisis Recomendados para los Suelos Chilenos*; Comisión de Normalización y Acreditación, Sociedad Chilena de la Ciencia del Suelo: Santiago, Chile, 2006; p. 113.
50. Wright, A.F.; Bailey, J.S. Organic carbon, total carbon, and total nitrogen determinations in soils of variable calcium carbonate contents using a Leco CN-2000 dry combustion analyzer. *Commun. Soil Sci. Plant Anal.* **2001**, *32*, 3243–3258. [CrossRef]
51. Anderson JP, E.; Domsch, K.H. A physiological method for the quantitative measurement of microbial biomass in soils. *Soil Biol. Biochem.* **1978**, *10*, 215–221. [CrossRef]
52. Jenkinson, D.S.; Brookes, P.C.; Powlson, D.S. Measuring soil microbial biomass. *Soil Biol. Biochem.* **2004**, *36*, 5–7. [CrossRef]
53. Horwath, W.R.; Paul, E.A. Microbial Biomass. In *SSSA Book Series*; Weaver, R.W., Angle, S., Bottomley, P., Bezdicek, D., Smith, S., Tabatabai, A., Wollum, A., Eds.; Soil Science Society of America: Madison, WI, USA, 2018; pp. 753–773. [CrossRef]
54. Linn, D.M.; Doran, J.W. Effect of Water-Filled Pore Space on Carbon Dioxide and Nitrous Oxide Production in Tilled and Nontilled Soils. *Soil Sci. Soc. Am. J.* **1984**, *48*, 1267–1272. [CrossRef]
55. Alef, K. Nitrogen mineralization in soils. In *Methods in Applied Soil Microbiology and Biochemistry*; Alef, K., Nannipieri, P., Eds.; Academic Press: London, UK, 1995; pp. 234–245.
56. Tabatabai, M.A. Soil Enzymes. In *SSSA Book Series*; Weaver, R.W., Angle, S., Bottomley, P., Bezdicek, D., Smith, S., Tabatabai, A., Wollum, A., Eds.; Soil Science Society of America: Madison, WI, USA, 1994; pp. 775–833. [CrossRef]
57. Kandeler, E.; Poll, C.; Frankenberger Jr, W.T.; Tabatabai, M.A. Nitrogen cycle enzymes. In *Methods in Soil Enzymology*; Dick, R.P., Ed.; Soil Science Society of America Book Series; Soil Science Society of America: Madison, WI, USA, 2011; Volume 9, pp. 211–244.
58. Tabatabai, M.A.; Bremner, J.M. Use of p-nitrophenyl phosphate for assay of soil phosphatase activity. *Soil Biol. Biochem.* **1969**, *1*, 301–307. [CrossRef]
59. Bello, D.; García-Carballal, S.; Muiño, F.; Salgado, J.; Trasar-Cepeda, C. A pH-adjusted method to measure urease activity in soils spiked with tetrafluoroborate 1-butyl-3-methylimidazolium [BMIM][BF4]. In Proceedings of the 20th International Electronic Conference on Synthetic Organic Chemistry, Santiago de Compostela, Online, 1–30 November 2016. [CrossRef]
60. Alef, K. Estimation of the hydrolysis of fluorescein diacetate. In *Methods in Applied Soil Microbiology and Biochemistry*; Alef, K., Nannipieri, P., Eds.; Academic Press: London, UK, 1995; pp. 232–233.
61. Green, V.S.; Stott, D.E.; Diack, M. Assay for fluorescein diacetate hydrolytic activity: Optimization for soil samples. *Soil Biol. Biochem.* **2006**, *38*, 693–701. [CrossRef]
62. Amacher, M.C.; O'Neil, K.P.; Perry, C.H. *Soil Vital Signs: A New Soil Quality Index (SQI) for Assessing Forest Soil Health*; RMRS-RP-65; U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: Washington, DC, USA, 2007; p. RMRS-RP-65. [CrossRef]
63. Dube, F.; University of Concepción, Chillán, Región del Bio Bio, Chile. Personal communication, 2019.

64. Nanzyo, M. Unique properties of volcanic ash soils. *Glob. Environ. Res.-Eng. Ed.* **2002**, *6*, 99–112.
65. Panichini, M.; INIA Carillanca, Temuco, Región de La Araucanía, Chile. Personal communication, 2020.
66. Nissen, J.; Quiroz, C.; Seguel, O.; Roberto Mac Donald, H.; Sch, A.E. Variación del potencial mátrico durante el movimiento de agua en andisoles. *Agro Sur.* **2005**, *33*, 36–47. [CrossRef]
67. Zadzawka, A.; Carrasco, A. *Quimica de los Suelos Volcanicos*; Biblioteca Digital INIA: Santiago, Chile, 1985; Available online: https://biblioteca.inia.cl/handle/123456789/26667 (accessed on 18 January 2023).
68. Qi, K.; Pang, X.; Yang, B.; Bao, W. Soil carbon, nitrogen and phosphorus ecological stoichiometry shifts with tree species in subalpine plantations. *PeerJ* **2020**, *8*, e9702. [CrossRef] [PubMed]
69. Schroth, A.W.; Bostick, B.C.; Graham, M.; Kaste, J.M.; Mitchell, M.J.; Friedland, A.J. Sulfur species behavior in soil organic matter during decomposition. *J. Geophys. Res. Biogeosci.* **2007**, *112*, G04011. [CrossRef]
70. Jones, C.A. Estimation of percent aluminum saturation from soil chemical data. *Commun. Soil Sci. Plant Anal.* **1984**, *15*, 327–335. [CrossRef]
71. Alfaro, M.; Dube, F.; Zagal, E. The Influence of Overmature, Degraded *Nothofagus* Forests with Strong Anthropic Disturbance on the Quality of an Andisol and its Gradual Recovery with Silvopasture in Southwestern South America. In *Agroforestry for Degraded Landscapes*; Dagar, J.C., Gupta, S.R., Teketay, D., Eds.; Springer: Singapore, 2020; pp. 67–85. [CrossRef]
72. Decker KL, M.; Boerner RE, J. Elevation and vegetation influences on soil properties in Chilean *Nothofagus* forests. *Rev. Chil. Hist. Nat.* **2003**, *76*, 371–381. [CrossRef]
73. Schnürer, J.; Rosswall, T. Fluorescein diacetate hydrolysis as a measure of total microbial activity in soil and litter. *Appl. Environ. Microbiol.* **1982**, *43*, 1256–1261. [CrossRef]
74. Reyes, F.; Lillo, A.; Ojeda, N.; Reyes, M.; Alvear, M. Efecto de la exposición y la toposecuencia sobre actividades biológicas del suelo en bosque relicto del centro-sur de Chile. *Bosque (Valdivia)* **2011**, *32*, 255–265. [CrossRef]
75. Muñoz, C.; Torres, P.; Alvear, M.; Zagal, E. Physical protection of C and greenhouse gas emissions provided by soil macroaggregates from a Chilean cultivated volcanic soil. *Acta Agric. Scand. Sect. B—Soil Plant Sci.* **2012**, *62*, 739–748. [CrossRef]
76. Casanova, M.; Salazar, O.; Seguel, O.; Luzio, W. (Eds.) Main Features of Chilean Soils. In *The Soils of Chile*; Springer: Dordrecht, The Netherlands, 2013; pp. 25–97. [CrossRef]
77. Prosser, J.A.; Speir, T.W.; Stott, D.E. Soil Oxidoreductases and FDA Hydrolysis. In *SSSA Book Series*; Dick, R.P., Ed.; American Society of Agronomy, Crop Science Society of America, and Soil Science Society of America: Madison, WI, USA, 2015; pp. 103–124. [CrossRef]
78. Ortiz, J.; University of Concepción, Chillán, Región del Bio Bio, Chile. Unpublished work. 2021.
79. Arshad, M.A.C.; Lowery, B.; Grossman, B. *Physical Tests for Monitoring Soil Quality*; Doran, E.J.W., Jones, A.J., Eds.; SSSA Special Publications (Soil Science Society of America): Madison, WI, USA, 2015; pp. 123–141. [CrossRef]
80. Pagliai, M.; Vignozzi, N. The soil pore system as an indicator of soil quality. *Adv. Geoecol.* **2002**, *35*, 69–80.
81. Vidal, I. *Fertirrigación Cultivos y Frutales*; Publications of Department of Soils and Natural Resources, Faculty of Agronomy, Universidad de Concepción: Chillan, Chile, 2007; Volume 1, p. 118.
82. Reynolds, W.; Yang, X.; Drury, C.; Zhang, T.; Tan, C. Effects of selected conditioners and tillage on the physical quality of a clay loam soil. *Can. J. Soil Sci.* **2003**, *83*, 381–393. [CrossRef]
83. Lal, R.; Elliot, W. Erodibility and Erosivity. In *Soil Erosion Research Methods*, 2nd ed.; Soil and Water Conservation Society (U.S.), Ed.; St. Lucie Press: Boca Raton, FL, USA, 1994; p. 340. [CrossRef]
84. Carter, M.R. Soil Quality for Sustainable Land Management: Organic Matter and Aggregation Interactions that Maintain Soil Functions. *Agron. J.* **2002**, *94*, 38–47. [CrossRef]
85. Villarroel, R.B. *Diagnóstico de la Fertilidad del Suelo*; Instituto de Investigaciones Agropecuarias–Centro Regional de Investigación Remehue Serie Acta: Osorno, Chile, 2000; p. 71.
86. Villaroel, R. *Análisis de Suelo, Metodología e Interpretación*; INIA: Osorno, Chile, 1989; p. 79.
87. Xu, X.; Thornton, P.E.; Post, W.M. A global analysis of soil microbial biomass carbon, nitrogen and phosphorus in terrestrial ecosystems: Global soil microbial biomass C, N and P. *Glob. Ecol. Biogeogr.* **2013**, *22*, 737–749. [CrossRef]
88. Moebius-Clune, B.N.; Moebius-Clune, D.J.; Gugino, B.K.; Idowu, O.J.; Schindelbeck, R.R.; Ristow, A.J.; van Es, H.M.; Thies, J.E.; Shayler, H.A.; McBride, M.B.; et al. *Comprehensive Assessment of Soil Health: The Cornell Framework Manual*; Cornell University: Ithaca, NY, USA, 2016; Edition 3.1; Available online: https://www.css.cornell.edu/extension/soil-health/manual.pdf (accessed on 18 January 2023).
89. Stott, D.E.; Andrews, S.S.; Liebig, M.A.; Wienhold, B.J.; Karlen, D.L. Evaluation of β-Glucosidase Activity as a Soil Quality Indicator for the Soil Management Assessment Framework. *Soil Sci. Soc. Am. J.* **2010**, *74*, 107–119. [CrossRef]
90. Nannipieri, P.; Kandeler, E.; Ruggiero, P. Enzyme Activities and Microbiological and Biochemical Processes in Soil. In *Enzymes in the Environment*; Burns, R., Dick, R., Eds.; CRC Press: Boca Raton, FL, USA, 2002; Volume 84, pp. 1–34. [CrossRef]
91. Maulood, P.; Darwesh, D. Soil Quality Index Models for Assessing WalnutOrchards in Northern Erbil Province, Iraq. *Pol. J. Environ. Stud.* **2020**, *29*, 1275–1285. [CrossRef] [PubMed]