Giltrap, D.L., Kirschbaum, M.U.F., Laubach, J., Hunt, J.E., 2019. Modelling the impacts of irrigation on carbon balances in a grazed dairy pasture. In: *Nutrient loss mitigations for compliance in agriculture*. (Eds L.D. Currie and C.L. Christensen). http://flrc.massey.ac.nz/publications.html. Occasional Report No. 32. Fertilizer and Lime Research Centre, Massey University, Palmerston North, New Zealand. 11 pages.

# MODELLING THE IMPACTS OF IRRIGATION ON CARBON BALANCE IN A GRAZED DAIRY PASTURE

**Donna L. Giltrap[1], Miko U.F. Kirschbaum[1], Johannes Laubach[2], John E. Hunt[2]**

*[1]Manaaki Whenua – Landcare Research, Palmerston North*
*[2]Manaaki Whenua – Landcare Research, Lincoln*
*Email:GiltrapD@landcareresearch.co.nz*

**Abstract**

In New Zealand, increasing areas of dryland farming are being converted to irrigated farming. There are conflicting findings whether this will lead to gains or losses of soil organic carbon (SOC). In this study, we used 2 years of eddy covariance data from an irrigated, grazed dairy pasture in Canterbury and compared observed gas exchange fluxes with those from the process-based CenW model.

We found that gross primary production (GPP) was substantially reduced after grazing, with rates ~30 kgC ha$^{-1}$ d$^{-1}$ lower than the expected reduction due to leaf area loss alone. GPP then gradually recovered over the subsequent 2 weeks. We therefore added an empirical modification to CenW to describe this post-grazing GPP reduction.

Simulations of the modified model agreed very well with observations for evapotranspiration, GPP, and ecosystem respiration rates with model efficiencies ~0.8–0.9 for daily and weekly mean values. Model efficiency for net ecosystem productivity was ~0.7–0.8 for daily and weekly averages.

CenW was then used to compare SOC levels after 50 years of irrigated dairy farming with unirrigated systems. However, one cannot assess the effect of irrigation in isolation. Changes in water availability are inevitably accompanied by changes in fertiliser use and grazing regimes. We dealt with changing nutrient needs and feed availability by using automated routines to apply fertiliser and initiate grazing in response to grassland conditions. For different scenarios, we used conditions to initiate grazing that were set to values that were typical for either cattle grazing or sheep grazing. We then ran three scenarios: dairy-grazed systems with and without irrigation, and sheep-grazing without irrigation.

After 50 years, the irrigated dairy system had only 4 ± 3 tC ha$^{-1}$ (mean ± se) more SOC than the unirrigated dairy system. However, under these scenario conditions, the unirrigated system had unrealistically low feed production (<0.1 tC ha$^{-1}$ y$^{-1}$ in animal products) as the grazing threshold was rarely reached. When we instead compared unirrigated sheep-grazed systems with irrigated cattle-grazed systems, SOC was 13.9 ± 1.5 tC ha$^{-1}$ lower under the unirrigated sheep system than under the irrigated cattle system. The mean C removed in animal products was 0.67 ± 0.02, 0.06 ± 0.02, and 0.32 ± 0.04 tC ha$^{-1}$ y$^{-1}$ for the irrigated dairy, unirrigated dairy, and unirrigated sheep-grazed systems, respectively.

## Introduction

Soil organic carbon (SOC) is important for maintaining soil health. In addition, changes in SOC lead to the addition or removal of $CO_2$ from the atmosphere. Increasing areas of dryland farming are being converted to irrigated farming and there are questions whether this will lead to gains or losses of SOC. On one hand, Laubach and Hunt (2018) observed significant carbon gain of about 1 tC $ha^{-1}$ $y^{-1}$ from an irrigated grazed grassland site based on continuous measurements of net $CO_2$ exchange over 3 years, combined with measurements of biomass removal by grazing. However, in a broader survey of 34 paired sites around New Zealand, Mudge et al. (2017) found that at most, but not all, irrigated sites, the C stocks within the top 0.3 m of soil were lower than at the adjacent unirrigated sites. Previous modelling work had shown that this trend can depend amongst other factors on the total amount available to plants before and after irrigation application (Kirschbaum et al., 2017).

Process-based modelling is an approach that can help better understand the key interacting factors influencing SOC change. In this study, we used the CenW model (Kirschbaum 1999) to simulate C pools and fluxes from an intensively-grazed, irrigated dairy farm. We tested the performance of this model against the carbon-balance dataset of Laubach and Hunt (2018). The model was then applied to estimate the 50-year impact of irrigation on SOC storage by comparing changes in SOC under an irrigated and two unirrigated scenarios. In the first unirrigated scenario, we assumed the same grazing management as for the irrigated scenario to illustrate a typical dairy-grazed production system while the other unirrigated scenario used a less intensive grazing regime more typical of sheep grazing as would be the more normal system used in this environment if no irrigation was available.

## Methods

### *Site description*

The site was located 50 km west of Christchurch (lat. 43.593°S, long. 171.929°E, elevation above sea level 204 m). The paddocks were predominantly perennial ryegrass (*Lolium perenne* L.) and white clover (*Trifolium repens* L.). The soil was a Lismore silty loam (Hewitt, 2010) and moderately stony (12% in top 100 mm, with stone content increasing with depth). In April 2013 (the first sampling year), the top 100 mm had a C content of 2.48 kgC $m^{-2}$ (Laubach and Hunt, 2018).

During the years of measurements, the farm operated with about 900 dairy cows (Friesian-Jersey crossbreds), managed in two herds over a total irrigated grassland area of 328 ha. The farm produced ~4 ML $y^{-1}$ milk, with the milking season running from mid- or late September through to late May. During the milking season, dairy cows were rotationally grazed on the farm’s paddocks. They were removed from the paddocks twice daily to be milked (Laubach and Hunt 2018). Detailed records of the grazing schedule, fertiliser application, and feed supplements were available from farm records. Irrigation was measured with rain gauges.

A centre-pivot irrigation system (890 m in length) supplied water to a circular area that was divided into 19 equal sized sectors. The measurement site was located within one of these sectors and, during grazing events, was surrounded by grazing cows. During the warmer seasons (Oct/Nov to Mar/Apr), irrigation was applied approximately once every 3 days to maintain soil volumetric water content above 0.2 $m^3$ $m^{-3}$. The grassland was intensively managed with additions of synthetic N fertiliser of ~225 kgN $ha^{-1}$ $y^{-1}$. Grazing events were short and intensive (>100 cows $ha^{-1}$) and typically lasted 1–3 days. Net $CO_2$ exchange was measured continuously by eddy covariance, but measurements during grazing events were not considered valid estimates because of the large variability and uncertain footprint attribution of respiration from the large herd of cows. The C removed by grazing was estimated from biomass measurements before and after grazing.

### *$CO_2$ flux measurements and partitioning between gross primary production and ecosystem respiration*

Heat, water vapour, and $CO_2$ exchanges of the irrigated grassland with the atmosphere were measured using the eddy covariance (EC) system described by Hunt et al. (2016). Procedures for quality-checking, filtering and gap-filling are detailed by these authors, too. The fetch was homogeneous in all directions for at least 130 m, ensuring that a sufficient fraction of the flux footprint was representative of the target paddock. The $CO_2$ exchange represents net ecosystem productivity (NEP). It was partitioned into gross primary productivity (GPP) and total ecosystem respiration rate (ER), with the same OzFluxQC software (Isaac et al. 2017) that was also used for the gap-filling. For this, the ER time series was constructed first, taking ER = −NEP at night-time. Daytime ER was then calculated by the gap-filling algorithm, with soil temperature, volumetric soil water content and NDVI as driver variables. Half-hourly GPP was obtained as NEP + ER.

### *Net ecosystem carbon balance*

The net ecosystem carbon balance (NECB) was calculated as:

$$C_{NECB} = C_{GPP} + \mathrm{C}_{in} - C_{ER} - C_{grazer\ resp} - C_{prod} - C_{loss} \quad (1)$$

Where $C_{NECB}$ is the net C change in the pasture-soil ecosystem (positive for a gain), $C_{GPP}$ is the C gained through photosynthesis, $C_{in}$ is C in imported animal feed and fertiliser, $C_{ER}$ is ecosystem respiration excluding grazer respiration, $C_{grazer\ resp}$ is grazer respiration, $C_{prod}$ is the C removed as product (milk) and stored in animal body mass, $C_{loss}$ is other C losses such as C excreted off pasture (in the milking shed and in transit), emitted as $CH_4$ through enteric fermentation, and the loss of dissolved organic carbon by leaching to groundwater. $CH_4$ losses from dung deposits account for <0.1% of the C content of dung (Pickering et al. 2016) and were ignored here.

$C_{in}$ was obtained from farm records of fertiliser and feed use. CenW does not simulate DOC leaching losses, so the assumed values from Laubach and Hunt (2018) were used in the calculations for both the observed and modelled C balances. The fraction of excretal carbon deposited off-site was assumed to be equal to the daily time fraction spent off-site during grazing days. This fraction was estimated as 17% based on half hourly observations of $CH_4$ levels during grazing events (data not shown).

$C_{grazer\ resp}$ and $C_{prod}$, as well as the C losses from enteric methane and off paddock excretion, were estimated from the C consumed by grazing animals. Pal et al. (2012) found that after a grazing event, ~5% of the dry matter grazed remained in the field in the form of unconsumed litter. Therefore, we assumed that only 95% of the biomass C lost by the plants during grazing was consumed by the animals. The remaining 5% was returned to the soil as litter. In CenW, the percentage of consumed C removed in animal products, respired, or lost as enteric $CH_4$ are user-defined parameters, with the remaining C being returned to the soil as animal excreta. Similarly, the user defines the percentage of consumed N retained in animal products and the percentage lost via leaching or gaseous emissions. According to the calculations of Hunt et al. (2016), 27% of consumed C (including feed supplements) were excreted as dung and urine, and 20% removed in milk solids (or animal growth). C lost in enteric $CH_4$ production was estimated from the methane conversion rate of 21.6 $gCH_4$ $(kgDM)^{-1}$ intake used in Pickering et al. (2016).

For N, it was assumed that 20% of the consumed N was retained in animal products, while 17.8% of excretal N was lost via leaching and gaseous emissions (Pickering et al. 2016). In addition, the urine and dung excreted during milking would not be returned to the soil (in this case the dairy shed effluent was applied to another paddock).

***Model modifications***

During initial simulations, CenW consistently over-estimated GPP immediately after grazing events (see Results). The magnitude of that over-estimate decreased over the following few weeks. It appeared that grazing was causing a decrease in GPP above that expected due to a reduction in leaf area index.

To account for this post-grazing downturn, we introduced an empirical grazing-damage factor and its recovery over time.

***Model parameterisation***

Measurements started in August 2012, and we used the first year of measurements to parameterise the model. The data from the second year were used for validation.

The soil C and N pools were initialised by running the model in equilibrium mode using automated grazing routines for unirrigated sheep-grazed conditions followed by 4 years of irrigated dairy. The biological N fixation parameter was adjusted until the 0-70 cm SOC at the end of the simulation matched the measured value at the start of the measurement period (75.2 tC $ha^{-1}$).

Irrigation events were simulated by adding additional water on the irrigation days according to the recorded time and amount of irrigation application. Fertiliser N was applied according to the recorded timing and amount of fertiliser application on the farm.

Over the 2-year period, there were 20 grazing events. The fraction of the above-ground biomass consumed was estimated using rising plate meter readings before and after grazing when available, otherwise the mean post-grazing biomass reduction for that year was used.

Where possible, we used measured parameter values. However, for several parameters, there were no direct observations. We started with estimated parameter values from previous studies (Kirschbaum et al. 2008, 2015) and then used an automated optimisation routine in CenW that alters parameter values to minimise residual errors between observed and modelled data. The routine aimed to optimise the mean of the model efficiencies (EM) for daily GPP, ER, and NEP. This optimisation was performed using only the measurement data from Year 1.

Model efficiency is defined as (Nash and Sutcliffe, 1970):

$$E_M = 1 - \frac{\sum_i (O_i - P_i)^2}{\sum_i (O_i - \bar{O})^2} \qquad (2)$$

where $O_i$ is the $i$th observed value, $P_i$ is the $i$th predicted value, and $\bar{O}$ is the mean of the observations. Model efficiency values can range from -∞ to 1. A value of 1 indicates a perfect fit, while a value of zero means that the mean square model error is equal to the variance of the observations (when calculated using the population variance formula). Positive model efficiencies indicate that the model has greater explanatory power than the simple model $P_i = \bar{O}$.

***Scenario analysis***

We investigated the long-term effect of irrigation on soil carbon by comparing the simulated SOC after 50 years for irrigated and unirrigated scenarios. The initial SOC pools were set to equilibrium under unirrigated sheep-grazing conditions. We used daily weather data between 1980 and 1999 for the area from the NIWA Virtual Climate Station Network (Tait et al 2006; Tait and Lilley 2009). To investigate the interaction of climate with management, each scenario was run 20 times using a single year's climate data repeatedly over 50 years.

For all scenarios, we used an automated fertilisation routine to ensure that relative nutrient limitations remained the same irrespective of irrigation application. Grazing events were also automated, with grazing initiated when plant biomass reached a given threshold value and grazing animals then consuming available feed down to a specified lower limit. All animal excreta were returned to the paddock, effectively assuming the dairy shed effluent was applied back to the pasture.

Three scenarios were considered: irrigated dairy (ID), unirrigated dairy (UD), unirrigated sheep (US). The UD scenario was identical to the ID scenario, except that irrigation was switched off. However, the grazing management of ID was unlikely to be appropriate in an unirrigated system. Therefore, in the US scenario animals started grazing at a lower biomass thresholds that would be more typical of a sheep grazed system. The proportions of grazed N and C returned to the paddock as dung and urine or removed in animal products was the same for all scenarios.

**Results**

***Model Modifications***

In the unmodified version of CenW, GPP was reduced through grazing, attributable solely to the reduction in leaf area. However, that mechanism alone reduced GPP to only about 70 kgC ha$^{-1}$ d$^{-1}$ (Fig. 1) which meant that following grazing, the observed rates were only ~60% of simulated rates. It was thus apparent that grazing might cause a short-term suppression of GPP through mechanisms not already included in CenW. At this stage, we can only include an empirical factor to account for the observed reduction in NPP as it is not yet known what the exact reason, or range of contributing factors, for that GPP reduction might be.

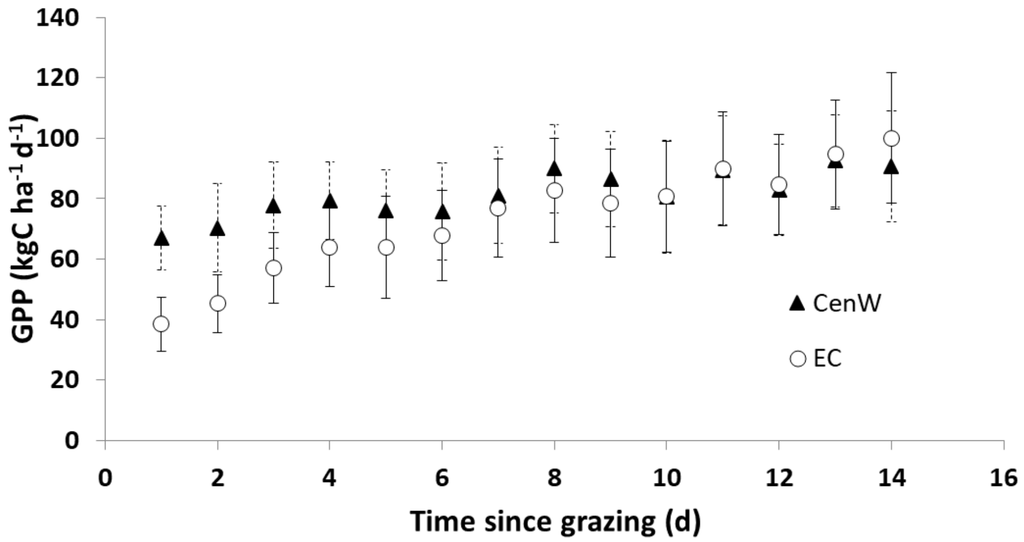



**Figure 1:** CenW (pre-modification) and eddy covariance (EC) derived GPP vs time since grazing. Symbols represent the mean across all grazing events. Error bars indicate 2 standard errors.

We, therefore, added a "grazing damage" function to CenW.

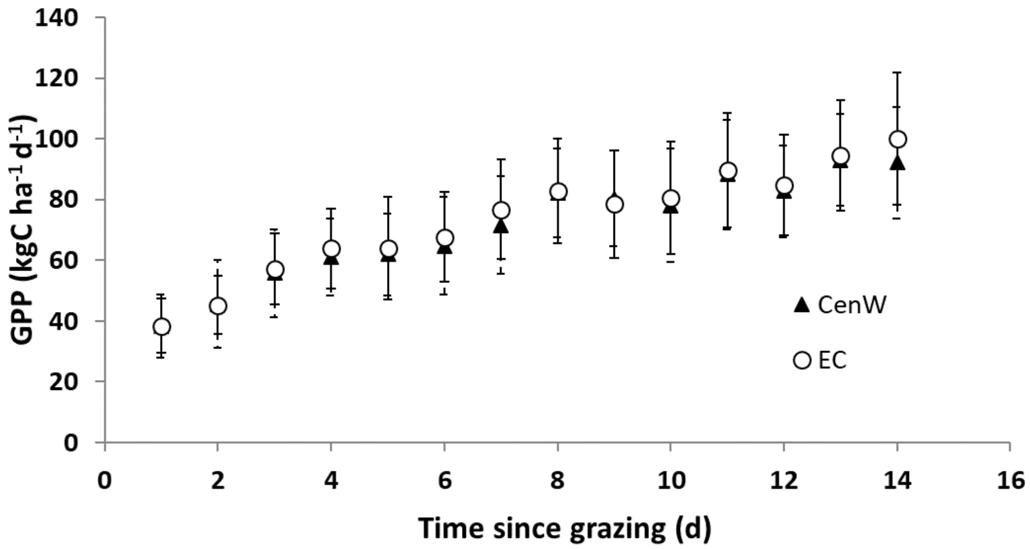



**Figure 2:** CenW (post-modification) and eddy covariance (EC) derived GPP vs time since grazing. Symbols represent the mean across all grazing events. Error bars indicate 2 standard errors.

The grazing-damage rate was set to 0.074% per kg DM consumed, and the grazing-damage recovery rate was 0.166 $d^{-1}$. This modification eliminated the 30 kgC $ha^{-1}$ $d^{-1}$ discrepancy between observed and modelled GPP for the first 5 days following grazing events. As a result, the evolution of modelled and observed GPP as a function of time since grazing tracked each other closely (Fig. 2).

***Model parameterisation and validation***

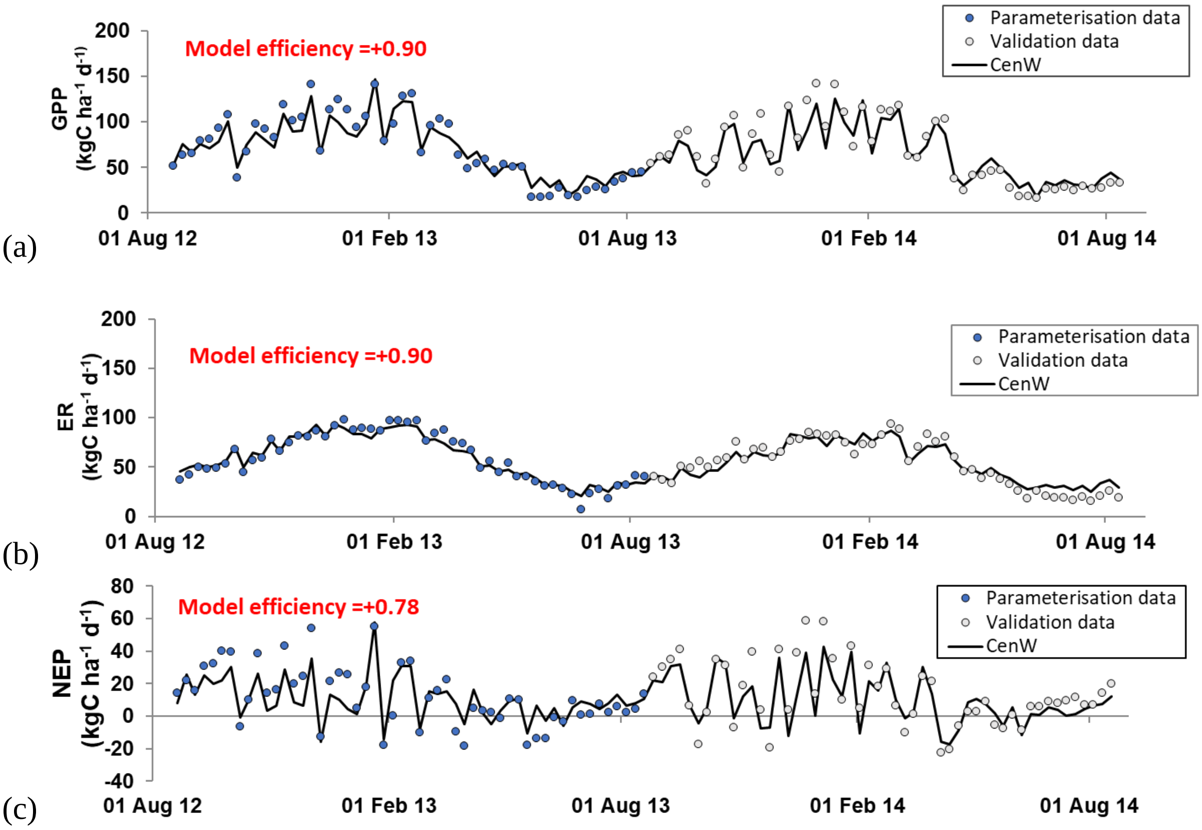



**Figure 3:** Time course of weekly averaged measured (symbols) and CenW (solid line) for (a) GPP, (b) ER, and (c) NEP. Model efficiencies were calculated using the parameterisation data only.

Figure 3 shows the time series for weekly averaged GPP, ER, and NEP for the modelled and EC derived results. The agreement was generally good as shown by the high model efficiencies.

***Carbon balances***

**Table 1:** Observed and modelled carbon balances. "Observed" values include estimates based on eddy covariance and biomass removal measurements or inferred from other observations. $C_{GPP}$ is gross primary productivity, $C_{in}$ is C imported in fertiliser and feed, $C_{ER}$ is the ecosystem respiration excluding grazing animals, $C_{grazer\ resp}$ the respiration from grazing animals, $C_{prod}$ the C exported in animal products, $C_{loss}$ is the combined carbon losses to enteric fermentation, leaching, and dung deposition in the milking shed and in transit. $C_{NECB}$ is the net ecosystem carbon balance (positive if the system gains carbon). "*" indicates that the same value was used for both the observations and CenW.

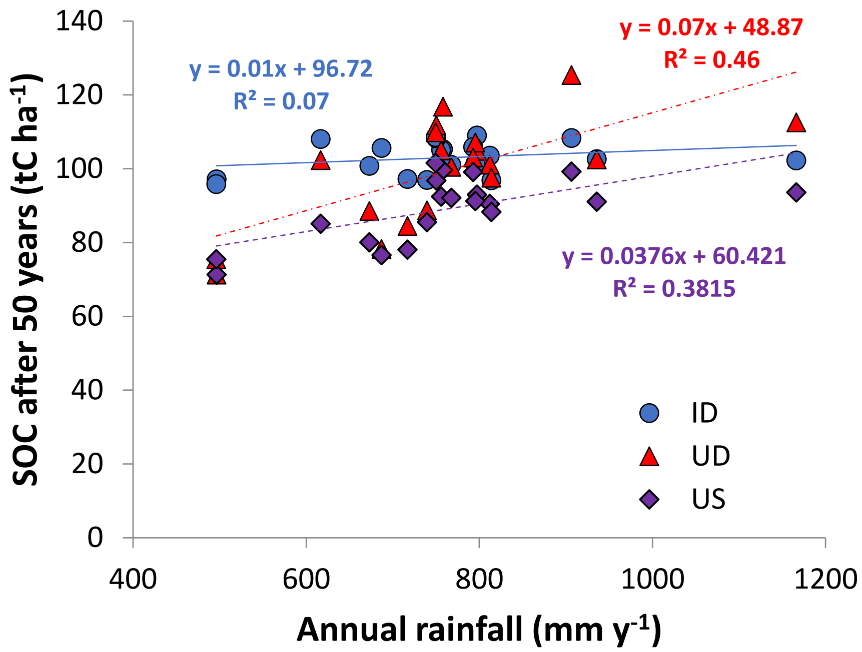
| | | **Observations (tC ha$^{-1}$ y$^{-1}$)** | | | **CenW (tC ha$^{-1}$ y$^{-1}$)** | | |
|---|---|---|---|---|---|---|---|
| | | **Year 1** | **Year 2** | **Ave** | **Year 1** | **Year 2** | **Ave** |
| C gains | $C_{GPP}$ | 26.0 | 24.2 | 25.1 | 25.7 | 23.5 | 24.6 |
| | $C_{in}$ | 0.6 | 1.0 | 0.8 | 0.6* | 1.0* | 0.8* |
| C losses | $C_{ER}$ | 21.8 | 19.5 | 20.7 | 21.7 | 19.8 | 20.8 |
| | $C_{grazer\ resp}$ | 2.4 | 2.9 | 2.6 | 2.5 | 2.6 | 2.6 |
| | $C_{prod}$ | 1.0 | 1.1 | 1.1 | 1.0 | 1.1 | 1.0 |
| | $C_{loss}$ | 0.5 | 0.6 | 0.5 | 0.5 | 0.5 | 0.5 |
| | $C_{NECB}$ | 0.9±0.4 | 1.2±0.5 | 1.0±0.3 | 0.6 | 0.5 | 0.6 |

Both the observations and modelled results found the system was gaining carbon. The modelled NECB was, on average, 0.5 tC ha$^{-1}$ y$^{-1}$ lower than the observed, which is within the estimated uncertainty of the observed NECB, of ~0.4-0.5 tC ha$^{-1}$ y$^{-1}$ (Laubach and Hunt 2018).

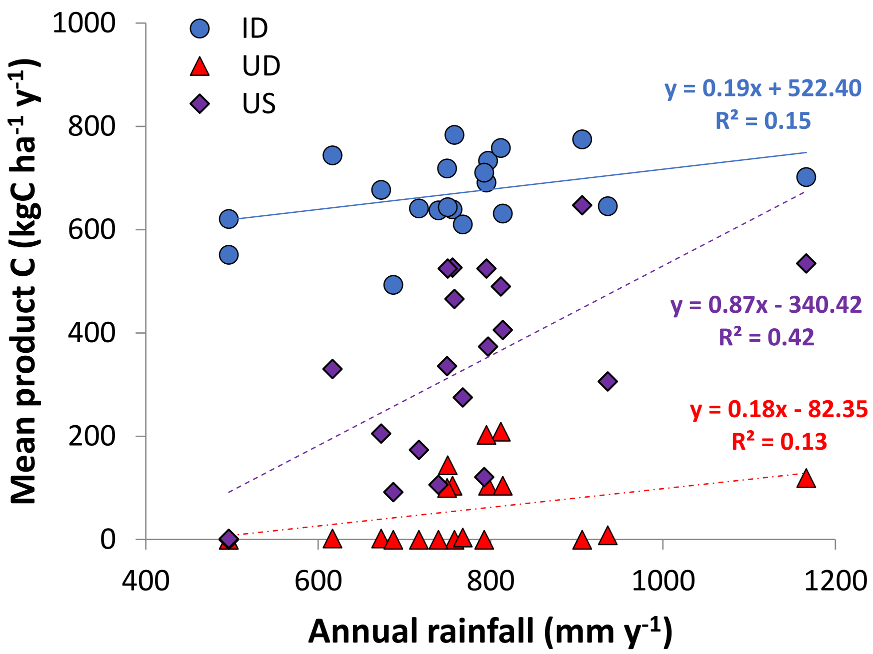
***Scenario Analysis: Long-term irrigation***

Short term changes in NECB do not indicate the total long-term SOC impact of a management change, particularly if the measurement is made when the system is near its new equilibrium. We investigated the long-term effect of irrigation on soil carbon by comparing the simulated SOC after 50 years for different management practices. We assumed that the soils would be near their equilibrium SOC values after this time. The ID scenarios had on average 4 ± 3 tC ha$^{-1}$ (mean ± standard error) more SOC than the UD scenarios and 13.9 ± 1.5 tC ha$^{-1}$ more than the US scenarios after 50 years. The SOC level in the ID scenarios was largely unaffected by the amount of natural rainfall while the UH and UL scenarios showed increasing trends in SOC with rainfall (Fig 5a). Therefore, the increase in SOC with irrigation is greater when natural rainfall is low.

While the reduction in SOC in the UD scenarios were small (and in some cases SOC increased), this was because the lower production in this system meant that the grazing trigger point was rarely achieved resulting in much lower C removal by grazing (Fig. 5b). This would not occur in practice. Instead a farmer would change other management practices to adapt to the lower biomass production. The US scenarios represent a more realistic situation where higher production levels are achieved, but at the cost of SOC. The mean C removed in animal products was 0.67 ± 0.02, 0.06 ± 0.02, and 0.32 ± 0.04 tC ha$^{-1}$y$^{-1}$ for the ID, UD, and US systems, respectively. In these scenarios "animal products" are assumed to be 19% of the consumed C and could represent milk or meat.




(a)




(b)

**Figure 4:** (a) SOC level after 50 years, and (b) mean C in grazing products plotted against rainfall for three different management systems.

**Discussion & Conclusion**

The CenW model was able to simulate weekly GPP, ER, and NEP values with high model efficiency (0.8-0.9). The CenW model also revealed that the downturn in GPP post-grazing was greater than that expected simply from the reduced leaf area. Understanding the cause of this downturn and whether it can be minimised by management practices could be a fruitful area for further study.

Both CenW and the observations were consistent in estimating that the irrigated dairy site was gaining C at an average rate of ~0.6–1.0 tC ha$^{-1}$ y$^{-1}$ (Table 1). Other studies in grazed dairy systems without irrigation also found that reported NECB values generally ranged from neutral to small gains (Rutledge et al. 2015; Felber et al. 2016).

However, the total impact of a management change on SOC depends on the difference between the equilibrium SOC levels of the two systems, rather than short-term changes in SOC. Our long-term scenarios showed that while plant production increases with irrigation, this does not necessarily translate into an increase in SOC. This is because other management practices (e.g. grazing management) affect how this additional C gets split between increasing animal products and increasing SOC. Therefore, irrigation cannot be considered in isolation, but all management changes need to be taken into account.

In some circumstances, we saw an increase in the SOC in the unirrigated dairy system compared to the irrigated dairy system, so our results do not conflict the observations of Mudge et al. (2017).

The long term SOC level depends on the balance between carbon inputs and removals from the system. A cattle-grazed system is relatively inefficient in removing all biomass, especially under water-limited conditions, as cattle tend to leave some ungrazed biomass behind. That lowers potential milk production, but the uneaten biomass left behind can contribute to soil organic matter formation. Sheep, on the other hand, are more efficient in utilising even small amounts of available biomass and convert it to animal produce. That, however, removes a larger proportion of stand biomass and leaves little behind for organic matter formation. This difference in biomass removal efficiencies accounts for differences in animal produce, and conversely, for inverse differences in organic matter formation.

The question as to whether inclusion of irrigation would increase or decrease SOC seemed a simple question to ask, but detailed modelling work has identified a range of serious complications. Because irrigation is applied to enhance biomass productivity, it inherently interacts with nutrient management and the amount and fraction of biomass removed through grazing. Grazing by sheep and cattle can be quite differently managed with regard to the amounts of biomass removed and the thresholds of biomass before and after grazing. If there are no physical barriers between irrigated and unirrigated portions of a paddock, there may also be carbon and nutrient transfers between them if animals preferentially spend time on one or the other portion of the paddock.

All these factors affect the dynamics of SOC, making it difficult to single out the effect of irrigation even in a model. Empirical observations are likely to be similarly affected by these confounding factors, in addition to spatial variation of influencing variables (such as soil properties).

**Acknowledgements**
This research was funded by the New Zealand Agricultural Greenhouse Gas Research Centre (NZAGRC). We would like to thank David Whitehead for reviewing an initial draft and Anne Austin for editing.

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