Assessing invasion impact: survey design considerations and implications for management of an invasive marine plant

Barrie M. Forrest* & Michael D. Taylor
Cawthron Institute, Private Bag 2, Nelson, New Zealand;
*Author for correspondence (e-mail: barrie@cawthron.org.nz; fax: +64-3-5469464)

Abstract

We use a three-year study of sheltered low shore assemblages colonised by the non-indigenous Asian kelp *Undaria pinnatifida* to explore survey design issues for assessing the ecological impacts of invasive species. The weight of evidence overall suggested little impact from *Undaria* on low shore assemblages, with control–impact contrasts that could plausibly be interpreted as impacts probably reflecting natural causes. We demonstrate that the potential for reaching incorrect conclusions regarding the impacts of invasive species using control–impact designs is greater than when such designs are used to assess traditional forms of anthropogenic impact. We suggest that a before–after control–impact framework is essential, but recognise that such an approach has a number of limitations. In particular, there is no assurance that the before–after impact site will be invaded at all, or to the extent that provides worst-case impact information for coastal managers. We discuss possible ways of assessing invasive species impacts, but suggest that the uncertainty inherent in extrapolating impact information to other places and times means that the precautionary principle should be applied, and 'worst-case' impacts assumed, until the level of scientific uncertainty is reduced. Such an approach should only be applied, however, after an evaluation of the feasibility, costs and benefits of managing a particular pest in relation to other priorities for invasive species.

Introduction

The nature and severity of impacts caused by invasive species, and the relative effects of one species over another, will be key considerations in setting management priorities for them. Comparative studies (e.g. Findlay et al. 2000), local-scale field surveys (e.g. Windham 1999), long-term data sets (e.g. Howe et al. 1997) and various experimental approaches (e.g. Olsen et al. 1991; Floc'h et al. 1996) have all been used to describe the effects of invasive species and identify mechanisms that may lead to significant impacts. For many invasive species, however, and for invasive marine species in particular, unequivocal evidence of impacts is generally lacking, even for those considered a significant threat (Blossey 1999; Parker et al. 1999).

Rather, the literature for many such species is primarily dominated by accounts of only their occurrence and spread. While evidence of impacts remains equivocal and largely speculative, rational management decisions cannot be made, and dissenting views from scientists are likely (Peterson 1993; Blossey 1999).

The Asian kelp, *Undaria pinnatifida*, typifies this situation. *Undaria* is a large (1–2 m length) canopy-forming species that can reach high densities in both artificial and natural habitats (e.g. Hay and Villouta 1993). It is considered a potential fouling nuisance (Sanderson 1997; Fletcher and Farrell 1999), and a threat to natural ecosystems and associated fisheries, for example through displacement of native species via the development of 'mono-specific' *Undaria* stands (Sanderson and Barrett 1989; Miller et al. 1997;

Stuart 1997; Battershill et al. 1998). While its basic biology (summarised in Sanderson and Barrett 1989), spread (e.g. Hay 1990; Sanderson 1990; Fletcher and Manfredi 1995; Casas and Piriz 1996; Forrest et al. 2000), population dynamics (Hay and Villouta 1993; Brown and Lamare 1994; Castric-Fey et al. 1999), and physiology (Campbell et al. 1999) are quite well understood, information on impacts is limited, and often speculative and polarised (e.g. Rueness 1989; Parsons 1994; Battershill et al. 1998; Miller et al. 1997; Stuart 1998; Walker and Kendrick 1998; Sinner et al. 2000).

Battershill et al. (1998), for example, made spatial comparisons of ecological assemblages in areas with *Undaria* at different infestation levels, with those dominated by native *Carpophyllum* spp. They suggested that significant ecological changes to the *Carpophyllum* sub-canopy community resulted from *Undaria*'s establishment, and concluded that *Undaria* may displace multi-species macroalgal communities characterised by *Carpophyllum*. In contrast, Hay and Villouta (1993), with reference to the same general locality, suggested that *Undaria* colonised bare areas outside beds of native *Carpophyllum*, rather than the beds themselves. Similarly, Hay and Sanderson (1999) considered that there was very little evidence that *Undaria* displaced native brown seaweeds in several New Zealand harbours where it had been established for many years.

In the climate of uncertainty regarding *Undaria*'s impacts, a precautionary approach to the seaweed's management in New Zealand has been advocated by some regional and central government agencies. In contrast, many private stakeholders (e.g. vessel operators, marine farmers), for whom *Undaria* management costs (e.g. for regular hull de-fouling) could be significant, are reluctant to be drawn into a management strategy when adverse effects have not been documented and hence the benefits of management are unclear.

The example of *Undaria* thus highlights a considerable need for defensible information on impacts. In the studies referred to above, the lack of a pre-invasion baseline, and hence the associated uncertainty regarding the level of ecological change caused by *Undaria*, clearly contributed to the dissenting opinions on impacts and the need for management. The limitations of control–impact surveys in studies of the effects of anthropogenic pollution have been recognised for some time, and the advantages of establishing baselines and inferring impacts based on before–after control–impact (BACI) designs and their variants have been widely promoted (e.g. Green 1979, 1993; Stewart-Oaten et al. 1986; Underwood 1991–1994).

This paper describes a three year investigation of rocky low shore assemblages in a sheltered New Zealand harbour, and examines the efficacy of BACI and control–impact designs in assessing *Undaria*'s impacts. We also consider the utility of these survey designs in assessing the effects of invasive species generally, and identify a number of areas where their application has significant limitations when compared with their more traditional use in anthropogenic impact studies.

Methods

Study sites and sampling

Our investigations were conducted in the low neap-spring tide zone at four sites in Lyttelton Harbour, New Zealand (Figure 1), in algal-dominated habitats consisting of stable boulders and bedrock. A combination of small tidal range (~2 m), moderate shore slope, and poor water clarity, confined *Undaria* to a narrow band (typically 1–3 m wide) in this zone. Sites consisted of: one infested locality (Cass Bay) where *Undaria* was already established; one uninfested locality (Diamond Harbour) which became infested during the study (as we had anticipated); and two uninfested control locations (controls 1 and 2) that were isolated from known vector pathways and beyond the likely range of natural spread via spore dispersal (Forrest et al. 2000).

In its native range *Undaria* is an annual species exhibiting a strong seasonal hiatus between the sporophyte which is dominant in spring, and the microscopic gametophyte that is present over late summer and autumn/fall during sporophyte senescence (Akiyama and Kurogi 1982). While such a marked seasonality is less evident in New Zealand, larger sporophytes are nevertheless more prevalent during late winter and spring (Hay and Villouta 1993), suggesting some potential for a seasonal difference in impact. To account for such possibilities, surveys at each of the four sites were carried out in spring (September–November) and autumn/fall (March–May), for the three years from spring 1997 to autumn 2000.
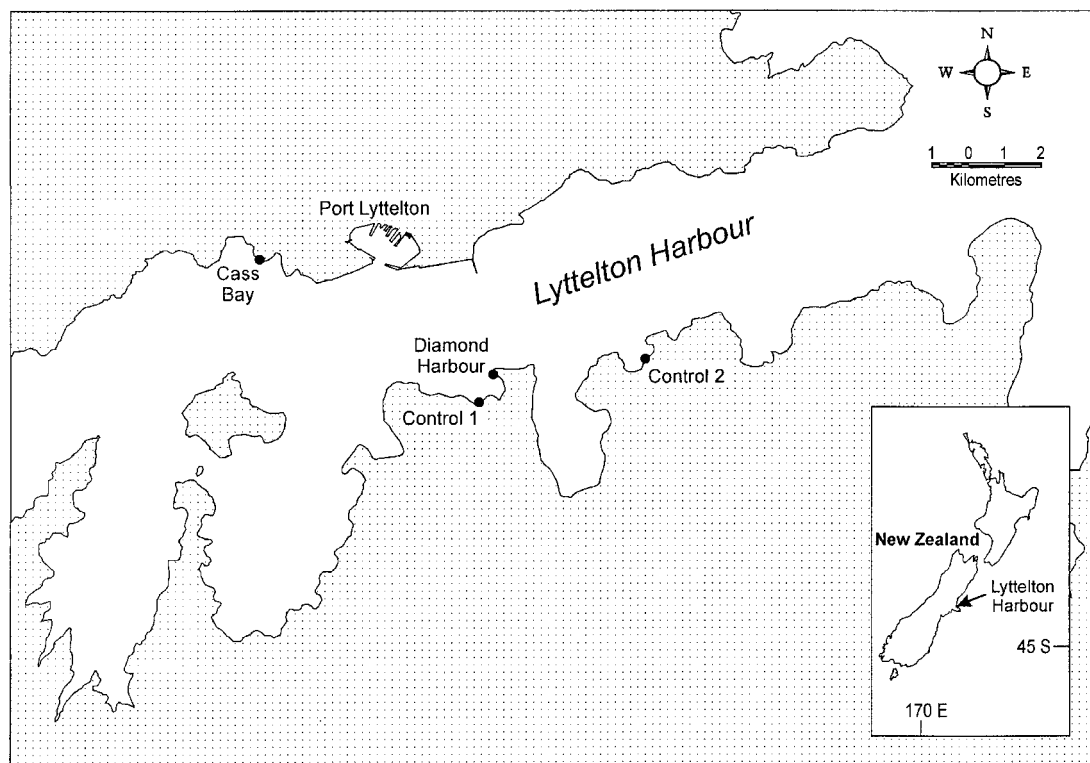
Figure 1. Map of Lyttelton Harbour, New Zealand, showing the four study sites.

Sampling was undertaken using transect and quadrat methods. Two long-shore transects (50 m length) were sampled at each site: one along the neap tide level corresponding to *Undaria*'s upper limit on the shore and one at the level of low spring tide where *Undaria* was most prevalent. Point-sampling on each transect was conducted at 80 randomly generated distances. Macroalgae, sessile invertebrates, or bare rock falling beneath each of the 80 points were recorded. Between the 2 transects (i.e. spanning the neap-spring tide zone) 8 quadrats (0.25 m^2) with 80 mm grid spacings were placed at pre-determined random distances. The number of *Undaria* within each quadrat was determined, and macroalgae, sessile invertebrates, or bare rock falling beneath each of the 49 intercept points formed by the grid were recorded. The time constraints of low shore sampling and the limited number of suitably low spring tides meant that only the canopy level of substratum cover could be sampled using this method. Changes to sub-canopy assemblages are nevertheless of interest in terms of assessing the ecological effects of *Undaria*, hence we also recorded (presence/absence) the conspicuous taxa in the quadrats that were not detected by the point count method.

Point counts generated from both the transect and quadrat sampling were later converted to percent cover. Taxon richness data were derived from the total number of different taxa recorded within quadrats irrespective of sampling method (point intercept, counts and presence/absence). Taxonomic identification in the field was made to species level where practicable, but voucher specimens collected as necessary.

Statistical analyses

A control–impact inference structure was based on planned comparisons between Cass Bay and the control sites, since Cass Bay was infested with *Undaria* from the outset. At Diamond Harbour, where *Undaria* was first recorded in spring 1998, there were two 'before' sampling times (spring 1997–autumn 1998) and four 'after' sampling times (spring 1998–autumn 2000). Thus, the inference structure was based on a BACI design and used the following planned comparisons: 'before' at Diamond Harbour *versus* 'after' at Diamond Harbour; 'before' at the two control sites *versus* 'after' at the controls; 'before' at Diamond Harbour *versus*

'before' at the controls; and 'after' at Diamond Harbour *versus* 'after' at the controls. Hence this BACI structure at Diamond Harbour also provided a 'control–impact (after)' contrast for direct comparison with the Cass Bay situation.

For univariate analyses (ANOVA and Pearson correlation), data were entered into SAS (SAS/STAT 1997) and $\log(X + 1)$-transformed (where necessary) prior to analysis to satisfy the independence and normality of error terms assumptions of the general linear model. Data were analysed using the MIXED procedure with site, sampling time and their interaction term included as main fixed effects. Quadrat and transect (spring and neap tide) were declared random effects nested within site, and evidence for quadrat effects and serial correlation (AR 1) were investigated using the restricted maximum likelihood (REML) method.

Multivariate analyses of quadrat data (pooled within each site and survey) were undertaken with the software package PRIMER V5, to examine spatio-temporal patterns in community composition. The dataset was derived by weighting each taxon by the number of quadrats in which it was recorded for any one site and survey, thus providing a measure of relative abundance on a 0–8 scale. For example, *Undaria* was recorded in six out of eight quadrats in spring 1997 at Cass Bay so is scored as six. Using this dataset, a two-dimensional non-metric multi-dimensional scaling (nMDS) ordination was produced from a Bray–Curtis similarity matrix. Using group average clustering, site groups that formed at a 60% Bray–Curtis similarity threshold were superimposed on the nMDS ordination pattern (Clarke 1993). The SIMPER procedure (Clarke 1993) was used to identify the major taxa contributing to the site groups, and one-way ANOSIM (Clarke 1993) used to examine the control–impact and BACI contrasts described above. Bray–Curtis similarity measures for pairwise combinations of sites were examined to describe temporal trajectories in site similarity.

Results

Undaria *infestation levels and impacts on canopy species*

Temporal changes in *Undaria* infestation levels did not follow any consistent seasonal pattern, in contrast to our expectations. The percent cover of *Undaria* in quadrats (Figure 2) and along transects (Figure 3)
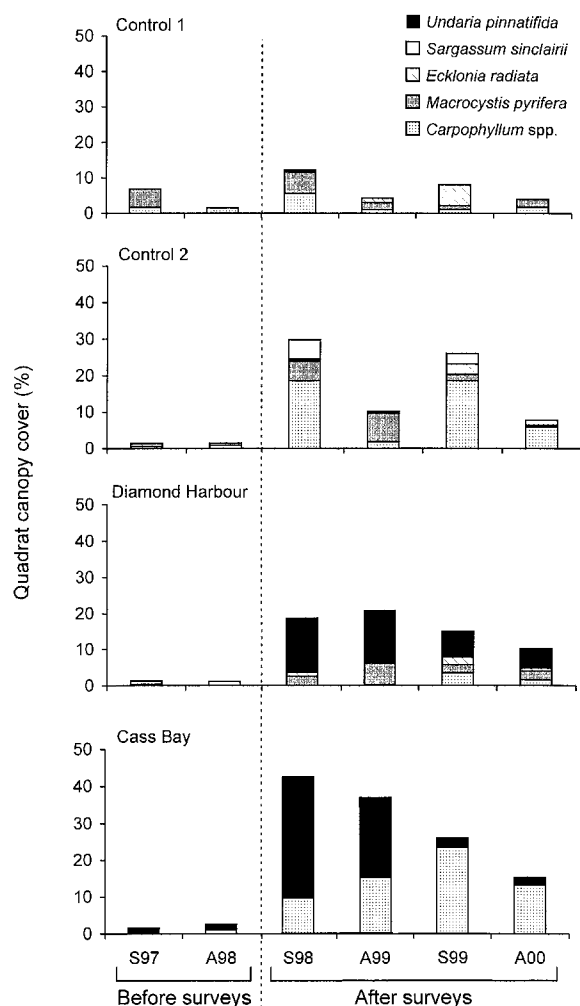

Figure 2. Mean percent cover of *Undaria* and other canopy-forming seaweeds within quadrats (0.25 m²) over the six surveys from spring 1997 (S97) to autumn/fall 2000 (A00). *Undaria* was first recorded at Diamond Harbour in spring 1998.

was greatest at both Cass Bay and Diamond Harbour in spring 1998, and steadily declined thereafter. Maximum percent cover levels, as recorded from transects, were approximately 45% and 19% for the two sites, respectively. The density of *Undaria* was notably high at Cass Bay ($\sim$130 sporophytes m⁻²) in spring 1998 but was otherwise less than half of this value, with higher density patches characterised by numerous small or immature sporophytes rather than mature-sized plants.

Native canopy species (defined in this study as *Sargassum sinclairii*, *Ecklonia radiata* and *Macrocystis pyrifera*) covered up to 40% of the substratum
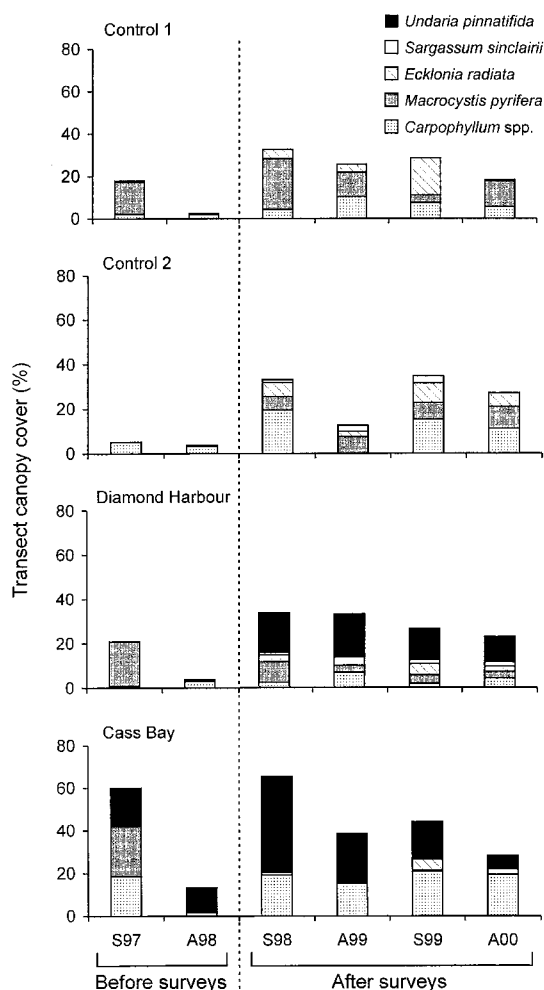
Figure 3. Percent cover of *Undaria* and other canopy-forming seaweeds along transects (data pooled over neap and spring tide level) over the six surveys from spring 1997 (S97) to autumn/fall 2000 (A00). *Undaria* was first recorded at Diamond Harbour in spring 1998.

largely suggesting a 'no impact' result (Table 1). The quadrat percent cover results are equivocal, however, owing to significant random effects. The most interesting contrast was the significantly lower native canopy cover at Diamond Harbour compared with control quadrats 'after' *Undaria* arrived. While displacement of the native canopy could be inferred from this spatial pattern, such an interpretation contrasts with the observation that the cover of native canopy species at Diamond Harbour significantly increased from 'before' to 'after' *Undaria*'s arrival (Table 1, Figure 2). In fact, Pearson correlation revealed a weak positive association ($r = 0.28$, $P = 0.06$) between the cover of *Undaria* and the native canopy at Diamond Harbour, rather than a negative effect. There was little association between these variables at Cass Bay ($r = 0.19$, $P = 0.20$).

Impacts on taxon richness

The richness of both macrofaunal and macroalgal taxa showed a high degree of year to year variation, although the sites showed similar temporal trends (Figure 4). Mean richness levels for macrofauna and algae were reasonably low, ranging from approximately 3–18 and 4–11 taxa per site respectively. There were no significant control–impact or BACI contrasts that would be consistent with the displacement of either macrofaunal or algal species by *Undaria* (Table 1). While algal richness at Cass Bay was significantly ($P < 0.05$) less than the control sites in the overall control–impact contrast (Table 1), this result does not appear to reflect an impact of *Undaria*, since algal richness was greater at Cass Bay than the controls on a number of occasions, including spring 1998 when the percent cover of *Undaria* was greatest. In fact, Pearson correlation provided evidence for a positive association between macroalgal richness and *Undaria*'s percent cover ($r = 0.24$, $P = 0.10$) and density ($r = 0.33$, $P = 0.02$) at the Cass Bay site. Similarly, algal richness at Diamond Harbour exhibited a strong positive correlation with *Undaria*'s percent cover ($r = 0.39$, $P = 0.006$) and density ($r = 0.49$, $P = 0.0004$).

Impacts on assemblage composition

The grazing snail *Turbo smaragdus* was common at all sites, but substratum cover outside the primary canopy was dominated by macroalgae – notably articulated corallines, *Ralfsia verrucosa*, *Cystophora* spp.,

and consisted primarily of *Carpophyllum maschalocarpum*, although juvenile *Ecklonia radiata* and *Macrocystis pyrifera* were sometimes more dominant (Figures 2 and 3). As for *Undaria*, a greater canopy cover was generally recorded along transects than in quadrats (e.g. spring 1997). In part this will reflect the placement of the spring tide transects in the lowest accessible part of the intertidal zone where algal cover was very high compared with the area between spring and neap where the quadrats were positioned.

There was no evidence for displacement of the native canopy by *Undaria*, with planned contrasts of percent cover between the controls and each of the infested sites

Table 1. Summary of mixed model analyses of variance for the control *versus* impact and BACI designs. *P*-values are shown with numerator/denominator degrees of freedom for fixed effects and 95% confidence limits [L_1, L_2] for random effects. AR 1 refers to serial correlation within random effects.

Survey design	Canopy cover (quadrats)	Canopy cover (transects)	Algal richness (quadrats)	Faunal richness (quadrats)
Control–impact				
Fixed effects				
Time	<0.001, 5/104	<0.001, 5/15	<0.001, 5/104	<0.001, 5/104
Site	0.244, 2/21	0.942, 2/3	0.013, 2/21	<0.001, 2/21
Time*site	0.02, 10/104	0.127, 10/15	0.372, 10/104	<0.001, 10/104
Cass Bay vs. controls	0.634, 1/21	0.831, 1/3	0.020, 1/21	0.160, 1/21
Random effects				
Quadrat/transect (site)	0.017, [0.01, 0.11]	0.127, [−0.08, 0.03]	0.135, [−0.19, 1.37]	0.963 [−0.74, 0.78]
AR 1 Quadrat/ transect (site)	0.005, [−0.54, −0.09]	0.517 [−0.61, 1.19]	0.975, [−0.28, 0.27]	0.159 [−0.41, 0.07]
BACI				
Fixed effects				
Time	<0.001, 5/104	<0.001, 5/15	<0.001, 5/104	<0.001
Site	0.032, 2/21	0.879, 2/3	0.060, 2/21	<0.001
Time*site	0.062, 10/104	0.096, 10/15	0.839, 10/104	0.001
Diamond Hbr vs. controls (before)	0.534, 1/104	0.321, 1/15	0.159, 1/104	0.454, 1/104
Diamond Hbr vs. controls (after)	0.019, 1/104	0.216, 1/15	0.616, 1/104	0.011, 1/104
Before vs. after: Diamond Hbr	0.019, 1/104	0.114, 1/15	<0.001, 1/104	0.001, 1/104
Before vs. after: controls	<0.001, 1/104	<0.001, 1/15	<0.001, 1/104	0.002, 1/104
Random effects				
Quadrat/transect (site)	0.044, [0.0, 0.08]	0.544, [−0.07, 0.13]	0.106, [−0.11, 1.13]	*
AR 1 Quadrat/ transect (site)	0.094, [−0.44, 0.03]	0.3, [−0.3, 0.96]	0.624, [−0.34, 0.21]	0.767 [−0.24, 0.18]

*Variance estimate = 0, however, *P*-value and confidence limits not calculable.

Hormosira banksii, and *Gelidium caulacantheum*. The cover of bare rock and sessile macrofauna outside the primary canopy was typically <20%, and was particularly low (or zero) at most sites in spring 1998. This not only reflected the arrival of *Undaria* at Diamond Harbour and the marked increase in its percent cover at Cass Bay, but also a far greater cover of other macroalgae at all sites in spring 1998 compared with other times. As was the case with the univariate measures above, the multivariate analyses of low shore assemblage composition provide no evidence of an ecological impact that could be attributed to *Undaria*'s invasion.

The nMDS site/survey ordination discriminated five groups of sites having a within-group Bray–Curtis similarity of approximately 60% (Figure 5). The infested Cass Bay site formed a distinct group for all six surveys. In spring 1998, each of the Diamond Harbour and two control sites formed individual clusters, while for all other surveys these sites formed a single group (hereafter referred to as the Diamond Harbour/control group). One-way ANOSIM revealed significant differences in composition between Cass Bay and the controls ($R = 0.535$, $P < 0.05$), but all BACI contrasts at Diamond Harbour were non-significant ($R = -0.036$–0.25, $P > 0.05$). Hence, from the two infested sites, opposing conclusions could be drawn from the ANOSIM results regarding the impacts of *Undaria*.

SIMPER analysis revealed that the Cass Bay group was primarily discriminated from the Diamond Harbour/control group by the relative dominance of *Undaria* and to a lesser extent *Gelidium*, and the relative paucity of *Cystophora* (Table 2). However, *Undaria*'s contribution to the average measure of dissimilarity between the two groups was low (~5%). As such, the ordination pattern that resulted when *Undaria*
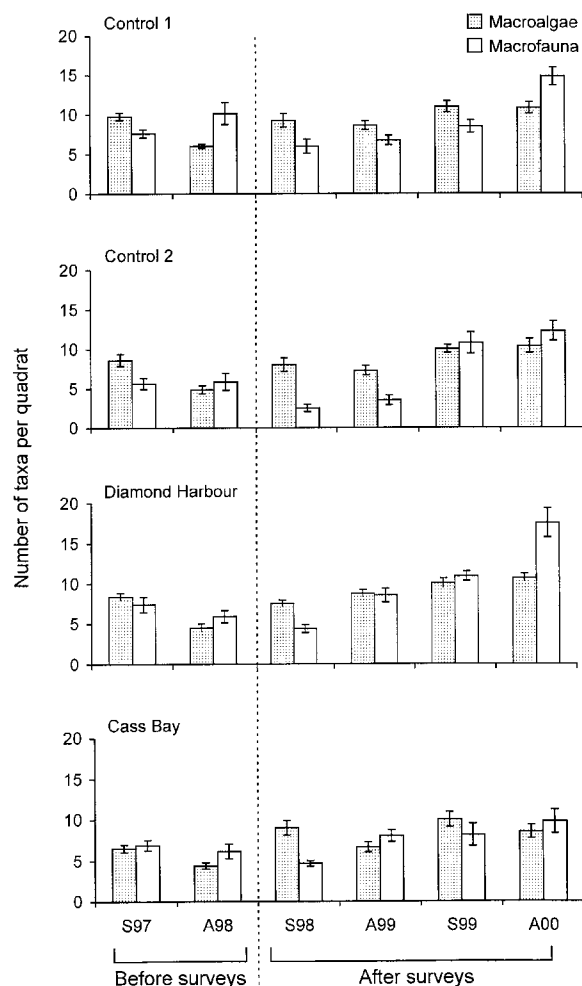
Figure 4. Mean number (±1SE) of macroalgal and macrofaunal taxa within quadrats (0.25 m²) over the six surveys from spring 1997 (S97) to autumn 2000 (A00). *Undaria* was first recorded at Diamond Harbour in spring 1998.
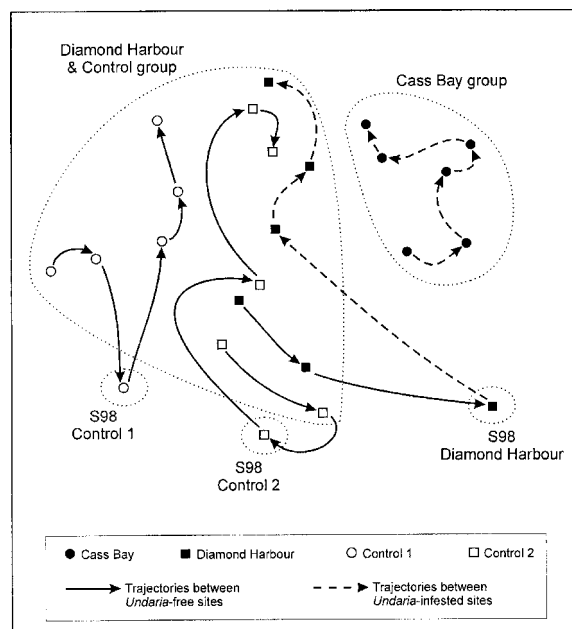

Figure 5. nMDS ordination (stress = 0.21) showing trajectories in assemblage composition at each of the four sites over the six surveys from spring 1997 to autumn 2000. The cluster analysis overlay indicates five groups of sites (encircled by a dotted line) having a within-group Bray–Curtis similarity of approximately 60%. *Undaria* was first recorded at Diamond Harbour in spring 1998.

was omitted from the data was strikingly similar to that shown in Figure 5, indicating that *Undaria*'s presence in the analysis does not mask other spatio-temporal patterns in the assemblage.

The site ordination trajectories (Figure 5) and the temporal trend of Bray–Curtis similarity scores for pairwise comparisons of sites (Figure 6) show a convergence in site similarity over time. While the time of greatest divergence of Cass Bay from the controls occurred when *Undaria* was most abundant there in spring 1998, this was also a time when dissimilarity among the two controls was relatively high. A marked spatial separation of Diamond Harbour was also evident at this time (Figure 5), coinciding with *Undaria*'s

first appearance there. Despite the fact that *Undaria* was reasonably prominent (up to 22% cover), however, more important determinants of the dissimilarity in spring 1998 were the dominance of the rhodophytes *Asparagopsis armata* and *Myriogramme denticulata* (Table 2). Hence, differences among sites in spring 1998 appeared to be a general phenomenon, rather than a pattern solely attributable to the proliferation of *Undaria* at infested sites.

Discussion

Our three year study of low shore assemblages in a sheltered New Zealand harbour has provided no evidence of significant ecological impacts from the invasion of *Undaria*. While impacts could be inferred from the differences between the infested Cass Bay site and the controls, our findings suggest that these differences reflect underlying spatio-temporal variation rather than effects from *Undaria*.

These results, and apparent effects such as the positive association between *Undaria* cover and algal richness, contradict what might be predicted, but are

Table 2. Summary of SIMPER analysis showing individual and cumulative contribution of the 10 most important taxa (rank 1 = most important) to average measures of dissimilarity between the Diamond Harbour/control group compared with the other site groups shown in Figure 5.

Taxon	Cass Bay group		Spring 1998, DH		Spring 1998, C1		Spring 1998, C2	
	Rank	Percent	Rank	Percent	Rank	Percent	Rank	Percent
Asparagopsis armata			1	6.1	1	4.3		
Aulacomya ater maoriana			6	(3.3)			1	(5.0)
Bryozoa (encrusting)	8	(2.4)			10	(2.3)		
Carpophyllum maschalocarpum	7	2.8					5	3.1
Ceramium spp.			5	3.4				
Chiton pelliserpentis							6	(3.2)
Cladophoropsis herpestica					7	2.6		
Cnemidocarpa bicornuata	9	2.4						
Codium dimorphum	10	(2.3)					10	(2.6)
Colpomenia spp.			8	(2.6)				
Cystophora distenta							8	2.9
Cystophora scalaris	3	(3.5)	9	(2.5)	4	(3.7)		
Ecklonia radiata			7	2.7	3	3.9		
Elminius modestus	6	(2.5)	10	(2.4)	5	(2.7)	7	(3.0)
Gelidium caulacantheum	2	3.6			8	(2.5)		
Hormosira banksii			4	(3.8)	2	(4.3)	2	(4.7)
Micrelenchus sp.	4	2.9						
Myriogramme denticulata	5	2.8	2	4.5			3	3.9
Mytilus galloprovincialis					6	2.6		
Ralfsia verrucosa					9	(2.3)		
Sargassum sinclairii							4	3.4
Trochus viridis							9	2.7
Undaria pinnatifida	1	4.9	3	4.0				
Average dissimilarity (%)		48.4		58.9		48.6		49.7
Cumulative percent contribution		29.9		35.4		31.2		34.4

Numbers outside brackets indicate situations where group discrimination was based on the specified taxon being less dominant in the Diamond Harbour/control group, whereas numbers inside brackets indicate the opposite. DH = Diamond Harbour, C1 = control 1, C2 = control 2.
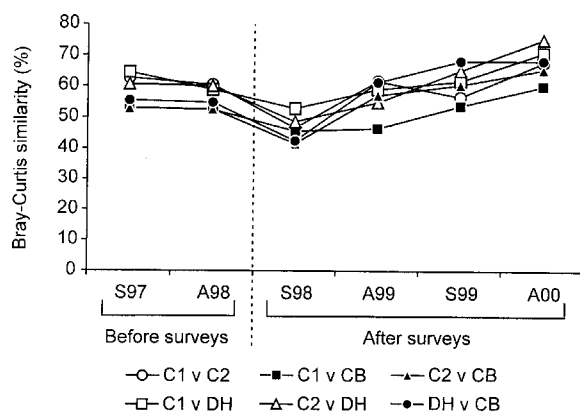

Figure 6. Trajectory of Bray–Curtis similarity values for pairwise combinations of the four sites over the six surveys from spring 1997 (S97) to autumn 2000 (A00). C1 = control 1, C2 = control 2, DH = Diamond Harbour, CB = Cass Bay. *Undaria* was first recorded at Diamond Harbour in spring 1998.

nonetheless plausible in this situation. For example, the increased canopy cover resulting from *Undaria*'s infestation could enhance sub-canopy low shore algal populations by providing greater shelter from dessication at low tide, as has been discussed in other studies (e.g. Leonard 1999; de Figueiredo et al. 2000). The fact that *Undaria*'s first appearance at Diamond Harbour and its proliferation at Cass Bay were associated with significant changes in the low shore assemblage (especially the algae) at all sites, suggests that *Undaria* was responding to the same favourable environmental variables as other species, thus tracking as opposed to causing the changes observed.

The lack of clear evidence of ecological impacts at *Undaria*-infested sites may partly reflect the fact that these areas already had an assemblage of canopy-forming species, albeit not spatially dominant. Although *Undaria* provided an addition to this, its level

of infestation would not have altered the physical structure of the habitat to the extent that might be expected from the formation of an enclosed canopy (e.g. Jenkins et al. 1999; Leonard 1999). It follows that dramatic changes to the structure and function of the resident assemblage would not necessarily be expected. Greater apparent ecological impacts from *Undaria* (Battershill et al. 1998) and marsh plants (e.g. Daehler and Strong 1996; Posey 1988), have been described where the invasions have occurred in relatively barren habitats. Battershill et al. (1998), for example, suggested that there was an increase in sub-canopy species diversity inside *Undaria* patches at shallow subtidal sites that had previously been largely devoid of native macroalgae.

Our conclusion of no appreciable impact, especially for Cass Bay, is weakened by the absence of 'before' data. In contrast, the pre-infestation baseline for Diamond Harbour greatly strengthened the inference we could make about *Undaria*'s impacts at that site. If, for example, the last four surveys at Diamond Harbour were analysed in isolation as part of a control–impact study, a plausible conclusion would have been that the cover of native canopy-forming algae at that site was 'reduced' by *Undaria*. The inherent assumption that underlies this conclusion (and seems quite reasonable) is that four surveys (i.e. two years) of control site data are representative of the natural range in levels of native canopy cover. In fact the native canopy cover significantly increased at both Diamond Harbour and the control sites from 'before' to 'after' the arrival of *Undaria*.

Our study of *Undaria* has thus reaffirmed the importance of a number of key survey design elements that have been widely promoted for studies of anthropogenic impacts, including the need to establish baselines, and incorporate temporal and spatial replication of control and impact sites. In reality, however, many studies of anthropogenic impact default to less ideal designs. The multiple control *versus* single impact site approach, for example, is still relatively common in pollution monitoring but can nevertheless provide convincing evidence for (or against) ecological effects (e.g. Smith 1994; Chapman et al. 1995; Roberts and Forrest 1999; Hindell and Quinn 2000).

This more simplistic approach could have been highly misleading in our study of *Undaria*, raising a question as to the necessary survey design requirements for investigating the ecological impacts of *Undaria* or in fact marine invaders generally. If it is assumed that worst-case impacts are of primary interest to managers,

then control–impact designs are an appealing prospect, since a site (or multiple sites) of greatest infestation can be targeted and results produced within a short time-frame. The weak inference structure provided by control–impact designs is clearly an issue with invasive species studies, however, especially where infestation levels are patchy as was the case for *Undaria* in this study. When the underlying causes of patchiness are unknown, the validity of any assumption that the control sites are invadable at all, or to the same degree as the impact sites, is questionable. Temporal replication, coupled with an evaluation of ecological changes associated with changing infestation levels over time does not adequately solve this problem. In the same way that the level of invadability may change spatially, it may also change over time as a result of external factors that similarly drive changes in the associated community. In both cases, questions of invadability and ecological impacts are confounded.

Where control–impact designs include temporal replication there are also practical issues to consider. Ensuring that control sites remain uninvaded for the duration of a study may be problematic, since it requires that they be selected from areas beyond the predicted dispersal range of a given invader (unless regular removal of new arrivals is an option). Where this leads to wide spatial separation between the impact and control sites (e.g. the invader has a lengthy planktonic larval stage), the controls are more likely to be subject to different environmental conditions and thus differ markedly from the impact locations at the outset, or follow different trajectories over time. In the current study, a few *Undaria* plants were discovered in the vicinity of both controls towards the end of the programme, but the founding populations either disappeared again or had not established along our transects before the completion of the study.

While baseline data for potentially infested sites appears critical in invasive species studies, the *a priori* prediction of areas of future worst-case infestation may be particularly difficult, even with a good understanding of invasion processes. In the present study, we successfully identified appropriate controls and an area of future infestation using knowledge of *Undaria*'s natural and human-mediated dispersal mechanisms. However, we were probably unsuccessful in describing the seaweed's worst-case effects at a harbour scale, since a subsequent infestation at a nearby reef appeared considerably more significant than at our two infested study sites.

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In light of such limitations, it is clear that the current study would have benefited greatly by the inclusion of *Undaria* removal experiments from plots within heavily infested sites. By also including heavily infested plots that were not cleared, this approach would have circumvented the question of the invadability of *Undaria*-free areas both spatially and over time. A spatial comparison of the assemblages of cleared plots with uninfested control plots, and evaluation of their trajectories over time, would have provided a valuable insight into the invadability hence utility of the controls.

A complementary approach, though one that may raise ethical concerns, would be to artificially introduce an exotic species (e.g. perhaps one already established in the general locality) to sites where a baseline had been established. Success is not guaranteed with such approaches, however. Floc'h et al. (1996), for example, inoculated the seabed with *Undaria* spores in areas from which native algae had been cleared, but few sporophytes appeared. Such results are not inconsistent with our own observations or artificial inoculation studies in and around the study area and elsewhere in New Zealand (Forrest and Taylor, unpubl. data). While *Undaria* possesses a number of the characteristics of a 'classic' invader (e.g. Fletcher and Manfredi 1995) its invasion patterns do not always reflect this. Even though a single *Undaria* sporophyte can in theory seed a new population, it is not a foregone conclusion that this will happen, or that conditions in the recipient habitat will favour the formation of high density canopy-forming stands.

Clearly, therefore, *Undaria*'s infestation levels and associated effects are likely to vary from place to place, and for reasons that may never be well understood. Hence even with compelling evidence of impacts (or lack of) from one general area or habitat, as *Undaria* spreads to different habitats and invades different assemblage types, the severity of its impacts may change. In terms of managing invasive species like *Undaria*, this caveat must always be kept in mind. Hence while defensible approaches to describing impacts can be developed, and information gathered accordingly, coastal managers and other stakeholders must seriously question the extrapolation of such information to other places and times. On this basis, we suggest that it is necessary to apply the precautionary principle to the management of pest species, and assume 'worst-case' impacts, until the level of scientific uncertainty is reduced. Such an approach should only be applied, however, after an evaluation of the feasibility, costs and benefits of managing the pest in question in relation to other priorities for invasive species.

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