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Trends in the effects of kelp removal on kelp populations, herbivores, and understory algae

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ABSTRACT

Kelp forests provide habitat and resources to diverse organisms and provide valuable ecosystem services. However, marine deforestation due to wild kelp harvesting, among other drivers, is being observed worldwide. Studies assessing kelp removal effects often focus on the effects on kelp populations, although deforestation also impacts the organisms that interact directly or indirectly with kelp, including herbivores and algal assemblages. Using a meta-analytical approach, we estimated the magnitude and direction of kelp removal effects on kelp, invertebrate herbivores, and understory algae. We also tested if responses varied among functional groups of understory algae and whether results were influenced by the subtidal or intertidal distribution of the removed kelp species and the time elapsed since kelp removal. We observed a substantial decrease in kelp abundance, remaining for up to 4 years following kelp removal, with a larger decrease in subtidal kelp but no recovery observed in intertidal kelp over time. Invertebrate herbivore abundance showed no significant change over time. Understory algae abundance responded positively, although this effect tended to slightly decline over time following subtidal kelp removal. Canopy-, turf-forming and foliose algae were the most benefited, which raises concern about their potential to outcompete kelp. The early succession patterns and cascading effects within kelp forests illustrated here highlight the need for long-term studies to elucidate the long-lasting effects of kelp fisheries, which are scarce at present. There is also a need to consider kelp forests' role in providing habitat and resources to improve predictive frameworks allowing kelp forest conservation and sustainable fisheries.

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1. Introduction

Marine deforestation, i.e., the loss of canopy-forming kelp, is being observed in multiple areas around the world largely due to the harvesting of natural kelp forests (Graham, 2004; Krumhansl et al., 2016; Carrere, 2021; Núñez, 2023). Intertidal and subtidal kelp species are targeted by fisheries to supply the global alginate market, and rising demands have led to increasing wild kelp harvesting in the last decades (Schiel and Nelson, 1990; Vásquez et al., 2012; Oyarzo-Miranda et al., 2023). In addition, other physical and biotic factors, such as climatic events (Thomsen and South, 2019; Carnell and Keough, 2020) and altered food webs resulting in increased overgrazing (Tegner and Dayton, 2000; Estes et al., 2004; Thomsen and South, 2019) are triggering or enhancing marine deforestation. Despite the regulations implemented for kelp fisheries which intend to guarantee the recovery of harvested kelp forests (e.g., harvest-fallow cycles, temporal bans, quotas; Vásquez, 2008; Christensen-Dalsgaard et al., 2020), diverse effects resulting from kelp removal can still be observed around the world (Anderson et al., 2006; Bularz et al., 2022; Norderhaug et al., 2020; Ulaski et al., 2020; Carranza et al., 2024; Minami et al., 2014). Kelp removal impacts not only kelp populations but also the organisms inhabiting kelp forests, such as herbivores and the algal assemblages that cover the seascape (Cárdenas et al., 2016; Pérez-Matus et al., 2017b; Wernberg et al., 2020). In addition, those impacts may change over time (Cárdenas et al., 2016; Carnell and Keough, 2020; Bularz et al., 2022). Although diverse effects of kelp removal have been assessed in some studies, the general direction and magnitude (i.e., effect size) of these effects and how they vary with the time elapsed since kelp removal have seldom been estimated. This information would contribute to the assessment of the trends and scope of kelp removal effects regardless of statistical significance (Borenstein et al., 2009; Sullivan and Feinn, 2012). A comprehensive assessment of the impacts of kelp removal on different components of the community and how these change over time is key to understand the impacts kelp exploitation and start assessing the recovery capacity of kelp forests to ensure their conservation and better determine the sustainability of wild kelp harvesting.

Canopy-forming kelps are foundation species (Dayton, 1972; Lamy et al., 2020) that provide complex 3D physical habitat and resources, fostering marine biodiversity and ecosystem functioning and services (Vásquez et al., 2014; Teagle et al., 2017; Shaffer et al., 2020; Bayley et al., 2021; Smale et al., 2022; Eger et al., 2023). As foundation species, the removal of wild kelp can trigger cascading and broad-scale effects throughout the seascape, as seen in terrestrial forests (Ellison et al., 2005). While some kelp species have annual life cycles, others are perennial, living up to 21 years (Rinde and Sjøtun, 2005). In addition to often being long-lived species, kelps increase the habitat complexity of the seabed through the structural heterogeneity of their holdfasts, stipes, blades, and even their epiphytes (Christie et al., 1998; Vásquez et al., 2001; Vega, 2016; Bué et al., 2020; Norderhaug et al., 2020). They also modify physical-biological interactions (e.g., irradiance, sediment deposition and water motion) that lead to kelp-defined environmental conditions (Angelini et al., 2011; Lamy et al., 2020), thereby influencing the cover, growth and survival of understory algae and other sessile species (Wernberg et al., 2005; Arkema et al., 2009; Flukes et al., 2014; Cárdenas et al., 2016). They are also one of the main food sources of invertebrate herbivores, including sea urchins and snails (Vásquez and Buschmann, 1997; Edwards et al., 2020). Furthermore, kelp forests support coastal food webs and valuable fisheries, as organisms benefiting from them often include species targeted by fisheries (e.g., Bracken et al., 2007; Bertocci et al., 2015; Pérez-Matus et al., 2017a), and provide a suite of additional ecosystem services including carbon storage and nutrient cycling (Smale et al., 2013; Bennett et al., 2015; Aller-Rojas et al., 2020).

Despite kelp forests' intrinsic value, kelp fisheries play an important cultural and socioeconomic role in multiple countries, mainly for the extraction of alginates and aquaculture feed (Anderson et al., 2006; Vásquez, 2008; Frangoudes and Garineaud 2015; Monagail and Morrison 2020). In the last decades, in order to respond to market demands, commercial wild kelp harvesting practices have evolved from the collection of shore-cast plants to other non-lethal or lethal practices, such as removing the surface canopy allowing regrowth or directly removing attached kelp plants, respectively (Schiel and Nelson, 1990; Vásquez et al., 2012; Geange, 2014; Ulaski et al., 2020). The changes in harvesting practices and the considerable increase in kelp landings raise questions about the habitat modifications likely to be triggered, such as changes in algal assemblages, along with the effects on herbivores (Siddon et al., 2008; Cárdenas et al., 2016; Pérez-Matus et al., 2017b; Bularz et al., 2022). Long-lasting effects could be detrimental to forest recovery and could result in the degradation of broader ecosystem goods and services provided by kelp forests (Smale et al., 2013; Edwards et al., 2020) and in the potential establishment of alternative stable states with different 3D structures (Vásquez, 1989; Filbee-Dexter et al., 2014; Christie et al., 2019; Reeves et al., 2022). However, whether these impacts on kelp and on associated taxa are transient or long-lasting remains largely unexplored (but see Christie et al., 1998; Lorentsen et al., 2010; Steen et al., 2016; Castorani et al., 2018; Bularz et al., 2022).

Numerous studies carried out around the world have demonstrated the effects of kelp removal on kelp populations and associated communities. The effects on kelp populations include a decrease in kelp abundance, as recovery to pre-harvested levels occurs at least two and a half years after harvesting (Anderson et al., 2006; Bularz et al., 2022); altered population structure, as even after kelp recovery plant age and size have been observed to remain below pre-harvest levels (Steen et al., 2016; Gouraguine et al., 2021); and increasing distances between plants (Vásquez and Santelices, 1990; Vásquez, 1995). These changes in kelp population dynamics due to their removal can result in bottom-up effects faced by other components of kelp forest communities, such as altered herbivore-kelp interactions (Vásquez and Buschmann, 1997). In fact, decreasing kelp density and increasing distance between plants reduce the whiplash effect of the fronds thus allowing greater herbivore access to plants (Santelices and Ojeda, 1984; Vásquez, 1989; Schiel and Nelson, 1990; Vásquez, 1995), which in turn can further impact kelp, modifying their morphology and making individuals more susceptible to removal by wave action (Vásquez and Santelices, 1990). As consumers, herbivores have the potential to exacerbate kelp forest deforestation by overgrazing, which has been documented around the world (Vásquez and Buschmann, 1997; Aguilera et al., 2015; Ling et al., 2015; Edwards et al., 2020; Reeves et al., 2022).

Other changes in different trophic levels and at the community level have also been observed following kelp removal. Understory algae can outcompete kelp on the short-, medium- and long-term, yet they do not provide the same habitat complexity, structural diversity, or ecosystem functions (LaRue et al. (2019); Bué et al. (2020); Edwards et al. (2020); Velasco-Charpentier et al. (2021). Higher sedimentation rates and increasing barren rocky substrate following kelp removal can also lead to changes in the understory composition of algae and sessile fauna (Cárdenas et al., 2016; Wernberg et al., 2020). Studies have shown changes in algal composition up to 3–4 years after kelp removal (Dayton, 1975; Toohey et al., 2007; Schiel et al., 2018). The abundance of some functional groups, such as foliose algae, turf-forming algae, and invasive kelp species (e.g., the Asian kelp *Undaria pinnatifida*) has been observed to increase significantly after kelp removal (De Leij et al., 2017; Schiel et al., 2018; Wernberg et al., 2020; Bularz et al., 2022). In addition, the influence of climate change may benefit these algae at the expense of kelp species, with the risk of turf-forming algae expanding even more under elevated temperature and CO_2, thus increasing the likelihood of large-scale shifts from structurally diverse kelp forests to less structurally complex habitats dominated by turf-forming algae (Vásquez, 1989; Smale et al., 2013; Smale, 2020).

Results from studies investigating the effects of kelp removal show, however, mixed trends. For example, while most studies evidence a decrease in kelp density and biomass in harvested areas (González-Roca et al., 2021; Gouraguine et al., 2021; Bularz et al., 2022), others have recorded an increase (e.g., Engelen et al., 2011; Pérez-Matus et al., 2017b; Ulaski et al., 2020). Furthermore, increases and decreases in invertebrate herbivore abundance (Reed, 1990; Siddon et al., 2008; Pérez-Matus et al., 2017b; Bularz et al., 2022) and understory algae abundance after kelp removal have both been documented (Schiel et al., 2018; Wernberg et al., 2020; Bularz et al., 2022). These mixed trends among kelp removal effects may be influenced by factors such as the time since kelp removal at which the effects were measured, or the intertidal or subtidal distribution of the removed kelp species, but studies exploring these dynamics across trophic levels are lacking. Given these inconsistencies between some studies, we sought to disentangle the general trends of different components of kelp forest communities with a meta-analytical approach (Borenstein et al., 2009).

Here, we assembled a global dataset including 197 case studies from 27 articles that comprised 16 kelp and pseudo-kelp species from 8 different countries (Fig. 1, Table S1, Table S2) to determine the effects of human-induced kelp removal by using a meta-analytical approach. We aimed to assess (1) the general trends of how kelp, invertebrate herbivores and understory algae respond to kelp removal, (2) if functional groups of understory algae respond differently, (3) if effects differ depending on whether the harvested kelp is intertidal or subtidal, and (4) if the removal effects observed varied depending on the time since the removal of intertidal or subtidal kelp. Studying the effects of the removal of different kelp species on their populations and communities, and how these effects vary over time is important to elucidate the global trends following kelp removal and to outline bottom-up effects and marine succession patterns, as they occur on land (Sousa, 1979; Vásquez, 1989; Guariguata and Ostertag, 2001). Assessing these trends is needed to improve predictive frameworks allowing kelp forest conservation and sustainable fisheries. To our knowledge, this is the first broad-scale meta-analytical study aiming to evaluate the magnitude of the effects of kelp removal not only on kelp populations but also on invertebrate herbivores and the understory algae assemblage, as well as on algal functional groups (Steneck and Dethier, 1994).

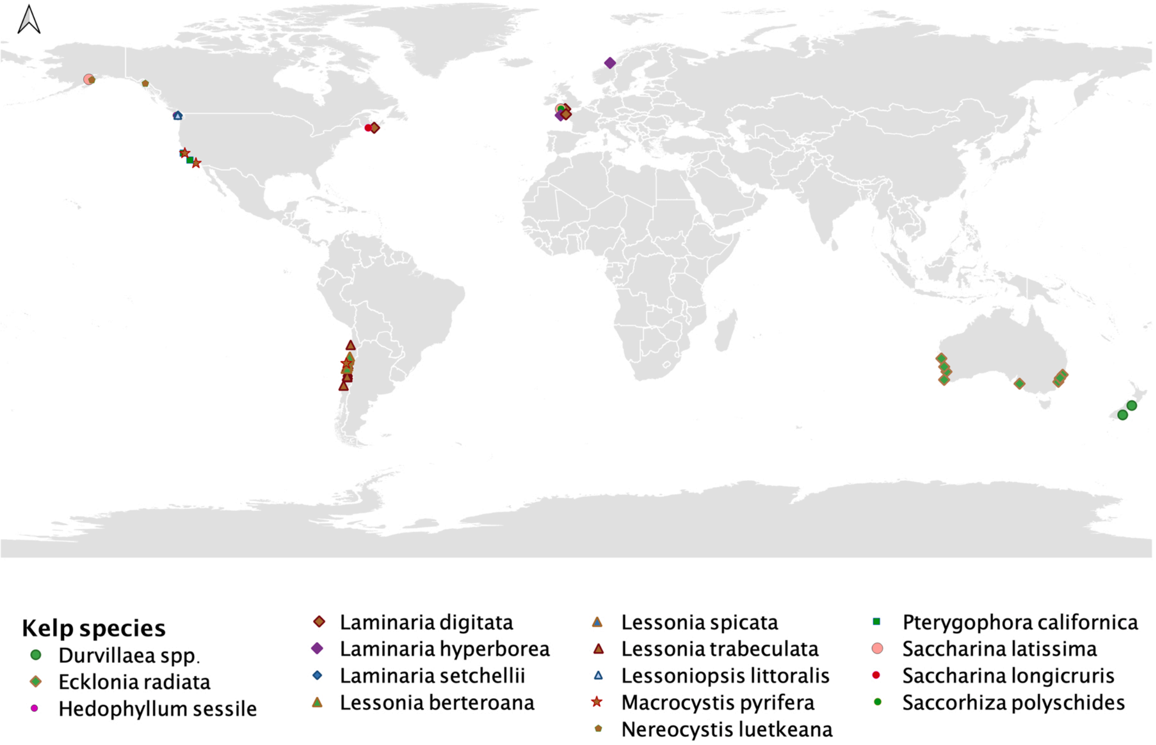

Fig. 1. Geographic distribution of study locations (193 case studies) by target kelp species.

2. Materials and methods

2.1. Study selection

To assemble an exhaustive collection of studies measuring the effects of kelp removal on kelp abundance, invertebrate herbivore abundance, and understory algae abundance we conducted a search in the ISI Web of Science using the following key words: ((ALL= (harvest* OR remov* OR clear*)) AND ALL= (kelp*)) AND ALL= (effect OR impact OR response OR consequence OR influence OR change OR relation OR abundance OR biomass OR density OR diversity OR species richness OR herbivor* OR understory OR alga*).

The study inclusion criteria were the following: (1) studies assessed the effects of the human-induced removal of brown macroalgae of the order Laminariales (kelps), and "pseudokelps" *Durvillea* spp. and *Saccorhiza polyschides* (Smale et al., 2013; Bertocci et al., 2015); (2) kelp removal was conducted as complete clearing or partial "thinning" of kelp in an area where it occurs naturally, we did not seek to compare areas with kelp canopy with areas naturally devoid of canopy; (3) kelp removal effects were assessed with manipulative experiments (i.e., those that described experimental kelp removal by mimicking commercial harvesting or another physical disturbance) or observational studies (i.e., those that made comparisons across space or time of different conditions in which kelp was either left intact or removed, e.g., inside and outside commercial harvesting areas) that had controls and replicates for their treatments; (4) kelp removal was done once during the study period (no repeated harvests), to monitor what would happen when alternating harvest-fallow periods (best case-scenario); (5) the studies reported at least one of these three response variables: total or adult kelp abundance, invertebrate herbivore abundance, understory algae abundance; (6) means, standard errors or standard deviations, and sample sizes for treatments were available in tables, text or graphical form. We focused specifically in the removal effects on invertebrate herbivores because of the many recorded cases of kelp overgrazing by sea urchins or snails (Vásquez and Buschmann, 1997; Sivertsen, 2006; Ling et al., 2015; Edwards et al., 2020; Reeves et al., 2022) and because their limited home rage compared to herbivorous fish (Hereu, 2005; Pillans et al., 2017; Lamy et al., 2018; Ruz et al., 2018;), which makes them stay longer in a forested area, unable to easily migrate to more distant kelp forest patches.

2.2. Data collection

Studies that reported more than one kelp species, populations from different locations, or harvests conducted in different seasons or years (separate treatments) were considered as separate case studies (Koricheva et al., 2013). Response variables were categorized into three categories: kelp abundance (e.g., total or adult density, biomass or cover), invertebrate herbivores abundance (e.g., density or biomass), and understory algae abundance (e.g., density, biomass or cover). We also categorized the understory algae by functional group to test if groups responded differently. When total kelp abundance could not be obtained, and studies reported kelp abundance by life stages (e.g., recruits, juveniles, adults), we recorded adult abundance, as adults account for most of the biomass. Recruits and juveniles do not always follow the same trends as total or adult kelp abundance (Gouraguine et al., 2021; Carranza et al., 2024). If the means and standard errors or standard deviations were in graphical form, they were extracted using the software Plot Digitizer (Huwaldt and Steinhorst, 2015). If more than two levels of removal treatments were reported, we used data from the two most extreme treatment levels (for example, 100% kelp clearing and 0% kelp clearing – control). Most studies were designed as simultaneous – control – experiment (SEC). In studies using a Before-After Control-Impact (BACI) design, we obtained data from the Control-Impact (SEC equivalent). In addition, for each case study we recorded if the harvested kelp species was subtidal or intertidal and the time elapsed since kelp removal at the time of the survey (or of the last survey for repeated measures studies because we wanted to consider the longest possible times), to evaluate their potential influence on the results.

We estimated effect sizes using Hedges' *g*, which is the standardized mean difference between treatments and control. It requires the mean value, standard deviation or error and sample size obtained in the treatment and control to be calculated, and transforms all effect sizes to a common metric, thus allowing to include different abundance outcome measures (e.g., density, biomass, cover) in the same synthesis (Borenstein et al., 2009; Stotz et al., 2021). Hedges' *g* was calculated using the *escalc* function in the *metafor* package (Viechtbauer, 2010) in R (R Core Team, 2022), version 3.6.3.

We calculated Rosenthal's fail-safe number to estimate how many studies would be needed to change the observed main effects (Koricheva et al., 2013), i.e., the number of additional studies with non-significant results that would be required to change the results of the general analyses. The number of cases was found to be well above the robustness criteria: 5 N + 10 (Koricheva et al., 2013).

2.3. Statistical analyses

To evaluate the general trends of kelp removal effects, we conducted random effects meta-analyses using the *rma.mv* function from the *metafor* package (Viechtbauer, 2010) in R (R Core Team, 2022), version 3.6.3. Random intercept models were run separately for each response variable (i.e., kelp abundance, invertebrate herbivore abundance, and understory algae abundance). Since we were looking for general trends and did not seek to evaluate differences between harvested kelp species, we specified the target kelp species as a random factor in each model to account for the non-independence of kelp species/populations under different treatments within a single study (Stotz et al., 2021). We also conducted a test of heterogeneity (Q). We found a few higher-than-average effect size values (i.e., outliers) for kelp and understory algae abundance but we found no compelling reason to exclude any of these outliers. Furthermore, results remained consistent when excluding those outliers from the analyses (Table S3).

We tested whether removal effects varied across functional group of understory algae (i.e., articulated coralline algae, canopy-forming brown algae, encrusting algae, foliose algae, and turf-forming algae) by including this factor as a moderator variable when testing the effects of kelp removal on understory algae abundance with random effects meta-analyses using the *rma.mv* function. Since this moderator was found to have a significant influence in model results, we then conducted the meta-analyses separately for each functional group (Table 2) to assess the summary effects of each group.

We also tested whether removal effects varied depending on the intertidal or subtidal distribution of the target kelp species by including kelp distribution as a moderator variable in the separate random effects models for kelp abundance and understory algae abundance using the *rma.mv* function. We could not test the differences in the effects of intertidal or subtidal kelp removal on invertebrate herbivore abundance as all our observations of this variable were conducted on subtidal kelp species. Since this moderator was found to have a significant influence on the responses to kelp removal, the following analyses were conducted separately for intertidal and subtidal kelp removal.

To assess the influence of the time since kelp removal on the magnitude of its effects, random intercept models were run separately for intertidal and subtidal kelp and for each response variable (i.e., kelp abundance and understory algae abundance) using the *rma.mv* function, including time since kelp removal as a moderator variable. Testing the influence of the time since removal on the magnitude of the effects on invertebrate herbivore abundance included only subtidal kelp species removal.

Heterogeneity tests (Q) were performed for all aforementioned models and target kelp species was specified as a random factor in all models.

3. Results

Our literature search returned over 1000 studies published between 1975 and 2022, out of which 27 (Table S1) met our criteria for inclusion. As studies could include multiple case studies, the meta-analyses were based on 193 case studies, comprising 16 kelp and pseudo-kelp species from 8 different countries, most having been conducted in Chile and the USA (Fig. 1, Table S1). Kelp removal effects were assessed between 1.5 months and 5 years after kelp removal.

Kelp removal showed significant effects on two of the three response variables that were analyzed, which followed opposite trends. Following kelp removal, summary effects showed that kelp abundance decreased, with an absolute value of effect size of nearly 1.34 (Table 1; Fig. 2). Fifty out of the 60 case studies recorded decreases in total or adult abundance of kelp (Fig. S1). Moreover, kelp removal had no effect on the abundance of invertebrate herbivores (Table 1; Fig. 2), which showed quite heterogeneous responses recorded in the few case studies ($n = 15$; Fig. S2). Opposite to kelp abundance, the abundance of understory algae increased in response to kelp removal (Table 1; Fig. 2), although responses were more heterogeneous than for kelp abundance, with 70 out of 119 case studies showing a positive effect of kelp removal (Fig. S3). However, different functional groups of understory algae species responded differently to kelp removal ($n = 102$, $Q_M = 86.35$, p-value < 0.0001), with some showing significant increases in abundance while others did not (Table 2, Fig. 3). The abundance of articulated coralline algae and of encrusting algae did not vary significantly with kelp removal, but the abundance of canopy-forming brown algae, foliose algae and turf-forming algae increased significantly following kelp removal with a magnitude of 1.24 ± 0.45 (SE), 0.37 ± 0.16, and 0.60 ± 0.29, respectively (Fig. 3). Canopy-forming brown algae included recruit, juvenile and adult understory kelp, as well as Asian kelp *Undaria pinnitafida*, and fucoids. Our research initially included removal effects on species diversity (e.g., species richness, Shannon index) as well, but as we found few case studies, the analysis was only included in the supplementary material (Fig. S4).

The intertidal or subtidal distribution of the removed kelp species showed significant differences in the effect of kelp removal on kelp abundance ($n = 60$, $Q_M = 19.88$, p-value < 0.0001) as well as on understory algae abundance ($n = 119$, $Q_M = 4.04$, p-value $=$ 0.0445), which included other non-dominant brown macroalgae. Following the removal of intertidal kelp, summary effects showed that kelp abundance decreased significantly, with an absolute value of effect size of 1.29 ± 0.35 (SE), while understory algae increased significantly with a magnitude of 1.42 ± 0.47 (Fig. 4). Following subtidal kelp removal, kelp abundance also decreased significantly (absolute value of effect size of 1.4 ± 0.55), but the increase in understory algae abundance was only marginally significant (p-value $=$ 0.0637) with a magnitude of 0.42 ± 0.23 (Fig. 4). Although there was a negative effect of kelp removal on kelp abundance of both intertidal and subtidal species, the removal of subtidal kelp species (e.g., *Ecklonia radiata*, *Laminaria hyperborea*, *Lessonia trabeculata*, *Macrocystis pyrifera*, *Nereocystis luetkeana*, *Pterygophora californica*) had a greater negative effect than the removal of intertidal kelp or pseudo-kelp species, such *Laminaria digitata*, *Lessonia* spp., *Saccharina latissima*, *Durvillaea antarctica* and *Saccorhiza polyschides* (Fig. 4).

We could not test if the season and the method of kelp removal influenced the responses of kelp abundance to kelp removal because we did not obtain balanced sample sizes for each season and different removal methods. Many studies did not report the season in which kelp was removed, and most experiments removed kelp using lethal methods (e.g., cutting the stipe just above the holdfast, removing the entire kelp plant from the rocky substrate, trawling). Most experiments mimicked commercial kelp removal methods or climate events or were done in commercial harvest areas (Pérez-Matus et al., 2017b; Ulaski et al., 2020; Norderhaug et al., 2020; Bularz

Table 1

Kelp removal effects, measured as the effect size (Hedges' *g*) per type of effect (i.e., kelp abundance, herbivore abundance and understory algae abundance). *Significant values.

Effect type (n)	Average effect size *g* (SE)	p-value	Confidence interval (CI) Lower limit	Confidence interval (CI) Upper limit
Kelp abundance (60)*	-1.3374 (0.2891)	**< 0.0001**	-1.6683	-0.6307
Herbivore abundance (15)	0.1088 (0.2915)	0.6838	-0.5321	0.8112
Understory algae abundance (119)*	0.7142 (0.2306)	**0.0024**	0.2747	1.2787

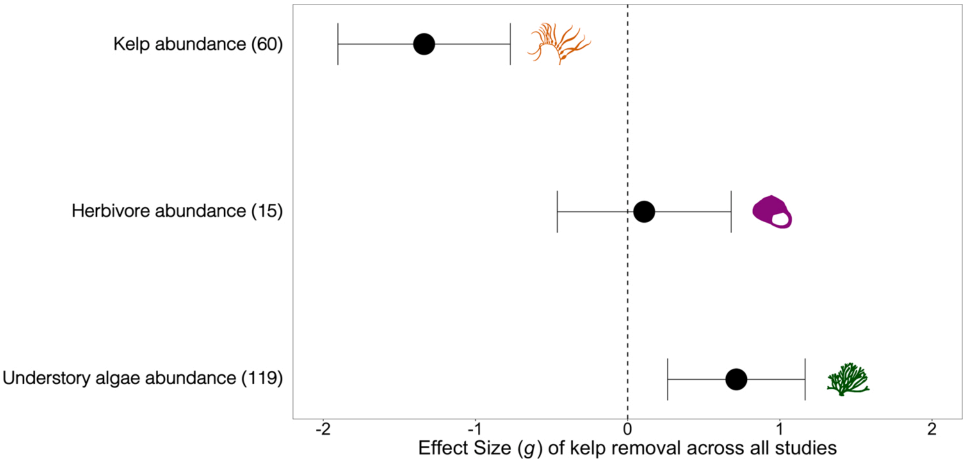

Fig. 2. Summary effects (Hedges' *g*) of kelp removal on kelp, invertebrate herbivore, and understory algae abundance across all studies, pooling the removal of intertidal and subtidal kelp (number of case studies by effect type in parentheses). Bars represent 95% confidence intervals. Silhouettes of the different categories were added using the *rphylopic* package (Gearty and Jones, 2023).
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Table 2
Understory algae categorized by functional group (e.g., Kennelly, 1987b; Schmidt and Scheibling, 2007; Siddon et al., 2008; Wernberg and Connell, 2008; Wernberg et al., 2020; Bularz et al., 2022).

Functional groups	*Included species*
Articulated coralline algae	*Amphiroa anceps*, *Corallina* spp., *Halimeda cuneata*, *Jania* spp., *Metagoniolithon stolinifera*, *Rhodopeltis australis*
Canopy-forming brown algae (juveniles and adults)	Fucoids (e.g., *Cystophora* spp., *Phyllospora* spp., *Sargassum* spp., *Scytothalia* spp.), understory kelp, invasive kelp *Undaria pinnatifida*
Encrusting algae	Encrusting corallines algae and encrusting non-coralline algae (e.g., *Mesophyllum* spp., *Lithophyllum* spp., *Ralfsia verrucosa*)
Foliose algae	*Callophycus oppositifolius*, *Caulerpa* spp., *Codium* spp., *Dictyomenia sonderi*, *Dictyopteris australis*, *Halopeltis australis*, *Hennedya crispa*, *Pterocladia lucida*, *Ulva* sp., *Zonaria-Lobophora* complex (e.g., *Zonaria turneriana*, *Lobophora variegata)*
Turf-forming algae	Filamentous turfs (e.g., *Acanthophora* sp., *Champia* spp., *Desmarestia* spp., *Feldmannia mitchelliae*, *Heterosiphonia* spp., *Polysiphonia* spp.)

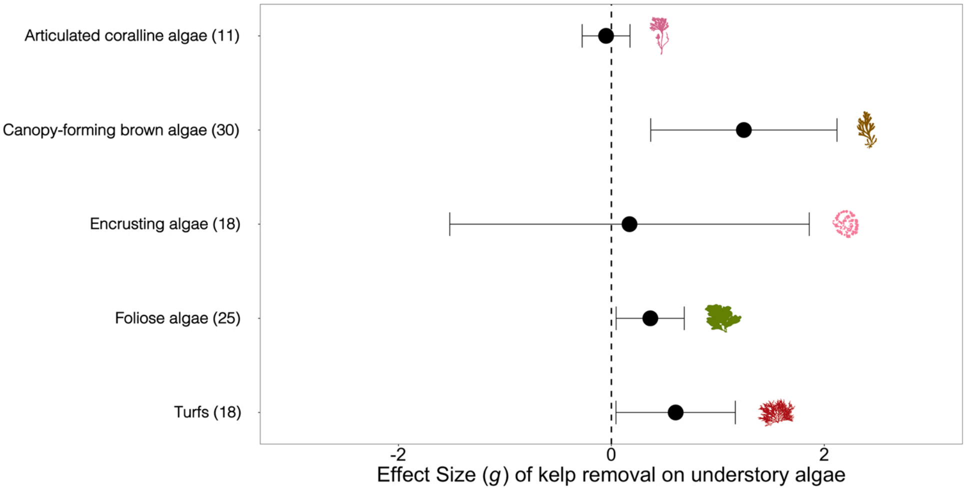

Fig. 3. Summary effects (Hedges' *g*) of kelp removal on different understory algae functional groups (number of case studies in parentheses). Bars represent 95% confidence intervals. Silhouettes of the different categories were added using the *rphylopic* package (Gearty and Jones, 2023).
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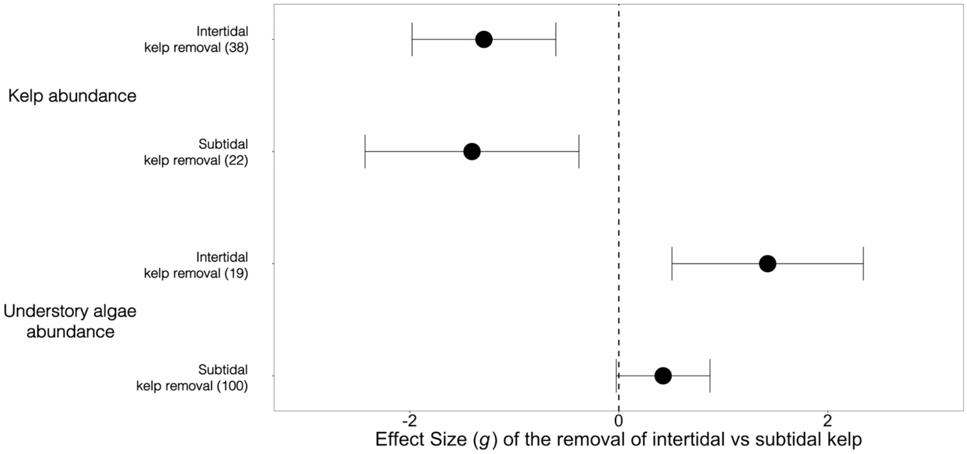

Fig. 4. Summary effects (Hedges' *g*) of the removal of intertidal or subtidal kelp on kelp and understory algae abundance (the number of case studies in parentheses). Bars represent 95% confidence intervals.

et al., 2022).

The time elapsed since the removal of intertidal kelp significantly influenced the magnitude of the effect of removal on kelp abundance (n = 21, $Q_M = 8.12$, p-value = 0.0044; Fig. 5A) but not on understory algae abundance (n = 19, $Q_M = 1.08$, p-value = 0.2977; Fig. 5B). On the other hand, the time elapsed since the removal of subtidal kelp significantly influenced the magnitude of the effects on both kelp (n = 17, $Q_M = 13.32$, p-value = 0.0003; Fig. 6A) and understory algae abundance (n = 82, $Q_M = 10.83$, p-value = 0.001; Fig. 6C). However, it did not significantly influence the abundance of invertebrate herbivores (n = 7, $Q_M = 1.2$, p-value = 0.2739; Fig. 6B), although the sample size for this analysis was small and therefore should be interpreted with caution. After intertidal kelp removal, kelp abundance decreased and remained lower than in the control treatments for up to 4 years, which was the maximum duration of some of the case studies (Fig. 5A), reflecting negative effects that remain on the long-term, whereas the effects on understory algae abundance showed no significant change over time (Fig. 5B). After subtidal kelp removal, kelp abundance was found to decrease, but it tended to increase towards control levels over time (Fig. 6A), indicating possible recovery of subtidal kelp abundance. The abundance of understory algae was found to initially increase following subtidal kelp removal, but this positive response diminished over time (Fig. 6C).

4. Discussion

This meta-analytical synthesis allowed us to quantify the magnitude of kelp removal effects on kelp populations and on two species groups that interact directly with kelp, i.e., invertebrate herbivores and understory algae, and to outline bottom-up effects and succession patterns that result from marine deforestation by human-induced kelp removal (Menge, 1992; Tegner et al., 1997). As kelp forests worldwide are also being threatened by overgrazing (Oróstica et al., 2014; Edwards et al., 2020), to which fragmented forests have reduced resistance (Reeves et al., 2022), it is important to assess the magnitude of changes and the trends in the abundance of kelp and invertebrate herbivores following kelp removal. In addition, as kelp forests become prone to invasion and collapse to turf dominance enhanced by climate change (Smale et al., 2013; Reeves et al., 2022), it is also crucial to evaluate how understory algae abundance changes in response to kelp removal. The use of a meta-analytical approach allowed us to estimate summary effects and to see if the effect sizes were consistent across studies (Borenstein et al., 2009). We found no evidence of intertidal kelp abundance recovery up to 4 years after kelp removal and subtidal kelp showed potential recovery over time, which highlights the need for strong regulations to ensure recovery of kelp, and thus the overall sustainability of the kelp industry (Vásquez et al., 2012; Steen et al., 2016; Hamilton et al., 2022). Importantly, we also showed that the removal of kelp had repercussions on different trophic groups and/or different functional groups, therefore accentuating kelps' role as foundation species with great influence in temperate coastal communities (Miller et al., 2015; Lamy et al., 2020). Thus, these results also indicate that the responses to kelp removal should be evaluated more comprehensively, since all these changes can also have other consequences at trophic levels not yet evaluated. There were few observations of invertebrate herbivore abundance, which showed no significant change over time. However, changes in the abundance of understory algae varied over time, which highlights the urgent need for further long-term studies, to elucidate the probable long-lasting effects of wild kelp harvesting, which are scarce at present. Furthermore, while the positive effect of subtidal kelp removal on understory algae was attenuated over time, the different impacts that kelp removal had on different understory algae functional groups suggests that there may be important changes in community composition, and thus in other associated species (Miller et al., 2015; Cárdenas et al., 2016; Bularz et al., 2022) and ecosystem functioning (Edwards et al., 2020), which is relevant to kelp forest conservation. More integrative studies would allow us to evaluate kelp forests' role in providing habitat and resources for other species, the interactions among the components of kelp forest communities and the seascape, and how they can be altered by wild kelp removal.

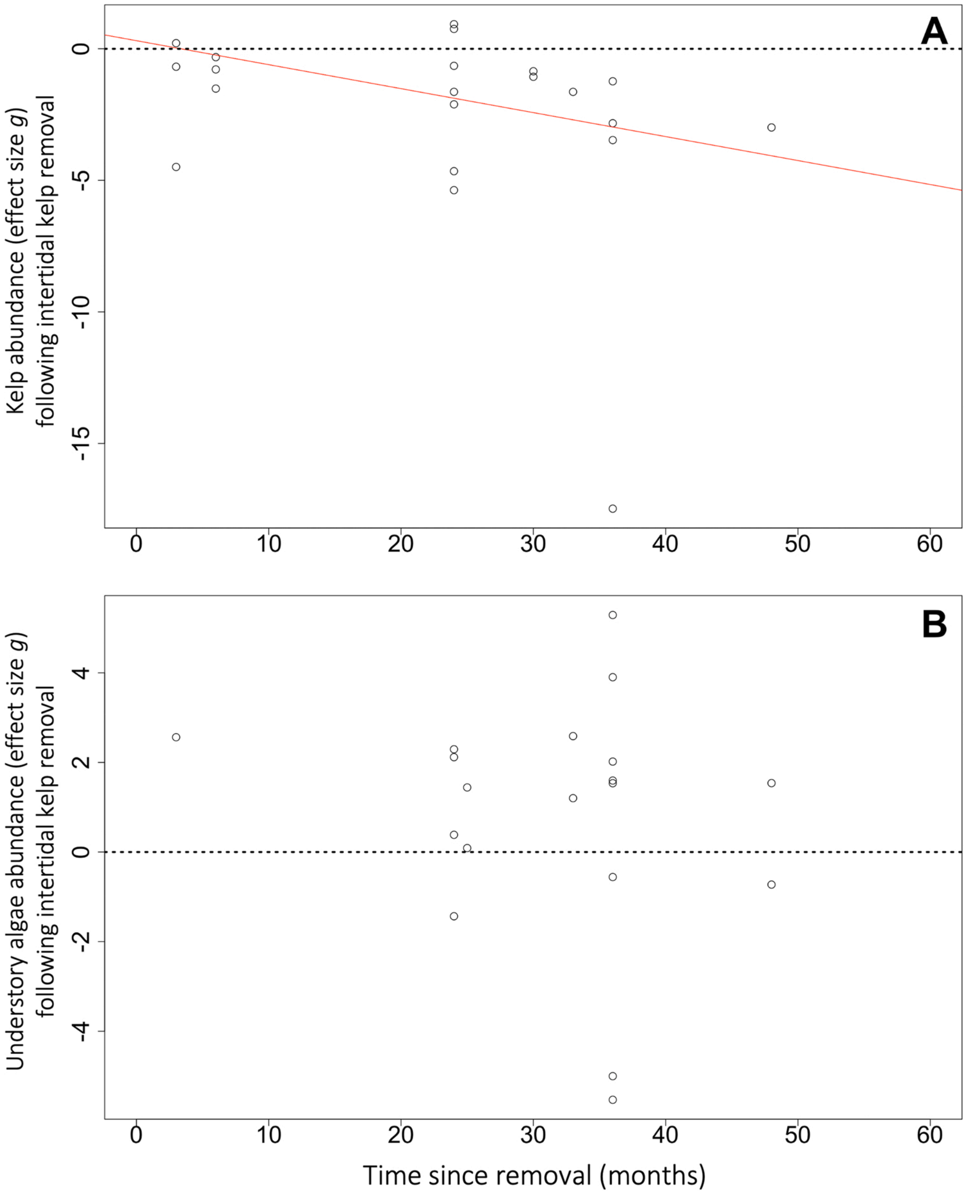

Fig. 5. Influence of time since removal of intertidal kelp on the response of kelp (A) and understory algae (B) abundance (effect size *g*). The red line represents the regression lines from significant models. Note the difference in scaling of the y-axes.

4.1. Changes of kelp removal effects over time

Analyzing the trends of these three interacting components of kelp forest communities (i.e., kelp, invertebrate herbivores, and understory algae) over time also allowed us to infer how their dynamics can be linked. Following kelp removal, kelp abundance decreased significantly, as expected (in high magnitude), whereas understory algae abundance increased, which could be linked to the freed-up space in the rocky bottom and the increase in irradiance resulting from removing the kelp canopy (Reed, 1981; Clark et al., 2004; Arkema et al., 2009). Invertebrate herbivore abundance showed no significant trends but, following kelp removal, their abundance can be altered by a decrease in the whiplash effect produced by kelp (Dayton, 1975; Santelices and Ojeda, 1984; Vásquez and Buschmann, 1997), and the increase in food access or availability (e.g., remaining kelp plants, increasing kelp recruits and increasing understory algae). Therefore, it remains crucial to analyze the course of these cascading effects over longer time periods.

4.1.1. Kelp-herbivore interactions

Most observations of kelp abundance following intertidal kelp removal showed that abundance remained lower in harvested areas than in unharvested areas for up to 4 years after kelp removal, which was the maximum duration of the analyzed studies, and followed

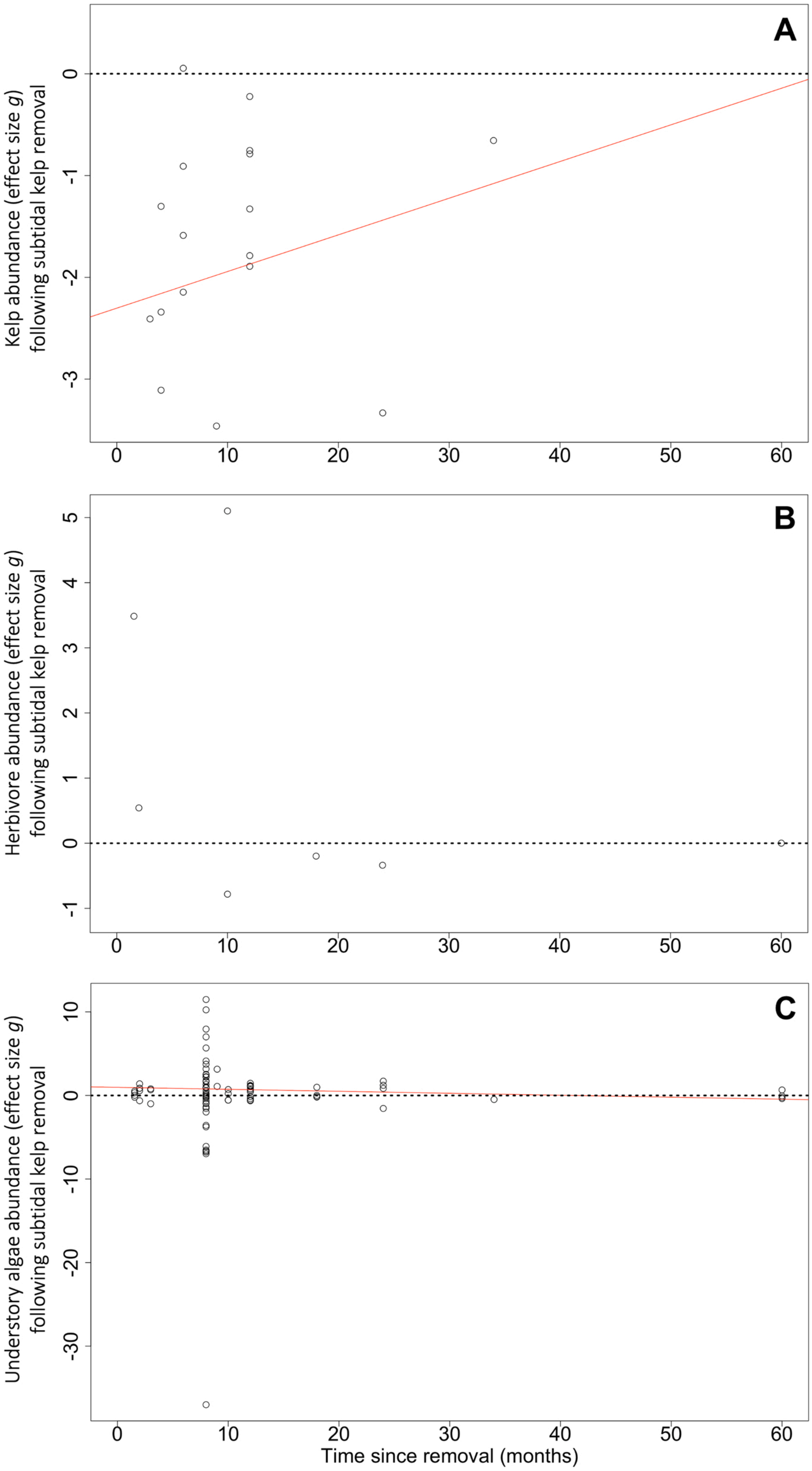

Fig. 6. Influence of time since removal of subtidal kelp on the response of kelp (A), invertebrate herbivore (B) and understory algae (C) abundance (effect size *g*). The red line represents the regression lines from significant models. Note the difference in scaling of the y-axes.

a decreasing trend over time. The limited number of case studies evaluating invertebrate herbivore abundance following intertidal kelp removal did not allow us to follow their trends over time, but other studies have found that invertebrate herbivores such as chitons can prevent algal colonization, even of large brown algae species, increasing bare space availability and variability through time (Duggins and Dethier, 1985; Aguilera et al., 2015). When removing subtidal kelp, kelp abundance in harvested areas also remained lower than in controls over time, but our results showed recovery trends towards pre-harvest levels. Other studies on subtidal kelp have however documented that these negative effects on kelp abundance can last even longer (Christie et al., 1998; Levitt et al., 2002; Bularz et al., 2022; Gouraguine et al., 2021), thus suggesting long-term losses in ecosystem functions and services. Further long-term monitoring is necessary to capture the long-term evolution of kelp populations following human-induced kelp removal more accurately, as it has been done with long-term simulations of repeated loss of giant kelp from destructive winter waves for example (Castorani et el, 2018), which is not necessarily equivalent. One of the reasons for the slow recovery of kelp abundance may be the presence and initially higher abundance of invertebrate herbivores (Vásquez and Buschmann, 1997; Norderhaug, Christie, 2013; Oróstica et al., 2014). Although our results showed no significant changes in effects on invertebrate herbivore abundance over time, the small sample size available for this analysis suggest that more studies on their response are required to better assess their trends over time with more confidence and understand how they respond to kelp removal. As invertebrate herbivores sometimes prefer sporogenous over vegetative tissue (O'Brien, 2018), intense grazing on kelp sori could impact reproductive potential and therefore population recovery, establishing a long-lasting influence of their grazing activity even after herbivore numbers decline (Ling et al., 2015; Edwards et al., 2020; Reeves et al., 2022). In turn, this can lead to the establishment of stable state sea urchin barrens when urchin abundance exceeds a threshold (Filbee-Dexter and Scheibling, 2014).

Since following kelp removal kelp abundance remains lower than in unharvested areas over time, the growing abundance of invertebrate herbivores (e.g., sea urchins and snails) may be limited by a threshold. Studies have found that density and grazing intensity of kelp grazers (e.g., sea urchins and snails) increase non-linearly with decreasing kelp biomass, which enhances direct and indirect kelp tissue loss, thus creating positive feedback between disturbances that cause deforestation (e.g., kelp removal) and grazing (Vásquez and Buschmann, 1997; O'Brien, 2018; Rennick et al., 2022). In addition, the general decrease in understory algae observed over time following kelp removal can also limit food resources for herbivores (Dean and Connell, 1987; Aguilera, 2011). Moreover, as kelp abundance decreases, the 3D physical habitat formed by kelps is modified, and their capacity to provide refuge from predators, such as sea otters, fish, and lobsters (Estes and Duggins, 1995; Hamilton and Caselle, 2015; Nichols et al., 2015), also decreases (Watanabe, 1984; Teagle et al., 2017; Christie et al., 2019b). In this synthesis, we sought to focus on direct interactions with kelp (i.e., herbivory and algal competition), thus we did not collect further information regarding predators. However, predator-prey-kelp interactions remain crucial to understanding kelp forest dynamics (Steneck et al., 2002) and would help to better understand the overall consequences of kelp removal. Some studies have found that although sea urchin biomass was negatively correlated with kelp biomass, it only explained 14% of the variation in kelp dynamics (Rennick et al., 2022). Thus, it is necessary to take the diverse interactions within kelp forests into consideration (Tegner and Dayton, 2000; Wallner-Hahn et al., 2015; Metzger et al., 2019) to be able to disentangle kelp forest dynamics and the consequences of their removal.

4.1.2. Competition with understory algae.

The abundance of understory algae, which compete with kelp for light and space (Reed, 1981; Arkema et al., 2009), remained mostly higher in areas where kelp was removed than in unharvested areas over time, even up to 4 to 5 years after intertidal or subtidal kelp removal. The abundance of understory algae did not show a significant general trend over time following the removal of intertidal kelp, which is contrary to the increase in abundance and diversity of most other algal groups, including invasive kelp species, that other studies have found that in the absence of competition from intertidal kelp (Duggins and Dethier, 1985; De Leij et al., 2017; Schiel et al., 2018). On the other hand, the abundance of understory algae decreased slightly over time after the initial increase following the removal of subtidal kelp. The slight decline in understory algae abundance could possibly be due to the few cases in which kelp canopy started recovering, which reduced irradiance despite not reaching pre-harvested canopy height (Clark et al., 2004; Lorentsen et al., 2010). In any case, increasing competition for space with other sessile species (e.g., algae or fauna) and invertebrate herbivore access are likely to influence understory algae abundance (Reed, 1981; Santelices and Ojeda, 1984). Furthermore, as not all functional groups of understory algae responded equally to kelp removal, it could be expected that this trend varies between functional groups over time as well. Understory algae respond differently to shading, for example, some red algae are light adapted while others thrive in the shade (Clark et al., 2004). Canopy-forming algae (e.g., fucoids, understory kelps, invasive kelps), foliose algae and turf-forming algae increased following kelp removal. It is worth noting that understory kelp did not necessarily include recruits or juveniles of the same removed kelp species but also other native species that live in the understory of another kelp's canopy, in particular of *Macrocystis pyrifera*, (e.g., *Ecklonia maxima* in South Africa and *Pterygophora californica* in North America). Therefore, an increase in their abundance does not necessarily imply the future recovery of the removed kelp's canopy.

Further monitoring is needed for algal species or sessile fauna that could potentially replace kelp without fulfilling the same ecosystem functions. Therefore, while our results indicate that subtidal understory algae abundance tends to decrease over time after their initial increase following subtidal kelp removal, which could eventually give way to kelp recovery, it will likely depend on the functional groups that are established. Foliose algae and turf-forming algae also have the potential to spread across the rocky seabed, reducing available space for kelp recruits, and without providing a canopy for other species. Turfs, in particular, can be a source of concern because of the multiple regime shifts from kelp-dominated to turf-dominated habitats observed worldwide (Burek et al., 2018; Christie et al., 2019; O'Brien and Scheibling, 2018; Vásquez, 1989a; Reeves et al., 2022). While turf-dominated habitats can shift back to kelp-dominated habitats over time, they could become permanent when the system also experiences increasing temperature, CO_2 or eutrophication (Smale et al., 2013). As turfs do not provide the same 3D complex habitat as kelps, other organisms depending on kelps, such as macroinvertebrate assemblages, would also be impacted (Bué et al., 2020). Other algae can also proliferate and lead to alternative community/habitat states, preventing or delaying recovery (Vásquez, 1989; Camus, 1994; Filbee-Dexter et al., 2014; Burek et al., 2018; O'Brien, 2018; Reeves et al., 2022). Although it was not one of the objectives of this synthesis, we believe exploring the effects of kelp removal on sessile fauna is important because they also compete with kelp for space, and estimating their response would also provide relevant information to understand the community consequences of kelp removal. In general, we did not see full kelp recovery over the monitored time in the available case studies, and other studies indicate that it may be the case over even longer periods (Christie et al., 1998; Levitt et al., 2002; Bularz et al., 2022; Gouraguine et al., 2021). Thus, our results already suggest that commercial exploitation of natural kelp forests – at least with the lethal methods used in most experiments – is not very sustainable. Long recovery times would need to be respected to ensure the sustainability of wild kelp fisheries.

4.2. Differences in kelp abundance following intertidal and subtidal kelp removal

The intertidal or subtidal distribution of the removed kelp species tested as a moderator variable showed a significant influence in the magnitude of the effect of kelp removal on kelp abundance, with an apparent lower absolute mean effect size when the removed kelp species was intertidal than subtidal, estimated by analyzing them separately. However, unlike subtidal kelp, intertidal kelp abundance showed a decreasing trend over time. The differences in kelp abundance following intertidal versus subtidal kelp removal could be partially explained by the interplay of abiotic factors and ecological interactions including herbivory and algal competition, which can differ between temperate intertidal and subtidal systems (Menge, 1983; Underwood and Kennelly, 1990; Moran, 1999; Kuklinski and Barnes, 2008). We expected subtidal kelp abundance to take longer to recover than intertidal kelp as the intertidal zone of rocky shores is characterized by strong fluctuations in abiotic factors, such as temperature, desiccation, and wave action (Thompson et al., 2002). Furthermore, light exposure and variation in environmental stress appear to confer some resistance and resilience to intertidal species since modifications in tissue strength and flexibility of intertidal species have been observed and spore attachment to rocky substrate is not affected by water motion (Demes et al., 2013; Parada et al., 2016; Millar et al., 2021). Moreover, Westermeier et al., (2017, 2019) tested the differences between autumn and summer kelp removals and the responses of intertidal *Lessonia berteroana* and subtidal *Lessonia trabeculata* and found that summer removal was unfavorable for subtidal *L. trabeculata*, while intertidal *L. berteroana* showed similar trends after autumn and summer removals. These results suggest that the subtidal species could be more prone to be limited by environmental seasonal variations such as nutrient depletion in summer, insufficient light and temperature to promote growth in winter, and natural blade erosion (i.e., biomass loss) caused by water motion that intensifies in spring and summer (Tala and Edding, 2005; Hurd et al., 2014; Westermeier et al., 2017). However, our results showed a stronger impact of intertidal kelp removal over time, suggesting that despite the tolerance of intertidal kelp to some environmental stress, other factors, such as herbivory and algal competition could likely inhibit the recovery of intertidal kelp, which was not observed during the monitoring time considered in our analyses. There may be differential predation of herbivores in intertidal versus subtidal systems which could enable herbivory to increase. For instance, in intertidal communities, some predators may be less effective, or gulls may more easily eliminate predators feeding on herbivores, and herbivores can find refuge along their vertical distribution (Menge, 1983; Petraitis, 1987; Ellis et al., 2007; Perez et al., 2009). Additionally, a difference in the response of understory algae abundance following the removal of intertidal or subtidal kelp was observed, with a higher mean effect size after intertidal kelp removal. Indeed, intertidal zones could be more prone to the establishment of invasive species replacing native kelp. For example, the invasive Asian kelp *Undaria pinnatifida* has an intertidal distribution, competing with native species such as *Laminaria digitata* and *Durvillaea* spp., depending on the location of the study (Valentine and Johnson, 2003; De Leij et al., 2017; Schiel et al., 2018), therefore posing an added threat after the removal of intertidal kelp. While these ecological interactions could partly explain these differences, further research would help to understand the stronger decrease in intertidal kelp, and the interplay of biotic and abiotic factors must be considered, as extreme climatic events are expected to increase (Kerr, 2011; Babcock et al., 2019).

4.3. Other factors influencing the magnitude of kelp removal effects

In this study, we could not test if the season of kelp removal influenced the magnitude of kelp removal effects on kelp abundance in part because not all studies recorded the season in which the kelp was removed, and the sample sizes of each season we could obtain from the studies that recorded this information were unbalanced. However, we believe that this factor could influence the effect size of kelp removal on kelp abundance as several studies have found different recruitment times and rates depending on the season of harvest, leading to different recovery patterns (Kennelly, 1987a; Engelen et al., 2011; Westermeier et al., 2014, 2017, 2019; Castorani et al., 2018). We could not test if the method of kelp removal (lethal or non-lethal) influenced the magnitude of the effect on kelp abundance either because most experiments removed kelp using lethal methods (e.g., cutting the stipe just above the holdfast, removing the entire kelp plant from the rocky substrate, trawling). Nevertheless, the potential differences in the magnitude of consequent kelp removal effects should be further investigated, as different outcomes depending on the method of removal, which can depend on kelp species, have also been described in experimental studies (Levitt et al., 2002; Rothman et al., 2007; Borras-Chavez et al., 2012; Westermeier et al., 2017; Ulaski et al., 2020). Possible differences in magnitude depending on the size of the removal area, and on the proportion that the removal area represents within a kelp forest patch, should also be considered. Responses to kelp removal and the recovery of kelp and of associated taxa could also be influenced by the morphology of the harvested species (Vásquez and Santelices, 1990), something that could not be assessed in this synthesis either.

Future studies should estimate the potential differences across species in their response to removal or to identify possible fast or slow recovery species, something we did not have enough data to assess (Table S2). Kelp recovery and that of associated taxa is likely to vary depending on the harvested species (e.g., their growth rate and other biological traits), as well as on the environmental conditions linked to their geographic location, as this influences their metabolic rates and therefore could influence their rebound capacity (Wernberg et al., 2020). For example, in Norway, stipe growth rate of *Laminaria hyperborea* has been observed to be higher in mid-Norway than in the southern and the northern parts of the country, and mortality decreased and longevity increased with increasing latitude probably because of the lower temperatures (Rinde and Sjøtun, 2005). In Australia, abundance of *Ecklonia radiata* was found to increase significantly with increasing latitude along both the east and the west coast, with reefs in the southern-most regions having greater kelp coverage generally related to cooler water temperature (Marzinelli et al., 2015). Water temperature is one of the most important factors influencing kelp distribution, and with their overall range size predicted to decrease with global warming (Steneck et al., 2002; Sudo et al., 2020). This highlights the importance of estimating kelp removal effects and their magnitude across locations to guide the decisions of the optimal harvesting regimes depending on the species and the region, also considering the persistence of the associated biodiversity. Standardized coordinated distributed experiments would be a valuable tool to further study kelp removal effects considering the diverse factors that could influence the recovery of kelp forests after harvest (Fraser et al., 2013).

4.4. Future directions

The dynamic interactions between kelp, invertebrate herbivores and understory algae that can be inferred from our results are an invitation to further monitor their trends and those of predators, sessile fauna, and other species of kelp communities over longer time periods after kelp removal. In addition, the general decline in kelp abundance should be considered not only from the biomass-available-for-fishery point of view but also as an indicator of seascape modifications and the reduction of ecosystem functions and services, which need to be quantified as well.

The ecosystem functions and services provided by kelp forests, thought of as structurally complex habitats where multiple ecological interactions take place, depend strongly on their abundance. Changes in the abundance and distribution of kelps dramatically reduce ecosystem functioning (Edwards et al., 2020). In addition to fisheries production, kelps provide a suite of ecosystem services including coastal defense, carbon sequestration and nutrient cycling, as well as cultural and hedonic services (Smale et al., 2013; Vásquez et al., 2014; Bennett et al., 2015; Thurstan et al., 2018; Eger et al., 2023). Given the numerous ecosystem functions and services linked to kelp forests, it is necessary to also monitor variables that are representative of the diverse roles they fulfill, considering not only a utilitarian point of view but also a habitat and seascape component perspective, thus reconciling utilitarian and non-utilitarian approaches (Loreau, 2014). Some of the ecosystem functions and seascape characteristics of kelp forests have been studied, such as habitat provision, primary production, ecosystem respiration, and seascape configuration and connectedness, which influence fish populations and other species (O'Connor and Anderson, 2010; Sievers et al., 2016; Reeves et al., 2022). However, other functions that are highly relevant in kelp forest trophic webs, have seldom been studied at all, such as their nursery functions (Lefcheck et al., 2019). Overall, there is a need to imitate studies of deforestation effects on land, which include multiple approaches, from local to global, assessing the impacts on wildlife and biodiversity as well as on humans, landscape, and climate, and on the long term (Sukumar et al., 1992; Laurance et al., 2000; Vogelmann et al., 2009; Lawrence and Vandecar, 2015; Betts et al., 2017; Decaens et al., 2018). This would allow to guide fisheries management and conservation efforts.

Although we did not focus on kelp removal effects on predators in this study, and as we were not able to properly quantify the effects on diversity, we did notice a paucity of information on how other interactions are modified by kelp removal and how it may alter the capacity of kelp forests to fulfill other roles other than fisheries production. As diversity is generally linked to ecosystem function and resilience (Risser, 1995; Isbell et al., 2015; Duffy et al., 2016), we initially sought to quantify how diversity (e.g., species richness, Shannon index) was influenced by kelp removal, but found scarce studies that provided the necessary primary data, comparable across studies, for a meta-analysis. Quantifying biodiversity metrics and how they are influenced by kelp removal is important if we want to further understand the impacts of kelp removal on their community and seascape in a comprehensive way. Indeed, community-level effects recorded in the literature include, for example, increased mortality of the invertebrate fauna within the holdfasts, decreasing invertebrate species richness when partially or totally removing the kelp, and an important reduction of the diversity of sessile invertebrates observed after deforestation by sea urching grazing (Vásquez and Santelices, 1990; Graham, 2004; Vega, 2016; Carbajal et al., 2022).

As most of the studies included in this meta-analytical synthesis did not separate kelp abundance by life history stages (e.g., recruits, juveniles, adults), we chose to use total or adult kelp density, biomass or cover as a measure of kelp abundance because adults account for most of the biomass and canopy cover. But assessing the trends of recruit and juvenile kelp would also help to outline the succession patterns following kelp removal. In fact, recruit and juvenile kelp usually follow opposite trends than total or adult kelp abundance, i.e., they increase after kelp removal (Gouraguine et al., 2021; Carranza et al., 2024), but this does not guarantee that they reach adult age, due to other biotic and abiotic factors like overgrazing, competition and climate events that cause natural dislodgment. Long-term studies of demographic dynamics would allow us to estimate the restocking capacity over time.

After reviewing the literature, we believe it is necessary to further monitor the short- and long-term effects of kelp removal on other species, in particular, the diversity and organisms directly and indirectly linked to kelp. These knowledge gaps strengthen the idea of needing permanent monitoring of a suite of interacting functional groups related to successional processes following harvests or natural deforestation over space and time, which has also been recommended for kelp restocking initiatives (Oyarzo-Miranda et al., 2023), breaking the paradigm of monitoring mainly kelp populations after removal. Furthermore, to be able to continue synthetizing future studies on kelp removal, it is crucial that published authors keep in mind to report the necessary information for meta-analytical syntheses (e.g., mean values, standard deviation or error, and sample sizes for estimating effect sizes based on standardized mean difference such as Hedges' g or Cohen's d, although other effect size measures can also be considered; see Borenstein et al., 2009), as they are valuable tools for estimating global trends for conservation and sustainable fisheries management (Fernandez-Duque, Valeggia, 1994). It is also important to keep in mind that the study of deforestation effects is most likely also being carried out in other areas of the world but were likely not included in this synthesis because of accessibility issues and being reported in other languages; this is likely the case for studies from East Asia (Graham, 2010).

5. Conclusion

This is the first study to test the effects of kelp removal across different geographic locations with a meta-analytical approach. Our study offers new insights into the magnitude of these effects on the abundance of kelp, invertebrate herbivores, and understory algae and how it changes over time. These patterns and the potential interactions between species that can be inferred from our results showed slow recovery of kelp abundance and shifting trends of increasing and decreasing invertebrate herbivore and understory algae abundance following kelp removal. This calls attention to the need for long-term monitoring of kelp removal effects and broadening the monitored effects and scales, including other species that interact with kelp and those that have an influence on its abundance and ecosystem functions and services, to consider them in current fisheries regulations, instead of only what guarantees a certain biomass for the next harvest (Carbajal et al., 2022). It is difficult to truly know the damages caused by human-induced kelp removal if they are not being monitored and the information collected to enact regulations seems incomplete. The effectiveness of harvesting extraction criteria in allowing kelp forest recovery, including other taxa associated with kelp, needs to be regularly assessed as well. Respecting fallow-harvest cycles to ensure recovery over time is important, as frequent repeated disturbances to a dominant habitat-forming species such as kelp can outweigh the influence of less frequent but severe disturbances for the surrounding community (Castorani et al., 2018). It is also crucial to leave some forested patches intact within kelp forests so that they can work as species refuge and spore sources and increase the probability of recovery (Fredriksen et al., 1995; Gaylord et al., 2004; Reed et al., 2006; Oyarzo-Miranda et al., 2023); and to determine and identify which kelp populations are source populations, to keep those forests intact to help lessen the effects of deforestation of surrounding forests. Kelp forest monitoring and fisheries regulations need to be adapted and regulated to ensure conservation and sustainable fisheries, by considering the diverse components and processes of kelp forests and their changes over time, allowing us to look at the system in a more comprehensive way.

CRediT authorship contribution statement

Carranza Daniela M.: Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Validation, Visualization, Writing – original draft, Writing – review & editing. **Stotz Gisela C.:** Data curation, Formal analysis, Methodology, Supervision, Validation, Writing – review & editing. **Vásquez Julio A.:** Validation, Writing – review & editing. **Stotz Wolfgang B.:** Conceptualization, Supervision, Validation, Writing – review & editing.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

The data that support the findings of this study are available from the corresponding author upon request.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.gecco.2024.e02805.

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